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Arthur N. Popper
Joseph A. Sisneros
Anthony D. Hawkins
Frank Thomsen
*Editors*

# The Effects of Noise on Aquatic Life

## Principles and Practical Considerations

Springer

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Arthur N. Popper • Joseph A. Sisneros •
Anthony D. Hawkins • Frank Thomsen
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## Principles and Practical Considerations

With 635 Figures and 170 Tables

Springer

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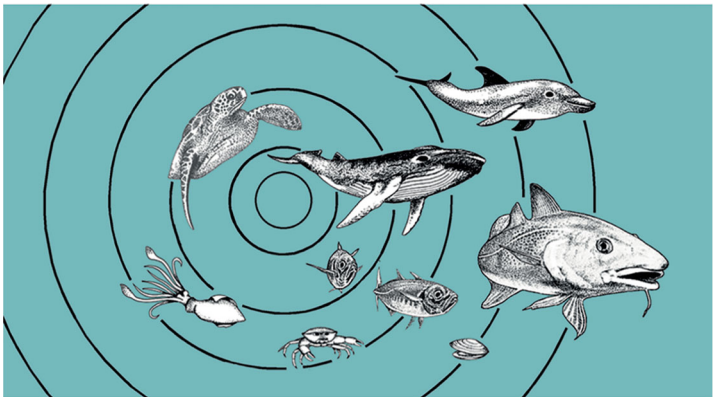
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*This volume is dedicated to our colleague and great friend Professor Anthony D. Hawkins. Professor Hawkins (Tony) has been a pioneer in the study of fish bioacoustics for over 60 years. He has conducted an immense body of research from basic science of hearing to very applied aspects of fisheries biology. But, from the perspective of Aquatic Noise, Tony is co-founder of the set of international conferences on Effects of Noise on Aquatic Life, from which this book is derived. Through these conferences, Tony has had immense impact on the field, and how it has moved forward. While not participating in future conferences, Tony's influence, both as grand scholar and founder of the conferences, will endure.*

# Preface

***The Effects of Noise on Aquatic Life***
**Berlin, Germany, 10-15 July 2022**



*Effects of Noise on Aquatic Life: Principles and Practical Considerations* is a compilation of chapters based on the papers presented at the sixth International Conference on "The Effects of Noise on Aquatic Life" that took place in Berlin, Germany, July 10–15, 2022 (https://aquaticnoise.org/2022/). This conference, generally referred to as AN2022, followed the very successful conferences on the same general topics held in Nyborg, Denmark (in 2007), Cork, Ireland (in 2010—see Popper and Hawkins, 2012), Budapest, Hungary (in 2013—see Popper and Hawkins, 2016), Dublin, Ireland (in 2016), and Den Haag, Netherlands (in 2019).

The primary goal of AN2022, like our past conferences, was to define the current state of knowledge on the impact of underwater noise on aquatic life and to explore the progress made in this field in the three years since the last conference. The continued need and value of these conferences, and the publications that come from them, are a result of ongoing human development that adds a wide range of noise to aquatic environments. These anthropogenic noise sources include (but are not limited to) exploration and extraction of oil and gas in offshore waters, increasing vessel traffic, various military activities, the prospect of industrial development of previously pristine waters (e.g., polar regions), ongoing and recent proposals to open

new areas for marine renewables around the world, as well as the development of other technologies, such as deep-sea mining. In all cases, there is increasing concern over potential impacts of these activities on aquatic species and their habitats

All this work has resulted in greater, and growing, interest in, and need to understand, the effects of underwater noise on aquatic life by investigators, regulators, industry, and the public. Moreover, the need for rigorous examination of the environmental impact of underwater noise has driven new research into the effects of anthropogenic noise on a variety of aquatic life including (but not limited to) aquatic invertebrates, fishes, and marine mammals, and on sound sources and on methods of mitigating their impact.

## Purpose of the Conference and This Book

Research on the impact of underwater noise on aquatic life is challenging. Major new technological developments are taking place within the offshore industry, and, quite often, the new technology produces sound. Those concerned with assessing the impact of sounds on aquatic organisms are investigating the biology, physiology, and ecology of animals not readily examined in the laboratory, and often must develop new methods that can be applied to free-living animals in the in underwater environments itself.

Thus, a very significant feature of the conference (as our past conferences) was to provide extensive opportunities for learning about current research and fostering interactions with colleagues who focus on different taxa, different noise sources, and different approaches to understand these issues and their mitigation. Indeed, there was a strong emphasis for communication of ideas between participants from different backgrounds and interests in the overall themes of the conference.

Consequently, a major feature of the conferences, and this book, is that they bring together individuals and the latest ideas from a great variety of disciplines, ranging from biologists interested in behavioral and physiological effects of exposure to sound, to acousticians and engineers with an interest in the physics of underwater sound generation, to regulators and conservation groups concerned with the protection of aquatic life. The fact is that surveys of participants in the Cork, Budapest, Dublin, Den Haag, and Berlin conferences showed that the overall opportunity for communication between people from different disciplines might have been the most important long-term outcome of the conference.

*Indeed, the fundamental goal for the Aquatic Noise conferences, and the books coming from the conferences, is effective communication and consultation between the breadth of people and groups with interest in the topic*.

While this book may not impart the personal interactions found in the conferences and which will certainly continue at our next conference in Prague in 2025 (www.an2025.org), the chapters do reflect the ideas and issues discussed. Moreover, the chapters bring together ideas, and "introduce" many of the current and future "players" who will be making future major contributions to this field today and in the future.

## Broader Impact of the Conference

The information and ideas presented at the conference, as reflected in the chapters in this book derived from that conference, will certainly help shape future research and understanding of effects of noise on aquatic life. In fact, the impact of our earlier Springer books based on earlier conferences (Popper and Hawkins 2012, 2016) is apparent from the more than 1 million (as of 1 December 2023) chapter views to date. Moreover, based on surveys and informal comments, it is apparent that the books and the interactions and networking that took place at the 2022 conference have already resulted in new collaborations between investigators, a broader understanding of the science used in their work by decision-makers, a better understanding of regulatory issues by scientists and industry, and substantial cross-fertilization of experimental design and approaches by investigators.

## This Book Is Like a Library of "Old"!

There are 142 chapters in this book, and it is very hard to classify them—indeed, as the editors compiled the book they realized that there were perhaps four or five ways to organize the chapters and that many of the chapters would fit into several of the categories. Thus, we encourage readers to "browse" the Table of Contents beyond the chapters in the section they think most fit their interests. We suggest this since readers are likely to find other chapters, in other sections, that are of considerable interest and value. Indeed, the editors like to think of the book as like going into a library of old where one did not just go to grab the book or journal that brought one to the library, but rather one looked around the shelves and often "bumped into" other books or journals that struck an interest and may have even helped answer the questions that brought the person to the library in the first place (or other questions that were not even the basis for the visit)!

College Park, USA — Arthur N. Popper
Seattle, USA — Joseph A. Sisneros
Aberdeen, UK — Anthony D. Hawkins
Horsholm, Denmark — Frank Thomsen
October 2024

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Popper AN, Hawkins A (eds) (2016) The effects of noise on aquatic life II. Springer Science+Business Media, LLC, New York

# Acknowledgements

There are many people who helped make a giant difference for the success of the conference, and this book. But since they are acknowledged on our web site (www.AquaticNoise.Org), we limit ourselves here to just a few people.

First, every one of the organizers and attendees owe a giant debt to Ms. Terri Cullinane, principal in the conference organizing group VenuesWorld. Terri has worked with the Aquatic Noise organizers since the second conference in Cork, Ireland, and she and her colleagues are integral to the success of our conferences. We could go on for pages about what Terri and her colleagues do, but suffice it to say that we are extremely grateful for their great contribution.

The other two people we want to acknowledge (and particularly by Tony Hawkins and Art Popper) are Sue Hawkins and Helen Popper. Helen and Sue were our (Tony and Art) best colleagues and supporters and guides when we conceived of the conferences in 2005 at a dinner in Aberdeen, Scotland. Sue and Helen provided wisdom, guidance, and invaluable work for many of the conferences. Even though Helen and Sue were not involved in the last few conferences, their legacy is integral to the success of the conferences even today.

For this book, we would like to express our sincere gratitude to our colleagues and friends who served as section editors—Paul A. Lepper, Marta Picciulin, Denise Risch, Amy R. Scholik-Schlomer, and Kathleen J. Vigness Raposa. These colleagues each handled dozens of papers with timely reviews and re-reviews and were instrumental in ensuring the highest quality from every paper.

We also are grateful to colleagues at Springer Nature. We thank Susanne Friedrichsen and Sonal Nagpal from the Springer Reference Team, the publishing program in which the book appears, and William Lamsback, the Springer Nature editor in New York, who first provided suggestions to include the book in this key program and who was always willing to provide guidance and solve problems.

Finally, we are very grateful to the many agencies and funders who supported the conference (all listed on our web site). One of the hallmarks of the Aquatic Noise conferences is having participation from younger colleagues who are just entering the field of aquatic noise (and related topics) and participants from developing countries. Our goal is to expand interest in aquatic noise, ensure that it is an issue recognized to be of importance worldwide, and to bring new people into the field.

Thus, a very substantial portion of the funds we receive supports these participants, and we are proud that we can do this.

Arthur N. Popper, College Park, MD
Anthony D. Hawkins, Aberdeen, UK
Joseph A. Sisneros, Seattle, WA
Frank Thomsen, Copenhagen, Denmark

# Overview of the Book

Because a reasonable number of the chapters in this book represent updated material and may take slightly different approaches than the meeting presentations, the book organization is somewhat different than the meeting organization. At the same time, we recognize that the book could be organized in any number of ways, depending on the organizers and their interests. Thus, readers are welcome to "reorganize" the book for their own interests and needs. Indeed, with the chapters online, it very easy to set up a different organization in a computer file and then link that organization to the specific chapters.

The book starts in Section I with chapters covering works on the history and evolution of the aquatic life meetings and a summary of the Berlin meeting. Following the introduction, the sections have been arranged in the same order as one would commonly perform an assessment of noise impacts. Thus, Section II, "Sounds and Sources," includes works on anthropogenic and biological sound sources, encompassing the physical properties of these sources. These chapters also comprise research on numerical sound modelling and other acoustical topics. Section III is devoted to "Soundscapes," which encompasses studies dedicated to sound sources and the acoustic characteristics of the biotic and abiotic ambient sounds present in a large variety of locations around the globe.

Sections IV through VII deal with "Hearing and Auditory Assessment." Sections V through VII include studies on hearing sensitivity and frequency range of hearing, as well as hearing anatomy and physiology.

Section VIII then deals with general issues of noise effects, while Section IX focuses on noise effects on fishes, Section X on marine mammals, and Section XI on other vertebrates. Section XII focuses on noise effects on invertebrates and plants. Management of noise in the form of specific frameworks for impact studies, noise criteria, and noise mitigation is covered in Section XIII. Finally, Section XIV deals with Monitoring, including marine animals using biological sound as well as ambient noise.

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**Arthur N. Popper** is Professor Emeritus and Research Professor in the Department of Biology, University of Maryland, College Park, MD (USA). His current research focuses on the effects of man-made (or anthropogenic) noise on aquatic life, and particularly on fishes. This interest follows Popper's decades of basic research on fish hearing capabilities and mechanisms. Recent studies have included effects of very intense man-made signals (pile driving, seismic air guns) as well as general increase in background noise levels (as from shipping and aquaculture). He also works with US and international institutions in evaluating overall effects of the increase in man-made noise on animals. This includes being organizer of six international conferences on effects of noise on aquatic life. Popper has edited 77 volumes of books on hearing and related topics in the *Springer Handbook of Auditory Research* series and edits the science and technology magazine *Acoustics Today.*



**Joseph A. Sisneros** is Professor and Chair of the Department of Psychology and an Adjunct Professor in the Department of Biology at the University of Washington (UW), Seattle, WA (USA). The primary focus of Dr. Sisneros' work is the investigation of seasonal reproductive-state and steroid-dependent changes in the adult auditory system of fishes. A secondary focus of his work is the investigation of sound source localization by fishes. Dr. Sisneros' work on sound source localization and reproductive-state-related changes in fishes has been primarily funded over the past 16 years by the National Science Foundation. He has published

over 90 scholarly papers and book chapters in top journals and book publications. Dr. Sisneros was editor of the Springer book *Fish Hearing and Bioacoustics* and he is senior editor of the *Springer Handbook of Auditory Research*.



**Anthony D. Hawkins** was Managing Director of Laughine Limited, Aberdeen, Scotland, a company dealing with fish biology. He was Director of Fisheries Research for Scotland (1987–2002) and an Honorary Professor at Aberdeen University. He was made a Commander of the British Empire (CBE) in 2000 for his services to fisheries science. His research interests have ranged from migrations of salmon to the effects of sound on fishes. In 1991, he set up the North Sea Commission Fisheries Partnership, bringing together fishermen and scientists, and he then set up and worked as Rapporteur for the North Sea Regional Advisory Council (NSRAC), providing advice from fishers and scientists to the European Commission on the management of North Sea fisheries. Together with Arthur Popper and others, Dr. Hawkins organized international conferences on the Effects of Noise on Aquatic Life bringing together scientists, engineers, regulators, and industry to discuss the latest scientific findings in this field.



**Frank Thomsen** is a Principal Scientist in Underwater Acoustics with DHI A/S, Denmark. His work focuses on the effects of anthropogenic underwater noise on aquatic life, especially looking at offshore wind farms and dredging. Dr. Thomsen's scientific work has also covered the acoustic communication in wild killer whales (*Orcinus orca*) and the distribution and abundance of marine mammals using aerial and acoustic monitoring. Dr. Thomsen has worked as chairman of working groups and advisor for a variety of national and international organizations such as the European Marine Board, the European Commission, OSPA, and the United Nations. He was chairman of the 6th International Conference of the Effects of Noise on Aquatic Life in Berlin (AN2022). Dr. Thomsen has published over 80 papers and reports, many in leading journals.

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# Section I

# Introduction

# Brief History of "The Effects of Noise on Aquatic Life" Meetings

1

Arthur N. Popper and Anthony D. Hawkins

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**Abstract**

The idea for holding The Effects of Noise on Aquatic Life meetings arose in 2005 when the two of us and our wives got together in Aberdeen, Scotland. We discovered a shared interest in the effects of noise on aquatic animals and also realized that others with interests in the topic often focused on a single animal group, a single sound source, or specific technical issues. Even though these people could learn from one another and likely have very useful and productive collaborations, they almost never came together to share ideas and knowledge. Thus, we agreed that having a meeting that encouraged participation by a wide range of individuals with diverse interests could have a major impact on the field. This led to our first meeting in Nyborg, Denmark, in 2007, which we co-organized with Magnus Wahlberg, and then to very successful subsequent meetings.

This chapter shares the history of the meetings from our perspective. We discuss the fundamental goals we set out for the meetings in 2007 and insights into how we planned several of the meetings. Finally, because we no longer anticipate being very involved in future meetings, we discuss how we would like to see the meetings evolve further.

**Keywords**

Sound · Noise · Regulation

## Introduction and Chapter Purpose

This chapter tells the history of The Effects of Noise on Aquatic Life, often referred to as Aquatic Noise, meetings from our perspective. We share our remembrances starting from the origin and early days of the meetings to the present. We discuss the fundamental goals we set out for the meetings in 2007 and insights into how we planned them. At this point in our lives, we anticipate that our participation in future meetings will be far be less active, but we still would like to provide some input into how the meetings might evolve, using this chapter.

Moreover, because the overall issue that the meetings consider is only getting larger, as the use of the oceans increases from shipping, and the quest for alternative energy sources increases, we suspect that there will continue to be the need for these meetings. Thus, we want to put on record how and why we came up with the idea for holding the meetings (Fig. 1).




**Fig. 1** The authors, Tony Hawkins (left) and Art Popper (right), checking out the tasting room at the Jameson Brewery in Cork to decide if this could be the site of the banquet for AN2010

## The Origins

The origin of Aquatic Noise can be easily traced to Helen and Arthur Popper taking a vacation in Scotland in 2005 and visiting Susan and Tony Hawkins. Art and Tony first met in 1980 at a meeting on hearing and sound communication in fishes in Sarasota, Florida (published in Tavolga et al. 1981). However, it was at the Scottish visit in 2005 that we discovered shared interests, leading to now almost 20 years of collaboration and close friendship.

Over a long dinner at the Tolbooth Seafood Restaurant in Stonehaven, Scotland, we realized that we shared deep interests in the potential effects of anthropogenic sounds on aquatic animals. Both of us had started to engage in research and writing about the subject, and many of our views were quite parallel.

In subsequent discussions during that visit, we also realized that we shared the view that one of the real problems in studies of aquatic noise was that there was very little communication between investigators asking similar questions but who were concerned with different animal groups, sources, or regulatory issues. For example, it was rare, other than perhaps at meetings of the Acoustical Society of America (ASA), that investigators interested in fishes and marine mammals got together to share ideas about anthropogenic sound or that those interested in seismic sources and pile driving would get a chance to hear about other work. And it was even rarer for those interested in issues such as sound mitigation would learn about work on the effects of sound on the animals they were trying to protect (Popper et al. 2020).

At the same time, we realized that these different groups really had a lot they could share with one another, ranging from the potential effects or acoustics of pile driving versus seismic sources versus sonar to how fishes and marine mammals responded to such sources. Our view was that by sharing knowledge, ideas, and experimental approaches, each field could move forward more than if they stayed isolated.

## Thinking About a Meeting

From our discussions and over a few months, mostly by email, we came up with the idea that there might be considerable interest in a meeting that would bring together individuals from the research, regulatory, conservation, and industrial communities to meet one another, share ideas, and form networks. Indeed, we saw a very important goal would be to meet at a location that would foster networking and conversation across species and sources.

As we continued to develop the idea for an Aquatic Noise meeting, we came up with several goals for the meeting. Indeed, although these were our guiding principles for the first meeting in 2007, they remain guiding principles in 2022 and for the future! They include:

- Focusing on sharing ideas and knowledge across taxa, disciplines, techniques, sound sources, regulatory issues, and conservation issues.
- Single plenary sessions so that every participant would be able to hear every oral presentation to broaden what they learn and to increase awareness of all issues.
- Promoting networking by holding the meeting in a very comfortable environments with lots of spaces to meet and mingle.
- Having meetings in places where the whole meeting, including talks, posters, and sleeping accommodations, would be in the same building so that people would constantly be "bumping into" one another. This would include providing breakfasts and lunches so participants would, again, be together as much as possible.
- Providing support for younger participants and for participants who come from countries where travel support may be particularly limited. Achieving these goals would be the focus of our fundraising for the meetings so that we could bring new people into our field.
- Producing an impactful publication so that the presentations would have the most impact well beyond the meetings.
- Holding the meetings in interesting places so that people would find additional things to do before and after the meetings and be able to bring families who would find lots to do while the participants were in sessions.

## Facts and Figures

The rest of this chapter deals with the individual meetings, focusing on vignettes and broad information. Because many readers will be curious about data such as attendance, Tables 1 and 2 summarize some information about all the meetings. Table 1 also provides links to the programs for each meeting where interested readers can find lists of the organizers, co-organizers, funders, and other information. Past programs are available and contain more data.

**Table 1** Meeting numbers

| Meeting | Number of delegates | Number of presentations (oral and poster) | URL of program booklet[b] | Publication | Additional notes |
|---|---|---|---|---|---|
| AN2007 Nyborg | 202 | 134 | Supplement AN2007 | *Bioacoustics 17(1-3) 2008* | Hawkins et al. (2008) |
| AN2010 Cork | 244 | 111 | Supplement AN2010 | Popper and Hawkins (2012) | |
| AN2013 Budapest | 243 | 125 | Supplement AN2013 | Popper and Hawkins (2016) | |
| AN2016 Dublin | 343 | 229 | Supplement AN2016 | *Proceedings on Meetings in Acoustics, volume 27* | Erbe et al. (2016) |
| AN2019 Den Haag | 312 | 214 | Supplement AN2019 | *Proceedings on Meetings in Acoustics*, volume 37 *The Journal of the Acoustical Society of America* Special Issue | Erbe et al. (2019) Thomsen et al. (2020) |
| AN2022[a] Berlin | 245 | 185 | Supplement AN2022 | https://link.springer.com/referencework/10.1007/978-3-031-10417-6 | |

[a]It is very likely that attendance in Berlin was lower than in 2016 and 2019 due to the COVID pandemic
[b]Program booklets are available under "Supplementary Electronic Material"

**Table 2** Meeting organizers

| AN2007 | AN2010 | AN2013 | AN2016 | AN2019 | AN2022 |
|---|---|---|---|---|---|
| A. Hawkins<br>A. Popper<br>P. Tyack<br>M. Wahlberg | A. Popper<br>A. Hawkins<br>R. Gentry<br>M. Picciulin<br>M. Tasker<br>D. Wartzok | A. Hawkins<br>A. Popper<br>C. Erbe<br>J. Miller<br>J. Sisneros | C. Erbe (chair)<br>A. Hawkins<br>A. Popper<br>J. Sisneros<br>F. Thomsen | C. Erbe (chair)<br>A. Hawkins<br>A. Popper<br>P. Lepper<br>J. Sisneros<br>F. Thomsen | F. Thomsen (chair)<br>A. Popper<br>A. Hawkins<br>P. Lepper<br>A. Scholik-Schlomer<br>J. Sisneros |

## The Aquatic Noise Trust

After the second Aquatic Noise meeting (Cork, referred to as AN2010) and as we started to work on AN2013 (Budapest), we realized that we needed a way to manage funds for the meeting and particularly to take in and distribute the funds we received from many international funding agencies and companies. Therefore, after getting legal advice, we set up the Aquatic Noise Trust as a charitable organization in the

United Kingdom. Some of the goals of the Trust were to: (1) organize lectures and conferences on the effects of noise on aquatic life, (2) facilitate the international exchange of information to enhance collaboration and research, (3) support the education of students and others by providing grants to facilitate attendance at lectures and conferences on aquatic noise and its effects, and (4) distribute funding for research into the effects of noise on aquatic organisms.

The original trustees of the Trust were Anthony Hawkins, Susan Hawkins, Arthur Popper, and Helen Popper. Paul Lepper is now the head of the Trust, and it continues in the United Kingdom and supports the meetings. The Trust is the owner of the meetings and associated websites.

## Aquatic Noise 2007: The First Meeting, Nyborg, Denmark

A critical question for the first meeting (Fig. 2) was location. We quickly realized that this is not a simple question because selecting a site requires getting a hotel, arranging meeting facilities, organizing space, and reserving hotel rooms. We also realized that dealing with all the details was way beyond our expertise or available time; indeed, this could have been a major barrier to organizing a meeting.

Fortunately, Art recalled attending a meeting of the International Society for Neuroethology in Nyborg, Denmark, organized by colleagues from the University of Southern Denmark, Odense. That meeting took place in a conference hotel, the Nyborg Strand. The appeal of this site was that the hotel was set up for conferences, and we would not have to deal with details such as reservations and food service.

We contacted Magnus Wahlberg, a colleague at Odense, about the meeting, and he agreed to co-organize with us (see Table 2 for a list of co-organizers). Indeed, Magnus graciously took over all work with the hotel and was a great collaborator in all aspects of the meeting planning. In fact, it turns out that Magnus had to sign the contract with the Nyborg Strand personally, putting him potentially in debt to over $150,000 (US) if the meeting did not go forward.

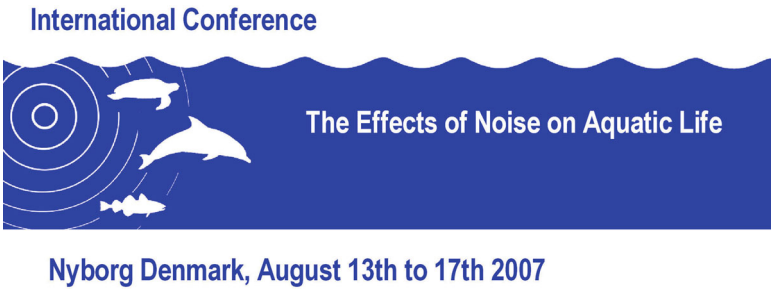



**Fig. 2** Logo for Nyborg meeting. Drawn by Tony Hawkins

In terms of the program, the three of us worked together along with major input from Peter Tyack (then at the Woods Hole Oceanographic Institution) and assistance in organization by Lydia Wysocki (then at the University of Vienna). Planning for the meeting took place not only via email but also at several other preceding meetings that various organizers attended in the United States.

Once we had figured out where to hold the meeting, we had to come up with a plan to let people know about it and how best to get funding. As we recall, publicizing the meeting was not too hard in that between us we knew many people in the field, and so we sent out announcements to hundreds of friends and colleagues and encouraged them to spread the word, and we used various mailing lists to further increase visibility.

The response was quite substantial; although others may not have thought about doing a meeting, there was clearly a sense among our colleagues that this kind of meeting was needed and would be a major step forward for the field.

## Funding

Getting funds was a bit of a challenge, but in our proposals, we focused on supporting students and participants from countries in which funds were not readily available. We were fortunate that we were able to get substantial funding from several US agencies including the US National Science Foundation, the US Navy (the Office of Naval Research and ONR Global), and the US Minerals Management Service (now BOEM). We also received substantial funding from the International Association of Oil and Gas Producers and from the Danish Carlsberg Foundation. We had several other smaller funders (see Supplemental Material in the program books for names of funders of this and other meetings).

Because we did not want parallel sessions, we could only have a limited number of talks over the 5 days of the meeting. In selecting talks from all the papers submitted, we tried to emphasize diversity in topic, and we focused on six major areas: (1) how animals use sound, (2) detection of sound by aquatic organisms, (3) underwater acoustics and ambient noise, (4) anthropogenic sources, (5) effects of anthropogenic sound on aquatic animals, and (6) regulation. We also had 84 posters.

## Lightening Sessions

We also created what we later called Lightening Sessions where many poster presenters would get a few minutes, and a few slides, to give an overview of their work. This proved to be very popular and was a major feature of all future meetings. At the first meeting, Lightening Session presenters tried to cram a 15-min talk into 5 min. We learned from these first presentations and subsequently changed things to give specific guidance on how to do a very short talk, including emphasizing that talks

should only include the research questions and the results, with minimal mention of background and methods.

## Keynote Speakers

An additional feature of the first, and all subsequent, Aquatic Noise meetings was to invite keynote speakers who could introduce new ideas and questions to our participants. Part of this was to share ideas and data from work with terrestrial animals (including humans) that is directly applicable to work on aquatic animals. Thus, for example, in Berlin, we invited Louise Roberts to talk about substrate vibration because this an area of growing interest to the aquatic community; and we asked Wei Qiu to talk about the applicability of kurtosis in assessing impacts of sound on animals. At earlier meetings, we had William Yost give a general overview of psychoacoustics and Colleen Le Prell discuss the latest findings on the effects of noise on human hearing and health. Many other colleagues have joined us in keynotes. These presentations and messages have resonated with attendees and have influenced the work and thinking of many.

By all accounts, AN2007 was a total success (Fig. 3). The "reviews" were excellent, and there was a clear interest in having another meeting. So, we started to think about another meeting.



**Fig. 3** Participants at the Nyborg, Denmark, meeting. This is the only meeting for which we have a photo of most participants

## Aquatic Noise 2010 (AN2010): The Second Meeting, Cork, Ireland

When we decided to have a second meeting (Fig. 4), we, of course, asked Magnus to work with us. But other obligations kept him from participating. So, we started out on our own, with the intent of developing an organizing group once we figured out where the meeting would take place.

In considering locations, the cost for participants and for holding the meeting was paramount to our thinking. But we also wanted to keep our goals in mind about meeting in a place that included conference rooms, hotel rooms, and a locale that was interesting. To help in our planning, we devised a set of additional specific criteria for site selection. These included:

- Easy access for international travelers, especially from the United States and Europe. Our preference was no more than one change in conveyance to get to the meeting city.
- Conference space and facilities for 200+ delegates, with poster display space. (In fact, due to an increase in size of the meetings, we now require space for at least 300.)
- Two meeting hotels for delegates, if possible. Most delegates would stay at the main meeting hotel, but we would also try to have a second hotel at a lower price for delegates for whom cost is a major issue.
- Accommodations to be at the meeting hotel and within easy walking distance of the town center to allow delegates to access areas of interest.
- Dates in July or August. Based on our survey from Cork, these seemed to be the most preferred times for the meeting for most delegates.

### Finding a Conference Organizer

We initially thought it would be good to hold the meeting in Scotland, and Susan Hawkins thus contacted Visit Scotland. They were very keen to be helpful, but the




**Fig. 4** Cork logo

only facilities that met our criteria, including time in the summer, were in Edinburgh and surrounding regions. Because our dates corresponded with the Edinburgh Festival, no facilities were available. The other places offered to us were golf resorts in the beautiful countryside but miles from any town, making transportation difficult, especially for accompanying families who might like to sightsee and travel around.

Visit Scotland then suggested that we consider the Republic of Ireland and gave Susan the contact details of the Irish Tourist Board. This group advised that we consider working with the conference organizing group Venues Ireland, a part of Venues World and, in particular, with Terri Cullinane.

Venues Ireland was incredibly welcoming and helpful and suggested that either Dublin or Cork should be able to meet our requirements. After considering our dates and needs, Terri specifically suggested we consider meeting in the city of Cork, in County Cork. Because Susan was more familiar with Cork, she decided that we should focus on it.

Thus, we concluded that we should look for locales in Cork because we anticipated reasonable costs. To make a final decision on meeting location, Tony and Sue Hawkins and Art and Helen Popper visited various hotels in Cork with Terri. Terri also took us around the County Cork region to show us places attendees could visit and how she could arrange tours and places for our banquet (Figs. 1 and 5). We even




**Fig. 5** Cork Banquet at the Jameson Brewery (also see Fig. 1)

got to meet Terri's parents and had a lovely several hours with them. We selected the Clarion Hotel Cork as our main meeting hotel.

Besides helping us pick and have Aquatic Noise 2010 (AN2010) in Cork, the decision to work with Terri may have been the most important single thing we did to ensure the quality of the Cork meeting and also the quality of future meetings. Indeed, Venues World took (and still takes) responsibility for everything from organizing our website to registering participants to collecting meeting registration to keeping our budget. Thus, the organizers can focus on meeting content, while Terri and her group ensure that everything works and that every attendee's needs are met.

## The Meeting

By the time we started to organize Aquatic Noise 2010, soliciting participants was far easier than in 2007. First, we had our participant list from 2007. But, equally importantly, the earlier meeting had been enough of a "hit" that when we announced Cork, people immediately made plans to attend and present.

We did decide to be a bit more formal, with an organizing group for the Cork meeting. Although the two of us did much of the overall organization, the organizing group was instrumental in helping design the program, soliciting speakers, and helping in a myriad of other ways. A list of organizers is in Table 2, while Table 1 has the links to the meeting programs that list all the people who helped with all the meetings.

Of course, we needed funding to hold the meeting, and we returned to the same groups that funded us in 2007. We received funds from most of those agencies but also funding from several other US agencies.

We also started to solicit support for specific parts of the conference from smaller organizations and companies, and this has turned out to be a substantial help in support of students and filling "niches" for things for which we could not get funds, such as paying for name badges, conference bags, and nicer coffee breaks. (These companies are listed in the program books; see Table 1 for the links.)

The organizing committee decided to require that all presenters do a chapter for the conference proceedings. We talked with several publishers and chose to work with Springer (now Springer Nature) because Art had a long history of working with them on other books and they were the most willing to produce the kind of volume we wanted. Most every presenter submitted a chapter. The chapters were meticulously edited by Helen Popper, a professional scientific editor, and the book was published as Popper and Hawkins (2012).

Every contributor received a copy of the book, as did our funders. Indeed, our insistence on a substantial publication from the meetings has been of major importance to many of our funders and has helped us get substantial support because there is a clear outcome from the meeting.

## Aquatic Noise 2013: The Third Meeting, Budapest, Hungary (Fig. 6)

### Finding the Location

With the success of the first two meetings, we started to think of Aquatic Noise as a triennial conference. Our thinking was that doing a meeting every year or every other year was just too much work for us and the meetings might be too close together to generate new material.

For this meeting, we again turned to Terri to get her thoughts on potential sites. She came up with several cities in Europe and found hotels in each that had the potential to hold the meeting. In addition to having our general criteria for meeting sites, one main hotel with conference and bedroom facilities in an interesting place, we added a request that most attendees should be able to get to the meeting with no more than one change in major transportation (e.g., one airport transfer).

After considering each of the locales, we eliminated some due to high costs, high summer temperatures, and other factors. We then decided that we would look at hotels in Prague and Budapest. Helen and Art Popper, Sue and Tony Hawkins, and Terri Cullinane visited both sites. One hotel in Prague was too far from the center of town and had no warmth; the second hotel was too expensive. Helen then remembered a hotel that the Poppers had stayed in that was in the old part of the city. Terri quickly arranged for us to see the hotel, and it was an instant hit. It had great meeting space and a meeting area that had spectacular views of the Old City. Indeed, this could easily have been our choice for the meeting, but the hotel refused to lower rates enough to make a meeting viable.

We then went to Budapest and looked at two spectacular hotels, one on the Buda side of the Danube and the other on the Pest side. We decided to go with the Corinthia in Pest because it was downtown and closer to restaurants and other venues. It also gave us rates that we felt were fair both for sleeping rooms and for the meeting. Importantly, the conference facilities were unsurpassed, providing a venue with style and gravitas yet with state-of-the-art facilities.

The hotel was a major "hit" with participants. Indeed, it has become the "benchmark" for all meeting sites since. The hotel was truly lovely, the sleeping rooms were




**Fig. 6** Budapest logo




**Fig. 7** Budapest plenary room. This was, by far, the most magnificent meeting room that we have ever had for an Aquatic Noise meeting

outstanding, and the main meeting room was perhaps the most magnificent that many participants had ever seen (Fig. 7). Delegates, thanks to participant Darlene Ketten, "discovered" the attached restaurant, the Bock Bistro, and made it a favorite during the meeting for its great Hungarian food and spectacular wine list.

## The Meeting

For this meeting, we decided to try to formalize the organizing group to provide more continuity and "corporate memory" to the planning process (Table 2). Funding for the meeting, which covered much of the support provided, again included the US National Science Foundation, several branches of the US Navy, and a myriad of smaller agencies and companies. Of the delegates, 49 students and postdocs received full or close to full support for travel, hotel, and registration. An additional 39 delegates received partial support.

Once again, all delegates were asked to contribute chapters to a volume published by Springer (Popper and Hawkins 2016). The volume was again edited by Helen Popper. It is worth noting that, as of this writing, chapters from the volumes from Cork and Budapest have been viewed over one million times, suggesting that the meetings, through the books, have likely had an extraordinary impact on the field.

## Aquatic Noise 2016: The Fourth Meeting, Dublin, Ireland

The organizing group for the 2016 (Fig. 8) meeting included the Art and Tony along with Christine Erbe, Joe Sisneros, and Frank Thomsen. Christine served as the nominal chair of the organizing committee, and she did a superlative job in moving us forward. We also started to formalize our approach to membership on the organizing committee by trying to diversify the disciplines and home countries of the participants.

We started to search for a venue for the 2016 meeting in 2014. Again, Terri came up with a list of suggested places that we winnowed down to Amsterdam, Den Haag, and Dublin. Art, Tony, Christine, and Terri visited the three sites. Our first visit was in Den Haag, and we fell instantly "in love" with the Grand Hotel Amrâth Kurhaus, and it wound up being the site of the 2019 meeting. However, the prices at this hotel for the 2016 meeting were somewhat higher than we wanted to pay. We visited several other venues in Den Haag, none of which offered the ambience we wanted. One of the sites was a conference hall that would have been good for the meeting, but delegates would have been spread over many hotels. In considering this site, we focused on our interest in keeping most delegates together as much as possible, and this site only reinforced that view.

We then went on to Amsterdam and visited several hotels. We were quite tempted by one hotel that told us that we could have use of the room where John Lennon and Yoko Ono stayed on their honeymoon and that it still had a Lennon guitar made it very tempting (and exciting). The hotel would have been great in many ways, but the management was not very accomodate.

We then went off to Dublin where Terri showed us the O'Callaghan Alexander Hotel, and we quickly decided that it was a great place to meet. We did look at several other places, but none had the same appeal in either facilities or location. During that visit, we also had the pleasure of meeting and getting to know Terri's husband John Burke, the co-owner of Venues World.

## The Meeting

A description of the meeting (Fig. 8) was published (Erbe et al. 2016) in Volume 27 of the *Proceedings on Meeting in Acoustics* (*POMA*), along with a large number




**Fig. 8** Dublin logo

of other papers. All the papers are open access. However, unlike the previous three meetings where we made submitting a paper to the meeting proceedings a requirement for all delegates, we only required this of people we funded to attend, although we did encourage others to submit as well. In fact, in hindsight, this approach did not achieve our goal of having meeting proceedings published. We tried a somewhat different approach in 2019 (see section "The Meeting"), but for the Berlin meeting, we again required papers be submitted for publication.

### Rodney Coates Student Award

We added a new piece to this meeting. One of the delegates at our first meetings was Rodney Coates. Rodney was particularly interested in seeing the success of younger attendees. However, he passed away in 2016 before the Dublin meeting. His family wanted to support Rodney's interests, and so they provided generous support, in his honor, for an award to the best student presentation. The Rodney Coates Student Award, still supported by his family, is now a strong tradition of the Aquatic Noise meetings.

In thinking about this award, we decided (with input from the Coates family) that every student or recent graduate could, if they wished, compete for the cash prize. However, we did not want judging solely based on posters, and we could not accommodate every student for a 15-min oral presentation. So we decided that candidates for the award would give a 4-min lightening (speed) talk and do a poster. We developed a rubric for judging so that every candidate would be evaluated in the same way, and we invited several senior delegates to judge the candidates and decide on the winner.

## Aquatic Noise 2019: The Fifth Meeting, Den Haag, The Netherlands

In considering the 2019 meeting (Fig. 9), Terri revisited the idea of using the Grand Hotel Amrâth Kurhaus in Den Haag, and this time, we were able to work out costs that would be fair to delegates and meet our budgetary needs. So, without looking anywhere else, Aquatic Noise 2019 went to The Netherlands.

Den Haag also had several features that made it appealing for our meeting, including being easily accessible from anywhere in the world (via a brief train ride from Amsterdam Schiphol International Airport). In addition, the village of Kurhaus, which is close to Den Haag, met all the requirements for our meeting. There were ample opportunities for evening outings at the end of a busy day at the conference, daytime activities for accompanying persons, and vacation destinations for families.

The organizing committee grew again. Our 2016 group of Art, Tony, Christine, Joe, and Frank was joined by Paul Lepper. We were able to get substantial funding from our regular contributors as well as from several new contributors and so were

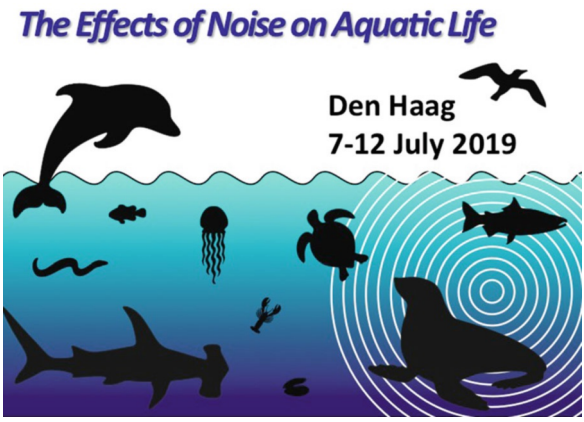



**Fig. 9** Den Haag logo

able to support the attendance of almost 100 delegates, most of whom were students or recent students.

### The Meeting

An overview of the meeting was published by Erbe et al. (2019). One of the issues that we encountered in the 2016 publication was that many presenters were reluctant to publish their work in *POMA*, a nonrefereed journal. So, through collaboration with the ASA, a strong supporter of the Aquatic Noise meetings from their inception to today, we worked out an arrangement whereby either delegates could publish in a special issue of *POMA* (https://asa.scitation.org/toc/pma/37/1) or they could submit papers for a special issue of *The Journal of the Acoustical Society of America* (https://asa.scitation.org/topic/special-collections/aqn2020?SeriesKey=jas) (see Thomsen et al. 2020).

## Aquatic Noise 2022: The Sixth Meeting, Berlin, Germany

Aquatic Noise 2022 (Fig. 10) proved to be a challenge due to the COVID-19 pandemic. We decided on a location and hotel before the start of the pandemic, but once the pandemic started, we were not confident that we could actually hold the meeting until early 2022. Indeed, we were greatly concerned about financial issues because we had financial obligations to the hotel and other groups that had to be booked in advance.

We spent most of a year planning and worrying about whether to hold the meeting or not. We decided that we did not want a virtual meeting because so much of a focus of Aquatic Noise is the opportunity for meeting people and networking. Moreover, a virtual meeting would have been very difficult to manage even if we focused on participants from Europe and the United States alone (something we did not want to do), and the up to 9-h time difference from the west coast of the United States to Europe would make scheduling of sessions very difficult.

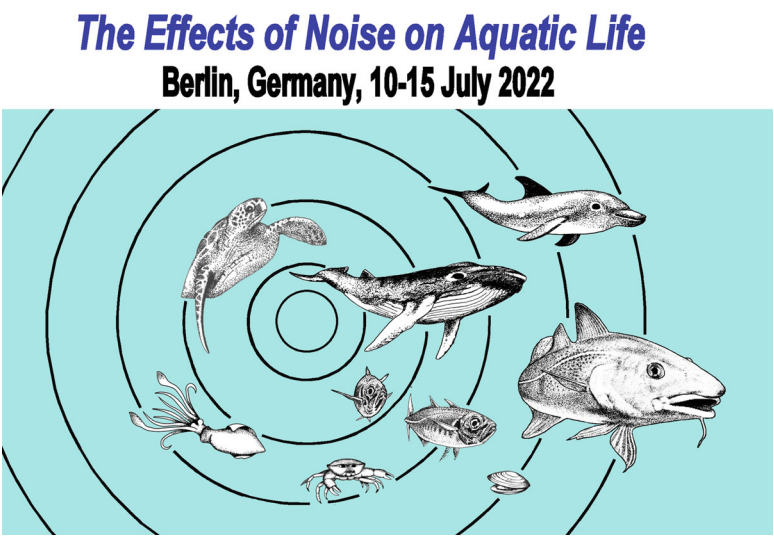



**Fig. 10** Some presenters at the Den Haag meeting

Thus, as the pandemic eased, we decided to hold the meeting, but we still had considerable concerns about whether people would register and come. To our delight, delegates were eager to attend in person, and although we did not have the same number of participants as in the past few meetings (Table 1), we still had 248 attendees.

One of the big changes for this meeting was in the function of the organizing group. Whereas in the past, the two of us were engaged in almost every detail of planning, we were less involved this time. We did stay on the organizing group, and we added Amy Scholik-Schlomer as a new organizer. Our friend Frank Thomsen took over as leader and did a brilliant job in moving us forward, particularly rising to the occasion in the face of the pandemic. And, of course, Terri and Venues World continued to be integral to the meeting.

Despite the problematic circumstances, Aquatic Noise 2022 turned out as much as a success as its predecessors. The scientific program was praised for its high quality and diversity of topics, including studies on lesser known taxa such as turtles, birds, invertebrates, and even seagrass. The participants seemed to be eager to interact in person again, and discussions were lively and engaged. Many participants made use of the many attractions and cultural events Berlin has to offer.

## The Future

As stated earlier, we anticipate that our involvement in Aquatic Noise will be minimal (if it exists at all) for any future meetings. However, we leave the future in excellent hands, with colleagues (and friends) who understand the discipline, appreciate deeply the value that the meetings hold, and appreciate the history and traditions of the meetings.

## Why No Meeting in North America?

But before making some recommendations, we want to answer a question that is often asked of us; why have the meetings been only in Europe and not in North America, especially considering the number of delegates from the United States and Canada. In fact, for almost every meeting, we have looked at various locations in the United States and Canada, but several factors have worked against these locations. When we found a city that met our travel expectations of reasonable access, hotel prices in the summer were far higher than in Europe, and hotels were not willing to make commitments to us 2 years in advance about rates. Indeed, we found a very suitable location in Baltimore, Maryland, for the 2019 meeting, but after several conversations, they would not commit to even holding rooms for us very far in advance.

Then, when we found places at reasonable cost, these were in locations that were hard to get to. For example, we found a great location in Canada and another in Maine, but getting to both would have taken three or four flight connections from Europe and often even several within the United States. Thus, for economic and practical reasons, Europe is still the most reasonably priced and accessible location. Hopefully this will change, and we encourage the organizers to consider the United States or Canada in the future.

## Our Recommendations

In closing, we do want to make several recommendations to the future organizers.

First, maintain the goals and principles on which we founded the meetings (and as discussed in section "Thinking About a Meeting"). We feel that these ideas have served us well and helped ensure that the meetings serve the community in very useful ways.

Second, in addition addition to focusing on support for younger colleagues, we feel strongly that an additional, equally important, focus of the meetings should be to reach out to groups who have not been well represented at our meetings, groups that in the United States are often referred to as underrepresented.

There are such groups in many countries, and ways need to be found to bring some of them to the meeting. Considering the breadth of the meetings, and the importance of the topic, it should be possible to use the meetings to excite young people from underrepresented groups to consider science and technology as career options and perhaps even set them on a path to such careers. So, our idea is that the focus not necessarily be on people already in the fields covered by the meetings, but the on "next generation," those students currently in university who are thinking about careers and who might be responsive to coming to the meetings, meeting with the participants, and being mentored by them.

We strongly support the idea of a publication from every meeting. And, after both requiring papers and only inviting them, we suggest that requiring papers provides the most value to the meetings and to our funders.

Finally, when we started the meetings, we were later in our careers and close to "retirement." Thus, we were able to find considerable time to handle much of the work associated with the meeting without impacting our other obligations.

However, the mantle of organization has passed to a younger generation who have substantial obligations to work, young families, students, and others, all of whom deserve a great deal of time. Thus, this generation of organizers, and we suspect organizers in the future, might need some more formal help in keeping the meetings going. We therefore suggest considering getting assistance from a part-time conference manager to work with the organizers to maintain to do a myriad of things, from communicating with potential participants to keeping databases to managing meeting finances. Such an arrangement would allow the organizers to focus on the content of the meetings and ensure that the high standards of past meetings continue.

**Acknowledgments** The Aquatic Noise meetings could not have taken place had we not had a cadre of extraordinary colleagues collaborating with us. We have mentioned many in this chapter, and we thank them all for their collaboration and for their friendship. We also thank the many people who have been on our advisory groups for their participation and advice.

We are also deeply grateful to the organizations and companies that have funded the meetings. In particular, we thank the leadership in many of these organizations, many of whom have become engaged with the meetings, who have assured us of continued support, and who work hard to ensure that we get their financial support on time. We apologize for not mentioning names, but those we refer to know who they are.

Finally, but most importantly, we thank Helen Popper and Susan Hawkins, not only for allowing us the adventures of conference organization but for being partners in developing and planning the meetings. Sue and Helen did not present papers, and we doubt they ever heard a single talk. But, their "fingerprints" are all over the meetings, and they were responsible for much of the "extras" that made the meetings so special. We also thank Helen for editing this chapter and Sue for providing a number of important memories.

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# Evolution of "The Effects of Noise on Aquatic Life" Meetings: 2007 to 2022

2

Amy R. Scholik-Schlomer, Christine Erbe, Anthony D. Hawkins, Paul A. Lepper, Frank Thomsen, Arthur N. Popper, and Joseph A. Sisneros

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**Abstract**

The effects of anthropogenic underwater sound on aquatic and marine life are an important environmental topic that has expanded considerably over the past decades. Since the first meeting in Nyborg, Denmark, in 2007. The Effects of Noise on Aquatic Life (AN) has emerged as a distinctive conference on this topic by bringing together experts from a variety of scientific fields, as well as those from the regulatory and industry communities.

**Keywords**

Sound · Noise · Marine mammals · Fishes · Invertebrates · International conference

## Introduction

The Effects of Noise on Aquatic Life (AN) conference in Berlin, Germany, in 2022 marked the 15th anniversary since the inaugural conference held in Nyborg, Denmark, in 2007 (Table 1). With this anniversary, it is an opportunity to discuss the evolution of AN, in terms of how the numbers of taxa and sound sources have changed, how international participation has expanded, and how these changes reflect the scientific advancement of the subject of aquatic noise over the past 15 years.

## Authorship and International Participation

The intent of AN has always been to offer a forum for scientists, regulators, industry, conservation groups, and others interested in this topic from across the globe to discuss and learn about the potential impacts of noise on marine life. Using the nationality of first authors as a surrogate means to gauge international participation over AN's 15-year history illustrates changing trends in participation (Fig. 1). For example, participants from the United States heavily dominated the earlier AN conferences (e.g., up to 51% in Cork 2010). However, over time, this has reduced

**Table 1** Locations and dates of past Aquatic Noise conferences

| Conference location | Date |
|---|---|
| Nyborg, Denmark | 2007 |
| Cork, Ireland | 2010 |
| Budapest, Hungary | 2013 |
| Dublin, Ireland | 2016 |
| Den Haag, the Netherlands | 2019 |
| Berlin, Germany | 2022 |

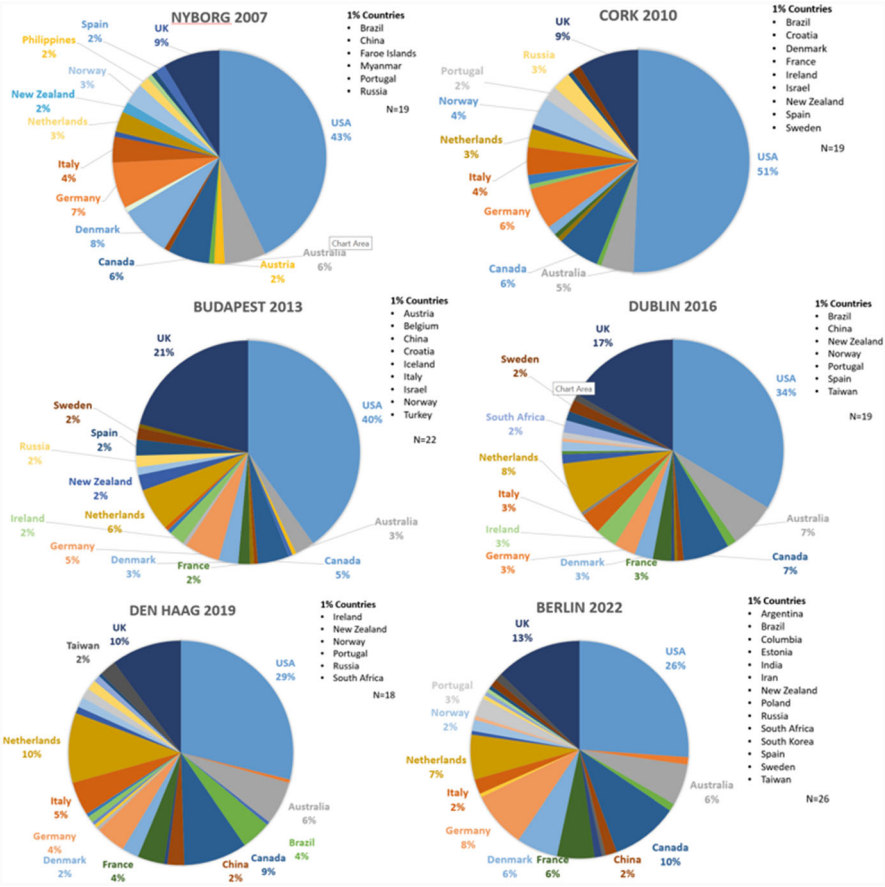



**Fig. 1** First-author participant by country over the 15-year history of Aquatic Noise showing countries with ≥2% participation. Countries with 1% participation are listed beside the corresponding pie graph. N indicates the total number of different countries for each conference

(e.g., 26% in Berlin 2022), and there has been a higher proportion of authorship (i.e., participation) from other countries.

Throughout AN's history, there has continually been a strong contingency of participants from North America (i.e., the United States and Canada), Europe (in particular, the United Kingdom, the Netherlands, and Germany), and Australia (Figs. 1 and 2). Additionally, overall the number of countries of first authors has gradually increased from Nyborg 2007 (n = 19) to Berlin 2022 (n = 26). However, participation from those in South America, Africa, and Asia notably lags behind. Whether this is because of other competing Asia- or Africa-specific bioacoustic conferences (e.g., Oceanoise Asia symposium or African Bioacoustics Community conference) or other reasons remains unclear. One notable reason may be financial. Thus, the preponderance of AN's funding has gone to support participation of students and of representatives from emergent countries. A recent review article (Becker et al. 2022) found that, at least in Africa,

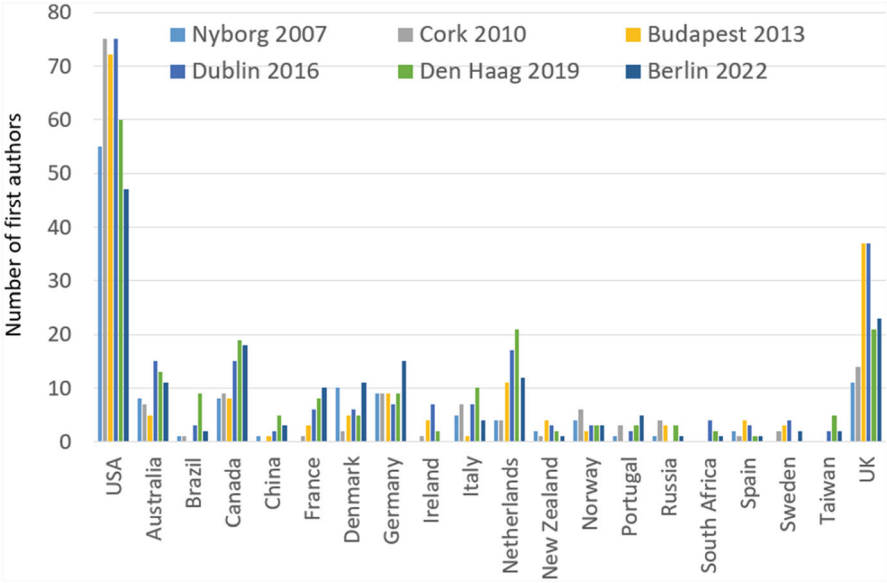



**Fig. 2** First authorship by top 20 countries over Aquatic Noise's 15-year history

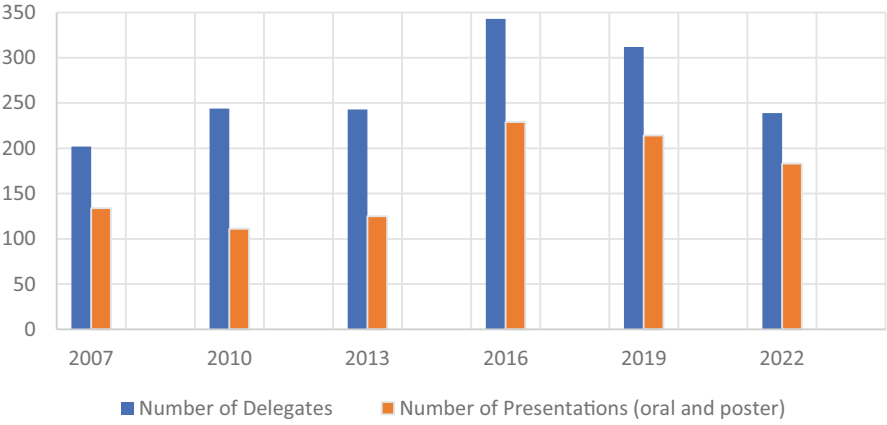



**Fig. 3** Statistics on number of delegates and presentations throughout Aquatic Noise's 15-year history

most bioacoustic research focuses on the terrestrial species, rather than aquatic or marine species, and that the majority of authors on these publications do not have African affiliations. Thus, there is an opportunity for the expansion of aquatic- and marine-focused bioacoustic research globally. Enhancing further international and diverse participation is a goal for future AN conferences.

Furthermore, the number of overall delegates and presentations has steadily increased from the first three conferences to the more recent conferences (Dublin 2016 and Den Haag 2019) (Fig. 3). There is a noticeable decrease in the number of

delegates and a slight drop in the number of presentations with the most recent conference in Berlin 2022, likely due to remaining concerns over COVID-19. Nevertheless, each of the AN conferences have sustained over 200 delegates and as many presentations since Dublin 2016.

## Taxa

The origins of early interest in the potential impacts of underwater noise on marine species date back to the early 1970s (Payne and Webb 1971). Momentum increased in the 1990s and 2000s with the Acoustic Thermometry of Ocean Climate (ATOC) program, North Pacific Acoustic Laboratory (NPAL) source, and the Heard Island Feasibility Test and the potential negative impacts particularly on marine mammals (e.g., Bowles et al. 1994; Au et al. 1997; Costa et al. 2003; Mobley 2005). Thus, while early studies may have focused on marine mammals, there has been increasing interest in understanding the effects of noise on other taxa such as fishes and invertebrates (Hawkins and Hawkins and Popper 2016; Popper et al. 2022).

Throughout AN's 15-year history, the number of taxa studied has increased. While marine mammal presentations, especially on cetaceans, dominated earlier conferences, the most recent conference in Berlin 2022 saw fishes being the dominant taxa in the overall number of presentations (Fig. 4). Presentations on invertebrates have also steadily increased over the years, especially since Budapest 2013, with a slight decrease for Berlin 2022. Marine bird species and sea turtles remain taxa with minimum representation in the conference program (< 5 presentations per conference). It is also interesting to see that later AN conferences included talks on plankton (since Dublin 2016), and the most recent conference in Berlin 2022 included a talk on the impacts of noise to seagrasses posidonia is impaired by human-generated noise.

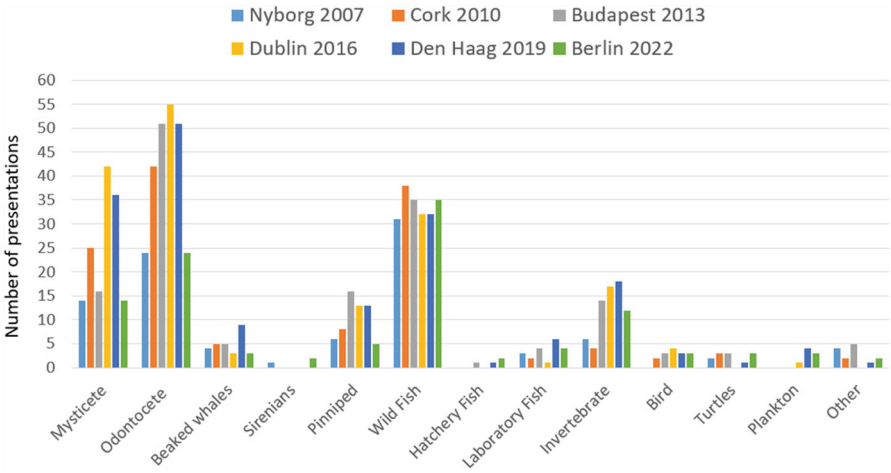



**Fig. 4** Statistics on the number of papers by taxa throughout Aquatic Noise's 15-year history




**Fig. 5** Evolution of Aquatic Noise logos (AN2007, 2010, 2013, 2016, and 2022 created by Anthony D. Hawkins; AN2019 created by Christine Erbe)

The logo over the years also reflects the expansion of a greater variety of taxa included in AN presentations (Fig. 5). The Nyborg 2010 logo only included a marine mammal, sea turtle, and fish, while AN 2019 and 2022 expanded to include numerous species of invertebrates.

The conference's evolution reflects a similar analysis published by Duarte et al. 2021. During their literature review (1973–2020), Duarte et al. 2021 reported on distribution of scientific studies by taxa, where odontocetes dominated, followed by fishes, mysticetes, invertebrates, pinnipeds, and sirenians (Duarte et al. 2021, indicated "a paucity of quantitative assessments for diving marine reptiles and seabirds"). In hindsight, with this AN analysis, it would have been useful to separate out harbor porpoise from other odontocetes, since this is one of the primary odontocetes studied in Europe and represents many of the presentations at AN (e.g., Tougaard et al. 2016; Wensveen et al. 2016; Schaffeld et al. 2020).

Finally, it is interesting to highlight the "other" category in Fig. 4, which has included past poster and oral presentations on the fishing cat (Malla et al. 2019), amphibians (e.g., Tennessen et al. 2016), sea otters (e.g., Ghoul and Reichmuth 2012), and humans Underwater Hearing and Noise Exposure Concerns in Humans.

## Sound Sources

Early attention to anthropogenic underwater sounds, especially from a management perspective, was primarily on those sources associated with serious acute impacts. Some examples include mortality associated with events, such as with certain

situations of beaked whale stranding and tactical sonar (e.g., Simmonds and Lopez-Jurado 1991; Cox et al. 2006; D'Amico et al. 2009) or fish barotrauma associated with coastal pile driving (summarized in Popper et al. 2019; Caltrans 2020). With this focus came the development of various acoustic criteria via scientific publications (e.g., Popper et al. 2014; Southall et al. 2019) and other management directives (e.g., Tasker et al. 2010; BfN 2012). Over time, the spotlight at AN has shifted from beyond acute noise sources to those that are capable of causing sublethal, chronic impacts, like commercial shipping.

Again, this AN 15-year analysis reflects the Duarte et al. 2021, literature review, in terms of the overall distribution of studies by sound source, with vessels (small and large) being a dominating sound source, followed by synthetic sound sources (e.g., laboratory-created tones and other signals), seismic air guns, sonar, and wind- and non-wind-related pile drivers (Fig. 6). With the AN conferences, there has been a steady increase over the years on papers especially related to large vessels, as well as presentations related to ambient noise. These increases are likely to also reflect the recent evolution of science that is now focusing more on chronic sources of noise, rather than noise that is typically more acute (e.g., tactical sonar, pile driving, and seismic air guns; although, it is important to point out that there is a peak in seismic presentations in Den Haag 2016, which likely reflects the special workshop on modeling seismic air guns held after the conference (Halvorsen et al. 2019).

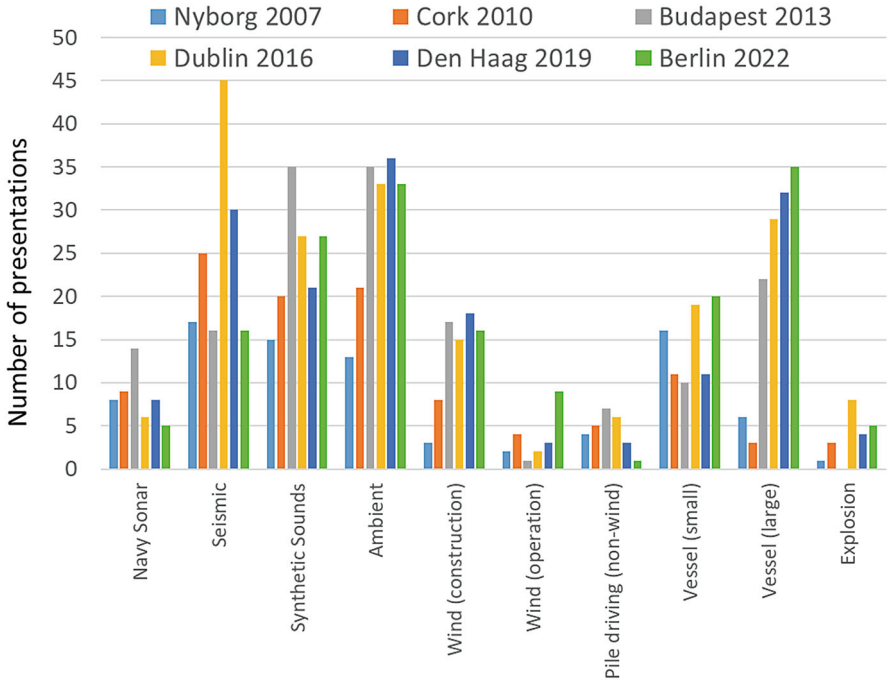



**Fig. 6** Statistics on the number of papers by sound source throughout Aquatic Noise's 15-year history

Specifically, in recent decades, there has been an effort to focus on broad-scale, long-term ambient noise measurements in Europe (e.g., Joint Monitoring Programme for Ambient Noise in the North Sea, https://northsearegion.eu/jomopans/; Joint Framework for Ocean Noise in the Atlantic Seas, https://www.jonasproject.eu/; The Baltic Marine Environment Protection Commission, https://helcom.fi/action-areas/marine-litter-and-noise/underwater-noise/; EU Interreg Italy-Croatia SOUNDSCAPE project, https://www.italy-croatia.eu/web/soundscape) and the United States (e.g., National Oceanic and Atmospheric Administration [NOAA]/National Park Service Ocean Noise Reference Station Network, https://www.pmel.noaa.gov/acoustics/noaanps-ocean-noise-reference-station-network; NOAA/US Navy SanctSound project, https://sanctuaries.noaa.gov/science/monitoring/sound/).

With offshore wind development increasing worldwide, presentations related to pile driving and wind farms have remained steady though AN's 15-year history. There was, however, a noticeable uptick in interest related to sound produced from operational wind farms that represents another form of chronic noise at the most recent conference in Berlin 2022. Interestingly, presentations on coastal pile driving (non-wind farm) have decreased since Budapest 2013.

Furthermore, other special topics, such as substrate vibration, particle motion, and kurtosis (Popper and Hawkins 2018; Müller et al. 2020; Hawkins et al. 2021) have steadily increased over AN's past 15 years, reflecting the increase in awareness and understanding of these topics especially to marine species (Fig. 7).

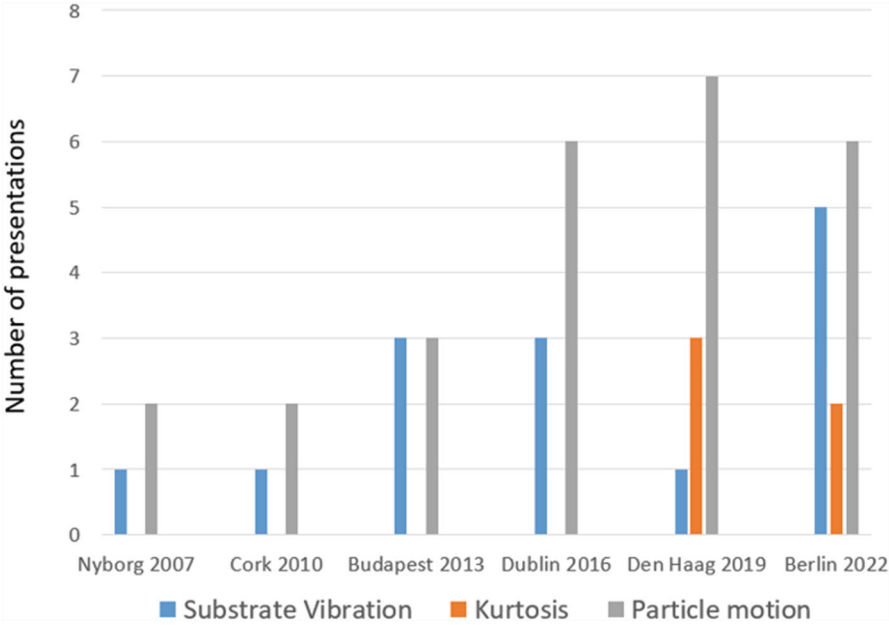



**Fig. 7** Statistics on the number of papers by special topic throughout Aquatic Noise's 15-year history

Kurtosis and substrate vibration even featured as keynote talks during the most recent conference in Berlin 2022. The Value of a Kurtosis Metric in the evaluation of Noise-induced Hearing Loss and Shaking up aquatic substrates: Taking lessons from biotremology

## Some Things Never Change

Despite the many aspects of AN that have changed and evolved over the past 15 years, some things have remained the same. For one, AN has been known for its scientific program, which has reflected the evolution of the science of this topic. Furthermore, the conference has not only focused on the science behind the issue of aquatic and marine noise but is one of the very few conferences that welcomes those from the regulatory, industry, and environmental communities to not only attend but also participate in presenting talks and posters. This valuable exchange between scientists and decision-makers is a unique feature of AN. Finally, AN encourages participation by students and postdoctoral researchers (and supports many of them financially), and in its very design, the meeting encourages networking between participants from all disciplines.

Furthermore, one of AN's strengths has been to have only one session at a time. This allows all participants to hear everyone's presentations and remain in the same room together to maintain the collaborative nature of talking and learning from one another before and after presentations. Again, this allows for a fruitful exchange between all disciplines ranging from underwater acoustics to animal physiology to regulation (Thomsen et al. 2020).

Finally, the unyielding support and effort by the conference's founding fathers Arthur Popper and Anthony Hawkins over the years have enabled AN to become one of the primary international conferences on the topic of underwater noise and aquatic and marine species. Today, AN is a conference many look forward to attending every 3 years. It will be interesting to see how AN evolves over the next 15 years.

**Acknowledgments** Many thanks goes to all the past Advisory Committee members who contributed to AN's continued strong scientific program throughout the years (in alphabetical order): Mathias Andersson, Michael Ainslie, Elizabeth Burkhard, Tom Carlson, Ben Colbert, Douglas Cato, Peter Cott, John Dalen, René Dekeling, Sarah Dolman, Rebecca Dunlop, Olga Filatova, Roger Gentry, Giacomo Giorli, Robert Gisiner, Mary Gray, Bruce Hanna, Vincent Janik, Craig Johnson, Ron Kastelein, Darlene Ketten, Anu Kumar, Christine Lamont, William Lang, Jill Lewandowski, Songhai Li, Klaus Lucke, Peter Madsen, David Mann, Nathan Merchant, Jennifer Miksis-Olds, Jim Miller, Lee Miller, Mirjam Müller, Sophie Nedelec, Michael Noad, Eric Parmentier, Marta Picciulin, Stephanie Plön, Roberto Racca, Vanesa Reyes, Denise Risch, Louise Roberts, Amy Scholik-Schlomer, Gail Scowcroft, Hans Slabbekoorn, Brandon Southall, Erica Staaterman, Jenni Stanley, Michael Stocker, Bill Streever, Alexander Supin, Mark Tasker, Jakob Tougaard, Sofie Van Parijs, Kathy Vigness-Raposa, Sander von Benda Beckmann, Magnus Wahlberg, Michael Weise, Jürgen Weissenberger, and Lidia Wysocki.

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# Effects of Noise on Aquatic Life: An Overview of the Berlin Conference

3

Joseph A. Sisneros, Frank Thomsen, Paul A. Lepper, Amy R. Scholik-Schlomer, Anthony D. Hawkins, and Arthur N. Popper

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**Abstract**

The "Effects of Noise on Aquatic Life" conference took place in Berlin, Germany, in July 2022, and the presentations have been revised to become the chapters for this book. The conference considered potential effects of anthropogenic sounds on aquatic species that ranged from whales, dolphins, seals, turtles, marine birds, fishes, and lesser-known taxa including zooplankton, mollusks, and seagrasses. It also considered the sources of the noises that might impact aquatic life and brought together researchers, regulators, industry, and environmental groups to consider, together, the potential impact of man-made sounds on aquatic life. Indeed, the meeting, as reflected in this book, brought together individuals and the latest ideas from a great variety of disciplines, ranging from biologists interested in behavioral and physiological effects of exposure to sound, to acousticians and engineers with an interest in the physics of underwater sound generation, to regulators and conservations groups concerned with the protection of aquatic life. Sessions at the meeting, as reflected in this book, included the measurement and management of underwater noise, an overview of the critical environmental issues related to offshore wind development as alternative sources of energy, and descriptions of changing soundscapes around the world. A social program was designed to encourage more leisurely discussions among the conference participants in order to facilitate networking and the strengthening of relationships among participants from academia, industry, and regulation. The feedback from conference delegates was very positive.

## Introduction

6th International Conference on the Effects of Noise on Aquatic Life

Berlin, Germany
10-15 July 2022

Meeting Overview: AN2022



Research on the potential impacts of underwater sound on aquatic life is challenging. New technological developments are taking place within the offshore industry, especially with respect to offshore wind development. Those concerned with assessing the impact of anthropogenic sounds on aquatic organisms are investigating the biology, physiology, and ecology of species not readily examined in the laboratory and often must develop new methods that can be applied to free-living animals in the sea itself. Since the issues are complex and cross lines between many disciplines that often meet separately, it becomes imperative to bring everyone together to consider the broad issues on the potential effects of underwater sound on aquatic life. In order to best tackle the issues, it is imperative to include scientists from a variety of disciplines, ranging from acousticians with an interest in the physics of underwater sound generation to biologists interested in physiological and behavioral effects of sound exposure. It is necessary for scientists and engineers to communicate effectively with one another on these topics, although they have often had difficulties understanding one another in the past and limited opportunities to network and discuss these issues with each other. Moreover, since the issues raised are significantly related to the protection of species and other environmental concerns, any discussion must also include colleagues from diverse areas such as regulators, environmental groups, industry, and other areas so that all share ideas and learn from one another.

To continue to raise and explore relevant issues, the "Sixth International Conference on the Effects of Noise on Aquatic Life" in July 2022 (AN2022) was convened in Berlin, Germany, to bring together scientists, regulators, and representatives from industry and environmental groups for formal presentations on the effects of anthropogenic sounds on aquatic life and numerous informal opportunities to "network." *The papers presented at that meeting are the basis for the chapters in this book.*

The meeting, as with past Aquatic Noise (AN) conferences (Erbe et al. 2019; Erbe et al. 2016; Popper and Hawkins 2016; Popper and Hawkins 2012; Hawkins et al. 2008), provided extensive opportunities for participants to learn about the effects of noise on aquatic life and the opportunity to interact with people, who focus on different taxa, different sound sources, and different approaches to understanding issues of unwanted sound and its mitigation to protect aquatic life and their habitats. There was a strong emphasis at the meeting for the communication of ideas between participants from different backgrounds (e.g., academia, industry, and regulation), and there was a strong interest by the attendees in the topics that composed the primary session themes at the conference (see Table 1). The AN2022 conference, like its five predecessors, was a great success in providing opportunities for cross-fertilization of knowledge, experimental approaches, and ideas.

The AN conference series was originally initiated by Anthony D. Hawkins, Arthur N. Popper, and Magnus Wahlberg in Nyborg, Denmark in 2007 (Hawkins et al. 2008; ▶ Chap. 1 , "Brief History of "The Effects of Noise on Aquatic Life" Meetings"). The overwhelming positive feedback and success of this initial conference led to additional conferences on the same topic (▶ Chap. 2, "Evolution of "The Effects of Noise on Aquatic Life" Meetings: 2007 to 2022"). See Table 2 for a list of the organizers for the previous conferences on the Effects of Noise on Aquatic Life.

**Table 1** List of topics/themes of the conference session at the Sixth International Conference on the Effects of Noise on Aquatic Life in Berlin, Germany

| Topic/Theme |
|---|
| Detection of sound: what turtles, birds, and humans hear |
| Soundscapes around the world |
| Noise effects on lesser-known taxa: seagrasses, zooplankton, and mollusks |
| Intense sounds and their effects |
| Priorities for regulation and management |
| Noise effects on the hearing and behavior of marine mammals |
| Acoustics and underwater sound emissions from windfarms, tidal stream turbines, and ultrasonic anti-fouling equipment |
| Management of noise |
| Hearing in marine mammals and fishes |
| Risk management of noise |
| Short- and long-term effects of noise on marine mammals |
| Oil and gas sounds and their effects on marine animals |

**Table 2** List of AN conferences and the conference organizers (listed by last name in alphabetical order)

| Conference | Location | Conference organizers |
|---|---|---|
| AN2007 | Nyborg, Denmark | Anthony Hawkins, Arthur Popper, and Magnus Wahlberg |
| AN2010 | Cork, Ireland | Anthony Hawkins and Arthur Popper |
| AN2013 | Budapest, Hungary | Anthony Hawkins and Arthur Popper |
| AN2016 | Dublin, Ireland | Christine Erbe, Anthony Hawkins, Arthur Popper, Joseph Sisneros, and Frank Thomsen |
| AN2019 | Den Haag, Netherlands | Christine Erbe, Anthony Hawkins, Paul Lepper, Arthur Popper, Joseph Sisneros, and Frank Thomsen |
| AN2022 | Berlin, Germany | Anthony D. Hawkins, Paul Lepper, Arthur N. Popper, Amy R. Scholik-Schlomer, Joseph A. Sisneros, and Frank Thomsen. |

Berlin was selected for the sixth AN conference (AN2022) for many reasons, including that it is a cultural center of Germany and that Berlin is easily accessible from anywhere in the world (via the Berlin Brandenburg airport). The Hotel Palace Berlin, in particular, met all the requirements for this meeting's venue, in addition to being near the epicenter of Berlin where there were ample opportunities for nearby outings.

The number of participants at the AN2022 conference in July 2022 was surprisingly strong given the recent COVID era. The number of delegates, countries represented, and presentations given at the conference were comparable to that of previous five conferences (see Table 3).

Forty-three people, including all students (undergraduate and graduate students) and postdocs who requested support, were funded thanks to our generous funders (see "Acknowledgments").

**Table 3** Aquatic noise conference information related to the number of delegates, countries represented, and presentations given at each of the conferences

| Conference | Location | Number of delegates | Countries represented | Number of presentations |
|---|---|---|---|---|
| AN2022 | Berlin, Germany | 240 | 26 | 186 |
| AN2019 | Den Haag, Netherlands | 312 | 23 | 214 |
| AN2016 | Dublin, Ireland | 323 | 23 | 229 |
| AN2013 | Budapest, Hungary | 243 | 24 | 125 |
| AN2010 | Cork, Ireland | 244 | 22 | 111 |
| AN2007 | Nyborg, Denmark | 202 | NA | 134 |

## Scientific Sessions at the Meeting

The scientific conference presentations were a mixture of 15-min oral presentations, 4-min speed talks (with posters), and posters. Each speed talk presented the highlights of an accompanying poster of the same title. In addition, there were also four 40-min keynote presentations. The following sections give brief summaries of oral and poster presentations by themes.

### Keynote Presentations

There were four keynote presentations, each on a different day (except for Wednesday, which was a half day at the conference). The intent of the keynote talks at the AN meeting is to share new ideas and information about broader aspects of hearing and behavior that can inform and broaden the perspective of participants.

The first keynote was by Christine Köppl (Carl von Ossietzky University Oldenburg, Germany) who talked about the evolution of vertebrate hearing. Dr. Köppl reminded us that all vertebrates hear using a sensory organ that resides in the labyrinth of the inner ear, but not all vertebrates hear the same way. Köppl detailed how the vertebrate inner ear and its hearing sensitivity evolved from fishes to early land vertebrates and how such changes in the inner ear led to very characteristically different morphologies and hearing sensitivities in lizards, birds, and mammals.

In the second keynote, Wei Qui (State University of New York at Plattsburgh, New York, USA) talked about the value of using kurtosis as a metric in the evaluation of noise-induced hearing loss. He gave examples from animal experiments and epidemiological studies on how kurtosis can be used to adjust noise-risk predictions and facilitate estimates of noise-induced threshold shifts.

Louise Roberts (Liverpool University, UK) talked on the biotremology of aquatic substrates. Biotremology is a new and emerging field in the biological sciences that involves the study of the production, dispersion, and reception of mechanical

vibrations by organisms and covers all aspects of animal behavior associated with substrate-borne mechanical waves.

Finally, Andrew Gill (Centre for Environment Fisheries and Aquaculture Science, UK) enlightened us about some of the critical considerations needed to understand the interactions and outcomes of renewable energy development on the marine environment.

## What Turtles, Birds, and Humans Hear

This session began with a short history of the "Effects of Noise on Aquatic Life" conferences from the first in Nyborg, Denmark (2007), to the current in Berlin, Germany (2022), given by Arthur N. Popper and Anthony D. Hawkins. Popper and Hawkins (▶ Chap. 1, "Brief History of "The Effects of Noise on Aquatic Life" Meetings") detailed the history of how the origin of the idea for the conference came about, how the conference has evolved (also see ▶ Chap. 2, "Evolution of "The Effects of Noise on Aquatic Life" Meetings: 2007 to 2022"), and its impact on the field of study that investigates the effects of anthropogenic sounds on aquatic life since the conference's origin to our present conference. Other talks detailed underwater hearing and noise exposure concerns in humans and temporary threshold shifts in turtles. Finally, a series of talks discussed in-air and underwater hearing in birds and the potential effects of noise and substrate vibrations on nesting penguins.

## Soundscapes from Around the World

Topics in this session included the contributions of vessel noise to ambient sound in the form of deep-water particle motion and how it can potentially mask hearing in fishes, how sound levels in a southern California antisubmarine warfare range change during a deep-water multibeam mapping survey, and how climate impacts can influence ambient sound on underwater reefs and animal call rates in different ways. Other talks included the measurement of underwater sound levels in the North Adriatic Sea, the characterization and identification of fish choruses and species contributions to ambient sound in Pamlico Sound in North Carolina, and the characterization of ocean sound and regional and global soundscape modeling.

## Noise Effects on Lesser-Studied Taxa

This session had a series of talks on the effects of noise on lesser-known taxa that included seagrasses, zooplankton, and mollusks. We learned that low-frequency human-generated noise can affect seagrass root statocysts, which are used to sense gravity and sound vibrations. Such alterations in statocysts can potentially affect the health of seagrass populations and other sessile organisms. In contrast, we also learned in certain cases that broadband noise can actually increase survival and

fecundity in freshwater zooplankton. Finally, a talk was given on how different sound exposures can affect mussels differently in terms of valve closing and filtration rates, which should be considered when assessing and mitigating noise effects on these and other potentially similar invertebrates.

## Intense Sounds and Their Effects

This session began with a talk that detailed a method called deflagration that involves an explosion where the flame speed is lower than the speed of sound for lowering noise levels from the clearance of unexploded underwater ordinance. Later, a series of talks were given that detailed the physical effects on Pacific Mackerel and another on the effect of explosive sounds on its inner ear. In addition, a study was presented that examined through numerical modeling the impacts of intense impulse sound on the fish inner ear. Finally, a talk was given that detailed underwater noise that is produced during percussive pile driving and the factors that influence pile-driving noise.

## Priorities for Regulation and Management

In this session, talks focused on detailed priorities for the regulation and management of underwater sound. Talks specified some of the research priorities for sound and vibration effects on fishes and aquatic invertebrates from offshore wind energy development and how adaptative management is critical for the noise management of offshore activities. Others talks described how moderate reductions in vessel noise source levels can dramatically reduce the negative impacts to marine wildlife and how managing technical and environmental uncertainty of subsea noise can impact project design.

## Effects on Hearing and Behavior

During this session, talks described the effects of sound on the hearing and behavior of marine mammals. The session began with a summary of the 16 years of behavioral response studies on sea mammals, sonar, and safety, which was followed by talks on how vessel noise and impulsive (and non-impulsive) sounds can affect energy balance and hearing thresholds in marine mammals. Other talks included the description of frequency-weighted thresholds for hearing loss and behavior disturbance in porpoises and phocid seals, and the use of behavioral experiments to estimate hearing sensitivity in humpback whales.

## Acoustics

Talks in this session included the underwater sound emission from floating wind turbines and the assessment of long-term offshore windfarm noise and how it

influences fish chorusing. Other talks included the comparison of underwater operational noise of two tidal stream turbines and the modeling of single turbines versus turbine arrays. Finally, an interesting talk was given that described the sound emissions from ultrasonic anti-fouling equipment and how it can substantially impact marine life.

## Management of Noise

Two sessions were dedicated to noise management. In the first session, talks included how to best assess impulsive underwater noise in Europe during the next European Union (EU) Marine Strategy Framework Directive cycles and a description of the joint monitoring program for ambient noise in the North Sea. Other talks included real-time monitoring of underwater radiated noise levels from ships using a cable hydrophone system in shallow water, revised Danish guideline for underwater noise from the installation of impact- or vibratory-driven piles, evaluating the benefits of noise reduction mitigation, and autonomous surface vessel passive acoustic baseline monitoring of marine mammals offshore in West Africa.

The second session's talks included a presentation on the best available technology and best environmental practice for mitigating three noise sources: shipping, seismic airgun surveys, and pile driving. Other talks included the development of underwater acoustic deterrents for invasive species control, the response of harbor porpoise to an acoustic deterrent and subsequent piling during the construction of offshore wind farms, and a talk on encouraging quieter shipping.

## Hearing

This session started with a presentation in honor of the late Dr. Sam Ridgway and his extraordinary career in marine mammal research. The presentation was given in an innovative and interactive format that allowed Sam to tell his stories using a new interactive video technology. Other talks included measures and models of dugong hearing, susceptibility of harbor porpoise hearing to intermittent sound pressure, zebrafish hearing in relation to hearing loss, physiological stress, and behavioral changes due to noise. Finally, a talk was given on the use of kurtosis to describe down-the-hole pile installation sounds related to the frequency response of marine mammal hearing.

## Risk Management

Talks in this session included a presentation on an alternative methodology to assess quantifiable noise level risk using a combination of noise pressure and species distribution maps based on a Bayesian-inspired estimation method. Other

talks included an ecological risk assessment approach for underwater sounds from dredging operations, regional noise management in the Northeast Atlantic, the management of noise in a busy harbor porpoise protected area, and the limiting of motorboat noise on coral reefs as a means to boost fish reproductive success.

### Short- and Long-Term Effects

Topics in this session included the adjustment of song unit frequencies among humpback whales in response to small boat noise; minke whale swimming behavior relative to the environment; conspecifics and Navy sonar; and examination of how shipping noise affects the abundance and distribution of harbor porpoise during high levels of marine traffic in the Baltic Sea. The final presentation in this session was on the cumulative effects of multiple stressors on marine mammals using elephant seals as a model system.

### Oil and Gas Sounds and Their Effects

In this session, presentations were on the measurement procedures for underwater sound sources associated with oil and gas exploration and production activities, the measurement of sound source levels of mobile offshore drilling units, and the ocean seismic survey impacts on the movement and distribution of nearshore temperate marine species. The final talk was on the behavioral responses of spawning cod to seismic airgun exposure.

## Rodney Coates Award

The Rodney Coates Award for the best student presentation was given on the last day of the conference. This award was made possible by the generosity of the Coates family in the memory and honor of the late Rodney Coates, who attended several of our conferences and was very interested in the research that was presented at the conference. Candidates for the Coates award either had to be a registered graduate student or had to have received their doctorates in 2022 from a recognized college or university. Each candidate was assessed for presenting both a speed talk (4 min) and a poster on their work. A team of 13 judges drawn from the scientific community at the conference were asked to judge both the talk and poster. The 2022 judges were Sander von Benda Beckmann, Benjamin Colbert, Hilary Kates Varghese, Klaus Kucke, Nathan Merchant, Jennifer Miksis-Olds, Mirjam Müller, Michael Noad, Marta Picculin, Denise Risch, Louise Roberts, Kathy Vigness Raposa, and Jürgen Weissenberger. All agreed that the quality and standard of the presentations were of high quality. The judges had a difficult time assessing the winners because of the high standard and quality of the

presentations. Three prizes were given: first place (£1000 prize), second place (£500 prize), and third place (£250).

The first-place winner was Johannes Veith from Charité Universitätsmedizin Berlin, Germany, for his presentation on "Directional Hearing in *Danionella cerebrum*." The second-place winner was Kees te Velde from Leiden University, the Netherlands, for his presentation on "Habitat-associated soundscapes of a lowland Dutch river system: Potential and vulnerability of auditory cues to migratory fish." The third-place winner was Emilie Hernes Vereide from the Institute of Marine Research, Vestland, Norway, for her presentation on the "Impacts of airgun blasts used in seismic surveys – Small effects on small copepods."

## Social Program

The social program began on Sunday night with a welcome reception in the Hotel Palace Berlin. The conference opened on Monday morning with two lovely arias sung by Oleksandra Diachenko. The program ended early on Wednesday afternoon (a conference tradition) so that the delegates could network and enjoy the local sights of Berlin with friends and family. On Thursday evening, the Hotel Palace Berlin set up a disco at the hotel bar for the delegates to enjoy music and dance. The conference concluded with the banquet in the Hotel Palace Berlin banquet room with live entertainment from a Marlene Dietrich impersonator and singer.

## Participant Feedback

An on-line survey was set up at the end of the conference, and 75 participants responded. We had more male (61%) than female (39%) participants in the survey, with the largest percentage of responders being students and postdocs (Fig. 1). The attendance of these young investigators at the AN2022 conference was enabled by the strong external grant support that was received. Attendees that were supported to attend by conference grants were from 13 countries including Argentina, Australia, Brazil, Canada, Colombia, Germany, Italy, the Netherlands, Norway, Poland, Portugal, the United Kingdom, and the United States.

Figure 1 shows the distribution of the professions of the survey responders. Academics, which include students (undergraduate and graduate), postdocs, and faculty, made up the majority (75%) of the responders. One goal of the conference was to attract a diversity of stakeholders, including those outside of research, which was achieved at the conference and was not necessarily demonstrated by the number of responders in the survey.

Table 4 summarizes the responding attendees' score regarding a number of specific aspects of the meeting. Based on the percentages of scores 4 or 5 (liked or liked greatly) from the survey, there was a broad satisfaction with the conference including

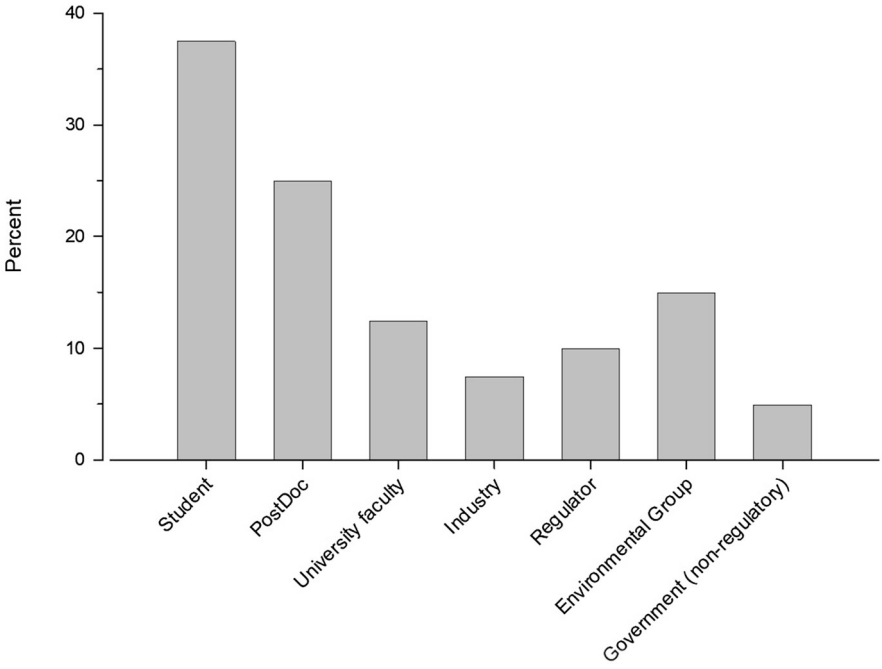



**Fig. 1** Profession of the survey responders. Note: the percentage of the responders exceeds 100% because some of the participants listed two professions, and both were counted

**Table 4** Scores for different aspects of the meeting (75 responders)

| | 1 (did not like) | 2 | 3 (neutral) | 4 | 5 (liked greatly) |
|---|---|---|---|---|---|
| Delivery of conference presentations (Oral, Speed Talks, Coates & Posters) | 0% | 1% | 7% | 44% | 48% |
| Meeting website | 3% | 3% | 17% | 53% | 24% |
| Abstract submission and site | 2% | 3% | 17% | 52% | 26% |
| Meeting registration (both website and in-person) | 0% | 0% | 23% | 41% | 36% |
| Information provided prior to the meeting | 0% | 3% | 19% | 33% | 45% |
| Welcome buffet and/or conference dinner | 2% | 3% | 28% | 19% | 48% |
| Sufficient break and lunches | 0% | 3% | 2% | 21% | 74% |

the delivery of conference presentations (Oral, Speed Talks, Coates & Posters) (see Table 4). We are very pleased that VenuesWorld, who handled the logistics of the meeting, also received very strong support from the attendees. Table 5 lists attendee-specific feedback on the scientific content of the meeting. As can be seen, there is a strong consensus that the scientific content was excellent, as further supported by the responses to several of the questions used to generate Table 5.

**Table 5** Verbatim responses (unedited other than spelling corrections and removal of duplicate responses from different delegates) to a question about the scientific content of the meeting

| | |
|---|---|
| Excellent | Very good |
| Very useful | Nice diversity |
| Good mix | Highly varied |
| Nice variety of subjects | Great content |
| Very high standard | Cutting edge and highly educational |
| Very interesting, complete and high level | Generally, talks were well done. |
| I enjoyed the breadth of the scientific content. | Mostly well-presented and researched |
| Inspiring!! So happy to learn a lot | Excellent across the board. |
| The best among the ones I've attended. | The scientific content was good |
| High level in general, pretty diversified | Talks were great! I learned so much! |
| Very useful and fits well with my research! | High, much better than most conferences |
| Good quality and variety | Great and very interesting topics |
| A good variety of topics/taxa/disciplines well covered. I learnt a few new things! | Good mix of recent research, rising topics/ issues, and practical topics. |
| Very cool. One of the best talks was the one about biotremology. | Great, sessions on turtles and birds, inverts, where good |
| Excellent and well organized. Matched well to the state of the field. | Very interesting, broad (animal group, kind of stressors, etc.) and relevant |
| It was a decent mix of research on biology and on underwater acoustics | I liked the flow of the program although the work presented was all very similar in method and content |
| Fabulous! the diversity was spot on, the continuum of subjects and how many of them built on each other and most importantly the single room aspect. my gosh what a great idea not to have to been torn between speakers. | Very good blend of topics. Good to include speakers from outside the field, very inspiring. |

## Conclusion

At the five-day AN2022 conference in Berlin we heard about new research on the effects of anthropogenic sound on marine animals including birds, turtles, fishes, and even humans. We also learned about how anthropogenic sound can impair seagrasses and invertebrates. Our understanding of the potential impacts on marine fauna and flora is growing, and we are beginning to have a better understanding of how anthropogenic sounds can affect aquatic life. At AN2022, we had the opportunity to discuss underwater anthropogenic sounds and their potential impacts on aquatic life with colleagues and fellow scientists, students and postdocs, industry representatives, government regulators and representatives, and members of non-government organizations. This AN conference, like the previous ones, has allowed our participants to get a better understanding of each other's concerns with regard to the potential impacts of anthropogenic sound and has allowed us to build relationships and grow as a community.

In conclusion, the organizers were very satisfied and pleased with the outcome of this year's conference and look forward to planning for the next AN conference in

2025. The venue for the next meeting in 2025 will be in Prague, the capital of the Czech Republic. Please keep an eye on aquaticnoise.org for updates.

**Acknowledgments** The conference leading to this book could not have taken place were it not for generous funders and numerous colleagues who graciously provided help, especially during uncertain times during the COVID era. We are indebted to our funders and sponsors, who enabled us to support the attendance of students, postdocs, and colleagues from developing countries. We thank our funding organizations: the Aquatic Noise Trust (UK), the Bureau of Ocean Energy Management (USA), the Marine Mammal Commission grant MMC21-036 (USA), the National Oceanographic and Atmospheric Administration grant NA20OAR4320271 (USA), the National Science Foundation grant OCE2136084 (USA), the U.S. Office of Naval Research award N00014-22-1-2814 (USA), the Naval Facilities Engineering Command (USA), and the U.S. Navy Living Marine Resources Program grant (USA).

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# Section II

# Sounds and Sources

# Effects of Array Design on Acoustic Propagation Modeling in Seismic Surveys

4

Alexander S. Douglass and Shima Abadi

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**Abstract**

Marine seismic surveys are large-scale tests used to study the seabed and sub-seabed by utilizing sound reflected from the ocean bottom, generated from airgun arrays. Airgun arrays used in these experiments often exceed Sound Exposure Levels (SEL) as high as 220 dB (re 1 μPa), and thus it is important to understand the potential impact of these surveys on marine mammals. To mitigate any harm that may be caused to nearby marine life, propagation models should be implemented to estimate the sound exposure level as a function of range for a given environment. These models are complicated by both the source array design, which effects the propagation modeling, and the receiver array design, which alters the data processing by misrepresenting the acoustic field. The purpose of this manuscript is to examine, through simulation, the potential effects of both of these arrays on both modeling acoustic propagation and interpreting acoustic data from the experiments.

**Keywords**

Seismic survey · Beam patterns · Mitigation ranges · Airguns

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## Introduction

Marine seismic surveys are experiments designed for studying and mapping the geological characteristics of the ocean, often up to multiple kilometers below the seabed. These surveys typically consist of an airgun array, capable of generating source signals above 220 dB *(re 1 μPa)*, broadcasting sound focused toward the ocean floor, with the acoustic reflections measured by a receiving array towed by the same ship towing the source array. Guidelines for sound exposure levels (SEL) corresponding to permanent threshold shift (PTS) and temporary threshold shift (TTS) in marine mammals are provided by the National Marine Fisheries Service (NMFS) (National Marine Fisheries Service 2018), and though not as well regulated, this sound exposure can potentially impact other marine life as well. However, propagation models used for understanding the impact of airgun pulses on marine mammals need improvement based on the NMFS 2018 standard. An understanding of the array designs for seismic surveys and their effects on the propagation and measured data are critical to these models.

Some work has looked at propagation in seismic surveys using both simulations and actual seismic data (Deruiter et al. 2006; Abadi and Freneau 2019a, b), including work on characterizing the spectra of the airgun arrays, again both in simulation and experimentally (Diebold et al. 2010; Guo-Fa et al. 2010; Sertlek and Ainslie 2015; Duncan and Gavrilov 2019; Prior et al. 2021). Little attention has been paid to the design of receiver arrays, which will be described in section "Description of MGL1212" of this manuscript, perhaps because the design details are not always immediately obvious (or even included) in the cruise documentation.

The intent of this chapter is to examine the effects of the source and receiver array configurations on transmission loss in acoustic propagation models that utilize parabolic equation methods. The impact of the receiving array design is particularly essential when considering actual acoustic data collected during these seismic surveys. The airgun array beam pattern is critical when trying to accurately simulate the acoustic field expected in seismic surveys. Section "Description of MGL1212" describes the experiment that forms the basis for this analysis, followed by a section of results and analysis, and finally a summary of the main conclusions.

## Description of MGL1212

The MGL1212 experiment (Holbrook et al. 2012b) forms the basis for the simulations completed in this manuscript – providing the parameters for array design and approximate geometries and sound speed profiles to use in simulation. Further, experimental data will be considered here to justify this work, but experimental data will not be utilized in the results presented.

The MGL1212 experiment took place off the coast of Washington State (USA) in summer 2012 to further study the Cascadia Subduction Zone, covering the regions shown in Fig. 1. The source and receiver arrays utilized by the ship, the R/V Marcus G. Langseth, consisted of a four-string airgun array and an ~8 km towed hydrophone

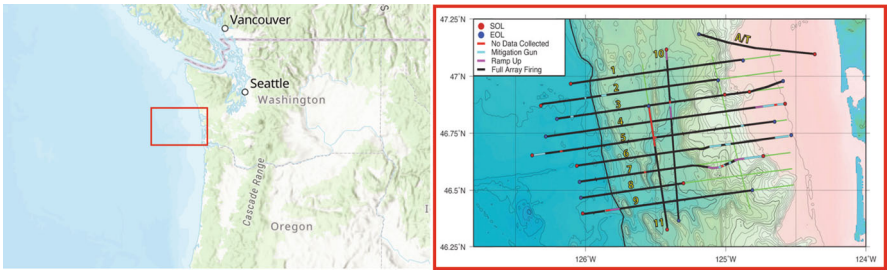



**Fig. 1** The approximate location and a detailed map of the MGL1212 seismic survey (Reproduced from Holbrook et al. 2012a, b). Black lines show the main track lines including nine east-west lines across the continental slope, two north-south lines along the strike of the continental margin, and one line northeast of the main survey area on the continental shelf

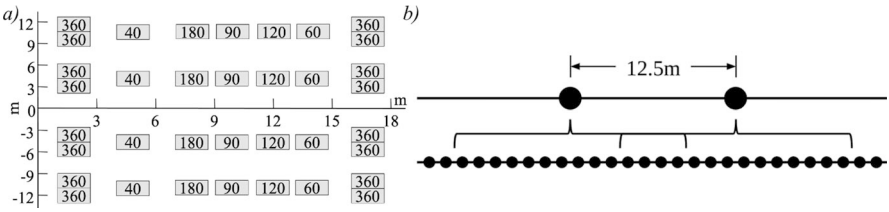



**Fig. 2** Array geometries from the MGL1212 experiment. The source array (**a**) consists of four identical lines of nine airguns with relative positions shown here. Each box represents an airgun with a volume in cubic inches corresponding to the number in the box. The receiver array (**b**) consists of groups of 14 hydrophones averaged together to give each channel output, with four hydrophones overlapping between each channel

array. Each string of the source array was identical, consisting of nine airguns of varying volumes, totaling 1650 $in^3$ per string (6600 $in^3$ for the full array). The receiver array consisted of 636 receiver channels, spaced 12.5 m apart, with the first receiver channel at 264 m from the center of the source array. Each channel, despite providing a single output, is an average of 14 hydrophones, spaced 1.25 m apart, with the channel position taken at the center of the group. Additionally, each group has four hydrophones overlapping with the neighboring groups. These geometries are laid out in Fig. 2a, b. The effects of these array geometries on the acoustic output are the primary focus of the remainder of this chapter.

The importance of understanding propagation characteristics of these experiments is shown in Fig. 3. Figure 3(a) shows a waterfall plot from a shot on line 6 (shown in Fig. 1) at roughly 2000 m depth, with range increasing with receiver channel, displaying the time domain output for every 15th receiver. Here the direct path, first reflection, and second reflection are all clearly visible. Figure 3(b) shows the corresponding sound exposure level (SEL) for this shot, as well as references for TTS and PTS levels for low-frequency marine mammals (as defined by NMFS 2018) and a line (red) for a cylindrical spreading model. While some aspects of the SEL curve are reasonably represented by a cylindrical spreading model, there are clearly

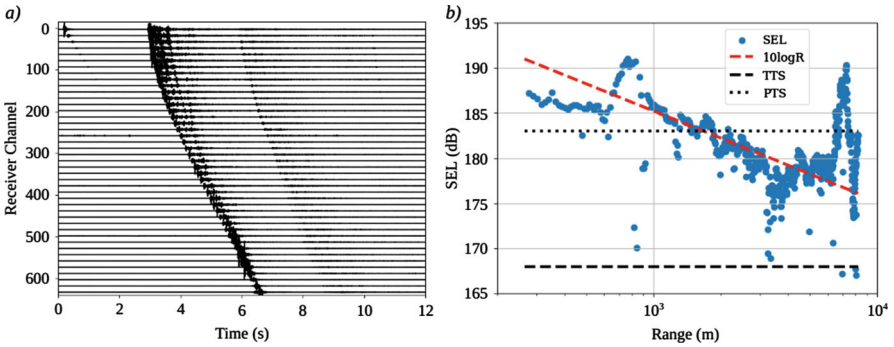



**Fig. 3** Results from a single shot in MGL1212. A waterfall plot (**a**) showing the time domain output for every 15th receiver, range increasing with channel number. Sound exposure level in dB (**b**) for the same data as a function of range, plotted on a log scale. The black lines represent the TTS (dashed) and PTS (dotted) thresholds for low-frequency marine mammals as defined in the NMFS 2018 document

some ranges, particularly between ~600–900 m and ~ 7 km, that vary significantly from this model, indicating that more complexity exists in the propagation than is considered by simplified spreading models. Thus, further analysis of experimental data and investigation of the impacts of various parameters on acoustic propagation is necessary. The first step to being able to analyze this data successfully is understanding the impacts of the experimental geometry (i.e. the source and receiver array designs) on the data itself. The next section considers these impacts from simulations of array geometries and acoustic propagation.

## Analysis and Results

The receiver array geometry is a simple linear array with equal receiver spacing; thus, the beam pattern can be calculated analytically (Kinsler et al. 2000). Figure 4(a) shows the directionality of the array for several frequencies, with 0° corresponding to horizontal propagation (along the array) and 90° corresponding to vertical propagation (perpendicular to the array and toward the sea surface and bottom). These patterns show that at low frequencies, such as 20 Hz, the beam pattern approaches an omnidirectional pattern. However, at higher frequencies, the difference in amplitude is highly angle-dependent, indicating that varying propagation angles will significantly impact the effects of receiver averaging. For example, a 100 Hz sound wave arriving at 50° relative to the horizontal will yield a channel output roughly 5 dB quieter than the physically present acoustic field. For a broadband SPL, applying appropriate weightings to the beam pattern and combining can yield a broadband beam pattern for analysis, shown in Fig. 4(b). The two curves show an unweighted (all frequencies of equal amplitude) and a weighted broadband beam pattern (a decay of 10 dB per order of magnitude of frequency is chosen for simplicity). In the weighted case, the low frequency components are much more significant, and thus a

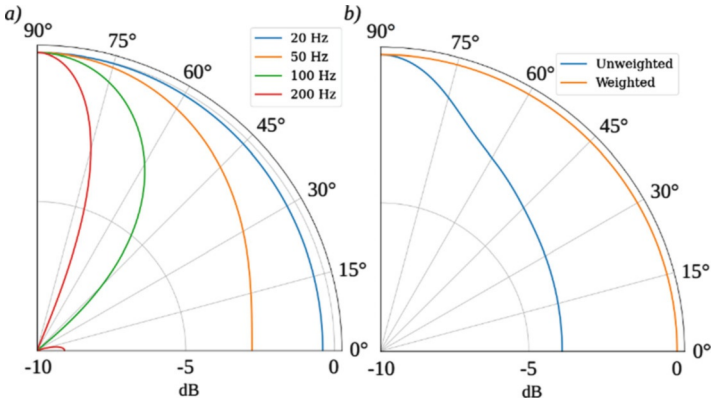



**Fig. 4** Beam pattern outputs for the MGL1212 receiver array geometry. Panel (**a**) shows the results for several individual frequencies, and panel (**b**) shows the result for a broadband signal between 20 Hz and 220 Hz with a flat spectrum (unweighted) and a spectrum with relative intensities decreasing by 10 dB per order of magnitude of frequency

nearly broadband beam pattern is achieved. In the unweighted case, the influence of the higher frequencies is clearly present. In reality, a more complex weighting exists that likely generates a beam pattern lying between these two curves.

For the source array, an analytic solution does not exist since each array element is of a different volume and the airgun spacing is not consistent. Thus, the beam pattern is calculated by choosing a point in the far field that corresponds to a given angle, calculating the acoustic field at this point from each airgun, and combining coherently, assuming each airgun is an independent point source with amplitude scaled by the cube root of its volume, and that all airguns are fired simultaneously. This is a notably simplified simulation, since the airguns are significantly more complex as sources, and the simulation ignores any interaction between airguns. Nevertheless, it provides insight into the importance of the array beam pattern. Figure 5 shows the same results as Fig. 4, for the airgun array rather than the receiver array, with notably different beam patterns resulting, due to the array complexity. In particular, side lobes start to appear at some of the higher frequencies, and the weighted and unweighted curves are actually more similar than one might expect, but still with non-negligible differences. As expected, the sound energy is focused in the vertical direction, as per the experimental design.

In order to evaluate the effects of these arrays on transmission through the water column, RAM simulations (Collins 1993) will be performed at multiple depths, incorporating environmental parameters specific to the MGL1212 experiment. Figure 6 shows the sound speed profiles used for the simulations, both in the water column (a) and the seabed (b). These are based on direct XBT measurements from the MGL1212 experiment for the water column (interpolated and extrapolated for deeper water), and an inversion result for the seabed. When the water depth was shallower than the profile presented here, it was simply truncated at the corresponding depth. The 0 value for the seabed profile corresponds to the water

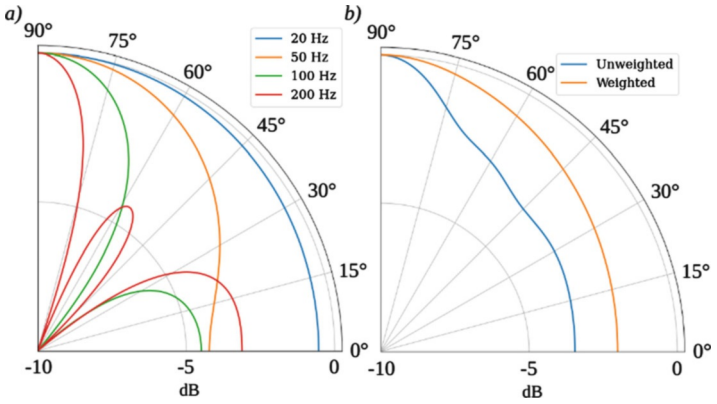



**Fig. 5** Same as Fig. 4 but for the source array. Airguns are modeled as point sources

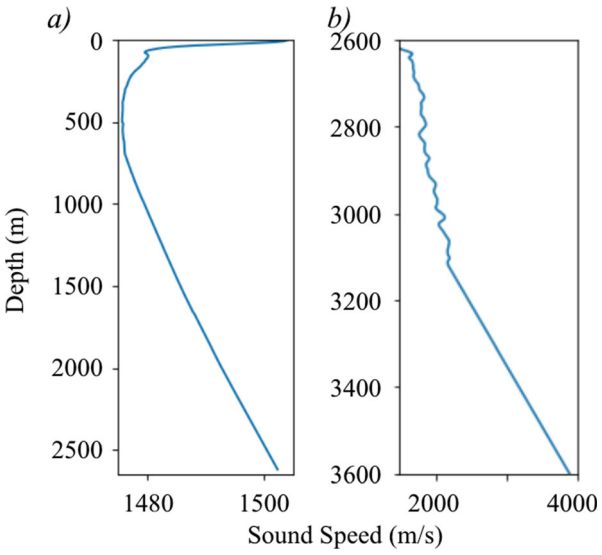



**Fig. 6** Sound speed profiles used for RAM simulations. The profile for the water column (**a**) is based on measurements taken during the MGL1212 experiment. The profile for the seabed (**b**) is based on an inversion result from MGL1212 data. The 0 m depth in the seabed profile corresponds to the water column/seabed interface

column/seabed interface. Here, the effects of the source array and receiver array geometry will be considered together. However, there is a notable difference between the effects of these two arrays. The airgun array, as an active source, will generate a beam pattern in the physical application; thus, its effects are important to consider for simulating propagation, but the received data does not need adjustment to represent the physical field. However, the receiving array output is the result of an array averaging process, and thus the effects of the array beam pattern are not physical but are rather a result of a signal-processing step. Thus, for array data to accurately represent the physical field, the beam pattern must be compensated for.

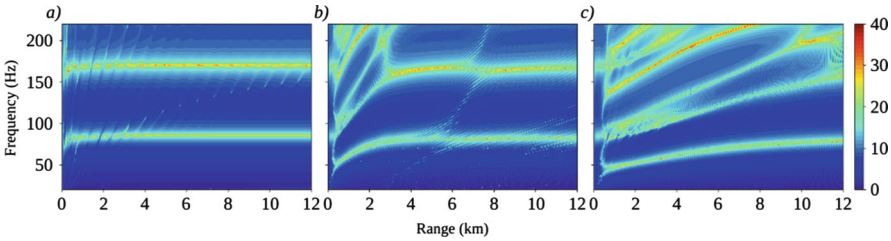



**Fig. 7** The differences in dB between two scenarios: the two arrays shown in Fig. 2 and a case with a single point source (with volume equivalent to the array total) and no group averaging for the receiver array. This is shown for three depths, 50 m (**a**), 670 m (**b**), and 2618 m (**c**)

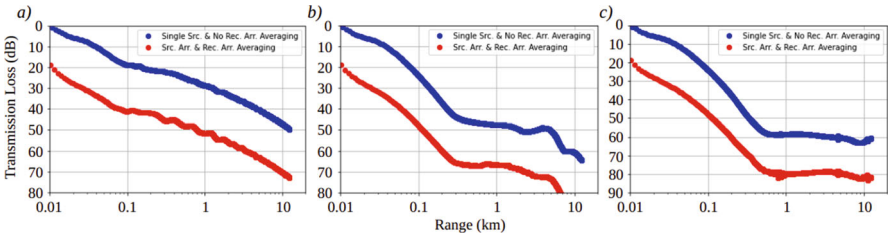



**Fig. 8** TL curves for two cases, one with the arrays shown in Fig. 2 (red curve), and a case with a single omnidirectional source (with volume equivalent to the array total) and no group averaging at the receiver array. This is shown for three depths, 50 m (**a**), 670 m (**b**), and 2618 m (**c**)

The impact of these beam patterns can be significant factors in calculating the acoustic field in simulation, especially at frequencies significant for the array geometries. This influence is dependent on the signal spectrum, so it is worth knowing the impact as a function of frequency. Additionally, since the beam pattern is angle-dependent, the range-dependence will vary as a function of bathymetry. Figure 7 summarizes the impacts of the arrays as a function of frequency and range for various depths. Here, the plots show the differences between a case with a single omnidirectional source (with the same amplitude as the total source array from MGL1212) and single omnidirectional receiver, and a case with source and receiver arrays based on the MGL1212 geometry. The panels show the results for depths of 50 m (a), 670 m (b), and 2618 m (c), each of which is based on geometry and sound speed profiles from the MGL1212 experiment. In each of these cases, it is notable that the influence of the array geometry remains relatively constant after ranges of roughly 7.5 times the depth of the water channel are reached. Prior to that distance, the influence of the array geometry is highly range dependent. Additionally, depending on the frequency, the differences can approach nearly 40 dB, having a potentially major effect on the overall SEL.

The differences between SEL and TL when accounting for these array effects can be quite significant. Figure 8 shows the TL differences for the same environments used in Fig. 7, for unweighted broadband signals. The results in this Figure show differences of up to 20 dB between the two cases, indicating the significance of

including these geometries in any models or simulations used to evaluate SEL in seismic surveys. This difference will vary significantly depending on the frequency weighting, with larger differences, when the frequencies corresponding to higher values in Fig. 7 have a higher weighting.

## Conclusion

The simulation results presented in this proceeding provide several key conclusions. First and perhaps most significant, the influence of array design can generate up to 10–20 dB differences in predicted versus actual TL or SEL. The effects of receiver array averaging are much more significant contributors to this difference and relevant when analyzing acoustic data collected during these surveys. The source array design is still an important factor as well, particularly for simulating the acoustic field. Second, the differences in received levels will vary as a function of frequency and range, up until a range of ~7.5 times the channel depth is reached, at which point the range-dependence becomes insignificant. Lastly, the effects of the array design can be captured as angle-dependent beam patterns, either at individual frequencies, or as a broadband beam pattern if information about the signal spectra are available.

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# Sound Fields in Two Small Experimental Test Arenas: A Comparison

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Amelia Holgate, Paul R. White, Timothy Leighton, and Paul S. Kemp

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**Abstract**

Lab-based bioacoustics experiments on fish enable control of confounding factors, yet these experiments often use small tanks that are spatially complex in terms of particle motion and sound pressure levels (SPL). One solution is to submerge an arena in water so the influence of the walls on the acoustic environment is greatly reduced. The experiment answered the following: (1) Can the sound pressure homogeneity in small tanks be improved? (2) Can the particle acceleration (PA) levels in small tanks be reduced? (3) Does submerging small

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tanks allow the sound field to be reliably described by progressive plane waves? The first setup consisted of a rectangular tank surrounded by air (in-air tank) and the second setup was a cylindrical arena submerged in water (submerged arena). Measurements of a 1 s, 125 dB tone at 400–2000 Hz were taken and mapped. The submerged arena possessed less heterogeneity in SPL and lower levels of PA than the in-air tank. The predicted PA (calculated to fulfill research question three) was a better approximation to the submerged arena than the in-air tank. This study demonstrates that the submerged arena gave greater control over the stimulus that a fish experiences in lab-based experiments.

**Keywords**

Tank· Acoustics· Particle acceleration· Sound pressure· Experimental· Methodology.

## Introduction

Both laboratory and in situ bioacoustics experiments on fish have benefits and disadvantages. Experiments that are either fully in situ or in a pen have high behavioral and acoustic validity but low experimental control (Slabbekoorn 2016). Conversely, tank experiments lack acoustic validity due to heterogeneous sound pressure levels (SPL) and particle motion, in which both the amplitude and the directional components vary substantially. While experiments in situ may be more favorable in terms of acoustic validity, indoor tank experiments have a much higher degree of experimental control (Akamatsu et al. 2002). This is advantageous for behavioral studies in which external factors, such as water temperature and ambient noise, may compromise the behavior more so in situ than in the lab (Akamatsu et al. 2002; Johnsson and Näslund 2018). However, the effect of confinement in a tank on the behavior of fish has long been a concern for its potential to generate deviations from wild behavior (Hawkins et al. 2015; Hawkins and Popper 2016; Popper and Hawkins 2018).

Small tanks have complex soundscapes owing to their small size, the elasticity of the tank walls, and the large impedance differences between water and air at the tank boundaries (Rogers et al. 2016). For example, consider a rectangular tank made from glass or acrylic which will have five hard surfaces (four sides and a bottom) and the water surface in close proximity to each other. In such circumstances there is a large degree of scattering from all the boundaries. The tank characteristics, particularly the tank size, shape, and the impedance difference, give rise to reverberation, in which the acoustic field cannot be regarded as homogeneous (Akamatsu et al. 2002).

In acoustic experiments, inhomogeneous sound fields mean that the sound level received by a subject vary according to their position within the experimental arena. In the problem we consider, the received SPL and particle motion experienced by a fish would be subject to a high level of uncertainty. For example, the addition of a fish may perturb the sound field to one that differs from that measured in a fish-less

tank, e.g. because of scattering from the swim bladder (which would, of course, also happen if the fish were in the field) (Stanton 1989; Diachok 2001; McKelvey and Wilson 2006). Additionally, the complex nature of the sound field makes it challenging to reproduce.

A practical way to improve sound pressure and particle motion fields, while using readily available equipment, is to alter the experimental methodology, exploiting the reduction in acoustic intensity as the wavefront propagates away from the source. This may involve setups in the field (Debusschere et al. 2014) or in a pen (Hubert et al. 2020). However, the use of tanks allows control of environmental factors which would be impossible in the field (Akamatsu et al. 2002; Gray et al. 2016; Currie et al. 2021). To increase the extent to which a tank is homogeneous and isotropic, large impedance differences should be removed. It would be impractical, and for many experimenters prohibitively expensive, to try to improve sound field homogeneity through the use of (very large) anechoic wedges on the arena walls. For the frequencies in question, these wedges would be of ≈1 m in length (Kolaini and Crum 1994), requiring a very large anechoic tank to provide a working space free of the niches provided by wedges in which the fish could be contained (e.g., by a net). Hence, this study tests a more affordable and practical method, exploiting the reduction in acoustic intensity as the wavefront propagates away from the source (rather than its conversion to heat by wedges) to homogenize the sound field. The experiment aimed to answer the following research questions: (1) Can the sound pressure homogeneity in small tanks be improved? (2) Can the particle acceleration (PA) levels in small tanks be reduced? (3) Does submerging small tanks allow the sound field to be reliably described by progressive plane waves?

## Methods

Two experimental setups were considered. The first setup consisted of a rectangular tank surrounded by air (in-air tank) and the second setup was a cylindrical arena submerged in water (submerged arena). For both setups, the sound field was mapped and then compared.

### Experimental Setup 1: In-Air Tank

Acoustic mapping was carried out in an experimental arena (86 cm length × 30.2 cm height × 30.8 cm depth) situated within a custom-made still water rectangular acrylic tank (300 cm × 30.8 cm × 30.2 cm; 12 mm thickness; 27 cm water depth; mean ± sd temperature 18.8 ± 1.4 °C) in September 2019 at the University of Southampton's International Centre for Ecohydraulics Research (ICER) facility, UK. The tank was situated on concrete blocks and surrounded by air on all sides (Fig. 1). Two underwater transducers (Electro-Voice UW-30; maximal output 153 dB re 1 μPa at 1 m for 150 Hz, Lubell Labs, Columbus, OH, USA) were

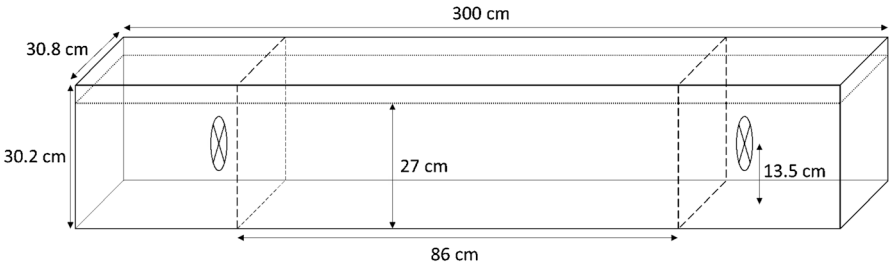



**Fig. 1** The rectangular in-air tank (300 cm length × 30.2 cm height × 30.8 cm depth) and experimental arena (86 cm length) in relation to the water level (dotted line, 27 cm water depth). Two transducers were suspended centrally at either end of the experimental arena and behind an acoustic baffle (dashed line)

suspended centrally, 13.5 cm from the bottom of the tank and 5 cm away from baffles that flanked the experimental arena.

## Experimental Setup 2: Submerged Arena

Acoustic mapping was carried out in a white medium-density polyethylene cylindrical arena (modified 100 L Round Water Tank; 55.5 cm diameter, 45 cm height; 4 mm thickness; 30 cm water depth; Direct Water Tanks, Retford, Nottinghamshire, UK) in November 2020 at the University of Southampton, UK. The submerged arena was mounted on a bespoke metal frame and suspended in a large tank (8 m length × 8 m width × 5 m depth) (Fig. 2). An underwater transducer (Electro-Voice UW-30) was suspended 70 cm below the arena.

### Acoustic Stimuli and Sound Mapping

Sinusoidal 1 s tones at five frequencies (400; 600; 800; 1000; 2000 Hz) were produced in MATLAB (Version 2019b, The Mathworks, Inc., Natick, USA) using a laptop connected to a DAQ (NI USB-6212; National Instruments, USA) operating at 25.6 kHz, transmitting the signal through an amplifier (Prosound Power AMP 200; frequency response: 20 Hz–20 kHz), and was emitted via the UW30 underwater transducer. Acoustic stimuli were standardized such that the SPL (125 dB re 1 μPa) was reached in the center of the experimental arena. Use of artificial stimuli allowed for control of the specific acoustic components tested.

The sound pressure field was measured and mapped using a calibrated hydrophone attached to a manual slider (for the in-air tank: 8103, manufacturer-calibrated sensitivity −211 dB re: 1 V μPa, Brüel & Kjær; for the submerged arena: 8105, manufacturer-calibrated sensitivity −205 dB re: 1 V μPa, Brüel & Kjær). Point measurements were recorded at three depths (in-air tank: 7 cm, 13.5 cm, 20 cm; submerged arena: 5 cm, 15 cm, 25 cm). At each depth, measurements were taken 5 cm apart (in-air tank: 80 measurements; submerged arena: 76 measurements). The

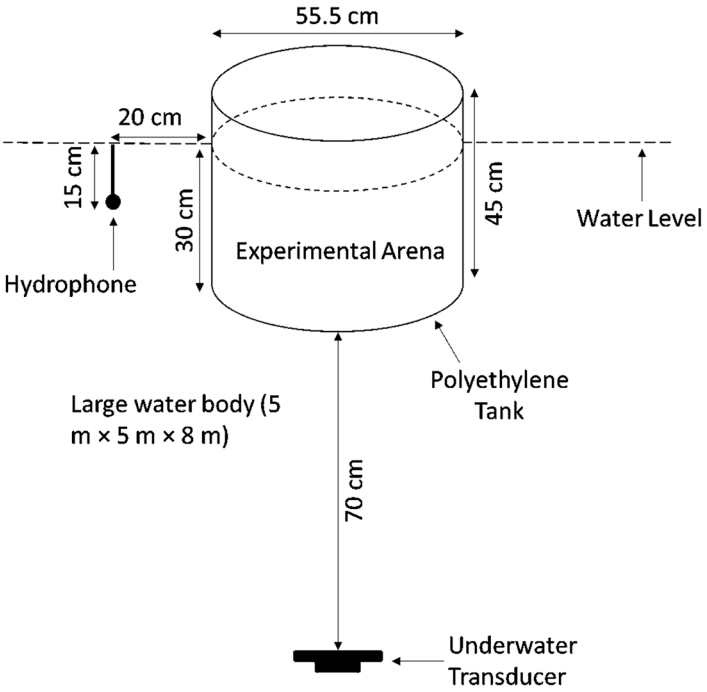



**Fig. 2** The submerged arena (55.5 cm diameter × 45 cm height) inside a large tank of dimensions (8 m length × 8 m width × 5 m depth) in relation to the water level (dotted line, 30 cm water depth). The transducer was suspended 70 cm below the tank, and a hydrophone placed 15 cm below the water level and 20 cm away

data capture and stimulus generation were synchronized to facilitate computation of the PA. Both SPL and PA were quantified to create maps of the sound field. The PA, $a$, may be calculated from the same dataset using a gradient-based approximation:

$$a = -\frac{1}{\rho}\nabla P, \tag{1}$$

where $\rho$ is the ambient density and $\nabla P$ is the pressure gradient (Kinsler et al. 1982).

The pressure gradient was computed using the measurements of the pressure signal. The root mean square (RMS) of the pressure difference was calculated in three directions (x, y, and z), in which the difference between two pressures was computed and then the RMS of that difference was calculated. The pressure gradient was calculated by dividing by the distance between measurements, while the PA RMS (Eq. 1), in all three directions, was calculated by dividing the pressure gradient by the water density. The total PA RMS was determined by combining the obtained values for all three directions and then expressing the results in decibels (dB re 1 mm $s^{-2}$).

## Data Analysis

All statistical analyses were performed in R (version 3.6.3: https://rstudio.com/), while acoustic maps were produced in MATLAB (version R2019b: https://mathworks.com/). Particle motion and sound pressure residuals followed a Gaussian distribution, determined by visual inspection of qq curves. Two-sided *F*-tests were conducted to compare variance of PA and SPL data between the in-air tank and submerged arena setups for each frequency. Similarly, two-sided, Welch's *t*-tests were used to compare the means of PA and SPL between the submerged arena and in-air tank for each frequency.

To determine if a sound field can be described using a plane wave, an expression for the PA under plane wave conditions was considered, (2). The predicted PA could then be compared to the experimental PA to determine which setup better allowed the stimulus to propagate as a plane wave. For a plane wave with no absorption (assuming no absorption in water over the frequency band of interest here is reasonable; Everest and Pohlmann 2015), then $u = p/(\rho c)$, where $u$ is particle velocity and $c$ is speed of sound underwater (Leighton 1994). If the plane wave assumption is satisfied, the gradient in acoustic pressure measured by paired hydrophone measurements in three orthogonal directions can be used to calculate the vector of acoustic particle velocity. For a plane wave, where $f$ is frequency, PA is

$$a = \frac{p2\pi f}{\rho c}. \tag{2}$$

The SPL and PA are expressed in dB using reference values of 1 $\mu Pa$ and 1 mm $s^{-2}$, respectively, via:

$$SPL = 20\log_{10}\frac{p}{10^{-6}}, \tag{3}$$

$$PA = 20\log_{10}\frac{a}{10^{-3}}. \tag{4}$$

The PA computed based on a plane wave assumption was then compared to the experimentally measured PA using the $\chi^2_{red}$ statistic such that (Andrae et al. 2010):

$$\chi^2_{red} = \frac{1}{\mathrm{k}}\sum\frac{\left(PA_e - PA_p\right)^2}{PA_p}, \tag{5}$$

where $PA_e$ was the experimental PA, $PA_p$ was the predicted PA (using a plane wave approximation), and $k$ is the degrees of freedom. The closer the $\chi^2_{red}$ value to one, the more unified the distributions, hence, the greater the agreement with the plane wave assumption (Andrae et al. 2010). Thus, for large values of $\chi^2_{red}$, it can be assumed that the sound did not propagate as a plane wave.

## Results

### In-Air Tank Setup

For the in-air configuration the SPL subjectively appeared heterogeneous across the horizontal plane for each frequency (Fig. 3), with greatest variation observed at the highest frequencies (shortest wavelength), i.e., at 1000 Hz and 2000 Hz (Table 1). The mean SPL differed by ≈6 dB between the top and bottom of the tank. The experimental PA was greater than that predicted based on the plane wave assumption, with 400 Hz exhibiting the highest levels of PA (Fig. 3). Had the sound propagated as a plane wave, the PA was predicted to increase with frequency (Table 2); however, the measured PA did not follow this trend. Hence, the two distributions were dissimilar, which resulted in large values of $\chi^2_{red}$.

### Submerged Arena Setup

In contrast to the in-air configuration the SPL in the submerged area was relatively homogeneous across the horizontal plane for each frequency (Fig. 4), with greatest variation observed at 2000 Hz (as expected, given the shorter wavelengths) (Table 3). The mean SPL differed by ≈11 dB between the top and bottom of the tank. The measured PA was lower in value than the in-air tank (Fig. 4). As with the in-air tank, had the sound propagated as a plane wave, the PA was predicted to increase with frequency (Table 2) and the experimental PA follow this trend (except for the lowest frequency considered, 400 Hz), resulting in smaller values of $\chi^2_{red}$ for 400–1000 Hz compared to the in-air tank (Table 4).

## Comparison of Setups

No difference in the variance of experimental PA was detected between the in-air tank and the submerged arena for frequencies of 400, 600, and 1000 Hz ($F = 1.450 – 0.788$; $p = 0.095 – 0.381$) but a difference in variance was detected for 800 and 2000 Hz ($F = 4.500 – 0.395$; $p = <0.001$). The type of setup had an effect on the variance of SPL for all frequencies and depths ($F = 5.533 – 2268.7$; $p = <0.001$). Results suggested that the SPLs of the in-air setup were more heterogeneous than the submerged arena.

Welch's *t*-tests were carried out, detecting experimental PA to differ between the two setups, with PA being higher in the in-air tank than the submerged arena for 400–1000 Hz ($t = 14.608–40.527$; $p = <0.001$) and lower for 2000 Hz ($t = -5.254$, $p = < 0.001$). Additionally, Welch's *t*-tests detected that the setup influenced the SPL at all frequencies and depths ($t = 6.559–28.299$; $p = <0.001$). The in-air tank setup exhibited higher mean SPLs at all depths compared to the submerged arena setup.

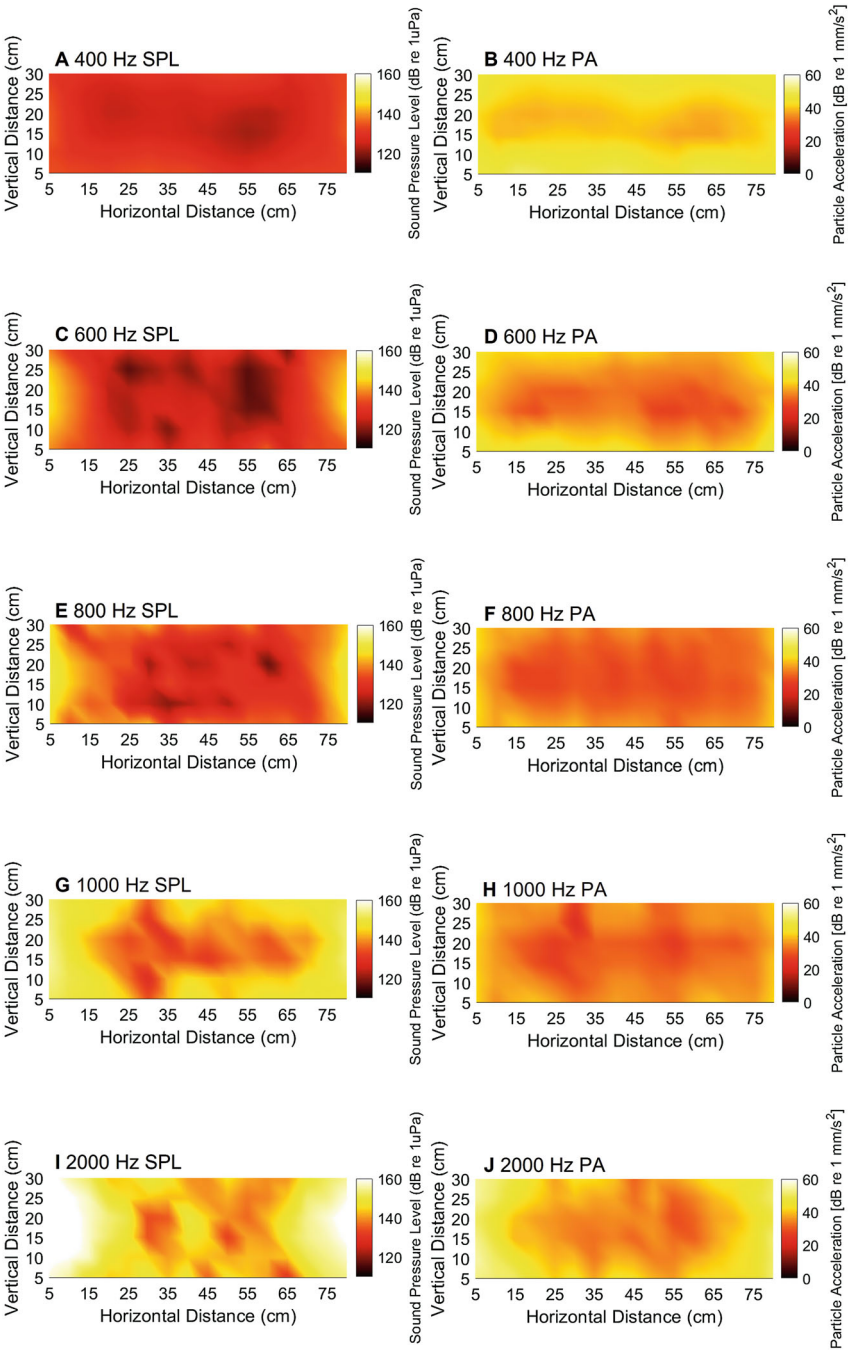



**Fig. 3** SPL maps (**a, c, e, g, i**) and PA maps (**b, d, f, h, j**) (13.5 cm depth) of a 1 s pure tone calibrated at 125 dB re 1 μPa in the center of a rectangular experimental arena (80 cm × 30.8 cm × 30.2 cm) within acrylic in-air tank (300 cm × 30.8 cm × 30.2 cm) at frequencies of 400; 600; 800; 1000; 2000 Hz

**Table 1** The mean SPL ± standard deviation of a 1 s pure tone calibrated at 125 dB re 1 μPa in the center of a rectangular experimental arena (80 × 30.8 × 30.2 cm) within acrylic in-air tank (300 × 30.8 × 30.2 cm) at frequencies of 400; 600; 800; 1000; 2000 Hz. Point measurements were taken at 3 depths (7, 13.5, 20 cm). Center SPL refers to the SPL in the center of the tank

| Frequency (Hz) | 7 cm (dB re 1 μPa) | 13.5 cm (dB re 1 μPa) | 20 cm (dB re 1 μPa) | Center SPL (dB re 1 μPa) |
|---|---|---|---|---|
| 400 | 124.3 ± 4.4 | 128.8 ± 3.4 | 131.4 ± 2.9 | 125.0 ± 0.5 |
| 600 | 128.5 ± 7.5 | 129.3 ± 7.4 | 134.8 ± 5.6 | 125.0 ± 1.4 |
| 800 | 134.6 ± 7.6 | 136.1 ± 9.2 | 138.6 ± 6.8 | 125.1 ± 2.8 |
| 1000 | 136.6 ± 8.2 | 137.7 ± 9.8 | 144.0 ± 6.2 | 125.1 ± 6.7 |
| 2000 | 136.5 ± 8.2 | 137.6 ± 9.8 | 143.9 ± 6.2 | 125.0 ± 6.7 |

**Table 2** The mean PA ± standard deviation of a 1 s pure tone calibrated at 125 dB re 1 μPa in the center of a rectangular experimental arena (80 × 30.8 × 30.2 cm) within acrylic in-air tank (300 × 30.8 × 30.2 cm) at frequencies of 400; 600; 800; 1000; 2000 Hz. Point SPL measurements were taken at 3 depths (7, 13.5, 20 cm) and PA (Experimental) was calculated from those values. PA (Predicted) was calculated using the plane wave approximation and the difference between experimental and predicted PA (Experimental – Predicted) was recorded

| Frequency (Hz) | 13.5 cm (dB re 1 mm/s$^2$) Experimental | 13.5 cm (dB re 1 mm/s$^2$) Predicted | $\chi^2_{red}$ In-air |
|---|---|---|---|
| 400 | 43.3 ± 3.4 | 12.9 ± 3.4 | 79.9 |
| 600 | 37.5 ± 5.1 | 16.8 ± 7.4 | 42.9 |
| 800 | 33.3 ± 4.3 | 19.3 ± 7.4 | 17.0 |
| 1000 | 34.3 ± 4.2 | 29.7 ± 9.8 | 4.72 |
| 2000 | 41.0 ± 6.6 | 35.6 ± 9.8 | 3.16 |

The predicted PA was calculated for both the in-air tank and the submerged arena setup, which determined the PA had the sound propagated as a plane wave. The predicted PA was then compared to the experimental PA using the $\chi^2_{red}$ statistic. For the in-air tank, the difference between the predicted PA and experimental PA was on average 15.0 ± 11.5 dB. The submerged arena had a smaller difference between the experimental and predicted PA at 10.9 ± 8.1 dB on average, depending on the frequency. The $\chi^2_{red}$ was closer to 1 for the submerged arena than the in-air tank for all frequencies except 2000 Hz. This suggests that the plane wave assumption provides a better approximation for the PA in the submerged arena than in the in-air tank.

## Discussion

Small tanks have been a subject of debate among underwater bioacoustics researchers, due to their sound field complexity, starting with the classic work of Parvulescu (1964, 1967; also see Rogers et al. 2016). If a laboratory study is necessary (due to lack of access to the field or to remove a variable that may be present in such conditions), the sound field used in this study is valid. This study consisted of two experimental tank setups and categorized a novel setup in terms

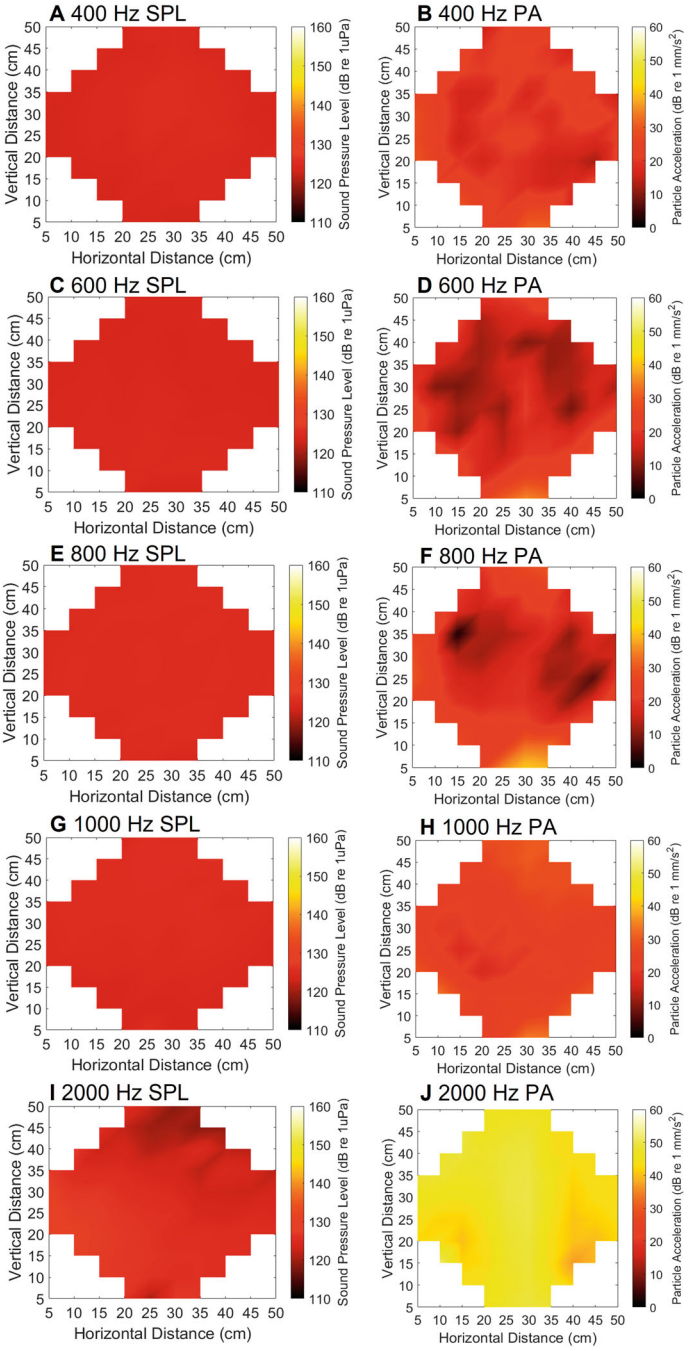



**Fig. 4** SPL maps (**a, c, e, g, i**) and PA maps (**b, d, f, h, j**) (15 cm depth) of a 1 s pure tone in the center of a cylindrical arena (55.5 cm × 45 cm) submerged in a tank (8 m × 8 m × 5 m) at frequencies of 400; 600; 800; 1000; 2000 Hz

**Table 3** The mean SPL ± standard deviation of a 1 s pure tone calibrated at 125 dB re 1 μPa in the center of a cylindrical arena (55.5 cm × 45 cm) submerged in a tank (8 × 8 × 5 m) at frequencies of 400; 600; 800; 1000; 2000 Hz. Point measurements were taken at 3 depths (5, 15, 25 cm). Center SPL refers to the SPL in the center of the tank

| Frequency (Hz) | 5 cm (dB re 1 μPa) | 15 cm (dB re 1 μPa) | 25 cm (dB re 1 μPa) | Center SPL (dB re 1 μPa) |
|---|---|---|---|---|
| 400 | 113.8 ± 0.5 | 124.4 ± 0.4 | 128.5 ± 0.6 | 125.0 ± 0.3 |
| 600 | 113.4 ± 0.4 | 124.3 ± 0.3 | 128.0 ± 0.6 | 124.5 ± 0.1 |
| 800 | 114.7 ± 0.4 | 125.0 ± 0.2 | 128.4 ± 0.3 | 125.0 ± 0.1 |
| 1000 | 114.9 ± 0.6 | 124.9 ± 0.5 | 128.3 ± 0.5 | 125.0 ± 0.3 |
| 2000 | 114.9 ± 1.8 | 124.7 ± 2.4 | 124.0 ± 2.6 | 125.4 ± 0.6 |

**Table 4** The mean PA ± standard deviation of a 1 s pure tone calibrated at 125 dB re 1 μPa in the center of a cylindrical arena (55.5 cm × 45 cm) submerged in a tank (8 × 8 × 5 m) at frequencies of 400; 600; 800; 1000; 2000 Hz. Point SPL measurements were taken at 3 depths (5, 15, 25 cm) and PA (Experimental) was calculated from those values. PA (Predicted) was calculated using the plane wave approximation and the difference between experimental and predicted PA (Experimental – Predicted) was recorded

| Frequency (Hz) | 15 cm (dB re 1 mm /s$^2$) Experimental | 15 cm (dB re 1 mm /s$^2$) Predicted | $\chi^2_{red}$ Submerged |
|---|---|---|---|
| 400 | 21.1 ± 3.8 | 8.5 ± 0.4 | 21.4 |
| 600 | 17.9 ± 5.6 | 11.9 ± 0.3 | 5.81 |
| 800 | 20.3 ± 6.8 | 15.1 ± 0.2 | 5.02 |
| 1000 | 24.8 ± 3.5 | 16.9 ± 0.5 | 4.57 |
| 2000 | 45.3 ± 3.1 | 22.7 ± 2.4 | 24.5 |

of its acoustic validity, as well as experimental control. The SPL heterogeneity was highest in the in-air tank, followed by the submerged arena, indicating that the submerged arena setup is an ideal substitute for creating a homogeneous sound field. The PA was also higher in the two in-air tanks than the submerged arena, suggesting that the submerged arena tank walls were acoustically transparent, hence reducing boundary reflections. When the experimental PA was compared to a PA predicted using a plane wave equation, the submerged arena provided a better approximation. This study provided an alternative experimental setup for performing bioacoustics studies on fish that gave greater control over the stimulus that a fish experiences in lab-based experiments in terms of SPL and PA.

For open-water fish species, experimental conditions should replicate far-field acoustics in which the PA and SPL increase proportionally (Slabbekoorn 2016). However, if in-air tanks were used for open-water species, the PA would be disproportionally larger than SPLs compared to those open waters (Parvulescu 1967; Rogers et al. 2016). Therefore, the small in-air tanks would not be representative of an open-water habitat and may lead to misleading results (Popper and Hawkins 2018). Nonetheless, fish species inhabiting shallow waters would encounter inhomogeneous sound fields due to turbulence (Tonolla et al. 2009), close top and bottom boundaries, and volume reverberation (Hawkins and Myrberg 1983;

Lynch and Newhall 2017). However, we recommend that if non-uniform sound fields were required for a particular study, it would be more valid to modify a homogeneous sound field to the desired inhomogeneity. This approach would ensure the replicability of the sound field.

The results of this study are of particular importance to hearing experiments that were historically undertaken in small, in-air tanks. However, it is also recommended that lower frequencies are also tested, since this study used frequencies above 400 Hz, which is higher than the sensitive frequencies for many fish. While the effect of small in-air tank acoustics on hearing studies has been discussed extensively elsewhere (see Rogers et al. 2016), the results from this study further suggest that some hearing experiments may be invalid due to the setup and the complexity of the sound field. In such conditions, the received level and source level may have been contradictory..

The submerged arena vastly improved the PM and SPL aspects of the sound field, primarily by reducing the average PM and reducing both SPL and PM variance. Submerging the tank in water removed air as a surrounding medium which meant the impedance and reverberation, factors that increase sound field complexity, were minimized (Rogers et al. 2016). When comparing the submerged arena to other setups, the behavioral validity remains low compared to field-based experiments but retains high experimental control, practicality, and feasibility (Slabbekoorn 2016). Compared to field-based methodologies, the acoustic validity is lower given that there was some scattering from the air-water interface, so it would be wise for a future study to compare the sound field as the submerged tank is placed further from the surface boundary (although this will make sound field mapping more difficult). An additional consideration is the effect of the swim bladder and the fish itself on the sound field (see Chapman and Hawkins 1973) and hence requires further investigation. However, there is a vast improvement to sound field homogeneity compared to small tank studies since a plane wave approximation was feasible. To be specific, the submerged arena used here, with its acoustically transparent walls, allowed the source to be placed outside of, and so at a greater range from, the working arena, which tended to decrease the radius of curvature of the wavefronts in the arena and so make the use of Eq. (2) (with its inherent plane wave assumption). This is an important consideration, especially if there is no independent way to validate the estimated PA, as is often the case in such studies. It is recommended that when cost, practicality, and experimental control limit the use of in situ studies, a submerged tank would be a suitable alternative.

## Conclusion

This study investigated two experimental setups: a ‘typical’ setup of a rectangular acrylic arena surrounded by air and placed on concrete blocks with two transducers situated in the arena and a setup of a cylindrical arena submerged in a large water tank with a speaker situated underneath. A submerged arena setup was chosen to

reduce sound field inhomogeneity so that the level received by the fish is constant and known to the experimenter, and the study is replicable. The in-air tank exhibited an inhomogeneous field with high levels of PA, while the submerged arena possessed a homogeneous sound field with PA measurements satisfying the plane wave approximation to a larger degree. While the behavioral validity of the setup does not compare with in situ studies, submerging an experimental arena greatly improves the sound field homogeneity essential for acoustic and behavioral replicability in bioacoustics studies with fish.

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# Measuring Directional Underwater Sound with Particle Motion Eccentricity

6

James A. Campbell and Sophie L. Nedelec

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**Abstract**

As acoustic particle motion has been shown to be an important sensory cue for many underwater animals, particle motion fields are frequently measured and reported on for studies on these taxa. However, few studies describe the biologically relevant directional information encoded within these fields. This chapter introduces measures of eccentricity in the context of particle motion, indicating how strong the directional cues in acoustic particle motion fields are. The measures of spectral and broadband eccentricity described here can be utilized for validating acoustic fields in experimental setups, in addition to providing insight into the ability of fishes and invertebrates to determine the directions of sound sources. With the development of measures – such as eccentricity – providing a quantitative language to report on particle motion directionality, researchers are better equipped to interpret the biological relevance of the particle motion fields animals are exposed to.

**Keywords**

Bioacoustics · Underwater sound · Particle motion · Directionality · Sound source localization · Eccentricity

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## Introduction

Over the past decade, commercial vector sensors for measuring underwater particle motion have become widely available to researchers. As all fishes and many sound-sensing underwater invertebrates are sensitive to particle motion (Popper et al. 2019; Dinh and Radford 2021; Day et al. 2019), measuring and reporting this sound field component is quickly becoming standard practice within studies on these taxa (Nedelec et al. 2021). However, while vector sensor measures of particle motion inherently record biologically relevant directional information, few studies report this.

The directional information coded within particle motion allows animals to estimate the direction of sound sources, thus aiding localization (Zeddies et al. 2012; Popper et al. 1973). This is critical for behaviors related to feeding, soundscape orientation, predator avoidance, and reproduction (Nedelec et al. 2016). While previous studies have examined directional responses to sounds (Holles et al. 2013; Rollo and Higgs 2008; Mirjany et al. 2011; Schuijf and Buwalda 1975; Nelson 1967), there has been a lack of methodology for measuring the strength of directional cues coded within these sounds.

Acoustic signals can be conceptualized as sinusoidal oscillations of particle motion or sound pressure. Where pure tones are composed of a single oscillation frequency – a sine wave – broadband signals can be thought of as a superposition of these sine waves across a frequency spectrum. When measuring particle motion, a pure tone summed across the axes of a vector sensor results in a particle motion ellipse with a shape that is defined by the phase difference between the sensor axes measurements. Broadband signals summed across multiple sensor axes can also be conceptualized as superpositions of these particle motion ellipses (illustrated in Fig. 1). For a theoretical, ideal plane wave, the particle motion summed across vector sensor axes is expected to result in a straight line – where all sensor axes measure the same phase. In practice, however, there will always be small phase differences between the axes measurements, resulting from interference from reflections, environmental background noise, or limitations of the vector sensor itself, producing measures of elliptical particle motion.

Eccentricity is a standard geometric measure which describes how circular or linear an ellipse is. It's defined as

$$e = \sqrt{1 - \frac{b^2}{a^2}},$$

where $a$ and $b$ are the lengths of the semi-major and semi-minor axes of an ellipse. A value of 1 indicates an ellipse resembling a line, while a value of 0 indicates a perfectly circular ellipse. In the context of acoustic particle motion, eccentricity can be applied to particle motion ellipses to measure how directional an acoustic signal is. Signals containing strong directional cues are expected to give high eccentricity values, while non-directional, ambient sounds are expected to result in low eccentricity values (see Fig. 2). The following sections detail how to apply eccentricity to particle motion measurements in both spectral and broadband contexts. The applications of eccentricity introduced in this chapter provide a quantitative measure of how reliably the direction of a sound source can be estimated.

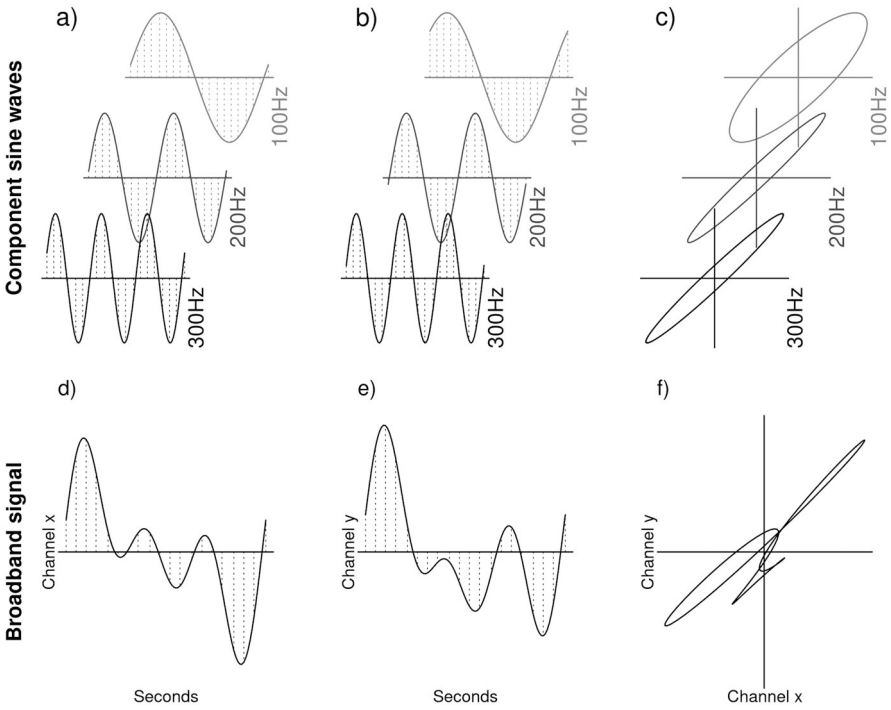



**Fig. 1** (**a**) and (**b**) show a series of sine waves which combine to make up the broadband signal measured on the horizontal channels of a vector sensor, (**d**) and (**e**), respectively. (**c**) shows the particle motion ellipses resulting from vector summing the sine waves in (**a**) and (**b**); (**f**) shows the broadband particle motion along the horizontal plane, composed of the superpositioned particle motion ellipses in (**c**). Broadband signals measured by vector sensors can be conceptualized as a spectrum of superpositioned particle motion ellipses

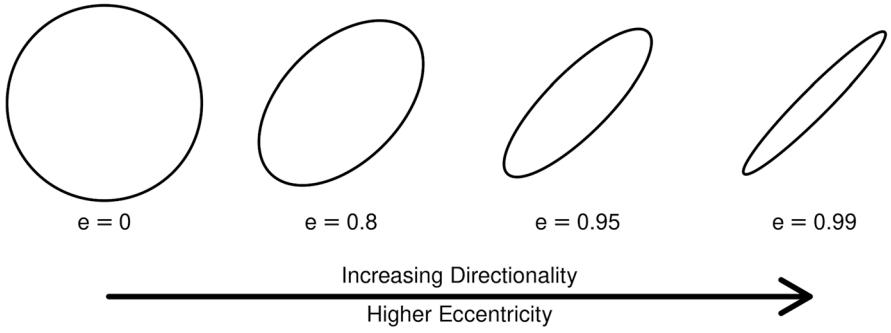



**Fig. 2** An illustration of ellipses covering a range of eccentricity values. Higher eccentricity results in more linear particle motions paths. Values of 0 result in non-directional, or perfectly circular, particle motion paths

## Spectral Eccentricity

To calculate the particle motion eccentricity for a particular frequency of a broadband signal – here termed spectral eccentricity – a discrete Fourier transform can be taken of the x and y channels, $X_f$ and $Y_f$, of the recorded sound. For a given frequency, $f$, the magnitude and phase can be used to construct a particle motion ellipse

$$\vec{p}(t) = \vec{c}_1 \cdot \cos(t) + \vec{c}_2 \cdot \sin(t),$$

where

$\vec{c}_1 = \vec{x}_0 + \vec{y}_0 \cdot \sin(\delta)$,
$\vec{c}_2 = \vec{y}_0 \cdot \cos(\delta)$,
$\delta = \arg(Y_f) - \arg(X_f) + \frac{\pi}{2}$,
$\vec{x}_0 = (|X_f|, 0)$, and
$\vec{y}_0 = (0, |Y_f|)$.

The above ellipse formula is parameterized by the vectors $\vec{c}_1$ and $\vec{c}_2$ which represent conjugate half-diameters. We can reformulate this in terms of the ellipse's semi-axes, $\vec{d}_1$ and $\vec{d}_2$, with

$$\vec{p}(t) = \vec{d}_1 \cdot \cos(t) + \vec{d}_2 \cdot \sin(t),$$

where

$\vec{d}_1 = p(t_0)$,
$\vec{d}_2 = p(t_0 + \frac{\pi}{2})$, and
$\tan(2 \cdot t_0) = \left(2 \cdot \vec{c}_1 \cdot \vec{c}_2\right) / \left(\vec{c}_1^{\,2} - \vec{c}_2^{\,2}\right)$.

Finally, the lengths of the semi-major and semi-minor axes can be measured:

$$a = \max\left(\left|\vec{d}_1\right|, \left|\vec{d}_2\right|\right) \text{ and}$$

$$b = \min\left(\left|\vec{d}_1\right|, \left|\vec{d}_2\right|\right)$$

Eccentricity can then be calculated from $a$ and $b$. Figure 3 shows an example application of this.

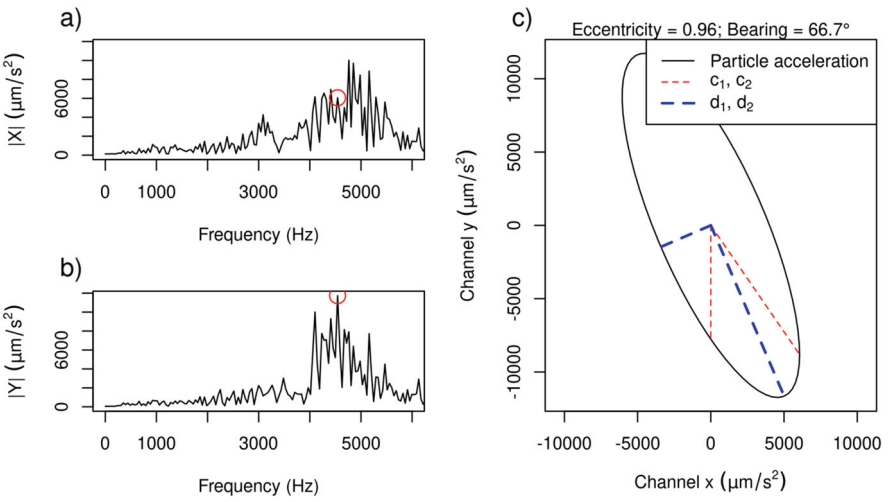



**Fig. 3** An example calculation of spectral eccentricity at 4542 Hz in the horizontal plane of a vector sensor. The signal is a particle acceleration recording of boat noise, bandpass filtered between 100 and 5000 Hz. Panels (**a**) and (**b**) show the complex modulus of the one-sided discrete Fourier transforms of the channels in the horizontal plane, x and y. Panel (**c**) shows the particle motion ellipse constructed from the complex modulus and arguments at 4542 Hz. $\vec{c}_1$ and $\vec{c}_2$ are conjugate half-diameters of the ellipse, while $\vec{d}_1$ and $\vec{d}_2$ show the semi-axes from which eccentricity can be calculated. The bearing reports the angle of the major semi-axis, $\vec{d}_1$, clockwise from the +x axis

## Broadband Eccentricity

Broadband sounds, of course, do not follow elliptical particle motion paths. In this case, eccentricity can be calculated from the confidence ellipse surrounding the particle motion measurements in the horizontal (or vertical) plane. Singular valued decomposition can be applied to the covariance matrix of the measurement axes to retrieve the axes of the confidence ellipse. Where $\vec{x}$ and $\vec{y}$ are the measured signals on the horizontal plane axes of the vector sensor, eccentricity is given by

$$e = \sqrt{1 - \left( \frac{\min\left(\text{diag}^{-1}(D)\right)}{\max\left(\text{diag}^{-1}(D)\right)} \right)},$$

where

$UDV' = \Sigma$, and
$\Sigma = \text{cov}\left(\vec{x}, \vec{y}\right)$.

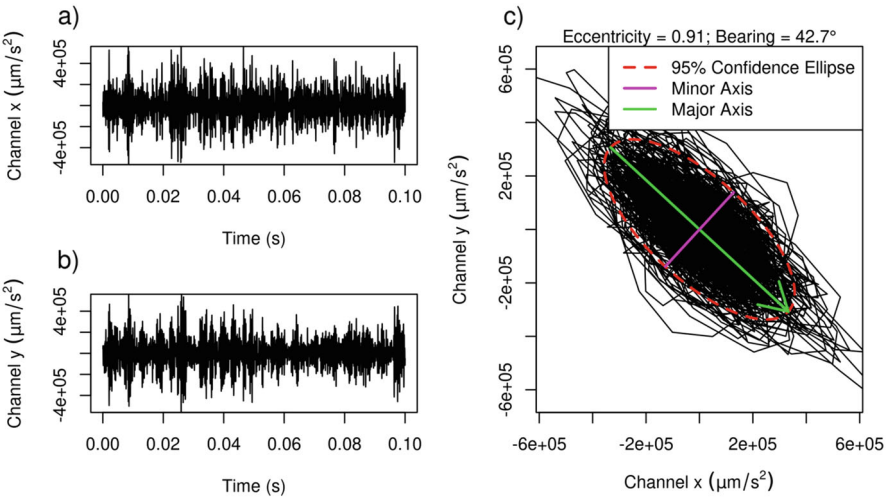



**Fig. 4** An example calculation of broadband eccentricity in the horizontal plane of a vector sensor. The example recording is the same used in Fig. 3. The red dashed line is the confidence ellipse of particle acceleration measures from which eccentricity can be calculated from the major and minor axes

*U*, *V*, and *D* represent the left singular vectors, right singular vectors, and singular values, resulting from singular valued decomposition of the covariance matrix, Σ. The maximum and minimum values of the diagonal entries of *D* give the squared major and minor axes, $a^2$ and $b^2$, respectively. Figure 4 illustrates eccentricity calculated from broadband signals.

## Discussion

As the use of vector sensors in underwater sound studies becomes commonplace, a new suite of bioacoustic measurements is called for that considers the directional nature of acoustic particle motion. To investigate ecological processes that make use of sound source localization, measurements which first describe the directional information coded within acoustic particle motion are required. The broadband and spectral eccentricity measures introduced here provide an intuitive measure of the strength of these directional cues.

The interpretation of particle motion eccentricity in the context of sound source localization is simple. Sounds with strong directional cues – loud sounds, relative to ambient, from a single source which travel along direct paths – result in more linear particle motion and high eccentricity. Sounds with weak or convoluted directional cues – resulting from interference from multipath reflections, ambient sources, or low signal to noise ratios – will give lower eccentricity values and are expected to be difficult to localize.

While eccentricity can be calculated from any particle motion component: displacement, velocity, or acceleration; particle acceleration is a natural choice for

calculating eccentricity as it is in accordance with best practice guidelines on particle motion measurements (Nedelec et al. 2021) and fishes are expected to directly sense particle acceleration (Fay and Edds-Walton 1997). Finally, eccentricity can easily be calculated for particle motion in 3D space.

A natural application of particle motion eccentricity is the validation for experimental sound fields. Acoustic conditions in experimental setups, particularly in small aquaria, have long been a focus of criticism (Parvulescu 1967; Rogers et al. 2016). Reporting values of eccentricity for directional sound stimuli, along with the particle acceleration and pressure fields in aquaria, provides a comprehensive description of acoustic conditions, facilitating comparisons to natural environments.

Additionally, little is understood about the challenges of localizing sounds in natural environments. Often, fish and invertebrates live in acoustically complex environments, surrounded by hard, reflective substrate. Hence, ecologically important sounds can be expected to contain confounding directional information. Examining the quality of naturally occurring or anthropogenic directional acoustic cues may provide insights

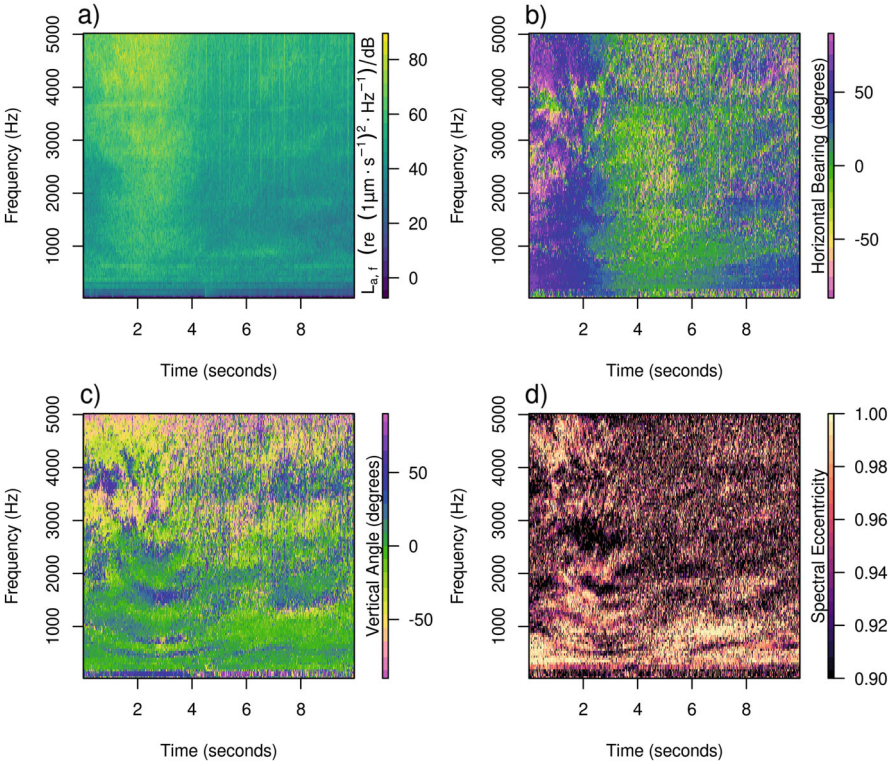



**Fig. 5** Directional bearings and spectral eccentricity calculated from three axes of a vector sensor. The example recording is the same used in Figs. 3 and 4. (**a**) shows a spectrogram of particle acceleration. (**b**) and (**c**) show the horizontal and vertical particle acceleration bearings – the major axis of particle acceleration – of the passing boat. (**d**) shows the particle acceleration spectral eccentricity measured in three dimensions. High vertical angles (40–50°) are associated with low eccentricity, suggesting that these values contain weak directional cues

into the behaviors and life history of animals residing in these complex environments. Highlighting the complexity of directional sounds, Fig. 5 shows a directional spectrogram (also termed "directogram") of a boat passing a coral reef, recorded by a vector sensor. The vertical angles reported are frequency dependent, varying from 0° to 50° as the boat passes. When taking into consideration spectral eccentricity, many of the high vertical angles are correlated with low spectral eccentricity – suggesting confounding directional cues at these high vertical angles. Here, a single anthropogenic sound source results in frequency-dependent localization bearings and eccentricity values, providing a convoluted sound field to localize.

In conclusion, while the reporting of directional sound in underwater bioacoustic literature has so far been sparse, these eccentricity measurements provide a quantitative language for discussing the strength of directional acoustic cues. It's hoped that this introduction of particle motion eccentricity will stimulate the development of further descriptive acoustic metrics.

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# Shaking Up Aquatic Substrates: Taking Lessons from Biotremology and Defining Terminology

7

Louise Roberts and Andreas Wessel

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**Abstract**

Sound pressure and particle motion form the basis of discussions regarding potential effects of anthropogenic signals on fishes and aquatic invertebrates. However, there is a growing need to acknowledge, and understand, a third class of stimuli involving substrates. The vibrational channel is a major sensory mode in the animal kingdom. Numerous terrestrial animals, from the fruit flies to the largest of land mammals, use substrate-borne vibrations to find prey, detect

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predators, sense environmental change, and to communicate. Yet although many aquatic organisms reside on, in, and around substrates, how animals use vibrational information in this environment has, to date, been largely overlooked. This is all the more important given that many anthropogenic activities directly contact the seabed, eliciting significant vibrations in the substratum. To better understand the role of vibrational signals to aquatic animals, lessons can be taken from the terrestrial environment, from the discipline of biotremology. This chapter provides a refined definition of biotremology to allow inclusion of aquatic systems and discusses the alignment of terminology required to apply vibrational principles to the aquatic world. Future discussions in this area are necessary to aid the progression of both underwater noise and aquatic biotremology research.

**Keywords**

Particle motion · Substrate-borne vibration · Bioacoustics · Vibrational noise · Anthropogenic noise

## Introducing Biotremology

### Vibrational Waves

This chapter discusses substrate-borne mechanical waves. These occur in the boundary between two media where particle motion is perpendicular to the direction of wave travel. These waves, detected by a nonpressure detecting sensor or structure, are utilized by a range of animals for vibrational sensing and communication, and are studied in a rather new discipline of study known as biotremology (Hill and Wessel 2016; Hill et al. 2019a, b) (Fig. 1). The strict definition of biotremology will be discussed in later sections. Bioacoustics as a discipline is here distinct, defined by the

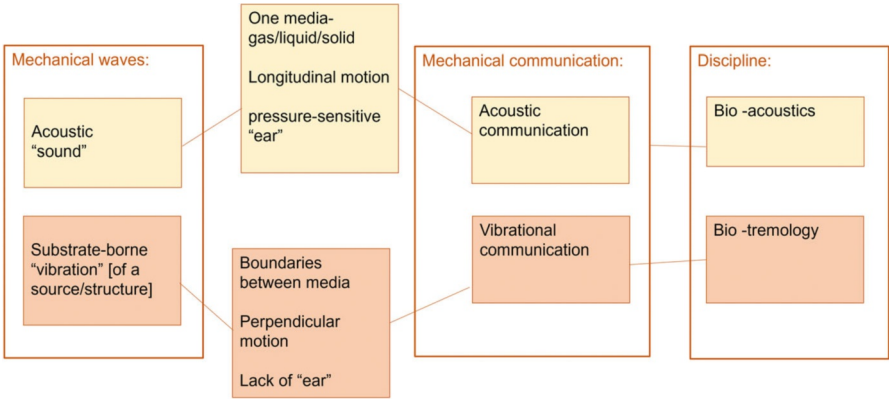



**Fig. 1** Distinctions between bio-oscillology disciplines: bioacoustics and biotremology (a table adapted from Hill and Wessel 2016)

study of acoustic (compressional) waves ("sound"), traveling in one medium, detectable by a pressure sensitive ear, or pressure difference, sensitive organ, the ear (Ewing 1989).

There are several different vibrational waves, characterized by wave speed and particle motion direction (Markl 1983; Aicher and Tautz 1990). These include compressional (sound) P-waves, transverse S-waves, and surface waves at boundaries. Studies have found, however, that terrestrial animals using vibrations are sensitive to surface waves, either Rayleigh (Sholte in aquatic) or Bending or Love waves (this last being only described as used for one situation, Aicher and Tautz 1990). These waves are termed "substrate-borne vibration." Vibrational waves propagate through a wide range of substrates), with the variability of substrates (e.g., soil, sand, clay, leaves, etc.) affecting vibrational transmission (e.g., Athanasopoulos and Pelekis 2000; Magal et al. 2000; O'Connell-Rodwell et al. 2001).

## Measurement of the Vibrational World

Vibrations, like sounds, are all around us. Substrate-borne vibrations form a vibrational landscape recently called the vibroscape (Roberts and Elliott 2017; Šturm et al. 2021). The vibroscape includes geophysical, anthropogenic, and biological vibrational components, in addition to indirect vibrations caused by airborne sounds. By pointing a laser at the stem of a red clover plant (*Trifolium pratense*), for example, researchers have eavesdropped on a world of vibrational signals, which vary with season, plant species, and location (Šturm et al. 2021). The acoustic equivalent of this is termed the soundscape (a collection of sounds, Pijanowski et al. 2011).

Measuring vibrations (and sound) depends largely upon the study animal or surface in question (Nieri et al. 2022). For example, the drumming of a ghost crab (Ocypode *spp,*) can be measured both as an airborne or ground-borne sound (microphone, Fig. 2 upper right), or as a vibration on or within a substrate (geophone, accelerometer, piezodisc – Fig. 2, left). Vibrations may also be measured via a noncontact laser doppler vibrometer, pointed at the crab body (Fig. 2 upper right).

## Terrestrial Animals Using Vibrations

Which animals are using substrate-borne vibrations? The most recent biotremology book documents animals from bees, planthoppers, beetles, ants, caterpillars, to frogs, mole rats, birds, and snakes (Hill et al. 2022), although the list of animals extends way beyond these (Hill 2008).

For many animals, vibration plays a significant role in communication (e.g., courtship, group co-ordination), as well as sensing the environment (e.g., source localization, predator–prey interactions, abiotic source detection) (Hill 2001, 2008, 2009). The chewing vibrations of caterpillars such as the beet armyworm (*Spodoptera exigua*), for example, can be recorded on leaves (Velilla et al. 2020). Other caterpillars such as the masked birch caterpillar (*Drepana arcuata*) produces

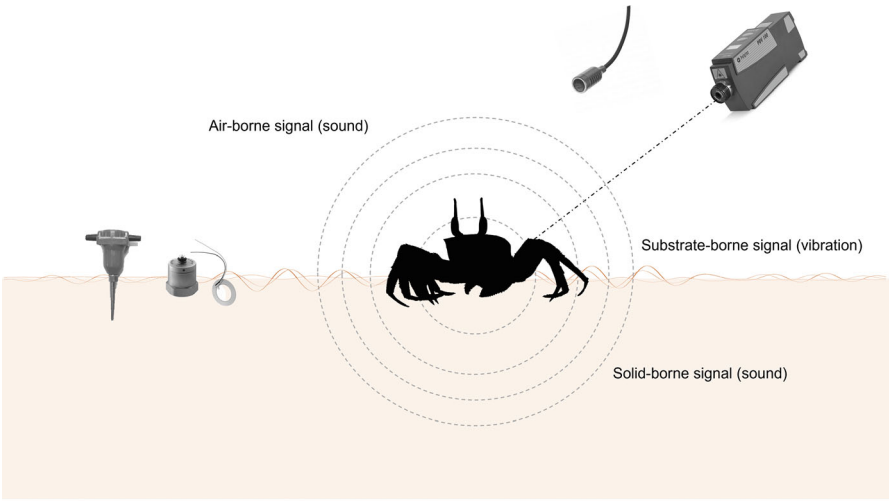



**Fig. 2** A ghost crab (Ocypode spp.) drumming on the sand producing signals that may be measured above, below, and at the surface of the sand as sounds and vibration

vibrational signals to defend its silk shelter on the leaf surface (Yadav et al. 2017). Underground, rhino beetle larvae (*Trypoxylus dichotoma*) actively beat against their pupal case when approached (Kojima et al. 2012), and white grubs (Scarabaeidae larvae) stridulate (Görres and Chesmore 2019). Other vibrational behavior is more visible and functions as a prey lure, such as the twitching of cane toad toes (*Chaunus marinus*) and the paddling of bird feet (Tinbergen 1962; Hagman and Shine 2008). More examples are explored in Roberts and Wickings (2022).

## Aquatic Vibrations

### Vibrational Waves in Aquatic Substrates

Much less is known about vibrations in aquatic environments (Roberts and Elliott 2017). Yet there are a range of substrates present, from biological surfaces (e.g., macroalgal fronds, sessile animals), abiotic surfaces (soft, hard sediments, the water surface), and even anthropogenic surfaces (supporting structures of offshore platforms, shipwrecks), which may transmit vibrations. As in terrestrial environments, vibrations may be produced directly, or indirectly (e.g., via water-borne sound), traveling long distances through denser surfaces making the seabed an additional communication medium. Animals may be in contact with vibrations when hidden inside substrates, traversing substrate surfaces, or both. The time in contact with substrates will vary according to life stage, time, and activity. While not traditionally charismatic, many of these species may be commercially relevant, and crucial to the ecosystem.

## Aquatic Animals Using Vibrations

The collection of evidence to support aquatic biotremology within the broader discipline of biotremology includes definite examples, plus some suggestions (reviewed in Roberts and Wickings 2022). Collated vibrational sensitivities of aquatic organisms to vibrations (unspecified mechanical wave types) show that a range of aquatic groups can detect vibrations (Frings and Frings 1967; Budelmann 1992; Roberts and Elliott 2017), though it is currently unclear *which* substrate waves are relevant to aquatic animals. It is hypothesized that Sholte (Rayleigh) waves are likely involved due to their propagating further than other substrate-borne waves and because they are also used extensively by terrestrial animals.

On the seashore, semiterrestrial crabs, the fiddler and ghost crabs (family Ocypodidae), are well-documented vibrational "drummers" and "rappers" producing substrate-borne signals (reviewed in Popper et al. 2001). Other crustacean producers include (terrestrial) hermit crabs residing on the seashore (*Coenobita* spp.) which produce "chirps" measurable as sand vibrations (Roberts 2021). Underwater, mantis shrimps (*Hemisquilla californiensis*) produce a low frequency "rumble" from within and around burrows in sediment (Staaterman et al. 2011), though so far measured only in the water but a strong contender for a vibrational signal.

Near the seabed, haddock (*Melanogrammus aeglefinus*) produce "knocks" and "hums" and other water-borne mechanical vibrations known as "sounds," while either on or close-by to the bottom (Casaretto et al. 2014; Hawkins 2022). Other fishes, such as the three-spined toadfish (*Batrachomoeus trispinosus*) "hoot" and "grunt," or "croak" and "drum" (e.g., the Sciaenidae), while directly contacting the bottom (Ramcharitar et al. 2006; Rice and Bass 2009).

Mollusk and urchins feeding directly on algae covered substrates use a specialized structure, the radula, to "scrape" and "rasp" (for example, in limpets, Barber et al. 2015). Vibrations of limpets scraping upon rocks are therefore measurable on the shell (Parpagnoli and Chelazzi 1995) and are likely measurable upon the substrate. Sounds of urchin feeding translate indirectly into the water column (Radford et al. 2008) yet the substrate vibrations of this feeding have not been measured directly.

## Humans as Vibration Producers

Another key contributor of aquatic vibrations are humans (Roberts and Elliott 2017; Hawkins et al. 2021). Just as any animal touching a substrate can create vibration (Aicher and Tautz 1990), so too can any human or human technology. In much the same way as sediment type and particle size is related to benthic communities (Sivapriya et al. 2022), sediment and substrate composition relates to vibrational wave propagation (Elias et al. 2010). There is evidence that vibrational noise has similar effects as acoustic noise, including physiological and behavioral changes, and in some cases damage (Roberts and Howard 2022). However, the range of species tested is limited so far in both terrestrial and aquatic environments (Roberts

and Howard 2022). The lack of research attention may be due to a human bias toward air or water-borne sounds, which they more easily detect.

Transport, energy development, harvesting, and construction, directly and indirectly, produce vibrations in the seabed and other substrates (Roberts and Elliott 2017; Hawkins et al. 2021). Crucially, these vibrations radiate outward from a source and travel kilometers trapped in the surface (Hazelwood and Macey 2016). Further details of these waves are provided in Hawkins et al. (2021). Example sources include pile driving, dredging, trawling, explosions, and seismic surveys. These sources produce low frequency Sholte waves known as seabed vibrations or ground roll, which can also cause evanescent pressure waves in the water above (Hawkins et al. 2021).

Estimates of the impact ranges of various vibrational sources related to flounder (*Paralichthys dentatus, Pseudopleuronectes americanus*), American lobster (*Homarus americanus*), hermit crab (*Pagurus bernhardus*), and mussel (*Mytilus edulis*) sensitivities have been undertaken (Roberts et al. 2015, 2016; Miller et al. 2016). As an example, blasting and dredging are detectable by hermit crabs up to c.a. 300 m and c.a. 200 m from source, respectively (Roberts et al. 2016). Note however, in a similar way to water-borne sound, impact ranges of vibrational sources are highly contextual and source-receiver dependent.

Perhaps this may seem too speculative, but it is accepted now that many fishes detect particle motion alone, despite only a few species being directly tested (reviewed most recently in Popper et al. 2019; Popper and Hawkins 2018; see also Chapman and Sand 1974; Hawkins and Johnstone 1978). In contrast, vibrational sensitivity thresholds have been experimentally defined for around 10 invertebrate species (Roberts and Elliott 2017) and including tests on some fishes (Berghahn et al. 1995). However, as with fish hearing studies, there is currently no standardized method for establishing vibrational sensitivity.

## Alignment with Biotremology Terms

### Aquatic Biotremology

The majority of the research within the discipline of biotremology so far has been based upon terrestrial animals residing on substrate-air boundaries. However, the same principles are applicable to water-substrate and water-air surfaces. Given the infancy of the aquatic research area, it seems helpful to begin a discussion of terminology and suggest an alignment of terms between terrestrial and aquatic environments before terms diverge. The discussion here should be viewed as a starting point for future discussions between both disciplines. A conversation analysis will be necessary to aid the progression of underwater noise and aquatic biotremology research.

**Textbox 1. The Discipline of Biotremology, as Defined by Hill and Wessel (2016)**
"Biotremology is the study of mechanical communication by surface-borne waves.

Surface-borne waves: waves (Rayleigh or bending [or Love]) at a boundary between two distinct media-particles oscillate in all three dimensions with the major amplitude perpendicular to the direction of the wave and the plane of the surface."

First, consider the original definition of biotremology (Textbox 1). Does this current definition incorporate aquatic animals and surfaces? How does this definition deal with the water or sediment surface? The answer depends on definitions of "surface," "surface-borne waves," and "boundary." For clarity, these are now defined more clearly in our proposed redefinition of biotremology, given in Textbox 2.

**Textbox 2. A Revision of the Current Definition of Biotremology, Based Upon Textbox 1**
"Biotremology is the study of organismal vibrational behavior and biocommunication in regard to mechanical *substrate-borne waves* at media boundaries (boundary and surface waves).

Substrate-borne waves: (Rayleigh, Sholte and Love waves) propagate along the *boundary between two media* with different properties in terms of density, stiffness and viscosity; particles of the more dense/stiff/viscous medium, i.e. the substrate, oscillate in all three dimensions with the major amplitude perpendicular to the direction of the wave.

The media boundary is a more or less steep gradient in the media properties (and is regarded as boundary if the gradient steepness in relation to wavelength allows for the development of surfaces waves) and has therefore a *depth."*

The main change here is a shift from "surface" to the broader term "substrate"-borne waves. Note that this change is in line with a previous suggestion (Hill et al. 2019a, b). Surface does not account for the "depth" of a substrate, and there are difficulties when considering the water column. Instead "substrate" is defined in Textbox 2 as the more dense/stiff/viscous medium of the substrate, within which particles undergo the elliptical oscillations of interface waves. This means, according to the definition, that water could be classed as a substrate at the boundary with air, and not as the substrate at the boundary with sediment.

Previously, substrate was defined as "the surface of solid matter an animal is living on, be it a wooden or soft part of a plant, terrestrial or underwater ground, or the surface of water to air" (Hill and Wessel 2016) in line with the ecological

definition of substrate. For example, aquatic substrates could encompass firm sediments, solid substrata, human artificial structures, algal fronds, and perhaps even sessile animals themselves, that is, those maintaining a structural integrity underwater, being denser than water. The gray area there, however, was "solid," which does not account for fine particulate soft-bottom sediments, for example.

In Textbox 2, the boundary of the media is defined, allowing inclusion of (a), but exclusion of (b):

(a) **Near-field "sound,"** i.e., in air or in sediment and water as "evanescent sound waves" (Hawkins et al. 2021); oscillation without the establishment of a P-wave, thus, no energy propagation (energy loss) occurs. The transformation of waveforms is accompanied by loss of mechanical energy, i.e., heat production, at the media boundary.
(b) **Substrate(-borne) sound**, i.e., the solid-borne compressional fraction of the signal, traveling as pure P-waves in homogeneous extended substrates; termed "solid compression waves" (Hawkins et al. 2021).

Use of "substrate" is also preferable in aquatic systems, rather than "sediment," "seabed," or "seafloor," since it is a broader term encompassing more breadth of physical location and habitat. The term "seismic" is avoided since aquatic substrates transmitting vibrations may include more than the ground alone. Within marine ecology, there are also terminology differences for this water-sediment boundary (e.g., deep sea benthic ecology versus benthic ecology), which the term "substrate" bypasses.

Given the novelty of this subject in aquatic systems, appending "aquatic" to biotremology is suggested for clarity, to emphasize the distinction from the terrestrial literature base (and therefore other interfaces) for the near future.

## Water-Borne Particle Motion

In aquatic bioacoustics the terms "water-borne particle motion" and the broad term "particle motion" are used extensively (Nedelec et al. 2016; Popper and Hawkins 2018). For the sake of this section these are termed "particle motion" (here discussing particle motion rather than flow). In the far field, particle motion and pressure are related (reviewed in Nedelec et al. 2016) with reflective boundaries changing this relationship. Currently, aquatic particle motion detectors fall within aquatic *bioacoustics* and *hearing* studies, although they are often described as being *vibrational* detectors. Is there a case for these animals being part of aquatic biotremology? From a biotremologists perspective, the question would be: What triggers depolarization of the detectors? If pressure, then study of these belongs to bioacoustics. If particle motion, then study of these *could* belong to biotremology.

Varying definitions of "hearing," "vibration," "particle motion," "boundary," and "substrate" between disciplines, however, make this discussion difficult. Substrate-borne vibration, for example, is used solely for boundary waves in biotremology (see

above), whereas in aquatic bioacoustics the term includes solid compressional waves. Often "vibration" is used synonymously with sound (especially in earlier literature, but even today), regardless of medium, while biotremologists are more specific. Along similar lines, particle motion could be "sound" or "vibration" depending on the discipline.

What biotremologists call sound and vibration is linked to historical usage, more than a strict definition based in physics. Sound and substrate-borne vibration are propagated as mechanical waves that can be described as both vibrational and acoustic waves in nature. The separation is based in the well-established use of "sound" for all mechanical waves that propagate as P-waves and are detected by animals with some sort of ear, using pressure or pressure-difference receivers. Biotremologists, who are acknowledged as mostly studying terrestrial animals (section "Terrestrial Animals Using Vibrations"), are attempting to develop a cohesive vocabulary at the same time as the discipline is expanding to include gray areas such as aquatic zones. Additionally, the vocabulary development is occurring at a time when publications within biotremology are increasing exponentially in number (Hill et al. 2022). Is there an argument that a watery medium can be a substrate? This is a new challenge still being explored, but it looks promising.

From a physical viewpoint, as described above, substrate-borne motion directly *at* media boundaries is more complex and mechanically different than particle motion away from boundaries. Substrate-borne motion cannot be calculated from pressure measurements. Is it then distinct from particle motion in a single medium?

Particle motion encountering a boundary may excite substrate-borne waves, and, in fact, across the air-solid boundary, waveforms in one excite waveforms of the other across the boundary. Thus, one event can initiate both sound and vibrational waves, detected as sound or substrate-borne vibration by animals at the boundary.

A benthic animal very likely detects both the water-borne particle motion and the substrate-borne waves using the same sensory structures, even though research focused on this distinction is still expanding. It is accepted that the majority of fishes and invertebrates detect particle motion rather than pressure, lacking a pressure-related organ/"ear" and also air-filled cavities (Budelmann 1992; Popper and Hawkins 2018). These receivers fall firmly in the *biological* definition of biotremology, due to using a nonpressure sensor to detect particle motion. As an example, flatfish such as the European plaice (*Pleuronectes platessa*) are sensitive to particle motion (Chapman and Sand 1974; Sigray et al. 2022) and are also responsive to substrate vibration from shakers (Berghahn et al. 1995). Use of the term particle motion to describe plaice hearing, then, would be an incomplete description of sensory capabilities, while also omitting substrate motion. Note, "hearing" in *biotremology* only includes animals detecting pressure from sound waves (but definitions of hearing vary between the disciplines – another area ripe for discussion). P-waves, as all mechanical waves currently known to us, also propagate via particle motion, but the detection is performed (exclusively) by pressure or pressure-difference receiver organs (tympana, swim bladders or other gas-filled organs in fishes: termed "acoustic transformers"; Popper and Hawkins 2018), which convert it to the particle motion in the inner ear.

## Anthropogenic Stimuli

A number of synonymous terms are used currently to describe vibrational stimuli of anthropogenic origin within an aquatic substrate (Table 1). At present, these are not typically aligned with particular substrate waves (e.g., compressional, shear, or Sholte/Rayleigh).

A workshop to discuss the use of these terms would be extremely valuable to the field. However, in the meantime the following is suggested. Although terms such as "ground, seabed, sediment and seismic" cover the applied side of aquatic biotremology (e.g., anthropogenic noise), they do not address all biologically relevant aquatic substrates and therefore should be avoided. The use of "particle" terms may cause confusion with water-borne stimuli, see previous section. "Vibration" and "underwater vibration" alone are too broad.

Given the common use of "water borne particle motion," a substrate-based version could be used, "substrate borne particle motion." However, given the previous explanation this should be avoided due to the following reasons: (1) the

**Table 1** Terms currently used in the aquatic literature relating to substrate-borne vibrations (Roberts et al. 2015, 2016; Spiga et al. 2016; Roberts and Elliott 2017; Fitzgibbon et al. 2017; Morris et al. 2018; Roberts and Laidre 2019; Day et al. 2019, 2020, 2022; Cote et al. 2020; Aimon et al. 2021; Hawkins et al. 2021; McCauley et al. 2021; Popper et al. 2022; Roberts and Howard 2022; Jézéquel et al. 2022). Organized by a first grouping term, and appended second term, e.g., "Ground acceleration," "Ground motion," "Ground vibration"

| First term | Second term |
|---|---|
| Anthropogenic | Vibration |
| Ground | Acceleration<br>Magnitude<br>Motion<br>Vibration<br>Roll |
| Ground-borne | Acceleration/velocity<br>Vibration<br>Signal |
| Interface waves | |
| Mechanical | Vibration<br>Wave |
| Particle | Motion in the seabed<br>Velocity/acceleration |
| Seabed | Vibration |
| Sediment | Vibration |
| Seismic | Noise<br>Interface waves |
| Substrate-borne | Particle motion<br>Energy<br>Vibrations<br>Vibrational noise |
| Substrate | Motion |
| Underwater | Vibration |
| Vibration | |
| Vibrational | Stimuli<br>Noise |
| | Wave |

term implies that particle motion of water and substrate are in the same direction of travel, and are the same wave types; (2) this may lead to measurement confusion and assumptions, such as that measuring particle motion may allow estimation of substrate-borne vibration, and vice versa.

It is suggested that aquatic terms are aligned to current biotremological terminology. That is, using the term "substrate-borne" appended accordingly, and "vibrational noise." Where a noise source is measured as a seismic interface wave, it is suggested that "groundroll" is appropriate (Hawkins et al. 2021) with the caveat that the term alienates the aquatic literature from the terrestrial. More precise terms such as Sholte waves, would be encouraged, as long as keywords of "substrate-borne" are additionally included.

## Conclusions

Here a necessary refinement and update of the definition of biotremology allows an incorporation of aquatic animals and associated surfaces. This brings the terrestrial and aquatic systems into alignment, increasing connectivity of both research areas.

Researchers are encouraged to develop approaches to analyzing sensitivities and responses to vibration in the aquatic environment and to measure substrate-borne vibration (alongside water-borne particle motion and pressure, where appropriate) in *any* noise exposure study involving animals residing upon, in, or just above a substrate. This is particularly important given that many anthropogenic activities directly contact the seabed, eliciting significant vibrations in the substratum.

Most importantly, this chapter highlights a need for standardization of terminology between the disciplines of bioacoustics and biotremology. However, to standardize terms, they must be discussed *within* and *between* each discipline. As an example, there is disagreement in terms between researchers studying different animal groups, as well as different stimuli.

What overall conclusions can be drawn? Only a cautionary note, rather than a recommendation; it is suggested that aquatic researchers consider the range of habitats that their study animal occupies. Should the animal encounter water-sediment or water–air interfaces at any time, then the use of "particle motion" in relation to hearing and noise creates confusion between the two disciplines and only elicits the question "which type of mechanical wave is being detected?" However empirical data are required to investigate whether such animals respond to substrate-borne waves, before they could be definitively classed within the realms of biotremology.

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# Modeling Underwater Noise in the North Adriatic Sea

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Michol Ghezzo, Robert Chavanne, Dominique Clorennec, Roger Gallou, Fantina Madricardo, Antonio Petrizzo, Marta Picciulin, and Thomas Folegot

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**Abstract**

The Adriatic Sea hosts numerous Natura 2000 sites and protected areas but it is an underwater noise hot spot in the Mediterranean Sea. This calls for managing measures. The SOUNDSCAPE project developed for a transnational assessment of the underwater sound in the Northern Adriatic Sea. Underwater sound modeling at the basin scale was performed by Quonops© by using oceanographic data developed by CNR-ISMAR as input data, together with automated information system (AIS) vessel data. Acoustic measurements based on nine monitoring stations were used to calibrate the model. A yearlong (2020) 3D simulation was run, and two sets of monthly maps were produced: the "natural ambient noise" based on the oceanographic data and the "baseline ambient noise" created by adding to natural maps the ship traffic (AIS data). Maps showing sound pressure levels for four one-third octave bands centered at 63 Hz, 125 Hz, 250 Hz, and 4000 Hz were calculated for three depth ranges. This first atlas of maps contributes to filling a relevant knowledge gap about the underwater sound distribution in the study area.

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**Keywords**

Underwater noise · Acoustic modeling · North Adriatic Sea · SOUNDSCAPE project

## Introduction

The attention on the underwater noise as emergent pollutant is growing in the last decades. Human activities have been recognized as relevant sources of underwater noise for the seas and oceans (Duarte et al. 2021). Dealing with such a pervasive, transnational pollutant is the future challenge for policymakers and environmental managers. Natural and anthropogenic noise sources are distributed along the coast and the open sea. The underwater sound levels are the result of the sources distribution and the sound propagation, which, in its turn, is influenced by environmental factors, such as bathymetry, seabed sediment type, water temperature, and salinity. Despite this high level of complexity, numerical simulation based on acoustic modeling allows to estimate the spatiotemporal distribution of sound levels at basin scale (Jensen et al. 2000).

The Marine Strategy Framework Directive (MSFD) 2008/56/EC, adopted by the European Union in 2008, directly addresses the underwater noise as new pollutant (Merchant et al. 2022) and prescribes to monitoring and mapping the underwater noise linked to maritime transport. The calculation of the underwater sound levels generated by anthropogenic activities using numerical models can support the legislative requests and the evaluation of potential impact on species of interest. Following MSFD guidance, several international programs dedicated to underwater noise have been developed in Europe covering important European seas (reviewed by Merchant et al. 2022). Nevertheless, long-term information on continuous underwater sound levels in the Mediterranean Sea is still poorly available. To fill this gap, the Italy-Croatia Interreg SOUNDSCAPE project developed a first basin-scale, cross-national underwater acoustic monitoring in the Northern Adriatic Sea (NAS) recording acoustic data from March 2020 to June 2021 at nine monitoring stations (Fig. 1; see also Petrizzo et al. in the present issue for further detail). The SOUNDSCAPE study area includes the northern part of the Adriatic Sea, as well as the northeastern part of the Mediterranean Sea, extending as north as 45°47’ N, until a transect from Split (Croatia) to Pescara (Italy). NAS has been recognized as an underwater noise hotspot, and it requires managing measures due to its high biodiversity, hosting numerous Natura 2000 sites and protected areas (Maglio et al. 2016). The in situ collected data, that have been recently publicly released (Petrizzo et al. 2022, 2023), were used to calibrate a 3D sound prediction model. In its turn, the modeling results provided a first statistical quantification of the spatiotemporal distribution of sound levels produced at a basin scale in the Mediterranean Sea for the whole year 2020.

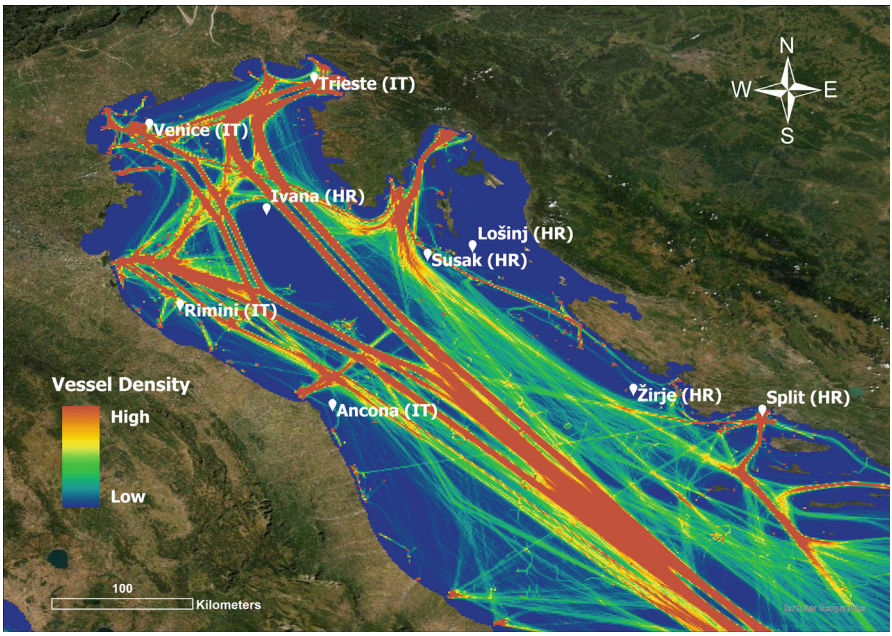



**Fig. 1** Location of acoustic monitoring stations in the North Adriatic Sea SOUNDSCAPE study area. Cargo vessel traffic is also represented as total number of cargo vessel passages per year (2020) as obtained from EMODnet Human Activities, Vessel Density Map. (Revision date 2022-03-21). Geographic map was created using ArcGIS software by Esri

## Sound Propagation Modeling

The Quonops© underwater sound prediction system (Folegot 2009), implemented over NAS, is built on the same principle as oceanographic or meteorological prediction systems (Jensen et al. 2000). The models embedded in the Quonops platform solve the Helmholtz equation of wave propagation based on parabolic equation (Collins et al. 1996) and, for the higher frequency range, on Gaussian energy distribution ray modeling (Porter and Reiss 1984), which reflects the geometric distribution of noise in the water column. The three-dimensional variability of the speed profiles of sound in water, depending on water temperature, salinity, and pressure (or depth), is considered in the model. To account for the variability, stochasticity, and uncertainties of key physical and environmental parameters, a statistical approach based on Monte Carlo simulation (Robert and Casella 2004) is used. This approach has the advantage of estimating the sensitivity of the model to both the variability and uncertainty inherent in input variables. Many noise sources can therefore be modeled and used to calculate the median values (50th percentile) of

noise at every given location. The other percentiles represent the description of the distribution of the noise in each point of the domain and can be associated to most frequent situations. For example, the underwater ambient noise is described by the percentiles from 25th to 75th, corresponding to noise levels occurring 25–75% of the time; *au contraire*, most intense noise events are described in the smallest percentiles, i.e., 1th to 25th percentile, with noise levels occurring 1–25% of the time. Natural noise is represented by the 95th and above percentiles.

The modeling has been carried out at the center frequency *fm* of each one-third octave in the frequency band of the acoustic sources. The SOUNDSCAPE project dealt specifically with monitoring continuous low-frequency sound as covered by the MSFD indicator 11.2.1. Therefore, the one-third octave band center frequencies of 63 and 125 Hz (re 1 μPa) were the focus of the modeling within SOUNDSCAPE. In addition, two other bands were investigated: the 250 Hz and the 4 kHz one-third octave bands. Petrizzo et al. (present issue) describe in detail the reasoning for the choice of these frequencies.

The Quonops© implementation covers the Northern Adriatic domain (Lat 42.8, 45.8; Long 12.1, 17.9), slightly larger than the study area to avoid numerical effects on the border. The calculation is performed on regular grid with a resolution equal to 181 × 179 m.

Two different sets of monthly maps were produced: (i) a mapping of the "natural ambient noise" using as input oceanographic data such as temperature, salinity, sea state, and bottom properties and (ii) the mapping of the "baseline ambient noise" obtained by adding also the information about ship traffic using automated information system network (AIS) data. Specifically, the oceanographic input data (temperature and salinity of the water column) were generated by a three-dimensional, high-resolution, calibrated oceanographic model SHYFEM developed by CNR-ISMAR for the study area (Ferrarin et al. 2019); the wind data used are consistent with the hydrodynamic input data; bathymetry and bottom grain size were obtained by consulting public databases such as EMODnet (https://www.emodnet-bathymetry.eu/) and the NAVOCEANO bottom-type database (Fleischer et al. 2011). In some specific areas along the coast the bathymetric data were updated using the data adopted by the hydrodynamic model or kindly provided by Dr. Pikelj (*pers. com*). AIS data was provided by marine traffic provider (www.marinetraffic.com). A fusion data processing has been carried out by integrating into QO-AIS database, which is connected to the modeling service. Acoustic measurements from SOUNDSCAPE stations have been used to calibrate the sound model.

The results of the modeling were presented as monthly and annual statistical sound maps, considering four frequencies (63, 125, 250, and 4000 Hz one-third octave bands), seven percentiles, and three layers of depth (0–15 m, 15–30 m, and 30 m bottom). An example of monthly statistical sound maps for natural and baseline conditions is shown in Fig. 2; it refers to the entire water column considering 125 Hz one-third octave band. According to the map, the noisiest hotspots are mainly located in the central area of the NAS basin, partly mimicking the shipping routes

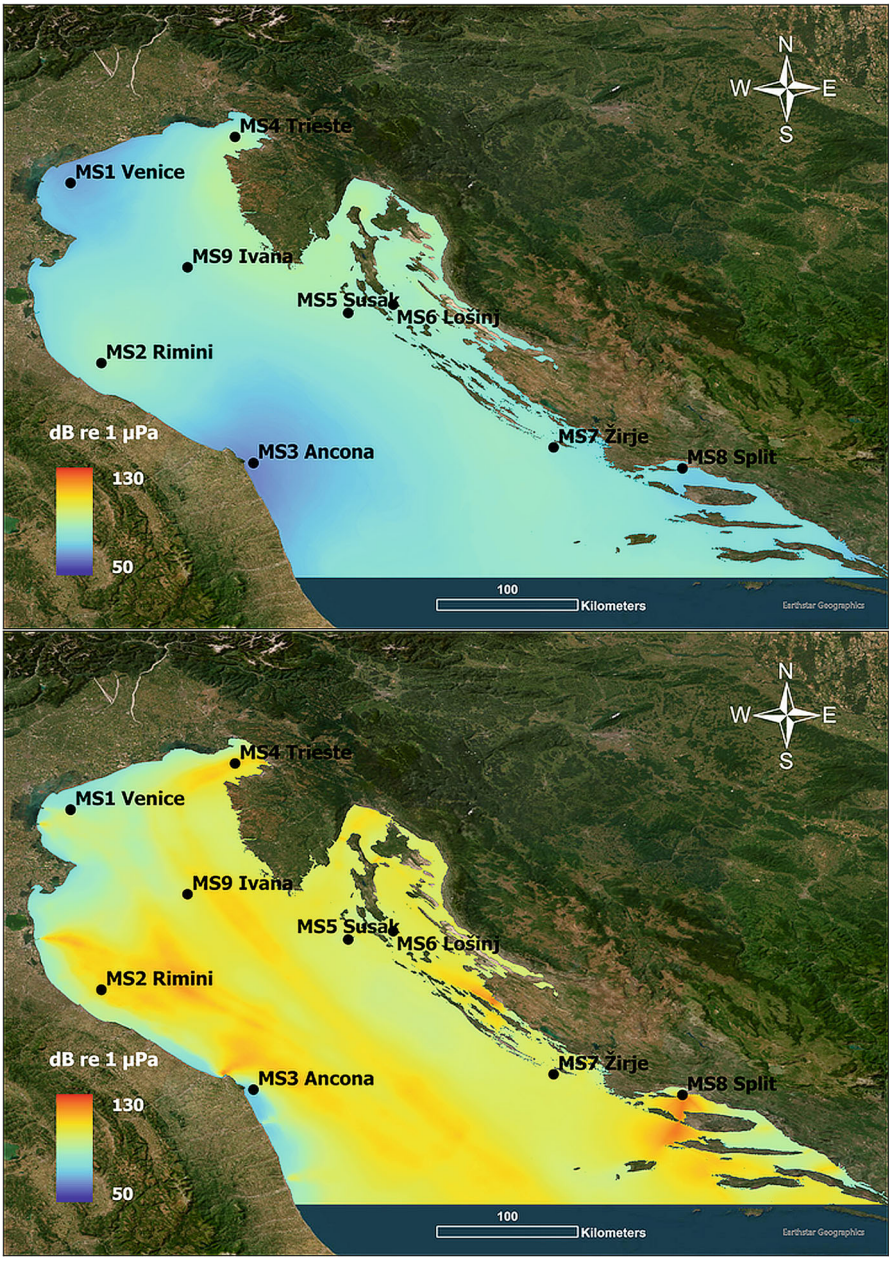



**Fig. 2** Statistical natural and baseline sound map (July 2020; 125 Hz 1/3 octave band, entire water column); 50th (upper panel) and 10th (lower panel) percentile, respectively. Statistical sound data have been obtained by Quonops© Online Services, by Quiet-Oceans (https://qos.quiet-oceans.com/). All right reserved. Geographic map was created using ArcGIS software by Esri

of the cargo traffic (see Figs. 1 and 2 for comparisons). Variations in the sound pressure levels can also be possibly related to: (i) the sediment composition, which is mainly sandy along the Italian coast and rocky along the Croatian coast, and (ii) the water depth, that is generally shallower on the Italian coast.

## Conclusions

A first atlas on underwater noise maps in the North Adriatic Sea was achieved by the EU Interreg Italy-Croatia SOUNDSCAPE project. The noise maps provide an estimate of the spatiotemporal distribution of noise levels in the Northern Adriatic Sea during the year 2020, which is also an exceptional period since it includes the pandemic lockdown time. The modeling results contribute to filling a relevant knowledge gap about the underwater noise distribution in the Mediterranean Sea.

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# Fish Sound Production Research: Historical Practices and Ongoing Challenges

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Audrey Looby, Kieran D. Cox, Santiago Bravo, Rodney Rountree, Francis Juanes, Amalis Riera, Sarah Vela, Hailey L. Davies, Laura K. Reynolds, and Charles W. Martin

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**Abstract**

Even with widespread evidence of the ecological importance of fish sounds for signaling and communication, and an increasing awareness of the negative impacts of anthropogenic sound, research into fish sound production still faces ongoing challenges limiting growth. To acquire quantitative data on research into fish sounds and assess topics related to their study, a systematized review was conducted to survey over 3000 references for published examinations of sound production by fish species. Additional information was further extracted on the study of marine, subtropical species from a subset of the assembled references. The review findings showed that the rate of soniferous fish species discovery remains inefficient for understanding even a sizeable portion of the behaviors of the 34900 extant fish species known to exist. In the literature surveyed, there was also no standardized sound nomenclature, fish species were not frequently tested in multiple environments, and sound visualizations were inconsistently provided. Advancements in these areas would improve the ability to document, describe, and synthesize fish sounds, which would in turn increase understanding of their global scope and the anthropogenic threats that they face.

**Keywords**

Acoustic communication · Fish behavior · Systematized review · Bioacoustics · Passive acoustics · Soundscape ecology

## Introduction

Over 1000 documented fish species produce active sounds in numerous behavioral contexts, including feeding, courtship, aggression, defense, and social interactions, and many more may be actively soniferous (Kasumyan 2009; Looby et al. 2022a, b;

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Rice et al. 2022). Among vertebrates, fish have the largest diversity of sonic organs and are one of the largest groups of active sound-producers (Ladich and Fine 2006; Ladich and Winkler 2017). Fish are capable of sound source segregation and auditory scene analysis, similar to humans, and can thus use the information embedded within acoustic cues (Bregman 1994; Fay 2009). Other organisms including birds (Hansen et al. 2017) and crustaceans (Hughes et al. 2014) may use fish sounds as a sensory cue to hunt prey, avoid predators, or find suitable habitat. Some fishes can be identified to the individual based on their calls (e.g., Vieira et al. 2015), and certain fish species may even have different call parameters between geographically distinct populations, suggesting the presence of dialects among fish (Fine 1978; Noel and Johnston 2015; Phillips and Johnston 2008). While the capacity to produce active sounds in fishes may not be taxonomically ubiquitous, the substantial extent of known actively soniferous fishes demonstrates an ecological importance analogous to other highly soniferous taxa, such as birds and marine mammals (May-Collado et al. 2007; Odom et al. 2014; Rice et al. 2022).

In addition to active sounds, all fish can produce passive sounds incidentally, which may also carry information (Kasumyan 2008; Rountree et al. 2018). For example, feeding sounds and hydrodynamic sounds can be acoustically distinguished, and they may both have varying characteristics with differing fish sizes and species (Kaparang et al. 1999). Swimming speed (Fujieda et al. 1994) and the type of food being chewed (Kim 1977) can also be differentiated acoustically. Fishes are attracted to both incidental feeding and swimming sounds, further demonstrating their ecological signal potential (Banner 1972; Kim 1977). Although only 219 species have so far been documented to make passive sounds, this is likely due to a lack of emphasis on the topic in research rather than being representative of the extent that they are produced and used for information acquisition (Looby et al. 2022a, b).

As the number of fish species known to use acoustic cues continues to increase, so too do the potential effects of anthropogenic sound (Looby et al. 2022a). In recent years, the capacity for anthropogenic activity to impact the ability of fishes to interact with soundscapes has become indisputable (Cox et al. 2018; Duarte et al. 2021; Rountree et al. 2020b). Anthropogenic sound now contaminates aquatic ecosystems globally, attributed to expanding commercial, recreational, and research sectors (Hildebrand 2009; Shannon et al. 2016). As a pollutant, anthropogenic sound has the potential to disrupt active and passive fish sounds, with consequences for their associated ecological functions and uses, including foraging ability, reproductive success, and predation risk (Cox et al. 2018; de Jong et al. 2020).

In addition to their ecological importance and anthropogenic threats, fish sounds can be used in remote monitoring applications through passive acoustics. Passive acoustic monitoring (PAM) of active fish sounds is used for a variety of applications, such as detecting and measuring fish spawning (Lowerre-Barbieri et al. 2013) or mapping the spread of invasive species (Rountree and Juanes 2017). Passive fish sounds, in contrast, have been primarily used in aquaculture operations, such as to monitor feeding (Mallekh et al. 2003), though they have other possible applications (Rountree et al. 2018, 2020b). Continued advances in the automated detection of fish sounds through machine learning and in online libraries of fish sound recordings will

further increase the value of fish sounds for ecological monitoring (Lindseth and Lobel 2018; Parsons et al. 2022; Sattar et al. 2016).

There is wide-ranging and ever-increasing evidence of the ecological importance of fish sounds, the anthropogenic threats they face, and their PAM applications. Nevertheless, the field of fish bioacoustics remains limited in the ability to document, describe, and synthesize fish sounds on the scales needed to comprehensively understand the soniferous behavior of the estimated 34900 extant fish species (Froese and Pauly 2022). In this chapter, data collected through systematized reviews are used to discuss the history of research into fish sounds and several ongoing challenges for the field, including an insufficient rate of soniferous species discovery, a lack of standardized sound nomenclature, limited testing for sound production in multiple environments, and the inconsistent provisioning of sound visualizations.

## Methods

### Basic Information on Global Fish Sound Production Research

In this review, the data compiled by Looby et al. (2022b) were used to describe global research into soniferous fish diversity. Looby et al. (2022a) conducted a systematized review (Grant and Booth 2009) to compile a dataset of published examinations of a fish species for sound production from 1874 to 2020. Looby et al. (2022b) used the same methods described in Looby et al. (2022a) to update the dataset to include the year 2021 as well as several studies missed by the original review methodology. Looby et al. (2022b) described basic sound production information (i.e., whether a species produces active and/or passive sounds) for 1214 species studied in 920 references. The Looby et al. (2022b) data were accessed through its associated online data repository (Looby et al. 2021).

### Additional Information on Marine, Subtropical Fish Sound Production Research

An additional systematized review was conducted of the marine, subtropical fish species that were determined auditorily to be actively soniferous in an English, peer-reviewed journal article or in Fish and Mowbray 1970. Using the 2018 version of the Looby et al. 2022a dataset that covered the years 1874–2018 (Looby et al. 2021), a follow-up review was conducted of descriptors, testing environments, and visualizations associated with sound production. In this review, only English, peer-reviewed journal articles were considered, to maintain consistency in language (e.g., to only assess sound descriptors in English) and scientific validation (i.e., through peer-review), respectively. Fish and Mowbray 1970, a book documenting surveys of sound production and associated behaviors of over 200 fish species, was also included as it is a highly regarded information source in

the field of fish bioacoustics (Rountree et al. 2002). Data were collected only on marine, subtropical fish species (Boettiger et al. 2012; Froese and Pauly 2022) to keep the data collection feasible. The species were then limited to include only fishes that were examined for sound production auditorily (i.e., documentation of the presence of audible sounds through listening live or on a recording) and found to produce active sound, as determined by Looby et al. 2021. The resulting data compilation comprised 186 fish species and 194 publications, with a total of 450 investigations (i.e., individual auditory documentations of actively soniferous behavior by a species in a reference).

Data were collected on what sound names or descriptors (i.e., one to a few words used to identify and describe different types of fish sounds) were reported as produced by the actively soniferous marine, subtropical species in the publications surveyed. Because numerous synonyms could be used to describe the same type of sound, each of the terms authors used to describe sound types was documented. The descriptors were standardized grammatically, so terms such as grunt, grunting, grunted, and grunts were all combined into one category. Similar descriptors, such as tick and fast repetitive tick (FRT) or crunch and scrunch, were also combined into singular categories. Species may have produced multiple types of sounds, which could have been described in the same or multiple publications. Sound descriptors used for active or passive sounds were not differentiated, as both may have been produced within the same study. Sound descriptors may have been tentative (i.e., if the authors speculated about a certain sound) and/or cited (i.e., as opposed to determine through testing within the study). For the purposes of this chapter, all tentative, nontentative, cited, and noncited entries were included in the subsequent data analyses and visualizations.

Data were also collected on the testing environments of the examined species. Testing environments were distinguished between three broad categories: captivity, semi-wild and wild. Captivity included any testing conducted in laboratories, tank environments, or other completely artificial, human, or captive environments. Semi-wild included any testing conducted in a natural environment with artificial/human enclosures for the study animals, such as outdoor pens, or relatively large human-made environments, such as artificial ponds. Wild included any testing conducted in a natural environment without any type of artificial or human enclosures for the study animals. The testing environment may have also been unclear or unreported by the study authors. Species may have been tested in multiple environments in the same or multiple publications.

Finally, data were collected on what types of sound visualizations were provided for a particular species. Whether any oscillograms (i.e., waveforms), spectrograms, and/or power spectra were provided for a species examined was determined for each reference. Visualizations included in the main text of the article or in electronically available supplementary material were all considered valid for the data collection.

All data collected were based on the reviewer's interpretation of the authors' intentions and meaning to mitigate reviewer bias. As such, data collection was conducted by a single person to maintain internal consistency. To validate the data collected, the data categorizations associated with over 100 of the 450 investigations

surveyed, including any that were ambiguous or uncertain, were assessed by a second reviewer.

### Data Visualization and Analysis

All data analyses and visualizations were conducted in R (R Core Team 2022) version 4.2.1 (2022/06/23). All taxonomic and water body information on the fish species were accessed through the R package rfishbase version 4.0.0 (Boettiger et al. 2012) with the FishBase version set to 22.02. The climate zone information was accessed from the FishBase website, which only designates a single primary climate zone per species as opposed to rfishbase which designates any climate zones the species is found in (Boettiger et al. 2012; Froese and Pauly 2019). A rarefaction and extrapolation analysis of actively and passively soniferous species discovery was also conducted, following the methods outlined in Looby et al. (2022a) with the data updated to those included in FishSounds Version 1.1 (Looby et al. 2020b). Other visualizations were created using the R packages ggplot (Wickham 2016) and wordcloud (Fellows 2018). All the data used to create the figures are available in a supplemental information file in the FishSounds Data Repository (Looby et al. 2021), and all the data collected in both systematized reviews are available on the FishSounds website (Looby et al. 2022b) and in the FishSounds Data Repository (Looby et al. 2021).

## Global Fish Sound Production Research

### History of Fish Sound Production Research

Research into fish sound production and detection has an extensive history. Pliny the Elder and Aristotle described fish sounds and possible fish hearing over 2000 years ago (Aristotle 1862; Pliny the Elder 1938). Contemporary scientific research on fish sound production began in the late 1800s (Abbott 1877; Dufossé 1874) then underwent a considerable expansion in the middle of the twentieth century due to innovations in underwater acoustic equipment. Technical advances and research were motivated in part by naval interests in distinguishing between sounds produced by marine organisms or by human activities (Fish et al. 1952), as well as increasing business investment in auditory attractants for fish ranching and acoustic fishing methods (Choo et al. 1988; Takemura et al. 1988). Scientific interest in fish bioacoustics was further driven by questions surrounding the physical mechanisms of sound production, the ecological implications of fishes' perception of and contribution to the prevailing soundscape, the impacts of anthropogenic activities, and the potential to monitor soniferous fish populations remotely (e.g., Carriço et al. 2020; Ladich and Winkler 2017; Rountree et al. 2020b). Consequently, the rate of studies examining fish species for sound production has steadily increased from the mid-1900s to the present (Fig. 1a).

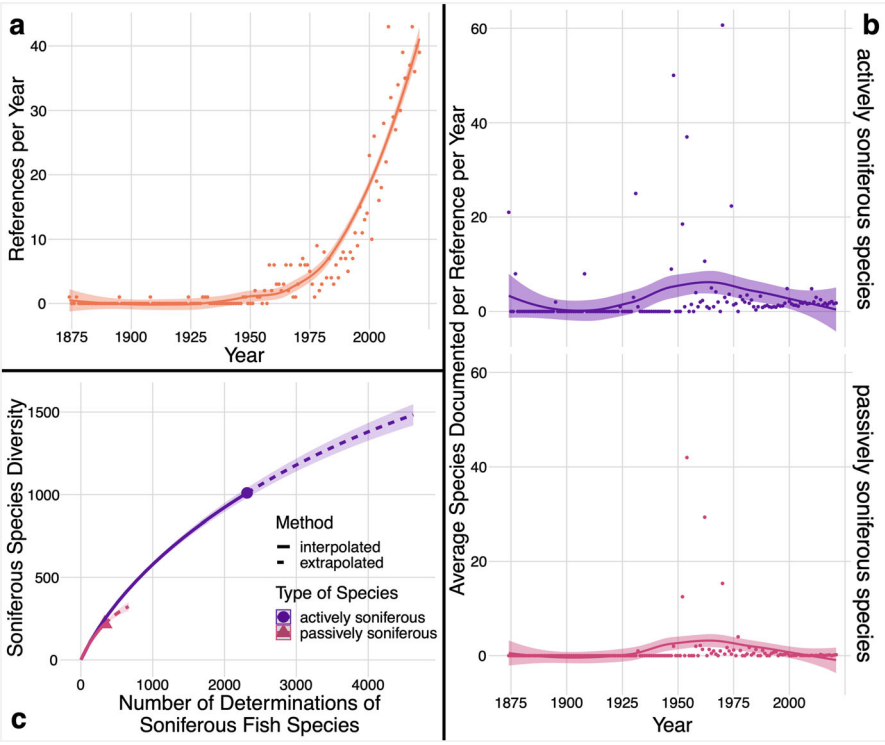



**Fig. 1** (**a**) The number of references per year with independent (i.e., not cited) fish sound production examinations and (**b**) the average number of actively and passively soniferous species documented per reference per year, with lines plotted using the local polynomial regression fitting method (Looby et al. 2022b; Wickham 2016); and (**c**) rarefaction and extrapolation curves of the number of actively and passively soniferous fish species if the numbers of determinations of each type of soniferous fish species were doubled (Looby et al. 2022a, b)

Despite considerable advancements in fish bioacoustics, the prevalence of sound production among extant fish species remains indeterminate. There have been numerous efforts to catalog soniferous fish species, such as within a particular region (e.g., Fish 1954; Fish et al. 1952; Rountree et al. 2019) or fish taxa (e.g., Colleye et al. 2011; Hallacher 1974; Kaatz and Stewart 2012). More recently, a review of known fish sound production was conducted using systematized review methodology to create a comprehensive global inventory of documented examinations of a fish species for sound production that was later made available on a public website (Looby et al. 2022a, b). Only about 4% of extant fish species have been examined for sound production, however, and active sound production in fishes has evolved independently multiple times (Looby et al. 2022a; Rice et al. 2022). As a result, the number of actively soniferous fishes globally remains difficult to estimate and many aspects of fish sound production behavior are still unknown (Parmentier et al. 2021; Rice et al. 2022).

## Rate of Soniferous Species Discovery

While the publication rate of studies about fish sound production has been increasing to the present day (Fig. 1a), the number of soniferous species documented per reference has decreased from its peak in the middle of the twentieth century (Fig. 1b). A relatively larger proportion of studies earlier in the contemporary scientific literature focused on surveying greater numbers of species for sonifery, including many publications that featured independent examinations of dozens if not hundreds of fish species (e.g., Burkenroad 1931; Fish and Mowbray 1970; Fish et al. 1952). In contrast, with increasing rates of publications in recent years, there were also increasing numbers of studies focusing on only one species to describe soniferous behaviors in greater detail (e.g., McIver et al. 2014; Noel and Johnston 2015; Phillips and Johnston 2008), with only three references since the year 2000 that documented examinations of more than 15 species for sound production (Kaatz and Stewart 2012; Kaatz et al. 2010; Tricas and Boyle 2014). This decrease is even more pronounced for studies of passive fish sound production compared to active fish sound production, with passive fish sounds remaining understudied despite their ecological signal potential and assumed widespread occurrence in fishes (Fig. 1b; Kasumyan 2008; Looby et al. 2022a).

Even with increasing numbers of scientific publications on fish sound production, and in part because of the decreasing numbers of soniferous species studied per reference, the current rate of soniferous species discovery remains insufficient to describe the thousands of soniferous species thought to exist (Fig. 1c; Looby et al. 2022a; Rice et al. 2022). FishBase lists 34900 extant fish species (Froese and Pauly 2022), of which only roughly 4% have been studied for sound production and even fewer of which have been documented as producing active and passive sounds (Looby et al. 2022a). Furthermore, an extrapolation and rarefaction analysis showed that if the scientific effort studying sound production in fish species from 1874 to 2021 was doubled, only a total of 1483 ($\pm$63) actively soniferous fish species and 324 ($\pm$27) passively soniferous fish species would be documented to exist. The low estimates of soniferous fishes produced through extrapolation reflect the paucity of studies on fish sounds relative to the number of extant fish species. Because active sound production is not an ancestral trait among fishes, active sonifery cannot be assumed among all fish taxa (Rice et al. 2022). Additionally, there has been evidence of possible losses of sound production behaviors in certain species, further signifying that active sound production may be difficult to speculate taxonomically (Boyle et al. 2014; Kaatz 1999; Rice et al. 2022).

The unique challenges in studying fish sound production have caused this research to lag behind other soniferous taxa. Documenting soniferous fishes can be highly uncertain, with many authors reporting doubt in at least some of their conclusions (Looby et al. 2022a). Fish sound production behavior can be highly context-dependent, and the fish sounds themselves may be rare, noncontinuous, or low in amplitude (e.g., Brantley et al. 1993; Lowerre-Barbieri et al. 2013; Wall et al. 2014). Community science efforts, like those that have been so successful for documenting bird and frog sounds (FrogID 2022; Macaulay Library at the Cornell

Lab of Ornithology 2022), have not yet been implemented as extensively to research fish sounds because of the technological limitations of recording underwater. There are also thousands of extant fish species – a much greater number than the roughly 200 species of marine mammals, almost all of which have been confirmed to make active sounds underwater (Parsons et al. 2022). Research into fish sound production is not unique in these challenges – the study of underwater invertebrate sounds faces similar difficulties (Hawkins and Popper 2017), – but considering the ecological, economic, and cultural importance of fishes globally, the rate of soniferous fish species discovery needs to increase to improve knowledge of fish behavior, signaling, and communication. This strengthened understanding is necessary to assess the effects of anthropogenic sound on fish acoustic communication and to effectively apply passive acoustics to monitoring, conservation, and fisheries management.

## Marine, Subtropical Fish Sound Production Research

### Sound Descriptors

There were 60 different terms used to describe the sounds produced by the 186 marine, subtropical fish species about which additional information was collected (Fig. 2). These included knocks, choruses, rasps, snitches, booms, erks, ticks/FRTs, wheezes, sneezes, chants, gurgles, moans, and complex calls. Sound descriptors were largely onomatopoeic in nature (e.g., boom), though a few were used to describe sounds based on frequency composition (e.g., tonal/harmonic) or calls composed of multiple smaller sound units (e.g., complex call, chorus). Pulse – as both a sound name and frequently used word to describe subunits of sounds – was the most common term found (Fig. 2). Other common terms included grunt, tonal/harmonic, knock, chorus, thump, and drum, all of which were used in over 50 investigations to describe sounds. Many sound descriptors were rarely used in the

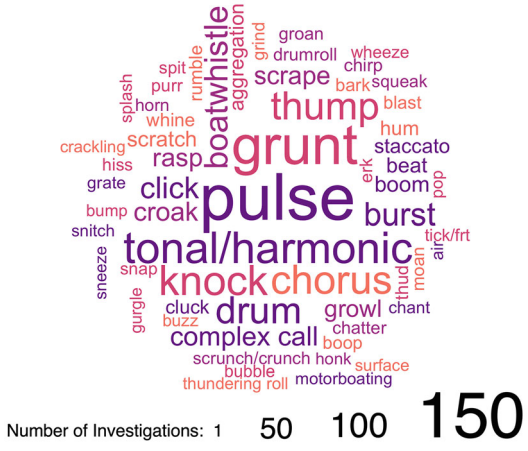



**Fig. 2** Word cloud showing the 60 sound descriptors identified in the data collection, with the font size of the words representing the number of investigations (i.e., individual auditory documentations of actively soniferous behavior by a species in a reference) in which a sound descriptor category was used

references assessed, with 9 terms only used once (e.g., motorboating, thundering roll) and 31 others used in less than five investigations (e.g., purr, bump). Beyond the short sound descriptors that were quantified, authors also often used longer phrases to describe particularly abnormal sounds, such as the "bubbling of a giant percolator" associated with the choruses of the hardhead catfish (*Ariopsis felis*; Kellogg 1955; Tavolga 1971), or the sustained clicks produced by the gray triggerfish (*Balistes capriscus*) that "resemble the tearing of canvas cloth" (Fish 1954).

The sound descriptors found were inconsistently used. Often, multiple sound names were applied to similar or identical sounds, creating redundancy and confusion. The terms grunt and thump typify this as both describe low-frequency sounds of relatively short durations, with little to distinguish between them, though they could still be used differentially between species even within the same reference (e.g., Fish and Mowbray 1970). Alternatively, multiple names could be used in different references to describe the same type of sound produced by the same species. For example, the plainfin midshipman (*Porichthys notatus*) produces three primary types of sounds (i.e., hum, grunt, growl; McIver et al. 2014) but had ten sound descriptors (e.g., buzz, groan) associated with it in the 11 publications surveyed. The overlapping nature of some passive and active sounds meant that they could be described using similar terms as well, making them more difficult to differentiate. For instance, scrapes could describe both the incidental sounds produced by parrotfishes (Family Scaridae) feeding on hardened substrates as well as the agonistic sounds produced for communication by the southern puffer (*Sphoeroides nephelus*; Burkenroad 1931; Sartori and Bright 1973). These inconsistencies affect the ability to understand the types of sounds being described as well as to compare findings among similar sounds or species.

Individual studies may do a thorough job in defining the different sound descriptors used within them (e.g., McIver et al. 2014; Sartori and Bright 1973), but the field of fish sound production research still lacks a widely applied, standardized sound-naming and description system. Naming systems exist for other soniferous taxa, such as the onomatopoeic naming system of bird songs (Pieplow 2017), the standardized division of whale songs into themes then sound units (Garland et al. 2013), or the norms around using dolphin sound types (Jones et al. 2020). Such systems allow for more effective communication about the characteristics of the sounds being described (Jones et al. 2020). To be successful, the development of such a system for fish sounds must consider their relatively simplified nature; coordinate among a global and diverse assortment of researchers, species, and sounds; and balance the needs for technical descriptions based on acoustic characteristics (e.g., sounds described in terms of their frequency, duration, or structure) and meaningful engagement of the public with more evocative terms (e.g., purr, boop).

## Testing Environments

Researchers may use different testing environments (i.e., captivity, semi-wild, and wild) when studying soniferous behavior, any of which can impact the results of

their studies. Testing for sound production in captivity, for instance, ensures correct species and associated behavior identification and allows for lab experimentation of environmental and situational variables of interest (Diamond 1983; Sprague and Luczkovich 2001). Some fishes, however, may not display soniferous behaviors in captivity, and others may behave differently in the wild and in the lab due to environmental and physiological changes (e.g., Bolgan et al. 2020; Holt and Johnston 2014; Rountree and Juanes 2020). Small tank environments can also have more ambient sound, reverberation, and resonance that may cause acoustic interference and distortions (Akamatsu et al. 2002; Bart et al. 2001). Though these effects can sometimes be counteracted, the results of fish bioacoustic studies conducted in tanks are still treated with caution unless validated in open water environments (Akamatsu et al. 2002; Slabbekoorn 2016).

As a result of the challenges with captivity testing, many researchers study fish sound production in wild (e.g., Fine 1978) or semi-wild (e.g., Brown et al. 2021) conditions, though these also present their own difficulties. Wild or semi-wild testing environments may provide a more accurate assessment of how sound production occurs and is used naturally (Diamond 1983). Testing for sound production in the wild, however, is often conducted without direct observation of sound producers or their behavior and relies on comparison with known sounds for identification or other more circumstantial evidence (Lindseth and Lobel 2018). Identification of unobserved sounds based on comparison with known sounds relies on the existence of comprehensive libraries of sounds, which are rarely complete for a given geographic area, giving rise to frequent calls for developing sound libraries (e.g., Lindseth and Lobel 2018; Luczkovich et al. 2008; Rountree et al. 2006). To identify sound sources more directly, wild studies may incorporate unaided visual observation (e.g., Abbott 1877) or underwater video (e.g., Rountree et al. 2018), among other methods. Even with direct observation, however, it may not be possible to provide conclusive proof of sound source identity if obvious sound production behavior is not evident (e.g., Rountree and Juanes 2010). In addition, fish behavior can be altered by the presence of observers or instruments (e.g., Lindfield et al. 2014). Some of these conundrums can be addressed using multiple modes of observation together with passive acoustic arrays, though these techniques have yet to be regularly implemented (Mouy et al. 2018; Rountree 2008; Rountree et al. 2020a). Similarly, semi-wild studies may aid in isolating the species and sound production behavior of interest while still reducing acoustic distortions and observer effects (e.g., Kaparang et al. 1999), but misidentifications are nonetheless still possible if another soniferous species is nearby (Moulton 1956; Fish and Mowbray 1970; Sprague and Luczkovich 2001). Because of the benefits and downsides of any individual fish sound production study method, testing in a variety of environments and scenarios is necessary to comprehensively describe the soniferous behavior of a fish species.

Despite the value in varied testing for sound production, these practices were not universal in the studies surveyed. For species tested within individual references, only 41 of the 450 investigations found in the review tested for active sound production behavior in more than one environment among captivity, semi-wild,

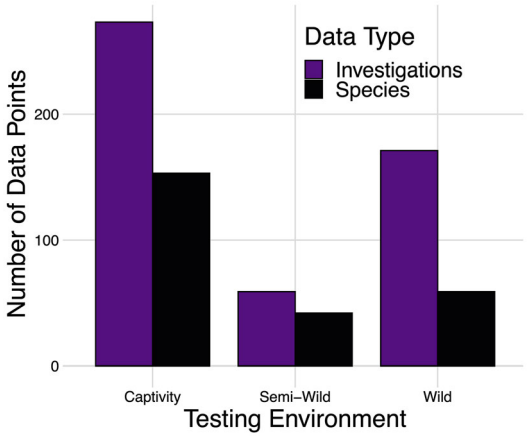



**Fig. 3** The number of investigations (i.e., individual auditory documentations of actively soniferous behavior by a species in a reference) and the number of species tested in each environment

and wild. Twenty-two investigations had at least one unclear or unreported testing environment, further complicating the reproducibility or comprehension of their research. Additionally, of the 186 species covered in this review, only 46 species were documented actively producing sounds in more than one environment, with captivity being the most common method among investigations and species, followed by wild, then semi-wild (Fig. 3). The field could therefore greatly increase its current rate of usage of multiple testing environments.

When possible, studies should utilize multiple testing environments to overcome the weaknesses of any individual approach (Diamond 1983) and to describe fish soniferous behavior more extensively (e.g., Bertucci et al. 2021; Fish and Mowbray 1970). As declaring a fish species unable to produce active sounds is challenging, the testing of multiple environments could also aid in determining the likelihood of soniferous behavior overall or at least in certain situations. The establishment of recommended fish sound production testing guidelines would further aid researchers in standardizing practices across studies and justifying the need for multiple testing environments in funding applications.

## Sound Visualizations

As the human ear may not be able to sufficiently perceive and differentiate between underwater sounds, bioacousticians commonly use visualizations to identify and compare sounds more effectively (Jones et al. 2020). There are many methods used to visualize sounds and soundscapes, but this review only focused on three: oscillograms, spectrograms, and power spectra. Oscillograms, also known as waveforms, show relative or absolute amplitude changes of sound over time. Spectrograms show frequency changes over time, with relative or absolute amplitude represented with a color or darkness scale. Power spectra, in contrast, summarize a time-varying sound by the relative or absolute amplitude of a range of frequencies. All have multiple

possible uses for fish sound production research. For example, oscillograms can be used to distinguish between complex sounds, such as pulse trains, more easily (e.g., McIver et al. 2014), spectrograms can be helpful in partitioning individual fish sounds (e.g., Carriço et al. 2020), and power spectra can be used to find acoustic metrics like peak frequency (e.g., Tu et al. 2018).

Despite the diverse uses of different methods of sound visualizations, they are not always included when discussing fish sounds in scientific publications. Among the 450 investigations reviewed for marine, subtropical soniferous fishes, 227 included at least one oscillogram, 244 included at least one spectrogram, 97 included at least one power spectrum, and 145 had none of the three sound visualizations. The investigations included in the review considered a total of 186 species for sound production. Of these, only 43 species had all three types of visualizations to illustrate their sounds, while 41 species had none of the three sound visualization types. Though the use of sound visualizations has increased similarly to the increase in fish sound production publications (Fig. 4a), the relative inclusion of sound visualizations in investigations per year has remained largely unchanged in recent decades,




**Fig. 4** (**a**) The number of investigations (i.e., individual auditory documentations of actively soniferous behavior by a species in a reference) per year; and (**b**) the percentage of investigations per year that included an oscillogram, spectrogram, or power spectrum

with any of the visualizations only occasionally provided in all references for a particular year (Fig. 4b). As data were not collected on whether visualizations were provided for each sound type produced by a particular species in a reference, the provisioning of sound visualizations could be even lower.

The inclusion of oscillograms, spectrograms, and power spectra in addition to representative recordings, when feasible, would facilitate knowledge sharing in scientific publications on fish sound production. For example, Fish and Mowbray 1970 provided spectrograms, oscillograms, and recordings for many of the soniferous behaviors of the fish species they studied – a lot of which are still used by researchers as a helpful identification guide and regional inventory of fish sound production to the current day (Looby et al. 2022a, b; Rountree et al. 2002). With the increasing use and availability of supplementary material, online data repositories, and other electronic options for figures and other files associated with publications, any feasibility barriers to including representative sound visualizations and recordings to accompany studies of fish sound production are substantially reduced. Researchers describing soniferous fish behavior should therefore be encouraged to include recordings and multiple types of sound visualizations to improve the applicability and understanding of their work.

## Concurrence with Previous Work

This publication is not the first to recognize these or other challenges related to fish sound production research. For example, Rountree et al. (2019) conducted a similar review of freshwater, North American fish sound production research that included 84 studies on 87 fish species, which highlighted the need for standardized sound nomenclature. They also found that of the species included in their review, 34% lacked any statistical description of sound characteristics, which are similarly crucial as visualizations for identifying and describing fish sounds. The comparable findings of research trends in both freshwater (Rountree et al. 2019) and marine systems (herein) highlight systemic gaps in research priorities, methodologies, and reporting patterns. Other works discuss issues that are not covered by this review, such as publications calling for the continued development of standardized acoustical analysis protocols (e.g., Gibb et al. 2019), comprehensive recording libraries (e.g., Parsons et al. 2022), or the consideration of particle motion in addition to acoustic pressure (e.g., Nedelec et al. 2016). There have also been innovative solutions proposed to some of the challenges in fish sound production research, such as synchronous video and audio recordings to validate sound producer identities in the wild (Lobel 1992; Mouy et al. 2018), ecosystem-level ocean observatories to integrate acoustic and other remote sensing technologies (Rountree et al. 2020a), or the establishment of online data and recording repositories (e.g., Looby et al. 2022b). This review of select challenges will hopefully contribute to the wider discourse on how to continue improving research into fish sound production.

## Recommendations for Future Work

1. Wide-reaching research surveys (e.g., Fish and Mowbray 1970) and community science initiatives (e.g., FrogID 2022) are needed to examine greater numbers of possibly soniferous fish species. As more data on soniferous species are collected and synthesized, then meta-analyses, ancestral state reconstructions, and other reviews and analyses can more effectively describe patterns and the scale of fish sound production globally (e.g., Looby et al. 2022a; Parmentier et al. 2021; Rice et al. 2022).
2. Standardized sound descriptors (e.g., Jones et al. 2020) should be developed and applied by the research community to promote effective communication and comparisons among fish sounds. Such a system would hopefully be able to provide technical information about the sound, such as its duration and frequency, while still being accessible to the wider public.
3. When practicable, fish species should be examined for sound production in multiple types of testing environments (e.g., both in the lab and in the field) and other conditions (e.g., at varying times of day) to ensure comprehensive documentation and understanding of their soniferous behavior (e.g., Bertucci et al. 2021).
4. Recordings and multiple types of sound visualizations should be provided in the main-text, supplementary files, and/or supporting data repositories when describing fish sound production in scientific publications (e.g., Fish and Mowbray 1970) to better facilitate the ability of researchers to understand and learn from each other's work.

## Conclusion

Contemporary scientific research into fish sounds, their ecological uses, and potential applications with PAM has been ongoing for almost 150 years, with thousands of fish species thought to use active and passive sounds for signaling and communication (Kasumyan 2008; Looby et al. 2022a; Rice et al. 2022). The systematized review data compiled here represent four general areas to ameliorate the efficiency and consistency of research into fish sound production. While the rate of studies on fish sound production has been increasing to the present day, the number of soniferous species documented per reference has decreased and the current rate of soniferous species discovery remains insufficient to describe the thousands of soniferous species thought to exist (Fig. 1). There are dozens of sound descriptors applied inconsistently for describing sound production (Fig. 2), generating confusion in the literature and limiting syntheses that could document trends in sound production across fish species. Fishes are not regularly tested in multiple environments (Fig. 3) despite widespread acknowledgment of the influence testing environment may have on sound production and behavior. Additionally, sound visualizations that are helpful for identifying and

describing sounds are variably provided (Fig. 4) despite their differing utilities. Advancements in these areas would greatly improve the ability to document, describe, and synthesize fish sounds for a wider variety of uses and studies.

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# Sounds of Walleye Pollock: A Quantitative Description

10

Amalis Riera, Rodney Rountree, and Francis Juanes

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**Abstract**

Rapid changes in the global distribution of walleye pollock (*Gadus chalcogrammus*) due to climate change have raised concerns of potential negative interactions with other Arctic and North Pacific fishes, particularly with the Arctic cod (*Boreogadus saida*). Since both walleye pollock and Arctic cod are known to be soniferous, passive acoustic monitoring (PAM) surveys can be used to document changes in the temporal and spatial distributions of the two species. However, PAM requires a detailed knowledge of the sound characteristics of each species. Therefore, this chapter presents the first quantitative description of walleye pollock sounds to enable future PAM surveys. Over 6000 sounds attributed to walleye pollock were detected in 69 h of recordings in April 2018 at the Hatfield Marine Science Center (Oregon). Walleye pollock sounds averaged 95 ms in duration, with a peak frequency

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of 83 Hz ($n = 635$). Sounds were classified into four types of grunts, based on frequency structure: broadband, low-frequency, mid-frequency, and two-frequency. A comparison with published data for Arctic cod suggests that the two species can be distinguished by the sound duration, peak frequency, and pulse rate of their calls.

**Keywords**

Sound · Fish behavior · Walleye pollock · Passive acoustics · Arctic ecosystem

## Introduction

Walleye pollock (*Gadus chalcogrammus*) is a widespread bentho-pelagic fish species that ranges from the coasts of Korea, Japan, and California in the North Pacific to Norway in the North Atlantic (Orlov et al. 2021). Walleye pollock is the second most extensively fished species in the world, with global annual catches over three million tons between 2015 and 2019 (FAO 2021). In addition, it plays a key role in the ecosystem as a prey for numerous marine mammals and birds and as a predator of zooplankton and fish (Springer 1992; Gaichas and Francis 2008; Hirota et al. 2014; Park et al. 2018).

Walleye pollock health and survival is tightly linked to zooplankton productivity, which is impacted by fluctuations in temperature (Kimmel et al. 2018). Rising ocean temperatures have caused a shift northward of walleye pollock distribution in the Northeast Pacific (Kotwicki et al. 2005; O'Leary et al. 2020), and this could in turn displace other fish populations (Fossheim et al. 2015; Stevenson and Lauth 2019), including Arctic cod (*Boreogadus saida*) (Koenker et al. 2018; Steiner et al. 2019). This concern warrants further monitoring of both species.

Methods to survey walleye pollock distribution include trawling and active acoustics (Honkalehto et al. 2011), which can be invasive. For example, the use of echosounders can elicit behavioral responses in cetaceans (e.g., Cholewiak et al. 2017; Burnham et al. 2022) and some fishes (e.g., Mann et al. 2005), while trawling physically disturbs seabed habitats and can cause declines in organism abundance and biomass (e.g., Pitcher et al. 2022). Here, the use of passive acoustic monitoring (PAM) is proposed as a non-invasive way to monitor walleye pollock populations. Sounds produced by Arctic cod have already been described (Riera et al. 2018), but very little is known about walleye pollock sounds (Park et al. 1994). Park et al. (1994) described soniferous behavior during reproduction but provide limited quantification of sound acoustic properties. They later described the sonic muscles and sound production mechanism of the species (Park et al. 1995). Herein, the sounds produced by walleye pollock are quantitatively described, so that they can be identified in PAM recordings and distinguished from the sounds produced by other gadids that co-occur in the same habitat.

## Methods

### Data Collection

Passive acoustic recordings were collected in April 2018 in a tank at the Hatfield Marine Science Center (Newport, OR) using a SoundTrap ST300 STD (Standard, Ocean Instruments, Auckland, New Zealand) on a continuous duty cycle at 96 kHz (16 bit). The circular tank was 4.5 m in diameter, 1.3 m high, and contained approximately 30 walleye pollock of mixed sexes, including some gravid females. These fish were wild caught by beach seine as age 0 in 2016. Water temperatures were kept between 8.0 °C and 8.4 °C. Salinity remained between ~30 and 32 PPT by pumping water at a high tide from Yaquina Bay into a large reservoir upstream of the holding tanks. The recorder was deployed at 2 pm on April 16 and recovered at 11 am on April 19, which yielded 69 h of acoustic data (almost three full 24-h periods).

### Data Post-Processing

The recordings were amplified by adding 30 dB. They were then analyzed with Raven Pro 1.5 acoustic software (Center for Conservation Bioacoustics 2014) by aurally and visually inspecting each spectrogram (10,500 FFT, Hann window, 85% overlap). Several parameters of frequency, duration, and energy were measured for sounds with good signal-to-noise ratios. Percentiles of each measurement are reported as they are more robust against measurement bias (Charif et al. 2010). The frequency measurements included the lowest, highest, peak, and center frequencies, the 5th (F. 5%) and 95th (F. 95%) percentile frequencies, and the first (Q1) and third (Q3) frequency quartiles (Charif et al. 2010).

The peak frequency is the lowest frequency at which maximum power occurs within the signal. The center frequency divides the sound into two frequency intervals of equal energy. F. 5% is the frequency that divides the selection into two frequency intervals containing 5% of the energy at the bottom and 95% of the energy at the top, while F. 95% is the frequency that divides the signal into two frequency intervals containing 95% of the energy at the bottom and 5% of the energy at the top. Q1 is the frequency that divides the sound into two frequency intervals containing 25% and 75% of the energy, while Q3 divides the sound between 75% and 25%. The interquartile (IQR) bandwidth is the difference between Q3 and Q1, while the 90% bandwidth is the difference between F. 5% and F. 95% frequencies.

Other measurements include energy, aggregate, and average entropy (see Charif et al. 2010, for calculation details). Energy is the total energy in the sound (within frequency bounds). Aggregate entropy is a measure of the disorder in a sound based on the distribution of energy throughout the sound. Higher entropy values correspond to greater disorder in the sound, whereas a pure tone with energy in only one frequency would have zero entropy. Average entropy is an average of the entropy calculated for each time interval within the sound.

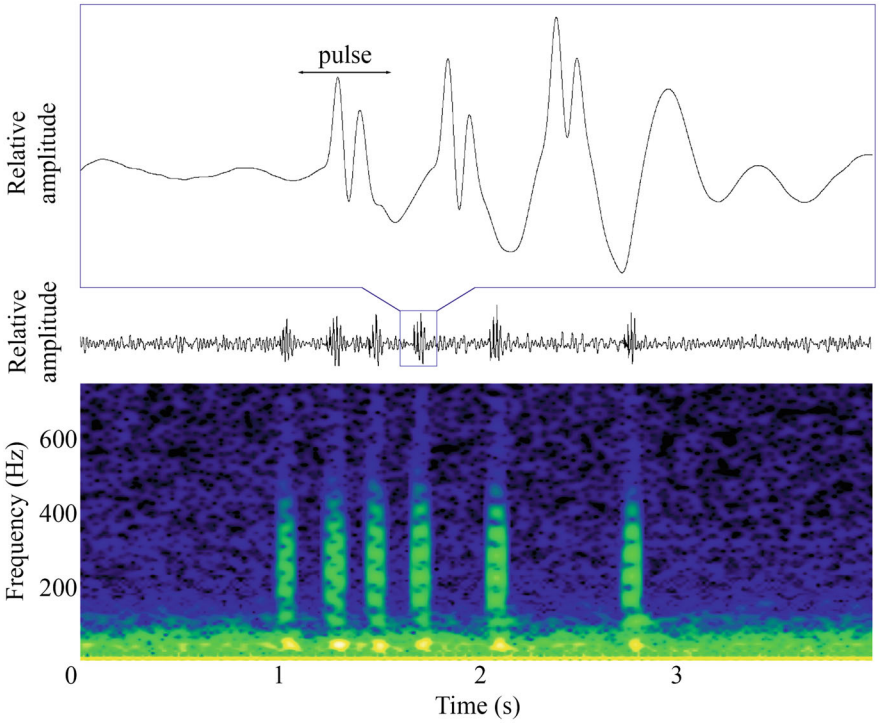



**Fig. 1** Waveform (top) and spectrogram (bottom) of a broadband grunt train produced by walleye pollock (*Gadus chalcogrammus*) recorded at the Hatfield Marine Science Center (10,500 FFT Hann window with 85% overlap). The top panel is an expansion of the fourth grunt in the middle panel, delineated with a box, to show the structure of each pulse

Given their similarity with Arctic cod vocalizations (Riera et al. 2018), walleye pollock sounds were referred to as grunts. Data were analyzed twice: first to count the sounds and then to select the best examples for detailed measurement.

A bandpass filter corresponding to the specific frequency range of each grunt was applied in order to obtain the number of pulses per grunt. As defined in Riera et al. (2018), a pulse is the unit that gets repeated in the amplitude-modulated oscillations that make out a grunt (see Fig. 1).

## Statistical Analysis

Canonical discrimination analysis (CDA) and analysis of variance (ANOVA) were used to test for multivariate and univariate differences in acoustic parameters among the four most common sound types. A stepwise discriminant analysis was used to reduce the number of input variables to CDA (Pielou 1984; Harris 2001; SAS Institute, Inc. 2012). The CDA tests whether the sound types exhibit significant

multivariate differences and, if so, which acoustic parameters contribute the most to the observed differences. All input variables were log transformed prior to analyses.

## Results

Walleye pollock were found to produce sounds as single grunts or trains of grunts. Although not quantified, sounds were produced much more frequently at night than during the day. A total of 6173 individual grunts and trains of grunts were found in the 69 h of recordings. Of these, 635 grunts were selected for detailed measurements from 168 grunt trains (Table 1).

The duration of walleye pollock grunts ranged between 29 and 340 ms (average of 95 ± 1.5 SE), and their frequency ranged from 14 to 838 Hz, with an average peak frequency of 83 ± 1.6 (Table 1). Trains averaged 4 grunts and 1.1 s duration (range 1–41 grunts, Table 1). Pulse structure was measured in 584 grunts, with an average of 5 pulses/grunt (range 2–20) and an average pulse rate of 57 pulses/s (Table 1).

Four types of walleye pollock grunts were distinguished based on frequency structure, which were named broadband ($n$ = 451), low-frequency ($n$ = 15), mid-frequency ($n$ = 95), and two-frequency ($n$ = 66) (Table 2). However, the disproportionate occurrence of the sound types was partially a result of selecting only the best sounds for measurement. Broadband grunts showed one strong energy band at the lower end of their frequency range and displayed harmonics up to 838 Hz (average bandwidth 558 ± 4.4 Hz; Fig. 1; Table 2). Low-frequency grunts presented

**Table 1** Acoustic variables for walleye pollock (*Gadus chalcogrammus*) and Arctic cod (*Boreogadus saida*) grunts recorded at the Hatfield Marine Science Center. *SE* standard error of the mean, *Min* minimum, *Max* maximum. Statistics are provided for each species, with all types of walleye pollock grunts pooled together. Arctic cod measurements were extracted from Riera et al. (2018). Arctic cod grunts did not appear to be grouped in trains

| | Walleye pollock (pooled) | | | Arctic cod | | |
|---|---|---|---|---|---|---|
| Acoustic variables | Min | Max | Mean ± SE | Min | Max | Mean ± SE |
| **Single sounds** | | | | | | |
| Grunt peak frequency (Hz) | 23 | 275 | 83 ± 1.6 | 59 | 234 | 107 ± 4 |
| Grunt low frequency (Hz) | 14 | 140 | 38 ± 0.5 | 36 | 157 | 79 ± 3 |
| Grunt high frequency (Hz) | 142 | 838 | 497 ± 6.8 | 184 | 851 | 400 ± 22 |
| Grunt duration (ms) | 29 | 340 | 95 ± 1.5 | 184 | 536 | 289 ± 13 |
| Sample size | 635 | 635 | 635 | 38 | 38 | 38 |
| **Pulse structure** | | | | | | |
| Number of pulses per grunt | 2 | 20 | 5 ± 0.1 | 6 | 12 | 9 ± 1 |
| Pulse rate (pulses/s) | 26 | 106 | 57 ± 0.4 | 14 | 64 | 30 ± 8 |
| Sample size | 584 | 584 | 584 | 6 | 6 | 6 |
| **Sound trains** | | | | | | |
| Number of grunts per train | 1 | 41 | 4 ± 0.3 | N/A | N/A | N/A |
| Train duration (ms) | 50 | 12,488 | 1093 ± 123.8 | N/A | N/A | N/A |
| Sample size | 168 | 168 | 168 | N/A | N/A | N/A |

**Table 2** Acoustic variables for four types of walleye pollock (*Gadus chalcogrammus*) grunts recorded at the Hatfield Marine Science Center. Statistics are provided for each grunt type: broadband, low-frequency, mid-frequency, and two-frequency. *SE* standard error of the mean, *Min* minimum, *Max* maximum. Variables were measured according to Charif et al. (2010) and are defined in the text

| | Broadband | | Low-frequency | | Mid-frequency | | Two-frequency | |
|---|---|---|---|---|---|---|---|---|
| Acoustic variables | Min-Max | Mean ± SE | Min-Max | Mean ± SE | Min-Max | Mean ± SE | Min-Max | Mean ± SE |
| **Single sounds** | | | | | | | | |
| *Frequency (Hz)* | | | | | | | | |
| Low | 14–140 | 35 ± 0.4 | 25–52 | 36 ± 1.5 | 22–96 | 52 ± 2.3 | 20–49 | 36 ± 0.9 |
| Fifth percentile | 23–170 | 46 ± 0.5 | 41–53 | 48 ± 1.0 | 23–111 | 71 ± 2.8 | 23–64 | 54 ± 0.8 |
| First quartile | 35–188 | 63 ± 1.0 | 53–59 | 56 ± 0.8 | 47–123 | 99 ± 2.3 | 41–123 | 70 ± 1.7 |
| Center | 53–246 | 91 ± 2.1 | 59–106 | 64 ± 3.1 | 70–129 | 118 ± 0.7 | 53–135 | 95 ± 3.1 |
| Peak | 23–275 | 74 ± 1.9 | 59–64 | 60 ± 0.6 | 53–135 | 121 ± 1.0 | 41–141 | 94 ± 3.9 |
| Third quartile | 53–299 | 141 ± 3.0 | 64–117 | 71 ± 3.4 | 111–141 | 126 ± 0.5 | 76–146 | 121 ± 2.0 |
| 95th percentile | 70–363 | 239 ± 3.2 | 70–135 | 99 ± 6.6 | 123–176 | 138 ± 1.0 | 123–217 | 144 ± 2.3 |
| High | 276–838 | 592 ± 4.3 | 142–421 | 232 ± 23.0 | 160–507 | 238 ± 5.6 | 200–462 | 301 ± 7.8 |
| Interquartile bandwidth | 6–223 | 78 ± 2.6 | 6–59 | 15 ± 3.2 | 6–88 | 27 ± 2.3 | 12–76 | 51 ± 2.0 |
| 90% bandwidth | 23–322 | 193 ± 3.2 | 23–82 | 51 ± 6.3 | 18–129 | 66 ± 3.2 | 64–158 | 90 ± 2.4 |
| Bandwidth | 242–812 | 558 ± 4.4 | 107–388 | 196 ± 22.7 | 67–477 | 185 ± 7.0 | 160–435 | 265 ± 7.5 |
| Sample size | 451 | | 15 | | 95 | | 66 | |
| *Duration and energy* | | | | | | | | |
| Duration (ms) | 29–340 | 84 ± 1.5 | 66–267 | 122 ± 14.3 | 57–232 | 114 ± 3.7 | 77–243 | 130 ± 3.4 |
| Aggregate entropy (bits) | 2.6–5.7 | 4.2 ± 0.3 | 2.5–3.4 | 2.9 ± 0.1 | 2.2–4.3 | 3.2 ± 0.1 | 3.2–4.4 | 3.7 ± 0.0 |
| Average entropy (bits) | 2.6–5.5 | 4 ± 0.3 | 2.5–3.2 | 2.8 ± 0.1 | 2.2–3.9 | 3.1 ± 0.0 | 3.1–4.2 | 3.6 ± 0.0 |
| Energy (dB) | 81–110 | 97 ± 0.2 | 94–105 | 99 ± 0.7 | 83–104 | 97 ± 0.4 | 93–112 | 103 ± 0.5 |
| Sample size | 451 | | 15 | | 95 | | 66 | |

| *Pulse structure* | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Number of pulses | 2–20 | 4.7 ± 0.1 | 4–16 | 7.2 ± 0.8 | 3–12 | 6.5 ± 0.2 | 5–12 | 7.3 ± 0.2 |
| Pulse rate (pulses/s) | 26–106 | 57 ± 0.5 | 52–64 | 60 ± 0.7 | 41–65 | 56 ± 0.8 | 29–71 | 57 ± 1.1 |
| Pulse period (ms/pulse) | 9–38 | 18 ± 0.2 | 16–19 | 17 ± 0.2 | 15–24 | 18 ± 0.3 | 14–35 | 18 ± 0.4 |
| Sample size | 446 | | 14 | | 55 | | 61 | |
| **Sound trains** | | | | | | | | |
| Duration (ms) | 50–5724 | 878 ± 87 | 124–1379 | 730 ± 363 | 114–11,331 | 1914 ± 665 | 791–12,488 | 3591 ± 1518 |
| Number grunts per train | 1–12 | 3.2 ± 0.2 | 1–5 | 2.7 ± 1.2 | 1–27 | 5.2 ± 1.6 | 5–41 | 11.6 ± 5.0 |
| Grunt rate (grunts/s) | 1.5–20.0 | 7.1 ± 0.4 | 2.9–8.1 | 4.9 ± 1.6 | 1.4–8.8 | 3.8 ± 0.5 | 2.0–6.3 | 3.6 ± 0.5 |
| Grunt period (ms/grunt) | 50–658 | 208 ± 10 | 124–343 | 248 ± 65 | 114–710 | 334 ± 36 | 158–509 | 317 ± 45 |
| Sample size | 141 | | 3 | | 17 | | 7 | |

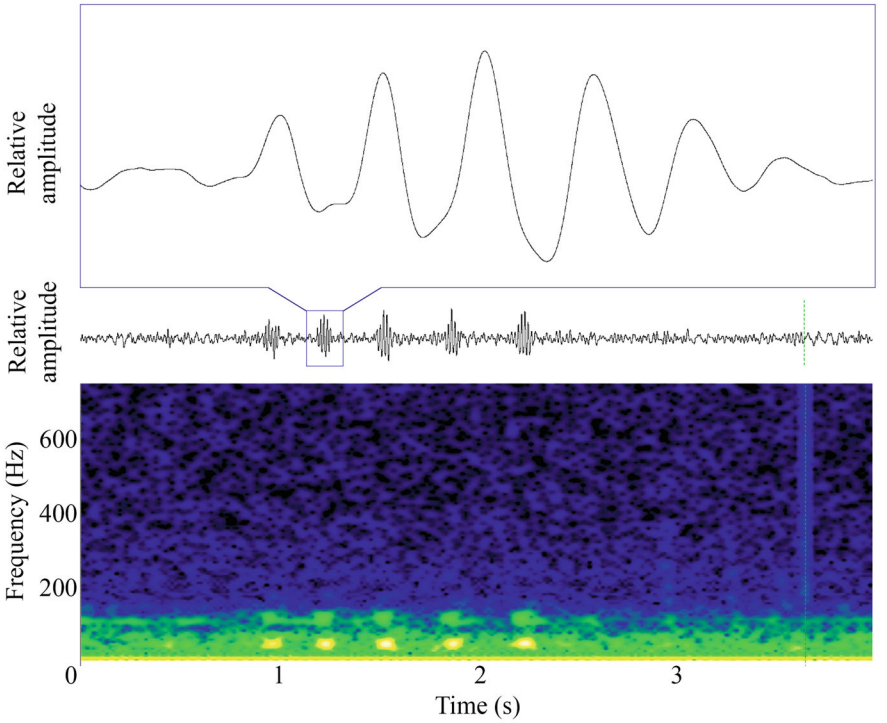



**Fig. 2** Waveform (top) and spectrogram (bottom) of a low-frequency grunt train produced by walleye pollock (*Gadus chalcogrammus*) recorded at the Hatfield Marine Science Center (10,500 FFT Hann window with 85% overlap). The top panel is an expansion of the second grunt in the middle panel, delineated with a box, to show the structure of each pulse

one strong energy band at the lower end of their frequency range, but fewer harmonics, with an average high frequency of 232 ± 23.0 Hz (Fig. 2; Table 2). Mid-frequency grunts were similar to low-frequency grunts, with an average high frequency of 238 ± 5.6 Hz, but had a wider 90% bandwidth (66 vs. 51 Hz, respectively; Fig. 3; Table 2). Two-frequency grunts looked like a combination of low- and mid-frequency grunts: with two strong energy bands and few harmonics (Fig. 4; Table 2).

Pulses tended to have a double peak structure, but differed among the four sound types. The first peak had a higher amplitude than the second peak for the broadband (Fig. 1) and mid-frequency (Fig. 3) type grunts, but the two-frequency grunts exhibited the opposite pattern (Fig. 4). Low-frequency grunts were most different, exhibiting a single peak pulse structure (Fig. 2). However, mid-frequency grunts tended to have pulses of varying structures; beginning with pulses with two peaks similar to broadband grunts, but transitioning to single peaks like the low-frequency grunts.

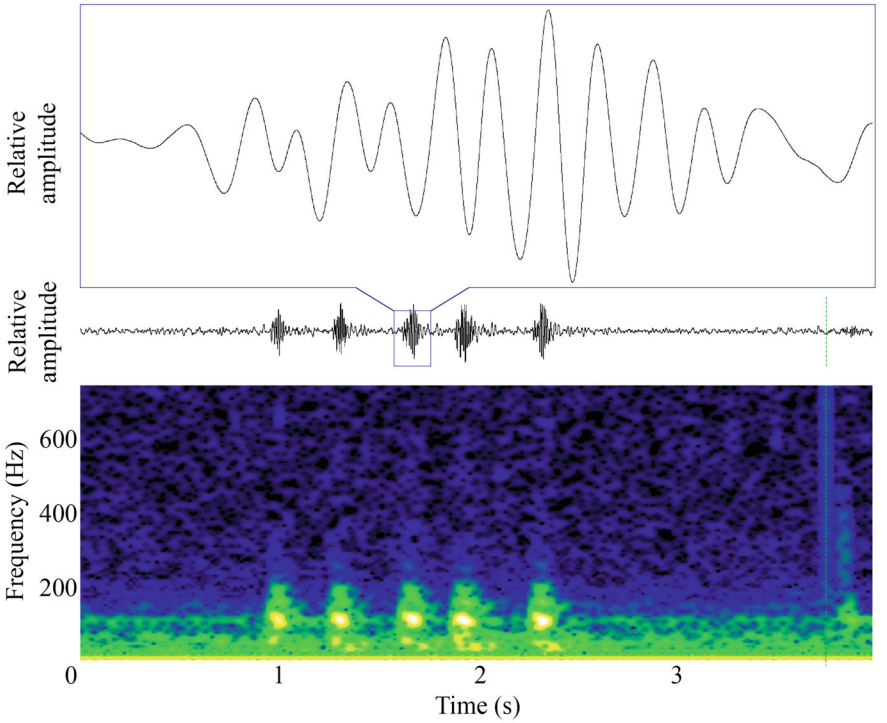



**Fig. 3** Waveform (top) and spectrogram (bottom) of a mid-frequency grunt train produced by walleye pollock (*Gadus chalcogrammus*) recorded at the Hatfield Marine Science Center (10,500 FFT Hann window with 85% overlap). The top panel is an expansion of the third grunt in the middle panel, delineated with a box, to show the structure of each pulse

Twelve variables were selected for input into the CDA analysis, which resulted in a strong discrimination among the four sound types ($p < 0.0001$; Table 3; Fig. 5). The eight variables that contributed the most to the group discrimination are shown as vectors drawn from the origin with lengths proportional to the variable's contribution to the group separation and direction indicating the variable's relative contribution to each canonical. The first two canonicals were significant ($p < 0.001$) and account for 98% of the variance. The four sound types are separated most strongly on the first canonical, which is driven largely by sound frequency composition. Sound types with lower frequencies (low-, mid-, and two-frequency types) cluster on the left, while the broadband type with higher-frequency content clusters to the right. The second canonical is driven most strongly by the sound energy content (positive) and frequency content (negative) and separates the mid- from the two-frequency sound types, with the low-frequency type intermediate. The low- and two-frequency types are most similar.

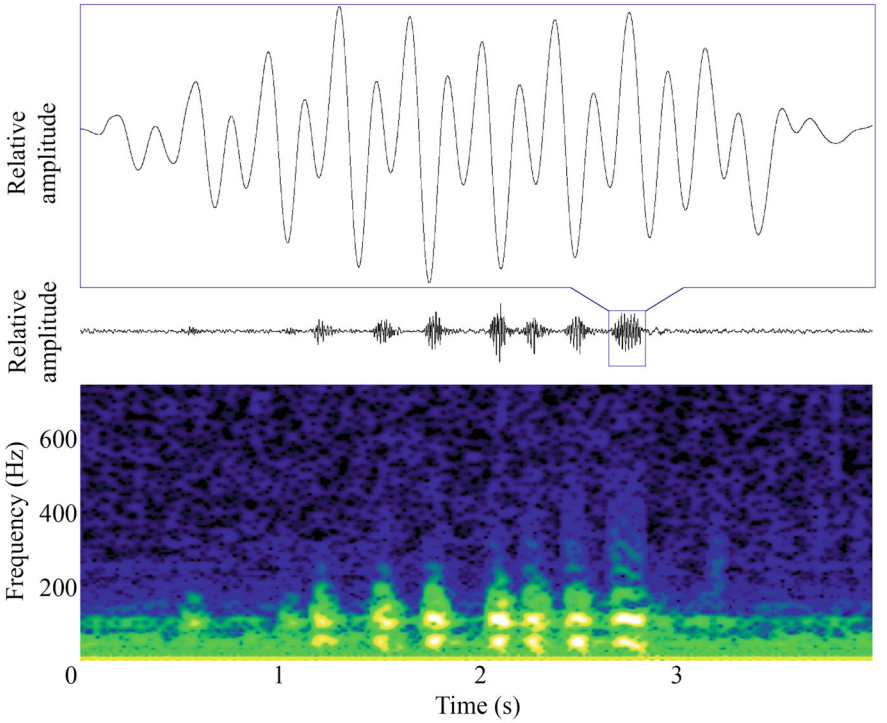



**Fig. 4** Waveform (top) and spectrogram (bottom) of a two-frequency grunt train produced by walleye pollock (*Gadus chalcogrammus*) recorded at the Hatfield Marine Science Center (10,500 FFT Hann window with 85% overlap). The top panel is an expansion of the last grunt in the middle panel, delineated with a box, to show the structure of each pulse

## Discussion

Walleye pollock produced at least four different types of sounds based on their frequency and pulse structures. Park et al. (1994) similarly reported four types of sounds, but presented only limited information on sound characteristics. They reported a high frequency of 800 Hz, similar to the 838 Hz maximum frequency reported herein. Park et al. (1994) highlighted four behavioral states in which sound production occurred: threatening, attacking, courting, and mating, each with a distinct associated sound type. Unfortunately, this study lacks visual observations of behavior, but the broadband grunts described here (Fig. 1) strongly resemble the spectrogram of aggressive-attacking calls presented in the Park et al. (1994) study. The other three types of grunts reported here had lower frequencies, similar to the courting and mating calls reported by Park et al. (1994), but more data are need to confirm the associated behaviors.

**Table 3** Canonical coefficients for all 12 variables input into a canonical discrimination analysis (CDA) among 4 types of sounds produced by walleye pollock based on transformed sound measurements (see Fig. 5). Pearson correlation coefficients (*$p < 0.05$; ***$p < 0.001$) of measurement variables with each canonical axis indicate variables that contributed the greatest to the discrimination among groups. Code = label for vector plotted in Fig. 5 indicating the relative contribution of each variable to the discrimination

| Acoustic variables | Can1 | Can2 | Code |
|---|---|---|---|
| Fifth percentile frequency | −0.53*** | −0.27*** | A |
| Peak frequency | −0.49*** | −26*** | B |
| Low frequency | −0.38*** | −40*** | C |
| Energy | −0.15*** | 0.49*** | D |
| Third quartile frequency | −0.09* | −0.14*** | E |
| Interquartile bandwidth | 0.40*** | 0.23*** | F |
| Aggregate entropy | 0.59*** | 0.17*** | G |
| Average entropy | 0.62*** | 0.14*** | H |
| 95th percentile frequency | 0.67*** | −0.15*** | I |
| 90% bandwidth | 0.75*** | 0.07 | J |
| Bandwidth | 0.96*** | 0.09* | K |
| High frequency | 0.97*** | −0.01 | L |
| Canonical $R^2$ | 0.86 | 0.38 | |
| Proportion of variance | 90% | 9% | |
| Canonical $p$ | <0.001 | <0.001 | |
| Wilks lambda | <0.001 | | |

Walleye pollock are being found in locations further North than their previously known range (e.g., Orlov et al. 2021), which means they now share a habitat with other Arctic gadids such as Arctic cod (*Boreogadus saida*). Being able to distinguish the two species acoustically would mean that PAM surveys can be used to help monitor these ecosystem-wide shifts and potential inter-species competition for habitat. In addition to Arctic cod, there are six other species of gadids that co-occur with walleye pollock in some areas of their range, such as Pacific cod (*Gadus macrocephalus*), saffron cod (*Eleginus gracilis*), Pacific tomcod (*Microgadus proximus*), Greenland cod (*Gadus ogac*), Atlantic cod (*Gadus morhua*), and polar cod (*Arctogadus glacialis*). Of these, Arctic cod (Riera et al. 2018) and Atlantic cod (Hernandez et al. 2013) are the only species with confirmed sound production. Pacific cod is the only other species that has been examined for sound production, but it has been found neither to possess sonic muscles (Hawkins and Rasmussen 1978) nor to produce sounds (Sakurai and Hattori 1996). Hawkins and Rasmussen (1978) did not find evidence of sonic muscles for saffron cod and Pacific tomcod, suggesting they are not soniferous, but direct studies of their behavior are needed to confirm that supposition. To date, the other two species (Greenland and polar cod) with distributions overlapping with walleye pollock have not been examined for sound production (Looby et al. 2022a, b).

A comparison between the walleye pollock sounds presented here and those previously described for other gadid species suggests clear differences that can be

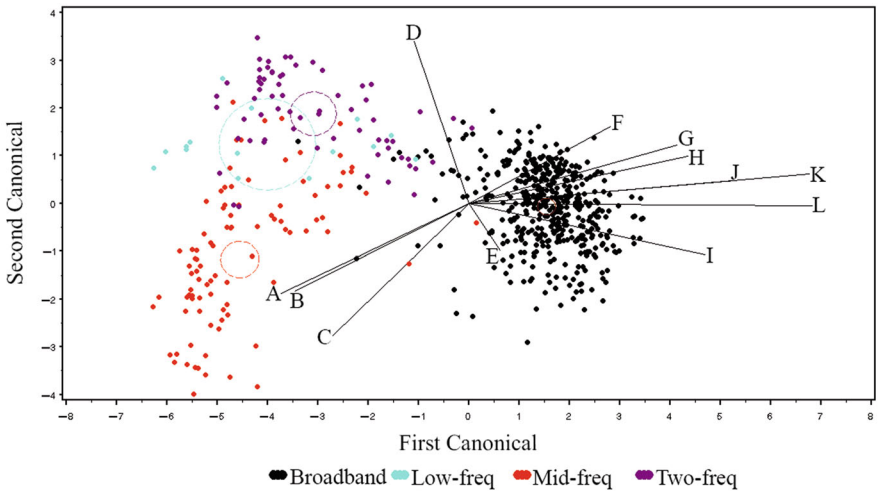



**Fig. 5** Comparison of acoustic properties of four sound types exhibited by walleye pollock: broadband (black), low-frequency (teal), mid-frequency (red), and two-frequency (purple). Canonical discrimination analysis of walleye pollock sound types for selected variables (see Table 3 for canonical statistics). Colored circles represent the 99th confidence interval for the group mean canonical. The nine variables most strongly influencing the group discrimination are shown as vectors drawn from the origin with lengths proportional to the variable's contribution to group separation and direction indicating the variable's relative contribution to each canonical. These variables are as follows: (A) fifth percentile frequency (F. 5%), (B) peak frequency, (C) low frequency, (D) energy, (E) third quartile frequency (Q3), (F) interquartile (IQR) bandwidth, (G) aggregate entropy, (H) average entropy, (I) 95th percentile frequency (F. 95%), (J) 90% bandwidth, (K) bandwidth, and (L) high frequency

used to distinguish among species where they occur together. The main difference between walleye pollock and Arctic cod grunts is the occurrence of grunt trains in the former, whereas Arctic cod are only known to produce single grunts (Fig. 6). The duration of walleye pollock grunts is shorter (average 95 ± 1.5 ms) than that of Arctic cod grunts (average 289 ± 13 ms) and is composed of fewer pulses (5 vs. 9, Table 1). The duration of grunts is still longer for Arctic cod when compared to the different types of walleye pollock grunts separately (Table 2). Although it was not possible to directly measure the fundamental frequency of either walleye pollock or Arctic cod due to high levels of low-frequency noise in the holding facilities, Arctic cod had a higher peak frequency (Table 1), but likely had a much lower fundamental frequency based on its pulse rate (30 vs. 57 pulses/s). Published data for Atlantic cod (Hernandez et al. 2013) suggest it is more similar to Arctic cod with a fundamental frequency of 50 Hz, duration of 168 ms, and average of 9 pulses/grunt.

Successful identification of underwater sounds to fish species supplies valuable proof of presence. In addition, being able to attribute specific sounds to a behavioral context can indicate the importance of a given habitat to the species (Rountree et al. 2006). This study confirms previous observations of sound production by walleye pollock during the spawning period and provides detailed data on their acoustic

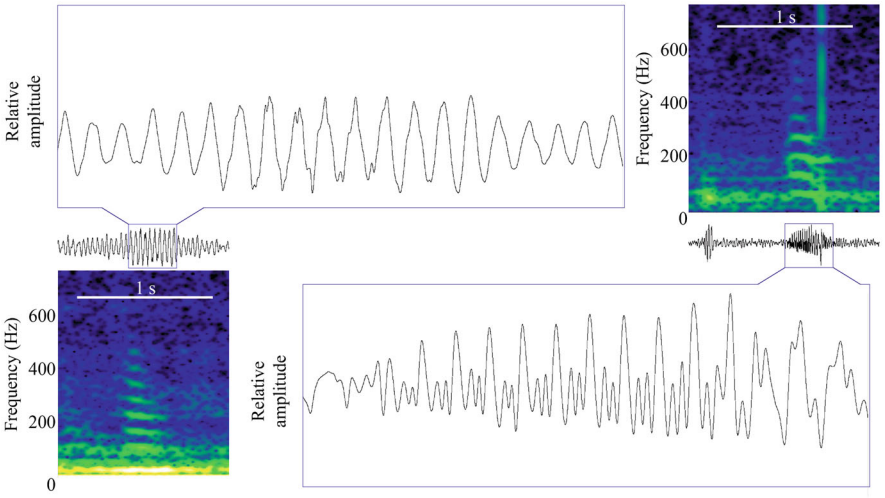



**Fig. 6** Waveform and spectrogram of Arctic cod grunts (*Boreogadus saida*) recorded at the Hatfield Marine Science Center (left) and University of British Columbia (right) (10,500 FFT Hann window with 85% overlap). The waveforms are further zoomed in to show the detail of the pulses. These grunts were quantitatively described in Riera et al. (2018)

characteristics, which will enable future PAM surveys for the species. More year-round studies will reveal whether sound production is exclusive to the reproductive period or if it also occurs at other times. Further observations with video recordings could help attribute each of the different sound types to specific behaviors. Knowing the behavioral context of different sounds could help identify important habitat for the species, for example, if reproductive grunts are reliably detected in certain locations, they might become protected at certain times of the year.

The number of existing long-term PAM datasets from the Arctic is rapidly increasing (Chou et al. 2019; Pine et al. 2020; Halliday et al. 2021), with corresponding fish sounds that have yet to be positively attributed to species. This quantitative description of walleye pollock grunts will help increase the number of fish sounds identified to species in existing recordings, as well as inform future studies with a focus on gadid presence, movement, and important habitats. However, additional studies are needed to describe sound production by other gadids in the region to ensure against false attribution of sound to either Arctic cod or walleye pollock and further enable PAM studies of interactions among all soniferous gadids in the region.

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# FishSounds Version 1.1: Data Archive, User Experience, and Online Resources

11

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**Abstract**

Fish sound production is taxonomically and geographically widespread. Produced actively or passively, acoustic contributions of fishes to aquatic soundscapes support a myriad of con- and hetero-specific interactions. Despite the ecological importance of fish sounds, the field of bioacoustics historically lacked a global inventory of fish species known to produce sound. FishSounds launched in 2021, cataloging examinations of active and passive sound production by 1185 fish species from 837 references, as well as 239 exemplary audio recordings. In April 2022, FishSounds released Version 1.1 to expand the website's data and improve its functionality. This new version updated the core dataset to include work published up until the end of the year 2021, adding 30 species, 25 recordings, and 85 references. It also added text to note the original species names used by the authors, updated all taxonomy data to FishBase version 02/2022, and included a webpage highlighting other resources that might be of interest to users. Additionally, FishSounds Educate launched a series of interactive lectures, classroom workshops, and field labs for primary, secondary, and university students. Future FishSounds versions will prioritize streamlining the data submission process, improving the user experience, and using bioacoustics to increase ocean literacy.

**Keywords**

Data portal · Bioacoustics · Soundscape ecology · Fish biology · Ocean literacy · Data archive · Passive acoustics · Education

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## Introduction

Sound is widely used for information dissemination and acquisition by aquatic taxa (Ladich 2000; Duarte et al. 2021). Acoustic signals travel faster and farther in aquatic environments than in air, often reaching greater distances than other sensory cues (Staaterman and Paris 2014; Ladich and Winkler 2017). Because of the unique benefits of sound as a sensory cue in underwater ecosystems, many fish lineages have undergone convergent evolution to increase their physiological capacity to perceive and produce sounds for communication and the assessment of habitats, resources, and threats (Ladich 2000; Kasumyan 2009). Consequently, fish sounds can be recorded passively for the purposes of ecological monitoring and fisheries management (Lindseth and Lobel 2018). Passive acoustic recording allows fish populations to be studied remotely, thereby reducing reliance on invasive sampling methods, increasing the potential spatial and temporal coverage of research efforts, and documenting novel contributions to aquatic soundscapes (Rountree et al. 2006; Rountree and Juanes 2017).

The prevalence of sound production among fish remains indeterminate despite considerable advancements in bioacoustics and global interest in how noise pollution impacts fishes (Cox et al. 2018; Duarte et al. 2021). Although there have been numerous efforts to catalogue soniferous fish species, these endeavors have been constrained spatially (Fish and Mowbray 1970; Rountree et al. 2019) or taxonomically (Colleye et al. 2011). Furthermore, currently published reviews synthesizing information on sound production in fishes lack explicit data collection methodology and reporting (Kaatz et al. 2017) or have primarily qualitative descriptions (Kasumyan 2009). Consequently, bioacousticians have identified the need to determine the prevalence of soniferous fishes and develop a comprehensive and updateable list of fish species that have been examined for sound production (Gannon 2008; Linke et al. 2018; Lindseth and Lobel 2018; Rountree et al. 2019).

To rectify this widely recognized constraint, the FishSounds project was launched (Looby et al. 2021, 2022b, 2023). To do so, it established an international collaboration between researchers and website developers from six organizations – the University of Victoria, the University of Florida, Dalhousie University, Simon Fraser University, Universidade de São Paulo, and the Marine Environmental Research Infrastructure for Data Integration and Application Network (MERIDIAN). This collaboration applied a cross-disciplinary approach, combining expertise in fish ecology, bioacoustics, and data management to the consolidation, management, and application of fish acoustics. The FishSounds project focuses on three primary objectives: (1) conduct a global, systematized review to catalogue the fish species that have been examined for sound production, (2) construct a comprehensive web portal amalgamating acoustic recordings and published information on fish sound production, which allows researchers to contribute observations and search archived examinations by taxa, sound characteristics, or reference, and (3) disseminate this information to acousticians, conservation managers, and the general public. This conference proceeding outlines the progress on these objectives, focusing on the release of FishSounds Version 1.1 as part of ongoing efforts to expand the website's

data and improve its functionality and launching FishSounds Educate as an avenue for increasing ocean literacy.

## FishSounds Version 1.1

The release of FishSounds Version 1.1 on 28 April 2022 included several extensive data and recording updates, improvements to the user experience, and additions to the web portal (Fig. 1). These updates are summarized below. Additional information on each development can also be found on the FishSounds.net website, as well as highlighted in tweets from the Twitter account (@FishSoundsWeb). When applicable, any changes made to the website are reflected in the Borealis data repository, which is versioned and archived (Looby et al. 2021).

### Data and Recording Updates

The most significant change to the website featured an update of the core dataset on literature documenting known soniferous fish species. Following the methods in Looby et al. (2022a), the updated FishSounds dataset includes any references published by the end of December 2021. This update surveyed 625 new references for potential relevance. From these references, 85 examined a fish species for sound

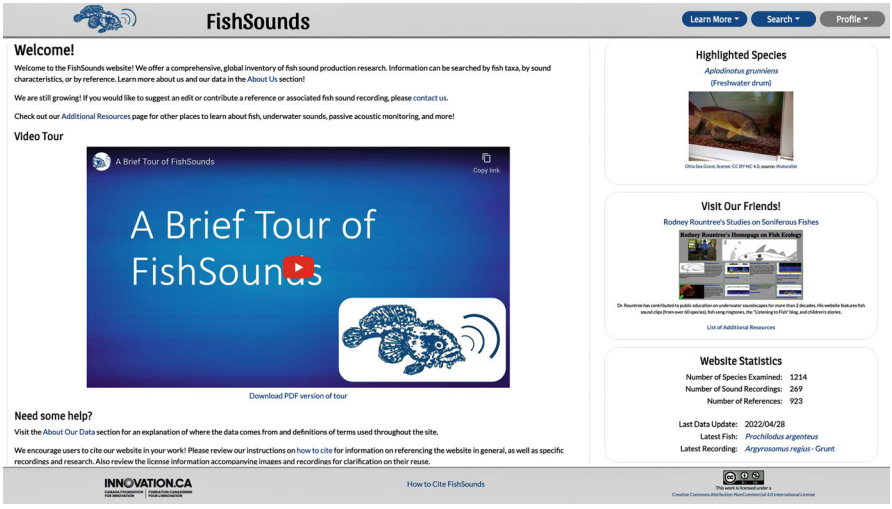



**Fig. 1** FishSounds Version 1.1 home page. The latest version of FishSounds includes the addition of 30 species, 25 recordings and 85 references, updated taxonomic information, and a web page highlighting other resources that might be of interest to users. As a result of this update, FishSounds now catalogues records for 1214 species examined across 923 references and allows users to search archived examinations by taxa, sound characteristics, or reference

production, 30 of which were new species not yet included on the website. Several of the new literature additions included work that had been published in the year since the initial Looby et al. (2021) review was completed (e.g., Parmentier et al. 2022) as well as studies that the original review had missed (e.g., Lowe and Skelton 2008). With the inclusion of these additional examinations, the website now catalogues records for 1214 species examined across 923 references, which were identified by a literature review that now spans over 3000 references.

In addition to updating the soniferous fish diversity data, 25 new recordings were added to Version 1.1 of the FishSounds website. These included several additional recordings linked to the SOUNDS table on FishBase (Kaschner 2012; Froese and Pauly 2022). Contributors generously donated new recordings, such as clips of grunt and pulse sounds produced by the meagre (*Argyrosomus regius*) that were recorded as part of Viera et al. (2019) (Fig. 2). FishSounds now provides a total of 269 recordings, making it one of the largest global repositories for fish sounds that currently exists.

The fish species information listed on FishSounds was updated as part of Version 1.1. FishSounds provides general information on fish species, such as taxonomy or distribution, taken from FishBase using the R package rfishbase (Boettiger et al. 2012; Froese and Pauly 2022). The latest update included data from the FishBase data version released in February 2022, which incorporated information from Nelson et al. (2016), among other taxonomic authorities. Changes to the FishBase data also included recognizing eight taxonomic classes instead of six, 600 families instead of 590, and well-defined suborders of Perciformes at uplifted rank orders (Bailly 2021). Numerous alterations to groupings, species, and scientific names were also included

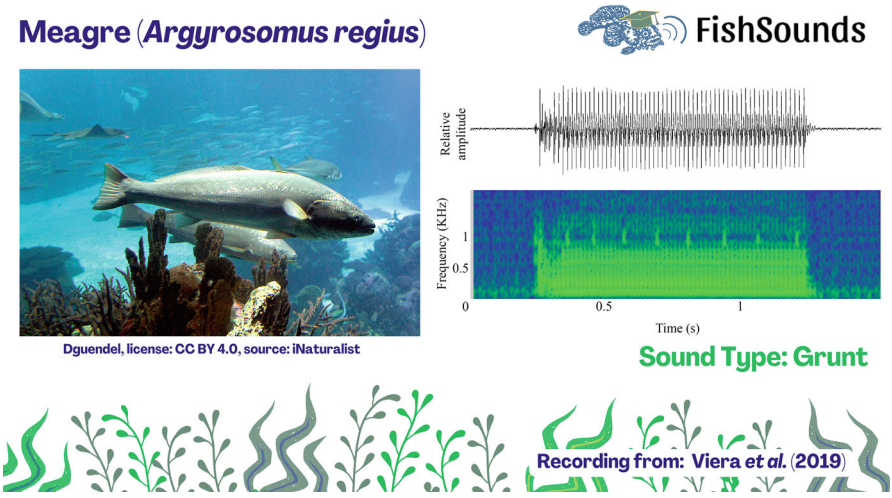



**Fig. 2** An image of the meagre (*Argyrosomus regius*) alongside a waveform and spectrogram of a representative example of its grunt sound as displayed on FishSounds (Looby et al. 2022b). Waveform and spectrogram created with permission from Viera et al. (2019). (Fish image used with permission from Dguendel (2022), license: CC BY 4.0, source iNaturalist)

in this FishBase update (Bailly 2021). Since the release of FishSounds Version 1.1, FishBase has changed to a June 2022 version, which will be integrated into FishSounds in its next data update. Incorporating the latest versions of FishBase data will ensure that FishSounds more accurately reflects contemporary thinking on fish nomenclature and taxonomy.

## Improvements to User Experience

FishSounds Version 1.1 added information to improve the user experience and website functionality. The most substantial addition included in this update was the provision of the species names originally used by the authors for the sound production examinations on the website. This will help users locate information within references for species names that have changed since their publication. For example, the scrawled cowfish was listed with the scientific name *Lactophrys quadricornis* in Fish and Mowbray (1970) but is currently listed on FishSounds as *Acanthostracion quadricornis* based on the latest FishBase classification (Froese and Pauly 2022); users may have previously had a hard time locating information on the species from its original reference because of the change. With the FishSounds Version 1.1, nomenclature previously used and currently accepted is now listed on the associated Reference Details page. The inclusion of changes in taxonomic classification will also improve the long-term sustainability of the FishSounds data, allowing it to adapt to ever-evolving fish nomenclature and making users aware of any reclassifications.

Editor's notes have also been added to the FishSounds website. Each note is indicated with a blue "" symbol and can be used by the FishSounds editors to denote potential contradictions in the literature or other notes of potential interest. For example, there is a so-called "chatter" sound that was misattributed in Fish and Mowbray (1970) to be produced by *Cynoscion regalis* (Sprague and Luczkovich 2001). Therefore, next to the Fish and Mowbray (1970) reference on the *C. regalis* species page, there is an editorial note that reads: "The chatter sound reported to be produced by this species was later shown to be produced by *Ophidion marginatum*." These editor notes will detail contradictions or corrections from the literature with more nuance than was possible during the Looby et al. (2022b) data collection.

FishSounds Version 1.1 also includes several additional features to increase users' awareness and use of external resources, web pages, and other bioacoustics repositories. All DOI numbers listed on the website, including those associated with each reference, are now hyperlinked to the source (e.g., published article). The inclusion of hyperlinks will more effectively encourage users to navigate from FishSounds to the original data sources. An "Additional Resources" page was also added to help users explore other related websites that may be of interest, such as the Ocean Networks SoundCloud or the Macaulay Library at the Cornell Lab of Ornithology, which also host impressive repositories of fish sounds and other recordings. A panel was added to the FishSounds homepage to further highlight

websites from the Additional Resources page (Fig. 1). These improvements will foster a sense of community among similar acoustics and fish websites while encouraging users to explore original research and related projects.

## Additional Changes

The FishSounds Version 1.1. update implemented other website alterations to ensure cohesion across the associated FishSounds datasets and increase data transparency. Several of the terms used to describe aspects of the references were altered during this process. For example, the term "physiological examination" was switched to "morphophysiological examination" to more accurately describe studies examining the morphological (i.e., relating to structures) and/or the physiological (i.e., relating to functions) aspects of fish sound production. Distinguishing morphophysiological examinations is also consistent with Looby et al. (2022b), which is the foundation of the FishSounds website. Other terms had their definitions clarified in the Website Definitions provided on the About page, such as the "*** frequency" term used in the acoustic measurements tables to describe the frequency that divides the spectral content above and below the percentage or quartile reported in the measurement, represented by the three asterisks. All data definitions are listed in a searchable table that allows users to easily navigate the myriad of definitions used in bioacoustics and on the website. Future FishSounds versions will continue to revise terminology based on user feedback. Other changes were made throughout the website as needed. These included listing additional citations for recording credits, adding new fish images that had been donated, and updating the website acknowledgments to include new recording contributors and research assistants, whose support is greatly appreciated. There is also an updated list of presentations and publications that have occurred since the initial website launch (e.g., Parsons et al. 2022; Looby et al. 2022a, 2023). Finally, the citation to be used when crediting FishSounds has been updated to 2022 instead of 2021 and Version 1.1 instead of 1.0.

## User Engagement with FishSounds

FishSounds has had success in garnering users and engagement. On 28 October 2022, FishSounds celebrated its 1-year anniversary (Fig. 3). Since launching, FishSounds has had over 13,000 users and 64,000 page views, with its highest user counts from the USA, Canada, and then Europe (Google Analytics 2022). The most viewed fish species on the website were the Bocon toadfish (*Amphichthys cryptocentrus*) and haddock (*Melanogrammus aeglefinus*) – fitting as the FishSounds logo is a toadfish. The FishSounds Twitter account (@FishSoundsWeb), which regularly highlights species and references from the website and shares other relevant news and papers (e.g., Fig. 4), achieved 690 followers and over 100,000 tweet impressions by December 2022 (Fig. 3).

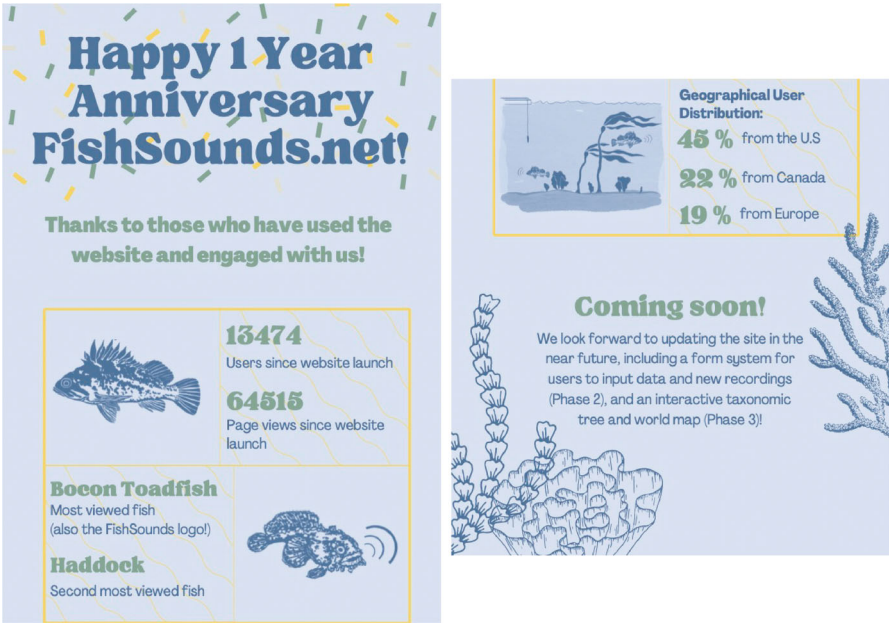



**Fig. 3** Image shared in celebration of the 1-year anniversary of FishSounds on the FishSounds Twitter account (@FishSoundsWeb) highlighting some of the website engagement metrics and future plans

## Upcoming Website Versions

Future versions of the FishSounds web portal have been scheduled for 2023. Led by Sarah Vela, these updates will continue to expand the functionality of the web portal and improve the user experience. Planned additions to FishSounds include the development of a form system for users to help contribute to FishSounds directly, an interactive map of fish passive and active sound production, and additional data visualizations. Future updates will also increase the number of species, recordings, and references in the core dataset to include work published in 2022. These data updates will follow the methods developed by Looby et al. (2022a). Version updates scheduled for the latter portion of 2023 and early 2024 will improve the extent to which users can search and download the FishSounds data and integrate a taxonomic tree illustrating passive and active sound production by fish. Future updates are the direct result of community input and aim to address the ever-expanding needs of bioacousticians, marine ecologists, and members of the public interested in fish sounds. Therefore, FishSounds is keen for feedback from the users of the web portal on how future updates can better meet the needs of the community that makes this work possible.

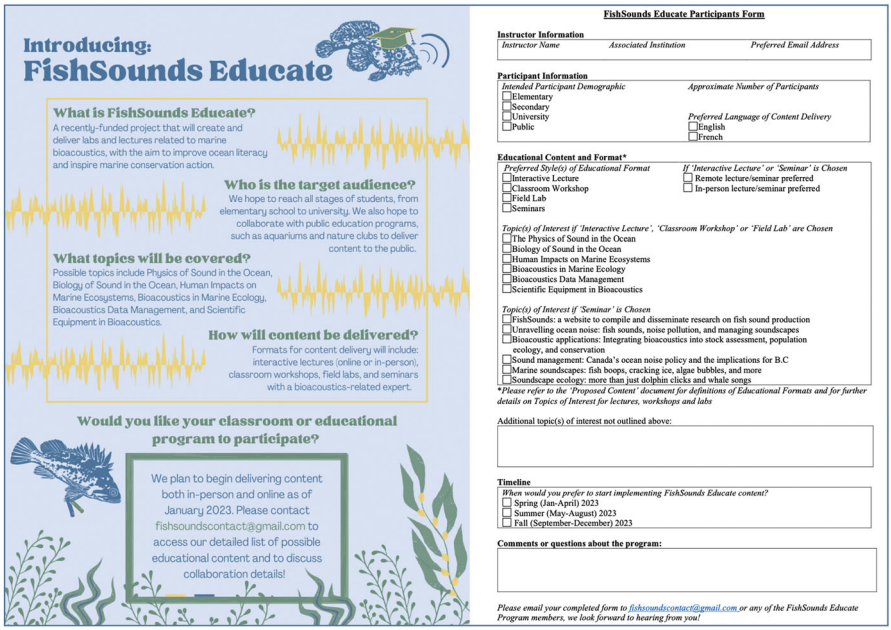



**Fig. 4** FishSounds Educate launch announcement and participation form. To increase ocean literacy in Canada, FishSounds Educate is developing and implementing interactive lectures, classroom workrooms, and field labs for primary, secondary, and university students, as well as seminars for members of the public. Instructors or institutions interested in integrating bioacoustics into their lesson plans should contact the FishSounds team

## FishSounds Educate

FishSounds Educate launched in September 2022 with the aim of increasing awareness of one of the most vital but understudied attributes of marine ecosystems – sound. This ambitious aim will be achieved through four primary objectives: establish educational partnerships, implement FishSounds Educate lectures and labs, disseminate resources equitably, and further develop FishSounds Educate online. The funding for FishSounds Educate was provided by Fisheries and Oceans Canada's Oceans Management Contribution Program, which supports outreach, monitoring, stewardship, and capacity-building initiatives to achieve Canada's commitment to marine conservation targets. The FishSounds team is currently establishing educational partnerships with instructors and institutions from across Canada to ensure that the materials reach a broad audience. Establishing partnerships involves connecting with instructors and institutions that have marine-focused programs (e.g., Bamfield Marine Science Centre), as well as those who have yet to integrate ocean sciences into their lesson plans. FishSounds Educate will implement four types of lesson materials: interactive lectures, classroom workrooms, field labs,

and seminars. Each of these will be developed for primary, secondary, and university students and then will be tailored according to classroom, institution or instructor's objectives, logistical constraints, and format. All FishSounds Educate materials will be available in English and French. FishSounds Educate will provide the training and equipment required for each type of lesson to be conducted, ensuring that participation in the program occurs at no direct cost to the instructor or institution, reducing any potential participation barriers. This will include providing access to software, hydrophones, and other acoustic recorders. FishSounds Educate will continue to support the instructors as they deliver the material or join the instructor either in person or remotely as guest presenters.

FishSounds Educate will be made freely available to the public by developing an education portal on the FishSounds website. This will include all lesson materials, instructional videos, and the information needed to become a FishSounds Educate partner. FishSounds Educate will be continually developed for the project's duration, including forming partnerships with public venues (e.g., aquariums) and non-governmental organizations. Finally, FishSounds Educate will develop the interactive lectures into a monthly seminar series on the importance of ocean noise as a tool for conservation, with the team and the broader research community presenting to academics, educators, and the public.

## Conclusions

FishSounds is one of the first publicly available and interactive web portals consolidating existing information on active and passive sound production by fishes, allowing users to search by taxa, geographical distribution (e.g., region or water body), sound type, or reference. Consequently, the FishSounds web portal provides researchers globally with a comprehensive species-level description of sound production in fish and continually updates these descriptions as the field progresses. This momentous feat was made possible due to the decades of dedicated and innovative research conducted by bioacousticians, whose examinations form the foundation of FishSounds. This web portal addresses multiple barriers restricting the progression of investigations into the ecological importance of aquatic soundscapes and extends FishSounds' utility beyond that of an archive, generating a platform for connecting researchers, facilitating collaborations, and promoting emerging research. FishSounds Version 1.1 represents an important step toward this objective. Expansions of the FishSounds data and recording archive, improvements to the user experience, and increases in the functionality of the web portal are the primary products of this update. These accompaniments coincide with the launch of the ocean literacy project – FishSounds Educate. Future updates will further prioritize increasing researchers' ability to explore and engage with the FishSounds data and foster public outreach opportunities. Despite still being in its infancy, FishSounds has the potential to bolster the field of bioacoustics' ongoing efforts to advance research into soniferous fish diversity, passive acoustic monitoring, and human

impacts on underwater soundscapes, as well as serve as a public resource for those interested in learning more about fish sounds and underwater soundscapes.

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# Underwater Noise of Two Operational Tidal Stream Turbines: A Comparison

12

Denise Risch, Brett Marmo, Nienke C. F. van Geel, Douglas Gillespie, Gordon Hastie, Carol Sparling, Joe Onoufriou, and Ben Wilson

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**Abstract**

This study measured underwater noise of two 1.5 MW tidal stream turbines (Atlantis AR1500 and Andritz AHH1500) at the MeyGen project site, Pentland

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Firth, Scotland, UK. Underwater turbine noise may be important for acoustically sensitive marine mammals to detect turbines and avoid collision, but it may also negatively affect them, as well as fish and invertebrates (e.g., injury, disturbance, and habitat exclusion). The turbines' source levels were determined, and a 3D-noise model predicted the acoustic footprints of the existing and future turbine arrays. Highest noise levels were between 50 and 1000 Hz for both turbines. However, noise levels (25 Hz–25 kHz) recorded from the Andritz turbine were generally lower in amplitude compared to those recorded from the Atlantis turbine. The results show that harbor seals (and probably other marine mammal species) will be able to detect operating tidal turbines at distances that are likely sufficient to avoid collision. The MeyGen project aims to build out to 400 MW in later stages of the project which will increase the predicted noise footprint ranges presented here, with unknown environmental consequences. This highlights the importance of including these results on underwater noise in strategic marine spatial planning of larger tidal turbine arrays.

**Keywords**

Tidal stream turbines · Underwater noise · Operational noise · Noise modelling · Detectability · Collision risk · Acoustic footprint · Turbine arrays

## Introduction

The United Kingdom (UK) has committed to reaching net zero greenhouse gas emissions by 2045 (Scotland) or 2050 (Wales and England) (Priestley 2019; Scottish Government 2020; Welsh Government 2021). To enable this, clean energy alternatives will have to replace fossil fuel-based energy generation. Tidal stream energy generation will likely play an important part in this transition and could provide an equivalent of 11% of the UK's annual electricity demand (34 TWh/year) (Coles et al. 2021).

As a relatively new renewable energy industry, it is important that tidal stream energy generation is developed sustainably and that next to economic, regulatory, and societal concerns, environmental impacts of large-scale tidal arrays are considered carefully when assessing the viability of future projects. While the primary environmental concerns about tidal stream energy relate to collision risk with turbine rotors and associated injuries (Band et al. 2016; Waggitt and Scott 2014; Wilson et al. 2007), underwater noise generated by devices may also negatively affect many marine animals. Potential impacts include auditory injury, disturbance, acoustic masking, barrier effects, and habitat exclusion (Pine et al. 2019; Polagye et al. 2011). In contrast, underwater noise from tidal turbines may provide an important sensory cue for animals to detect turbines and avoid collisions.

Regarding the potential of habitat exclusion study, results have been mixed and are species and scale dependent. For example, evidence from a single turbine demonstration project suggested no large-scale area avoidance for harbor seals (*Phoca vitulina*) (Sparling et al. 2018). Harbor porpoises (*Phocoena phocoena*) have also been detected within 30 m of an operational tidal stream turbine (Gillespie

et al. 2021); however, porpoises have also been shown to avoid operating turbines at the same site, with interactions with the tidal flow affecting this behavior (Gillespie et al. 2021; Palmer et al. 2021). Additionally, harbor seals have also been shown to avoid operating tidal turbine arrays on spatial scales of a few hundred meters to a few kilometers (Hastie et al. 2018; Onoufriou et al. 2021).

Whether marine mammal behavior observed around tidal energy demonstration projects can be applied across different tidal energy sites with different numbers of devices and in varying habitats is currently unknown. This uncertainty has the potential to delay or curtail licensing of future large-scale array deployments, which may include tens or hundreds of tidal turbines,[1] especially if these are sited in important transitory corridors or close to important foraging sites. To better understand potential effects of tidal arrays of this size, it is therefore important to characterize the spatial and temporal variation in the noise signatures of different types of turbines along with site-specific ambient sound and propagation characteristics of these challenging environments. This study compares the underwater noise signatures of two different types of tidal stream turbines: the Atlantis AR1500 and Andritz Hydro Hammerfest AHH1500 turbines. Both turbines are 1.5 MW three-bladed horizontal axis turbines and were measured, independent of each other, at the MeyGen[1] project site, located in the Pentland Firth, Scotland.

Underwater noise measurements near the operating turbines were used to estimate the turbines' source level and to develop and validate a three-dimensional noise propagation model for each of the turbines. The models for individual turbines were combined and used to predict the spatial extent over which turbine noise is likely to be above average ambient sound for the current array configuration in the study site (1 × Atlantis AR1500 and 3 × Andritz Hydro Hammerfest AHH1500 turbines). This information was then used to assess likely ranges over which turbine arrays may be audible to harbor seals, the marine mammal species of primary conservation concern in this area given their dramatic decline in population size in the project area over the last 20 years (Thompson et al. 2019; NERC Special Committee on Seals 2021[2]). In addition to the currently deployed MeyGen Phase 1 four-turbine array, a range of hypothetical future array scenarios for MeyGen Phase 2, based on information from SIMEC Atlantis Energy, were modelled.

## Methods

### Site Description

The MeyGen project is in the Inner Sound, a channel between the Island of Stroma and the Scottish mainland (Fig. 1); this area has a maximum water depth of 80 m and tidal currents which are among the strongest in the world, frequently exceeding

[1] https://simecatlantis.com/projects/MeyGen/

[2] http://www.smru.st-andrews.ac.uk/files/2022/08/SCOS-2021.pdf

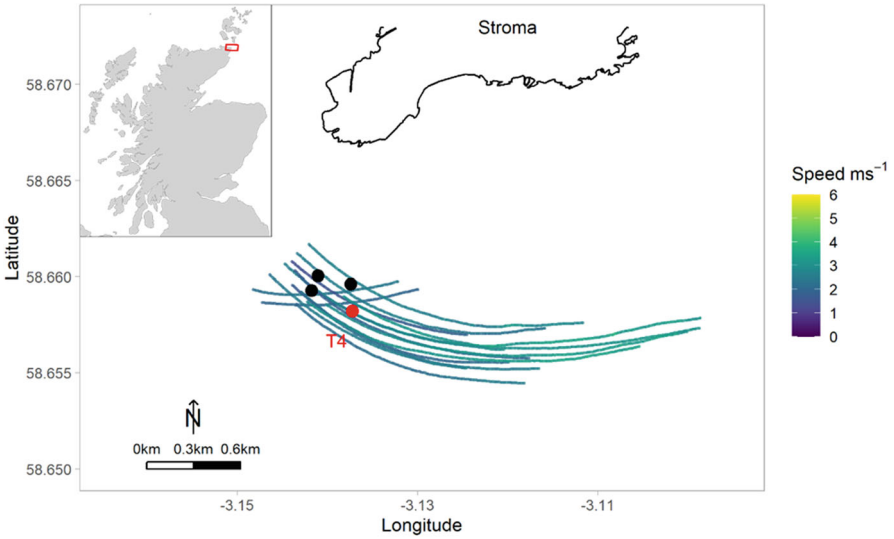



**Fig. 1** Map of the study area in the Inner Sound, showing drifter paths while Atlantis AR 1500 turbine (T4; red dot) was active and monitored (06/08/2018). The other turbines of the Phase 1 array (black dots), which were not deployed at the time of monitoring, are shown for reference. Inset shows study area (red rectangle) in relation to map of Scotland

5 ms$^{-1}$ (Shields et al. 2009). The MeyGen array was installed in 2016 and is currently the largest tidal-stream turbine array (6 MW) in the world, comprising of four 1.5 MW turbines. The MeyGen project aims to build up to 80 MW capacity during Phase 2, with the view of supporting up to 400 MW in later stages of the project[1]. The current four-turbine array contains one Atlantis AR 1500 (T4) and three Andritz Hydro Hammerfest AHH 1500 turbines (T1–3) which are deployed at water depths of about 40–50 m and spaced approximately 200 m apart.

## Sound Measurements

The Atlantis AR1500 turbine T4 and Andritz Hydro Hammerfest AHH 1500 turbine T3 were acoustically monitored on 06/08/2018 and 12–13/04/2021, respectively (Figs. 1 and 2).

To reduce the effects of water flowing over the hydrophone which can introduce flow- or pseudo-noise into sound recordings made in fast flowing waters (e.g., Martin and Popper 2016), three freely moving Lagrangian drifters were used for sound measurements. Each drifter was equipped with an acoustic recorder (RTSYS EA-SDA14) and hydrophone (Reson TC4032 and TC4014; sensitivities TC4032: −162.6 and −169.5 dB re 1 V/1 µPa; and TC4014: −179.9 dB re 1 V/1 µPa), which was suspended approximately 6 m below the surface (Risch et al. 2020; Wilson et al. 2014). The drifters were deployed upstream of the turbine from a

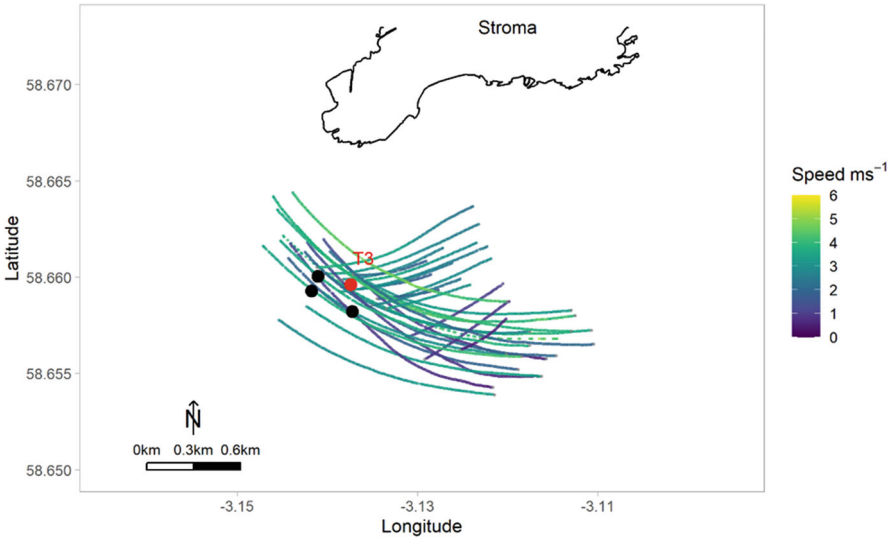



**Fig. 2** Map of the study area in the Inner Sound, showing drifter paths while monitoring the active Andritz Hammerfest AHH 1500 turbine (T3; red dot) (12–13/04/2021). The other turbines (black dots) of the Phase 1 array, which were not deployed at the time of monitoring, are shown for reference

small vessel, which then cut its engines and both the monitoring system and vessel drifted past the turbine. The drifters were then recovered, taken back upstream and redeployed.

Only one turbine was operational during each respective noise monitoring period. The Atlantis turbine was monitored during 15 individual drifts (13 during flood and 2 during ebb tide conditions), at a mean flow speed of 2.6 ms$^{-1}$. The closest point of approach of the drifting hydrophones to the turbine was at a horizontal distance of 7.6 m. The Andritz turbine was monitored over 2 days and 34 individual drifts (17 during flood and 17 during ebb tide conditions), at a mean flow speed of 2.8 ms$^{-1}$. The closest point of approach to the Andritz turbine was recorded at a horizontal distance of 23 m to the turbine.

Recordings were amplified (gain: 14.7 dB) and digitized at a sampling rate of 312,500 Hz with a resolution of 24 bit and the recording chain fully calibrated by the National Physics Laboratory (NPL). Acoustic baseline surveys covering an area of approximately 10 km$^2$ around the turbine locations prior to turbine deployment were carried out in October 2015 and May/June 2016 (Fig. 3; see Risch et al. (2020) for details). During these ambient baseline surveys, a total of 54 individual drifts were performed using 3 drifters (36 during flood and 18 during ebb tide conditions). The mean flow speed for these drifts was 2.7 ms$^{-1}$ (Fig. 3). All acoustic measurements used in this study were recorded at sea states of Beaufort wind force scale 3 or lower. Acoustic spectra were manually screened using RavenPro 1.6.4 (K. Lisa Yang

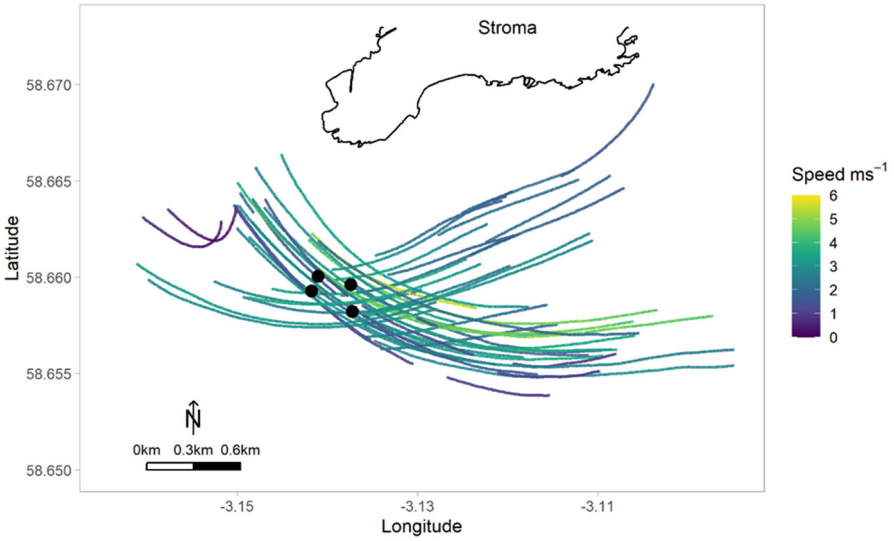



**Fig. 3** Map of the study area in the Inner Sound, showing drifter paths while monitoring ambient sound at the study site before the four turbines were installed (01/10/2015; 24/05/2016; 23–24/06/2016). Phase 1 turbine locations (black dots) are shown for reference

Center for Conservation Bioacoustics at the Cornell Lab of Ornithology 2023) and sections with vessel noise or drifter self-noise were excluded from further analysis.

## Acoustic Analyses

Spectrograms (10–1000 Hz; 10–5000 Hz; 10–40,000 Hz–1-s Hann window and 0% overlap) of the data recorded during each drift were generated and viewed using PAMGuide (Merchant et al. 2015). After initial inspection of the acoustic raw data, sound pressure levels were quantified in one-third octave bands with center frequencies from 25 to 125,000 Hz over a 1-s time window, using the one-third octave level (TOL) function in PAMGuide. Sound measurements were georeferenced, and each transect was divided into 10 m nonoverlapping segments. The median sound pressure level (SPL or TOL; dB re 1 μPa) was then calculated for each third octave band and each 10 m segment. These data were subsequently aggregated by tidal status (e.g., ebb, flood and slack water as assessed based on current directional flow in the field) and turbine rotations per minute (RPM), to evaluate noise level variations with and without operational turbine signals present.

Turbine operational data (RPM) were provided by SIMEC Atlantis Energy and ranged from 3 to 10 rotations per minute for the Atlantis AR1500 turbine (T4) and from 0.5 to 16 rotations per minute for the Andritz AHH1500 turbine (T3). However, the Atlantis turbine was not operating at full power at the time of monitoring, which was carried out during neap tides. Expected RPMs under full operation of the turbine

would be higher and more closely aligned with those of the Andritz turbine (pers. comm. F. Johnson, SIMEC Atlantis Energy).

## Turbine Noise Modelling

### Modelling Approach

A three-dimensional structural acoustic finite element model of the Atlantis AR 1500 and Andritz Hammerfest AHH 1500 tidal turbines was used to derive the source levels for turbine operational noise. The model was developed using the commercially available modelling package COMSOL Multiphysics (Comsol 2023) and has been applied successfully to the development of models of onshore wind turbine noise and tidal energy converters in the past (Carruthers and Marmo 2011). The model produces a three-dimensional sound field, from which source level as a function of angular position from the turbine main axis can be derived.

Estimated turbine source levels were calibrated using the measured acoustic data recorded in situ when the Andritz turbine (T3) was operating with a rotation speed greater than 14 RPM. This rotor speed is close the maximum speed of the Andritz device (maximum recorded RPM: 16.7) and was taken as a proxy for the peak power output. In contrast, the source level for the Atlantis turbine (T4) was derived by calibrating the finite element model to recordings of the Atlantis turbine operating at greater than 8 RPM (maximum recorded RPM: 10). The turbine was not running at full power when sound measurements were taken due to the monitoring taking place during neap tides.

The source levels derived using finite element models were extrapolated into the far-field (up to 9.3 km west of the center of the tidal array and 11.0 km east of the center) using a boundary element method at lower frequencies ($\leq$200 Hz) and a three-dimensional ray trace model at higher frequencies (>250 Hz) to increase computational efficiency. The propagation of noise was modelled at one-third octave frequencies bands from 25 Hz to 25 kHz. The tidal turbines were represented as point sources emitting ranges with a spherical spread. In the ray trace models, 10,000 rays were released from each point representing each turbine; thus, 40,000 rays were used to model the array of four turbines. The rays were allowed to propagate for 15 s, applying a 10 μs time step. Rays were terminated upon leaving the model space. Rays were also terminated when the power become less than $1e^{-20}$ W to improve the numerical efficiency of the model. The water was modelled as saltwater with properties listed in Table 1. The attenuation coefficient of the water was modelled as being frequency dependent based on the method of Ainslie and McColm (1998) and using the properties listed in Table 1.

The water surface was taken to be a perfect reflector with a reflection coefficient of 1. The seabed was modelled as sandy sediment, which is the predominant sediment type in the Inner Sound (e.g., Fairley et al. 2015) using an impedance condition based on a sediment density of 2040 kg/m$^3$ and speed of sound of 1720 ms$^{-1}$ (LeBlanc et al. 1992).

**Table 1** Materials properties of seawater modelled in the Humber Estuary (Ainslie and McColm 1998)

| Material property | Value | Unit |
|---|---|---|
| Speed of sound | 1500 | m/s |
| Density | 1026 | $kg/m^3$ |
| Salinity | 35 | Ppt |
| Temperature | 15 | °C |
| pH | 8 | |

The modelled sound pressure levels presented in this report are calculated assuming maximum power output, as approximated by peak RPM for the Andritz turbine. In contrast, the Atlantis turbine was not fully operational during monitoring, and the noise model of this turbine was calibrated with the available acoustic measurements when the turbine was at peak RPM on the day of monitoring (06/08/2018), which is not equal to peak power for this turbine. Measured and predicted noise levels presented in this report for the Atlantis turbine must therefore be treated with caution.

## Frequency-Weighting of Predicted Noise Levels

Impacts of underwater noise on marine mammals are generally estimated using auditory weighting functions (e.g., National Marine Fisheries Service 2018). Such weighting functions are based on species (or species groups) hearing range and sensitivity, and data on physiological or behavioral responses to noise. Animals are not equally sensitive to all frequencies, and auditory frequency weighting is therefore an important concept for the assessment of audibility and potential impacts of different sources of underwater noise (Tougaard and Dähne 2017). Based on phylogenetic relationships and best available knowledge, marine mammals have been separated into functional hearing groups for which similar auditory weighting functions should be applied (Southall et al. 2007, 2019). The group phocid carnivores in water (PCW) comprises all *Phocidae* in water, including harbor and gray seals. Predicted noise levels presented in this study will be frequency-weighted using the weighting function for this group to focus the analysis of turbine noise on harbor seals, which due to their declining population status (Thompson et al. 2019; NERC Special Committee on Seals 2021[2]) are the most vulnerable marine mammal species inhabiting the MeyGen project area.

## Modelling Turbine Array Scenarios

The first modelling scenario explored in this study focused on the existing four-turbine array (1 × Atlantis and 3 × Andritz turbines) deployed during Phase 1 of the MeyGen project. The noise propagation models for each individual turbine were calibrated with empirical field measurements of both turbines (see above) before integration into a cumulative far-field noise field. The MeyGen project will build up to 80 MW (= 30 turbines) in Phase 2 of the project[1]. Although the specific details of Phase 2 (in terms of turbine numbers, type, and locations) are yet to be finalized, four realistic yet hypothetical future 30-turbine array scenarios were modelled. Scenarios varied in turbine type (all Atlantis vs. all Andritz turbines) and lateral spacing of

turbine blade tips (4-m vs. 16-m). Array layouts were informed by information provided by SIMEC Atlantis Energy.

### Spatial Extent and Signal Detection Ranges

Soundscape measurements carried out in October 2015 and May/June 2016 were used to estimate the audibility of the turbine signal to harbor seals in the context of site-specific ambient sound conditions. In general, a sound signal can be detected by an animal up to the distance where the received sound level matches its hearing threshold at a certain frequency band. However, if the ambient sound in respective frequency bands is higher than the animal's hearing threshold, the maximum detection distance is determined by the distance at which the received sound levels equal ambient sound over a similar frequency range (Madsen et al. 2006). This simple approximation of signal audibility is generally applicable for broadband sounds. It should be noted, however, that due to the shape of their auditory filters, it is likely that harbor seals can detect the discrete tonal frequencies present in the noise produced by tidal stream turbines at a few decibels below ambient sound levels (see: Au 1993; Madsen et al. 2006; Erbe et al. 2015).

For simplicity, in this study, the minimum signal detection distance for harbor seals is assumed to be at the distance where predicted frequency-weighted turbine noise levels match the average ambient sound levels recorded during flood tides. In the Inner Sound, median ambient sound levels under these conditions are in the range of 100–105 dB re 1 μPa at 100 Hz–1 kHz (Risch et al. 2020), which is above the harbor seal hearing threshold at these frequencies (Kastelein et al. 2009). The 100 dB noise contour is therefore used to estimate the turbine signal detection distance for harbor seals. Detection distances are likely larger during periods of decreasing tidal flow when ambient sound levels are generally lower (e.g., Willis et al. 2013) and will vary spatially due to turbulence related variability in underwater sound during different tidal phases (Risch et al. 2020).

## Results

### Turbine Signal Characterization and Comparisons Between Atlantis and Andritz Turbine Sound Signatures

Example spectrograms of the closest drifts to the active Atlantis and Andritz turbines are presented in Fig. 6. Sound from both turbines is clearly discernible over ambient sound levels and reaches its maximum at the closest point of approach (CPA; measured horizontally) to the turbine (CPA Atlantis: 7.6 m; CPA Andritz: 23 m; Fig. 4).

Both turbines showed oscillating tonal signatures below 1 kHz (Fig. 4). Underwater sounds recorded from the Andritz turbine showed less pronounced tonal sounds and harmonic structure, when compared to recorded sounds from the Atlantis turbine. A strong tonal signal at approximately 20 kHz which was recorded from the

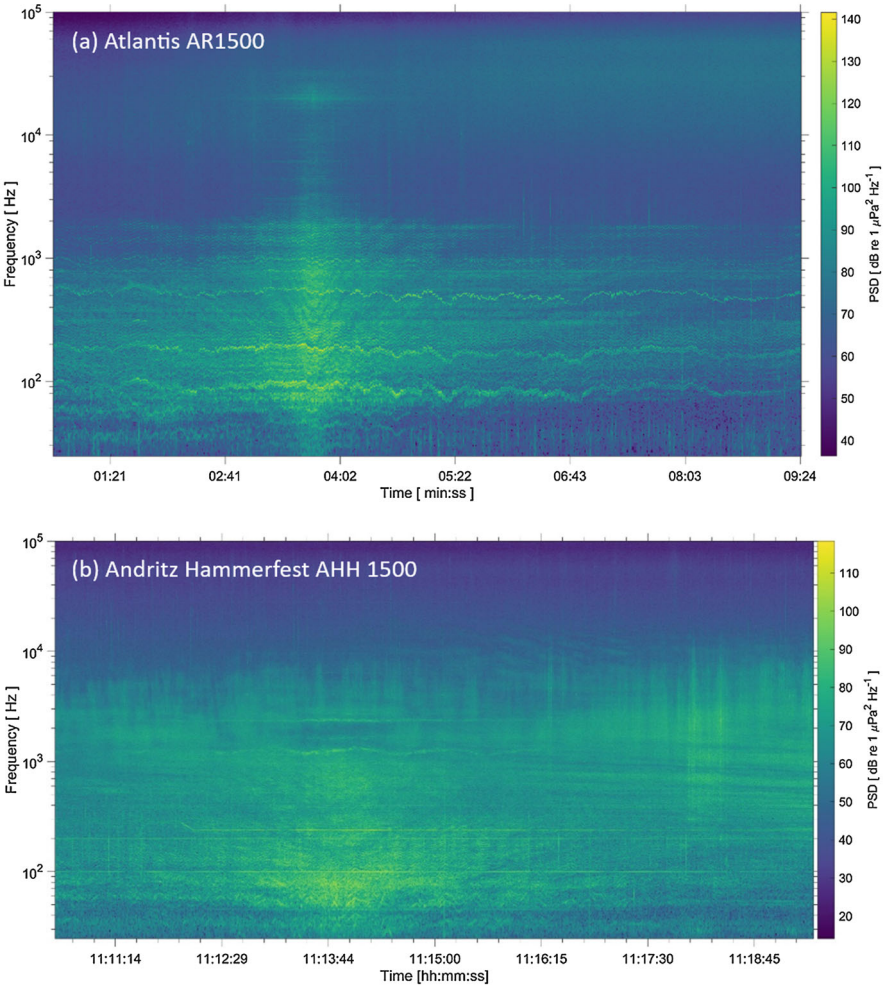



**Fig. 4** Power spectral densities (PSD; dB re 1 $\mu Pa^2 Hz^{-1}$) using a 1 s Hann window and 0% overlap for two of the closest drifts monitoring (**a**) the Atlantis AR1500 turbine (Closest Point of Approach (CPA) = 7.6m) and (**b**) the Andritz AHH1500 turbine (CPA = 23m)

operating Atlantis turbine (Fig. 4a; Risch et al. 2020) was not recorded from the Andritz turbine (Fig. 4b).

Highest broadband noise levels were observed in the lower-frequency range from 100 Hz to 1 kHz for both turbines. When comparing median noise levels, recorded at similar distances and under comparable environmental conditions, noise levels from the Atlantis turbine were generally higher than those measured for the Andritz turbine, particularly in the lower ($\leq$100 Hz) and higher (>1 kHz) frequency ranges (Fig. 5).

For both turbines, sound levels across the full spectrum were approximately 10–20 dB higher when measured within 20–30 m from the active turbines compared

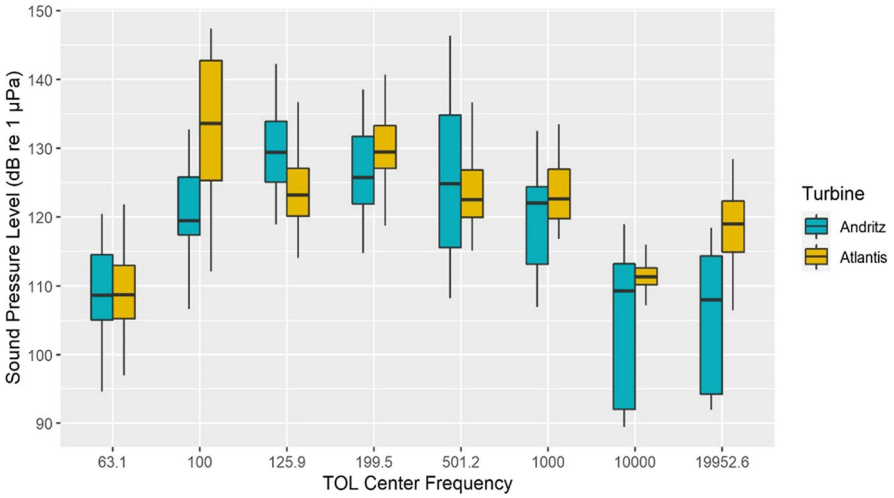



**Fig. 5** Grouped boxplot of one-third octave sound pressure levels (TOL in dB re 1 µPa) measured within 0–100 m from the Andritz Hammerfest AHH1500 (blue) and Atlantis AR 1500 (yellow), respectively. All measurements were taken during flood tide, sea state ≤2 Beaufort and maximum RPM (which differed between turbines and was higher for the Andritz turbine, see section "Methods"). Lower and upper bounds of boxes represent lower and upper quartiles, respectively. Solid lines represent medians, and whiskers indicate furthest data points within 1.5 × interquartile range. TOL center frequencies (x-axis) in Hz

to those measured between 100 and 120 m from it under similar conditions (Figs. 6 and 7). For the Atlantis turbine, median noise levels for frequencies between 100 Hz and 1 kHz were between 125 and 140 dB re 1 µPa, when measured within 20 m from the turbine (Fig. 7a). Measured sound levels for the Andritz turbine were more variable (Figs. 6 and 7), but maximum noise levels for individual one-third octave-bands were in a similar range for both turbines (Fig. 7a).

Sound levels were generally lower when RPM was lower, for both turbines (Figs. 6 and 7). As described above, the Atlantis turbine was not operating at full power during acoustic monitoring of the turbine; maximum observed RPM for this turbine were between 7 and 9 RPM, while for the Andritz turbine, highest recorded rotational speeds were between 14 and 16 RPM. This limitation of the available data set needs to be carefully considered when interpreting the results of this study and the predicted noise levels in subsequent analyses.

## Noise Propagation Modelling and Noise Predictions for Multi-turbine Arrays

Turbine noise level measurements were used to estimate source levels for both turbines and build individual turbine noise propagation models, treating each turbine as a point source. Propagation models were compared to measurements at a range of

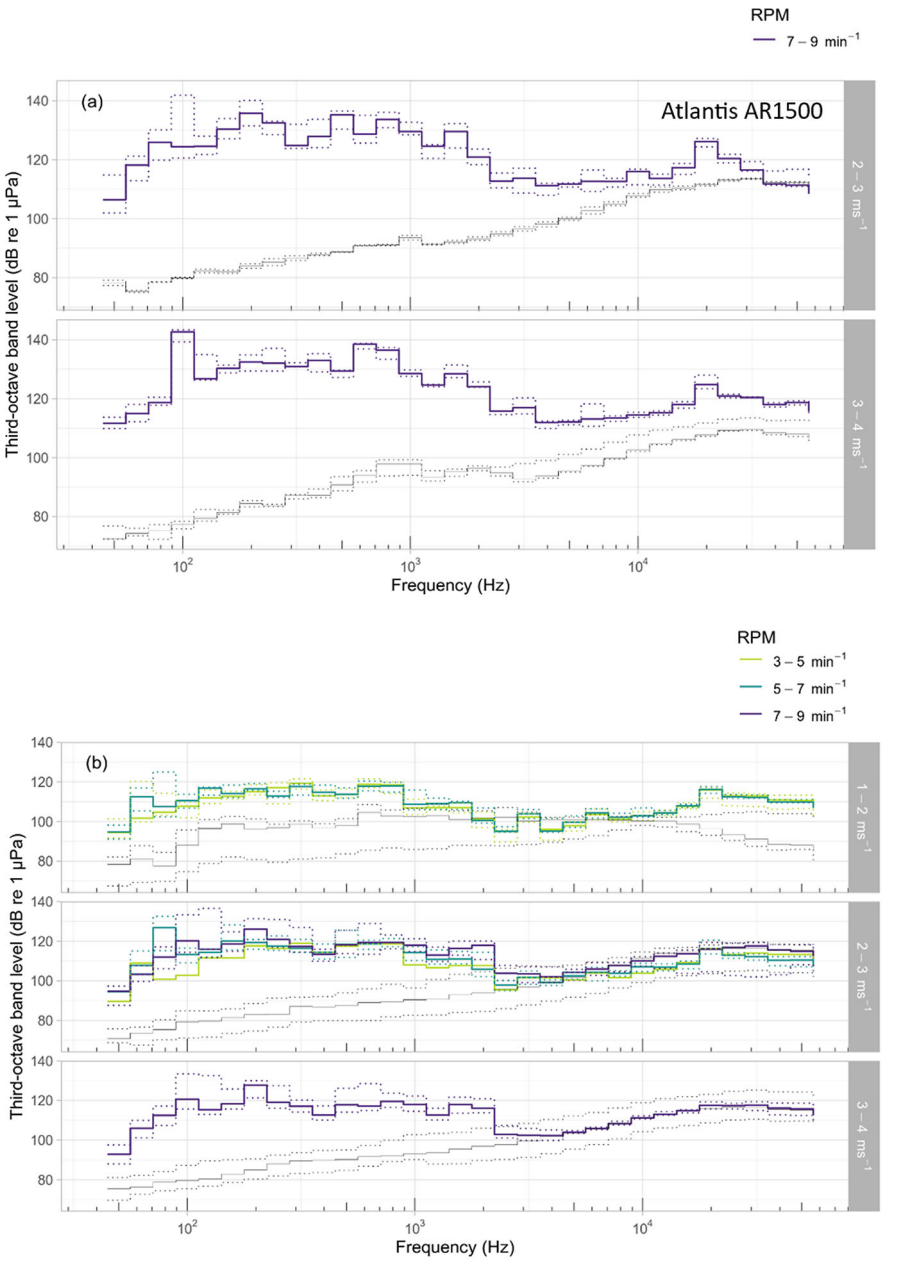



**Fig. 6** Atlantis AR1500: Median one-third octave sound pressure levels (50 Hz–63 kHz; TOL in dB re 1 µPa) (thick lines), and 5th and 95th percentiles (dotted lines), calculated over a 1 s window. Plots are faceted by current speed in $ms^{-1}$, as measured by drifters, and colored by turbine rotational speed in $min^{-1}$ (RPM). All data collected during flood tides and Beaufort 2 or below. (**a**) Data

distances from the operational turbines. Validation results for the Andritz Hammerfest AHH 1500 turbine showed reasonable agreement between measured and modelled data, with most model predictions falling within one stand deviation of the mean measurements at 100 and 1000 m from the turbine.

Cumulative, broadband (25 Hz–25 kHz) turbine noise levels modelled for the currently deployed four-turbine array (1 × Atlantis, 3 × Andritz), assuming maximum power generation,[3] are presented in Fig. 8. The overall larger acoustic footprint for noise generated by the Atlantis turbine is discernible. This difference is likely driven by the additional noise at 20 kHz and higher levels in the lower frequencies (e.g., Figs. 4, 5, 6, and 7). Given average broadband ambient levels of about 100 dB re 1 μPa (Risch et al. 2020), unweighted noise levels for this scenario extend to an area of 1.5 $km^2$ above ambient sound, with a median and maximum radius (calculated from the centroid of the 100 dB contour) of 714 m and 845 m, respectively (Fig. 8a, Table 2). Frequency-weighted (for phocid carnivores in water (PCW); Southall et al. 2019) turbine noise levels for this scenario are above ambient across an area of 0.2 $km^2$ with a median radius of 259 m (Fig. 8b, Table 2).

The MeyGen project will build out to 80 MW (= 30 turbines) in Phase 2 of the project[1]. Four hypothetical but realistic future 30-turbine array scenarios were modelled. Scenarios varied in turbine type (all Atlantis vs. all Andritz turbines) and lateral spacing of turbine blade tips (4-m vs. 16-m). Array layouts were informed by information from SIMEC Atlantis Energy.

Model results for the 30-turbine scenario with all Atlantis AR 1500 turbines for the 4-m array are shown in Table 2. PCW frequency-weighted turbine noise levels are above ambient across an area of 0.8 $km^2$, with a median and maximum radius of 540 m and 657 m, respectively (calculated from the centroid of the 100 dB contour). This compares to an area of 1.0 $km^2$, with a median and maximum radius of 588 m and 824 m, respectively, for the array with 16-m spacing between Atlantis turbines (Table 2).

For a 30-turbine array of all Andritz Hammerfest AHH 1500 turbines, PCW frequency-weighted noise levels of the 4-m spaced arrays are above ambient across an area of 0.3 $km^2$, with a median and maximum radius of 330 m and 434 m, respectively (Table 2). The area of turbine noise above ambient sound for the 16-m array is 0.4 $km^2$, with a median radius of 356 m and a maximum radius of 669 m from the centroid of the 100 dB contour (Table 2).

**Fig. 6** (continued) recorded between 0 m and 20 m and (**b**) between 100 m and 120 m from the operational turbine. Gray solid lines are median ambient one-third octave band levels collected under similar conditions and across the same spatial range. Gray dotted lines indicate 5th and 95th percentiles of ambient noise levels

[3] The Atlantis turbine was not fully operational yet at the time of monitoring (pers. comm. F. Johnson, SIMEC Atlantis Energy), which will affect measured and predicted noise levels.

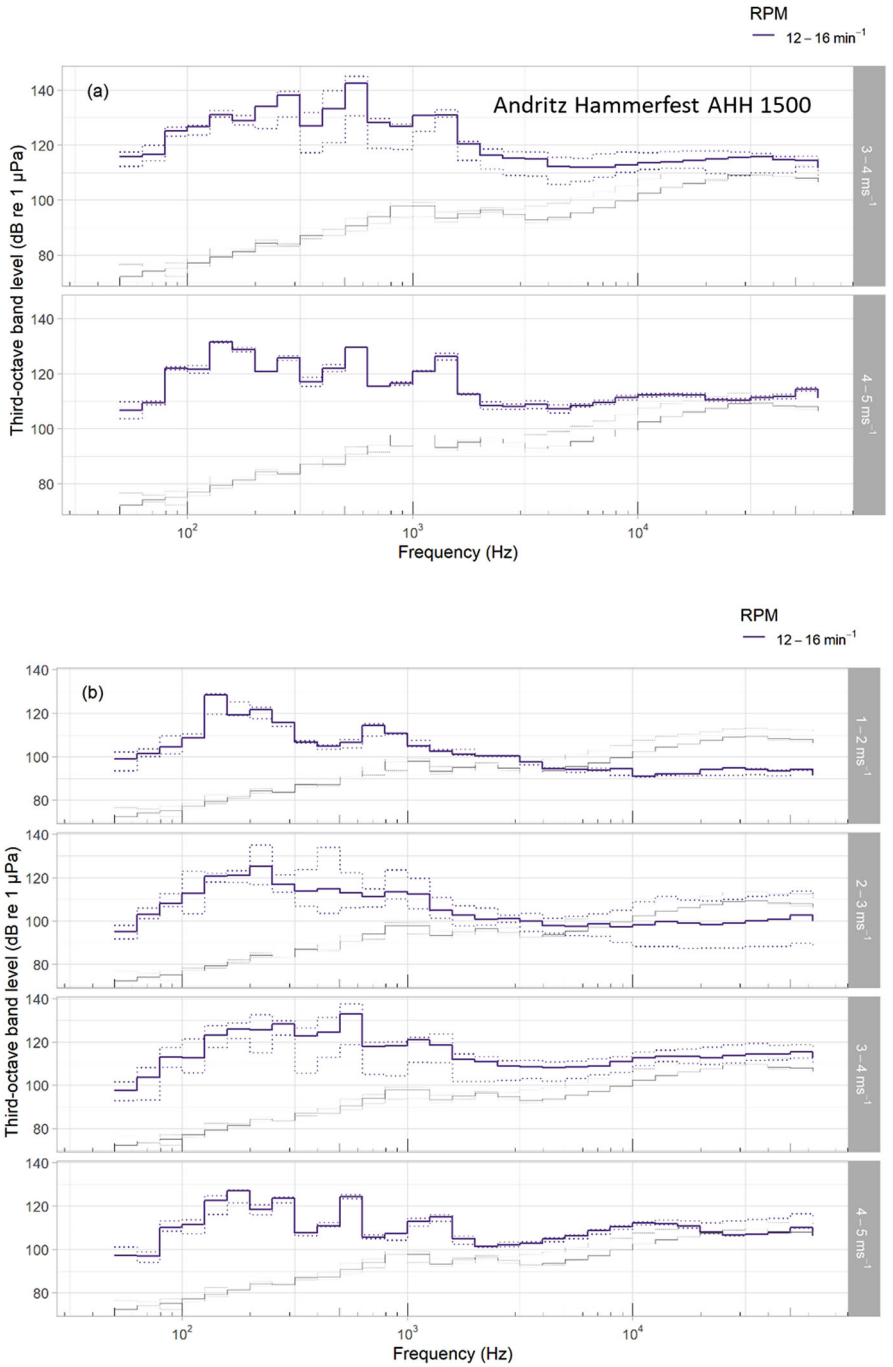



**Fig. 7** Andritz AHH1500 turbine: Median one-third octave sound pressure levels (50 Hz–63 kHz; TOL in dB re 1 µPa) (thick lines), and 5th and 95th percentiles (dotted lines), calculated over a 1 s

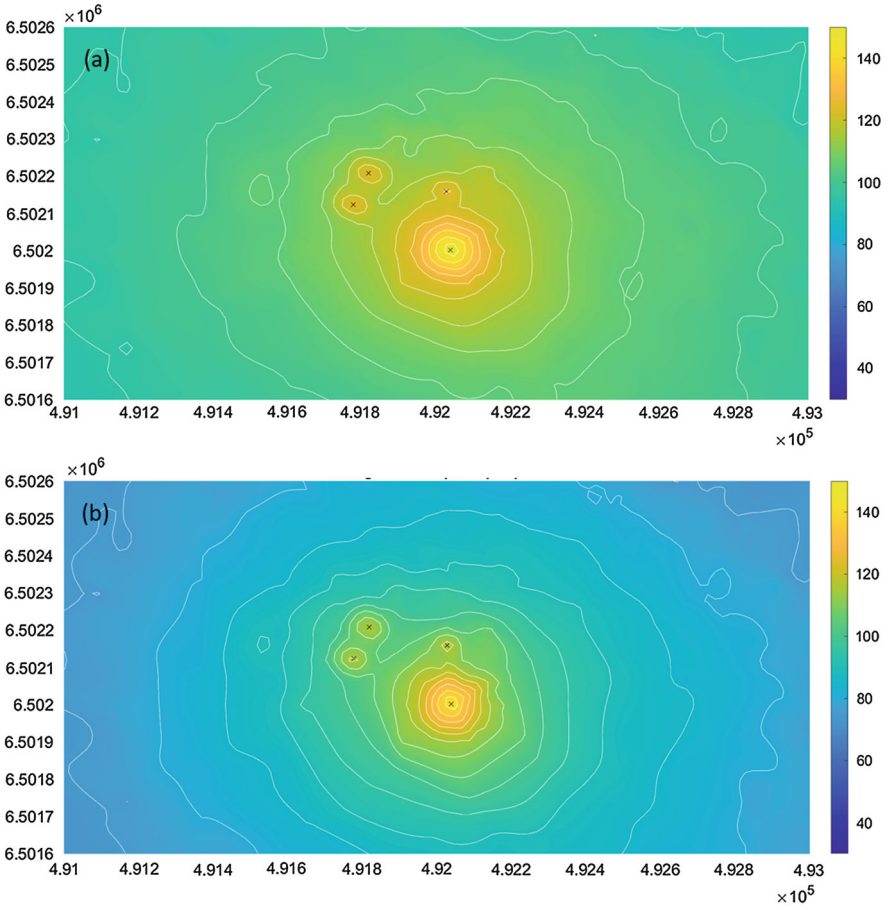



**Fig. 8** (**a**) Unweighted and (**b**) phocid carnivores in water (PCW) weighted sound pressure level map (25 Hz–25 kHz) for MeyGen Phase 1: four-turbine array (1 × Atlantis AR 1500, 3 × Andritz Hammerfest AHH 1500). Scale: one-third octave sound pressure level (dB re 1 μPa)

When comparing cumulative noise levels of the 4-m Atlantis vs. Andritz turbine arrays, the louder Atlantis array extends the area over which weighted received levels are above ambient sound by 0.5 $km^2$ in area and 223 m maximum radius compared to the Andritz array (Table 2). When spacing individual turbines by 16-m

**Fig. 7** (continued) window. Plots are faceted by current speed in $ms^{-1}$ (as measured by drifters) and colored by turbine rotational speed in $min^{-1}$ (RPM). All data collected during flood tides and Beaufort 2 or below. (**a**) Data recorded between 0 m and 30 m and (**b**) between 100 m and 120 m from the operational turbine. Gray solid lines are median ambient levels collected under similar conditions and across the same spatial range. Light gray dotted lines indicate 5th and 95th percentiles of ambient noise levels

**Table 2** Area, maximum and median distances to centroid of modelled 100 dB sound pressure levels (SPL) contour for the different scenarios presented in this report. PCW weighted: 100 dB contour calculated based on frequency weighted (for phocid carnivores in water) SPLs (Southall et al. 2019)

| Scenario | Area ($km^2$) | Max Dist (m) | Med Dist (m) |
|---|---|---|---|
| **4 turbines – unweighted** | 1.5 | 845 | 714 |
| **4 turbines – PCW weighted** | 0.2 | 334 | 259 |
| **30 turbines; 4-m lateral spacing (Atlantis); PCW weighted** | 0.8 | 657 | 540 |
| **30 turbines; 16-m lateral spacing (Atlantis); PCW weighted** | 1.0 | 824 | 588 |
| **30 turbines; 4-m lateral spacing (Andritz); PCW weighted** | 0.3 | 434 | 330 |
| **30 turbines; 16-m lateral spacing (Andritz); PCW weighted** | 0.4 | 669 | 356 |

between rotor tips, the difference between the Atlantis and the Andritz arrays is 0.6 $km^2$ in area and 155 m in maximum radius from the centroid of the 100 dB contour (Table 2).

## Discussion

This study characterized the underwater sound characteristics of two types of tidal stream turbines, the Atlantis AR1500 and Andritz Hammerfest AHH1500 turbines. The peak frequencies of both measured turbines were between 100 Hz and 1 kHz. Both turbines produce a similar oscillating tonal signal with a fundamental frequency of about 100 Hz, although oscillations were more clearly discernible in the signal of the Atlantis turbine. Harmonics were also more pronounced for this turbine.

Underwater noise produced by tidal stream turbines is typically related to the mechanical vibration in the machinery; it is created by imbalances of the rotating components, including the teeth in the gearbox encountering each other (referred to as *gear meshing*) and electromagnetic interaction between the spinning poles and stationary stators in the generator (referred to as *cogging forces*). Rotational imbalances often occur at very low frequencies (<50 Hz), while gear meshing and electromagnetic interactions tend to occur at low to moderate frequencies (50 Hz–2 kHz) (B. Marmo, *pers. comm.*). All these sources occur in discrete frequency bands related to the rotational speed of each component, which likely creates the tonal signatures observed here and in other studies of operational horizontal axis tidal-stream turbines (Lossent et al. 2018; Schmitt et al. 2015). It has also been shown that sound levels vary in relation to operational mode of turbines (e.g., constant spinning, free-wheeling, and braking) (Schmitt et al. 2015). In the current study, only rotational state of the turbines was known and compared against recorded turbine noise levels. In future acoustic studies, it would be useful to

record turbine noise during different operational states to assess relative changes in noise levels throughout the tidal cycle. When interpreting results from this study, the relatively low rotational speed of the Atlantis turbine, when the current recordings were made, needs to be considered, as noise levels can be expected to be higher when the turbine is operating at higher rotational speeds (Risch et al. 2020). Fundamentally, predicted noise levels for scenarios including this turbine type are an approximation and should be adjusted once new data collected from a turbine of this type operating at full speed become available.

With these caveats in mind, results from this study showed that within 100 m from the operating turbine, median noise levels in the peak one-third octave band (centered at 100 Hz) measured from the Andritz turbine (120 dB re 1 μPa) were about 10 dB lower than those for the Atlantis turbine (133 dB re 1 μPa) (Fig. 5). These noise levels are higher than those reported for operational noise from offshore wind turbines of comparable size (1.5 MW) recorded at the same distance (102–112 dB re 1 μPa; see Tougaard et al. 2020). This is expected, as in the case of tidal turbines all noise sources are fully submerged, whereas wind turbines produce airborne noise, some of which is carried underwater via the turbine tower. Turbine noise levels are slightly lower than those recorded from a passing ship (130–140 dB re 1 μPa; Hermannsen et al. 2014; Tougaard et al. 2020). However, compared to a passing ship, the contribution of tidal turbines to the local soundscape is more constant, albeit variable depending on tidal state (Risch et al. 2020).

When comparing median noise levels, recorded at similar distances and under comparable environmental conditions, levels from the Atlantis turbine were generally higher than those measured for the Andritz turbine (Fig. 5). Further, a 20 kHz tone was detected when monitoring the Atlantis turbine (see Fig. 4 and Risch et al. 2020) but was not present in the recordings of the Andritz turbine. The origin of this component of the signal is currently unknown. However, it is most likely produced by electromagnetic interactions inside the turbine, as it is recorded only when the turbine is generating power and not during times of slack water when the turbine is not generating (Gillespie et al. 2020).

Predicted noise levels in the subsequent array modelling exercises are dominated by the contribution of the louder Atlantis turbine (Fig. 8). There are many possible causes of the difference in noise levels, including differences in the way the drivetrains are coupled to their support structures (B. Marmo, *pers. comm.*). However, more data collected under various operational states of both turbines would help to elucidate further causes for the observed differences as well as their relative impact on the noise footprint of future turbine arrays.

Recordings of both turbine types were used to calibrate and validate the turbine source levels, and noise propagation modelling, using available peak RPM as a proxy for peak power generation. Modelled broadband (25 Hz–25 kHz) noise levels for all four turbines of the MeyGen Phase 1 array at peak power resulted in an acoustic spatial footprint of 1.5 $km^2$ (maximum radius: 845 m) above average ambient sound. This area was 0.2 $km^2$ (maximum radius: 334 m) when noise levels were weighted for best hearing of seals in water using recommended auditory weighting functions for this species group (Phocid Carnivores in Water (PCW):

National Marine Fisheries Service 2018, Southall et al. 2019). These findings imply that the operating turbine array is detectable to harbor seals over a range of at least 300 m. However, given the tonal components of the Atlantis turbine signal, animals are likely able to detect the turbines at larger distances, since tonal signals can be detected at received levels a few dB below ambient sound levels (e.g., Au 1993; Madsen et al. 2006; Erbe et al. 2015). Detection distances will also be influenced by changing ambient sound levels across the tidal cycle, with quieter periods during slack water and minimum tidal currents likely providing conditions for better signal propagation (e.g., Carter 2013; Willis et al. 2013). Localized turbulence (e.g., standing waves, boils, and downwellings) also contribute to a dynamic variation in the acoustic soundscape and may also influence detection distances of tidal turbines (Bassett et al. 2014; Willis et al. 2013). Possible avoidance of turbine arrays or other behavioral changes might also be extended to larger areas under quieter ambient sound conditions.

Overall, these data show that harbor seals (and likely other marine mammal species, such as gray seals, harbor porpoises, and delphinids) will be able to detect operating tidal turbines at distances that are likely sufficient to avoid collision. This aligns with previous studies of animal behavior around tidal stream turbines which showed that both harbor seals (Hastie et al. 2018; Joy et al. 2018; Onoufriou et al. 2021) and harbor porpoises (Sparling et al. 2018; Tollit et al. 2019; Gillespie et al. 2021; Palmer et al. 2021) avoid operating tidal turbines on spatial scales of a few hundred meters to a few kilometers.

Regarding the potential of habitat exclusion, evidence from a single turbine demonstration project suggested no large-scale area avoidance for harbor seals (Sparling et al. 2018). On the other hand, the species has also been shown to avoid operating tidal turbines on spatial scales of a few hundred meters to a few kilometers (Hastie et al. 2018; Onoufriou et al. 2021). Results from the current study show that the noise footprint of turbine arrays depends on turbine type, array size, and spacing of individual turbines, which will also affect avoidance distances. In addition, environmental factors (width of channel, number of entrances, flow speed, etc.) and individual and population level behavioral differences could also contribute to differences in avoidance between sites.

Comparing predicted noise levels of the MeyGen Phase 2, 30-turbine array scenarios of all Atlantis and all Andritz turbines showed radial differences in frequency-weighted sound levels (above ambient sound) of 233 and 155 m for the array with 4-m and 16-m spacings between turbine blade tips, respectively (Table 2). Due to their smaller size, noise levels in the center of the 4-m arrays are about 2 dB higher than in the 16-m arrays (Table 2). However, given that the two array sizes are still relatively similar in size (about 1.5 difference in overall array size), the noise fields of these (realistic) Meygen array scenarios are similar when compared at a few kilometers distance from the array center (Table 2).

Despite relatively small differences in noise extent between the 4-m and 16-m 30 turbine arrays compared here, results from this study demonstrate that, in general, it can be expected that the noise field of arrays with larger spacing between individual turbines will result in larger overall noise footprints, which might increase

behavioral response distances and the likelihood of habitat loss even in highly dynamic tidal environments that show comparatively high ambient noise levels (Risch et al. 2020). This is of particular importance, given that after the second phase of the MeyGen project, which will build the array out to the consented 86 MW (about 30 turbines added to the existing 4), the view is to support 400 MW in later stages of the project.

Results from this study extend an earlier acoustic study of the Atlantis turbine (Risch et al. 2020) by adding information about operational noise of the Andritz Hammerfest AHH 1500 turbine deployed at the same location. This allowed for comparisons between the turbine types and array configurations with different spacing between individual turbines which highlighted the importance of taking turbine type, array size, and configuration into account when predicting the spatial extent of underwater noise produced by tidal stream energy generation. Specifically, results from this study showed that different tidal stream turbines produce different noise signatures that vary with turbine rotational speed, which in turn is influenced by tidal current speed. This type of knowledge is vital when considering mitigation of collision risk as well as potential for avoidance, barrier effects, or large-scale habitat exclusion (e.g., Hastie et al. 2018; Gillespie et al. 2021; Palmer et al. 2021). In future, these data could be combined with locations of harbor seals tagged with GPS tags in the project area to relate observed behavioral responses to received noise levels, to then develop noise response functions, which may further inform current collision risk models (Onoufriou et al. 2021).

## Conclusion

In conclusion, this study highlighted the importance of considering cumulative noise effects of increasingly larger tidal turbine arrays. The current four-turbine array is likely detectable for harbor seals across an area of 0.2 $km^2$ (radius: 334 m). A 30-turbine (86 MW) array of all Atlantis turbines increased the area to 0.8 $km^2$ (radius: 657 m). An identical array of Andritz turbines creates a noise footprint of 0.3 $km^2$ (radius: 434 m). The MeyGen project aims to build out to 400 MW in later stages of the project which will increase these predicted ranges. We recommend that these results are included in regional strategic environmental impact assessments, which should also include other sources of noise nearby (e.g., the presence of shipping lanes).

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# Underwater Sound Emissions from the Moorings of Floating Wind Turbines: HYWIND Scotland Case Study

13

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**Abstract**

The nature of underwater sound generated by floating offshore wind turbines (OWTs) is of increasing interest to the renewable energy industry. This chapter presents the findings from the measurements of the first operational floating OWF, HYWIND Scotland.

Two acoustic recorders were deployed for 4 months: one at a control site and one within the OWF. Continuous tonal sounds associated with the turbine operations and transient sounds originating from the moorings were identified. The characteristics of the latter and the relevant implications for impact assessment are described. A kurtosis analysis of the sounds from the moorings of the HYWIND turbines indicated that the signals were more non-impulsive than impulsive. Directional analysis of the acoustic data revealed that different types of sounds (i.e., creaks, snaps, rattles) were generated from each of the three monitored turbines despite these having the same design. The loudness of their signals also varied according to the source. These findings show the importance of collecting in-field measurements to inform impact assessment studies. Due to the variability of these signals, characteristics from mooring noise are difficult to predict in underwater noise modeling studies.

**Keywords**

Sound · Offshore wind · Impact · Operational · Floating · Wind turbine · Kurtosis

## Introduction

Understanding the nature of underwater sound from floating offshore wind turbines (OWTs) is a topic of increasing interest in the renewable energy industry. There are ambitious plans to develop large floating offshore wind farms (OWFs) around the world. OWFs up to 54 GW are currently proposed for construction. Each will be required to submit detailed impact assessment studies that estimate the effects of sound on marine life, among other effects.

Sound emissions from fixed-structure OWTs, such as monopiles, have been extensively measured and evaluated during installation and operation. Fixed-structure OWTs are appropriate for water depths up to ~60 m; however, in deeper locations, other approaches are planned, with a leading design being floating foundations secured to the seabed with mooring lines. No public literature is available on measurements from floating foundations because only a few have been installed to date.

HYWIND Scotland is the world's first operational floating OWF and comprises five wind turbine generators (WTGs) for a total installed capacity of 30 MW. Each



**Fig. 1** Diagram of the HYWIND Scotland WTG. (Source: Equinor ASA)

WTG is mounted on a spar buoy/pillar moored to the seabed by a ballasted catenary system employing three mooring cables (Fig. 1).

JASCO conducted a sound source characterization (SSC) study for Equinor (previously Statoil) in 2011 on a similar mooring system deployed as part of the HYWIND DEMO system off the coast of Stavanger, Norway. Those measurements led to the identification of tonal and transient sounds (e.g., snapping and clicking) generated by the turbine; the latter may be associated with the mooring system (Martin et al. 2011).

The focus of this chapter is to present an operational noise profile for the HYWIND Scotland system, describing the characteristics of the sound signals associated with the moorings of these systems and discussing the implications for their impact on marine fauna.

## Sound from Operational OWTs

Underwater sound from any type of wind turbine in operation originates in the moving mechanical parts in the nacelle, almost exclusively with energy emitted at low frequencies below 1 kHz and typically with strong tonal elements at the frequencies corresponding to the rotation of the gears and their harmonics

(Pangerc et al. 2016). Noise is transmitted through the tower and radiates into the water from the submerged section. The underwater operational sound levels might therefore depend on the type of foundation, but this has not been demonstrated by the studies conducted to date (Tougaard et al. 2020). Two factors that influence the sound level are the turbine size and the wind strength. As the size of blades increases, the mechanical forces acting on the gears and bearings also increase, resulting in higher noise levels; the same is true for an increase in wind speed (Tougaard et al. 2020). The underwater sound levels emitted by operational turbines have the potential to impact marine fauna, particularly when considering the sound fields generated by several turbines within a wind farm (Stöber and Thomsen 2021).

With regard to floating WTGs, additional potential sources of underwater sound have been identified (Martin et al. 2011). These include the mooring systems and equipment that may be project specific (this may vary according to the float solution chosen), such as pumps that may be used to ballast semi-submersible platforms. As these elements are project specific, predicting their sound signature and associated levels may be difficult to predict, especially in the environmental impact assessments when the detailed design may not have been defined. For this reason, collecting in-field measurements to characterize the signals emitted by the WTGs once installed as full-scale systems is useful.

Sound emissions associated with ballasting pumps are expected to be of a continuous nature, given that they use the same type of propellers as ballasting ships. These will also generate low-frequency tones according to their dimensions and rotational speed specifications; however, little literature is available that studied this specific component as it is not a ship's main source of underwater radiated noise (URN). Nevertheless, the pumps are activated when the wind changes directions; therefore, depending on the volume of water to be transferred, they are active for a few minutes when such an event occurs, rather than pumping constantly.

The sound emissions from the moorings of floating turbines are comparable to chains and catenaries regularly used for offshore moorings, for instance, for ocean buoys collecting long-term data. While it is common knowledge that such moorings generate sound as they move vertically and horizontally in the water column with the movement of waves, current, and tide, their signature has not been described in the literature. Typically, this source is referred to in the available literature as an "undesired" source because studies are dedicated to characterizing anthropogenic activities and mooring noise is not the target of the investigation. Moorings on buoys dedicated to acoustic monitoring are therefore designed to be as quiet as possible to ensure that this source does not impact the quality of the data by masking the target source(s) or clipping the recordings (Robinson et al. 2014). Sounds from moorings are transient, but they can occur very regularly throughout a mooring's deployment. Therefore, characterizing and assessing the potential impacts of these elements requires further investigation. Transient noises (at ~23 second intervals) with $L_{0\text{-}p}$ greater than 160 dB re 1 μPa likely related to tension releases in the moorings from floating WTGs were described for the HYWIND demo project (Martin et al. 2011), which was a single, small-scale, floating turbine installed offshore of Norway.

## Acoustic Data Collection

All underwater sound data was measured using JASCO's Autonomous Multichannel Acoustic Recorders (AMARs). One recorder (an AMAR G4) was positioned in the west-northwest area of the HYWIND site (Fig. 3). This recorder was 642 m from the turbine HS-1.

A second recorder (an AMAR G3 with a single omnidirectional hydrophone) was deployed approximately 13 km from the HYWIND site to collect "control" data for ambient sound.

## Recording Specifications

Both AMARs were fitted with GeoSpectrum Technologies Inc.'s omni-directional hydrophones (i.e., M36-V35-100 with a nominal sensitivity of −165 dBV/μPa with pre-amplification). Flow noise minimization was achieved by using an acoustically transparent shroud around the hydrophone cade.

The four hydrophones of the HYWIND mooring were mounted in an orthogonal arrangement with a nominal 50 cm spacing to allow the time of arrivals from different directions to be determined (Fig. 2). Precise measurements of the relative distance between hydrophones were taken and used as inputs for the directional calculations.

The AMARs recorded continuously at a 64,000 Hz sample rate to return a recorded bandwidth of 10–32,000 Hz. The recording channels had a 24-bit resolution with a spectral noise floor of 32 dB re 1 $\mu Pa^2$/Hz and a nominal ceiling of 165 dB re 1 μPa. Acoustic data were stored on 13 × 512 GB flash memory cards for each instrument. JASCO field staff calibrated the AMARs before deploying and after retrieving them. Calibrations were performed using a pistonphone type 42 AC precision sound source (G.R.A.S. Sound & Vibration A/S). At both sites, the AMARs logged data 24 h per day from 21 October 2020 to 24 January 2021 (Table 1). Approximately 6.6 TB of data were collected.

## Acoustic Data Analysis

The acoustic analysis methods and the terminology used were in accordance with ISO 18,405:2017 (ISO 2017).

The bearing angles of each hydrophone were calculated based on their relative distances and positions with respect to each wind turbine (Table 2). The correct orientation of the array was further confirmed by correlating the bearings obtained from the analysis of the signature of the vessel used for deploying and retrieving the recorders, *SWATH 1* with actual bearings calculated from the positions of the vessel's GPS track log.

The orthogonal hydrophone array allowed for the spatial discrimination of bearing and elevation of detected sound. Such data was stored for individual events

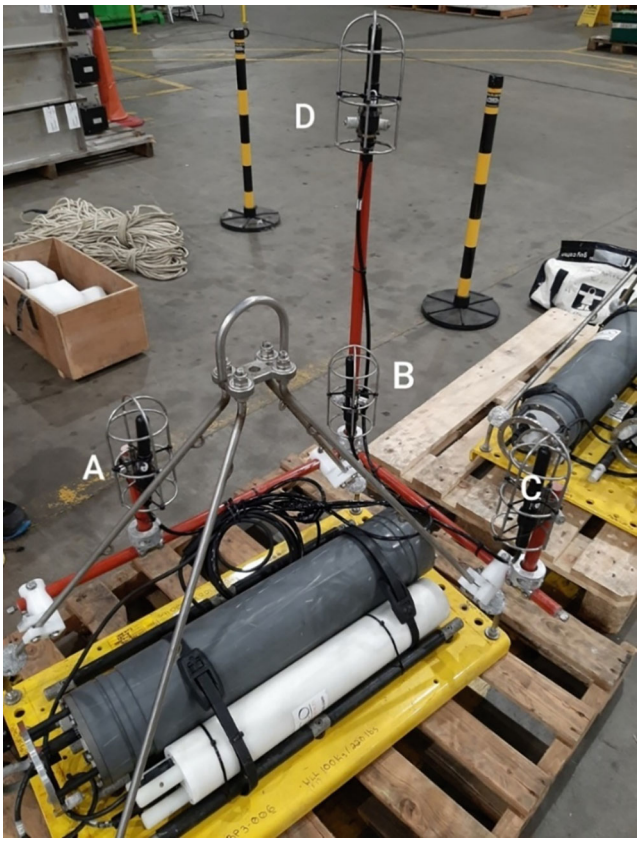



**Fig. 2** Photo of the HYWIND equipment before it was deployed. The assembly includes an AMAR G4 (white tube) fixed on a baseplate, a battery back (gray tube), and the four hydrophones mounted on a fiberglass frame (red tubes)

for later analysis. The bearing to each time/frequency cell is color coded and displayed using directograms (Fig. 3); this type of image shows the direction of a sound source relative to the array. The radial resolution obtained was 22.5 degrees. The color wheel represents sound originating from different directions. For instance, sound propagating from the north of the AMAR would be presented in a shade of red, while from the south would be colored in light blue.

Impulses were detected using a Teager-Kaiser energy detector (Kaiser 1990, Kandia and Stylianou 2006). Two distinct tonal continuous sounds below 100 Hz were identified during the analysis; these were identified as part of the floating turbines' signature. Both tones would affect the performance of the impulse detector and were therefore filtered out of the transient signals analysis. For the latter, a high-pass filter was applied (stop band $f = 85$ Hz, passband $f = 100$ Hz). A 100 ms energy time series was obtained by filtering the squared time series, then summing over a 100 ms window, and then dividing by the number of samples in the window. The

**Table 1** Dates and locations the two AMARs deployed (UTM 30 N, WGS84). Retrieval times (UTC) indicate the time when each retrieval started

| Location | Deployment | Retrieval | Latitude | Longitude | Easting | Northing | Water depth (m) |
|---|---|---|---|---|---|---|---|
| HYWIND | 21 Oct 202013:44 | 15 Jul 2021 10.30 | 57° 29.109′ N | 001° 20.571′ E | 599348.894 | 6372604.308 | 112 |
| Control | 21 Oct 202011:43 | 24 Jan 2021 13:05 | 57° 31.801′ N | 001° 07.580′ E | 612189.991 | 6377935.635 | 95 |

**Table 2** Distances and bearings of each HYWIND WTG from the AMAR deployed within the Hywind Scotland OWF from Burns et al. (2022)

| WTG | Distance (m) | Bearing (°) |
|---|---|---|
| HS-1 | 605 | 095 |
| HS-2 | 880 | 318 |
| HS-3 | 2242 | 308 |
| HS-4 | 951 | 1799 |
| HS-5 | 1799 | 269 |

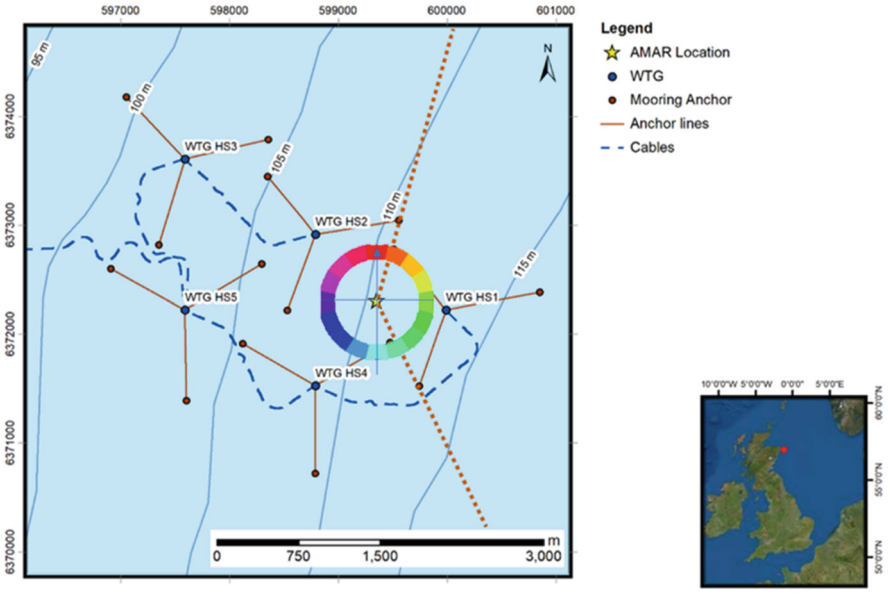



**Fig. 3** Map showing the layout of the wind turbines. The color wheel is oriented relative to north

time series was divided by its mean value for each 20-s long data buffer that was passed to the Teager-Kaiser operator (Kaiser 1990, Kandia and Stylianou 2006). Normalizing this buffer by its mean value allowed for the use of a fixed threshold that was independent of the absolute magnitude of the raw time-series data. A detection threshold of 15 was chosen for the Teager-Kaiser operator. The detector

was set with a 1.0 s "lock-out" after each strike was detected; this minimized false alarms on multipath arrivals.

## Kurtosis Estimation

Understanding whether signals are impulsive or non-impulsive is important when selecting which hearing threshold shift regulatory criteria to apply (NMFS 2018). In recent years, kurtosis has been suggested as an objective metric to determine the impulsiveness of a sound source (Martin et al. 2020, Müller et al. 2020). Kurtosis was therefore calculated for the evaluation of the broadband transient signals identified in the recordings to determine the most appropriate applicable criteria for impact assessment.

Kurtosis ($\beta$) is defined as the ratio of the fourth moment to the squared second moment of the instantaneous sound pressure:

$$\beta = \frac{\frac{1}{N}\sum_{N}(p_i - \bar{p})^4}{\left[\frac{1}{N}\sum_{N}(p_i - \bar{p})^2\right]^2} \quad (1)$$

where the $i$th sample of the instantaneous sound pressure is $p_i$, the sound pressure's arithmetic mean is $\bar{p}$, and the number of data samples in the analysis window that affects the resulting value for kurtosis is $N$. As Martin et al. (2020) suggest, this project applied a 60 s analysis window. A kurtosis value of 3 represents random Gaussian noise. When deciding if a soundscape is impulsive to help determine whether an impulsive or non-impulsive hearing threshold shift threshold is exceeded, kurtosis value of 40 is used as the threshold (NFMS 2018). A kurtosis value of approximately 3 also represents wind-driven underwater ambient sound.

## Acoustic Modeling

Sound propagation modeling was performed for two purposes: to compute the source level of the sounds emitted from the turbines (i.e., back-propagation) and then to forward propagate the source levels to determine sound levels as a function of distances from the turbines. The modeling assumed that a point source could represent the turbine spar (cylinder shaped). The reported nominal draft of the spar was 78 m, with the point source being modeled at the mid-point (i.e., source depth of 39 m).

JASCO's Marine Operations Noise Model (MONM)(Austin and Chapman 2011), which is based on the parabolic equation (Collins 1998), was used for low-frequency propagation (less than 1 kHz) combined with BELLHOP for frequencies above 1 kHz.

Decidecade bands in the range 10 Hz–25 kHz were modeled. The algorithm developed by Harrison and Harrison (1995) was used to consider sounds across each frequency band (as opposed to individual frequencies). This algorithm uses a

range-dependent smoothing factor to simulate a frequency smearing in frequency, eliminating peaks and troughs in the sound field.

### HS-2 Source Levels Estimation

Source levels ($L_s$) were estimated for a single turbine (i.e., HS-2, the dominant in terms of received levels among the three turbines) according to:

$$L_S = L_p + N_{PL}. \quad (2)$$

where $L_s$ is the source level, $L_p$ is the sound pressure level (1 min), and $N_{PL}$ is the propagation loss, as per (ISO 2017) terminology.

It is only possible to back-propagate the sound levels recorded at the HYWIND monitoring station. As such, the full frequency spectra from 10 Hz to 25 kHz were included. In the low frequencies (i.e., below 100 Hz), the sound signature of the wind turbine also included the tonal components (peaking around 24 and 75 Hz) that are associated with the electrical power generation and other unknown mechanical sources within the turbines. These are not described in detail in this publication.

The source levels presented may be overestimated when the difference in sound levels between the Control and Hywind Scotland sites at specific frequency bands is small. This is because the ambient sound in that frequency band would be back-propagated.

Figure 4 shows the propagation loss ($N_{PL}$) calculated

The source levels calculated for the turbines at the different wind speeds are shown in Fig. 5 for the median (50th percentile) sound levels and Fig. 6 for the 5th percentile sound levels. For each analyzed wind speed, Table 3 shows the broadband back-propagated source levels in the 5th, 25th, 50th, 75th, and 5th percentiles.

There was a trend of increased sound levels with increased wind speed, as well as changes in the spectral characteristics. Noticeable peaks appeared at 20–25 Hz, 40 Hz, and 63–80 Hz and also at 1.0–1.25 kHz, especially at 20 knots and with the addition of a peak at 200–250 Hz at 25 knots (Burns et al. 2022).

## Results

Analyzing the acoustic data revealed a substantial amount of mooring noise. Several energy peaks were observed between 100 and 400 Hz. These appear to be associated with the regular transient sounds of the HYWIND mooring system components. The sounds are described as “snaps,” “creaks,” and “rattles.” These generally broadband transients occur irregularly, and their duration also varied, with typical durations between 0.2 and 1.0 s.

The transient sounds from the moorings are broadband, repetitive, and considerably less impulsive. These sounds are three types, described as “bangs,” “creaks,” and “rattles” during aural analysis. Aural analysis reveals that an action of tension

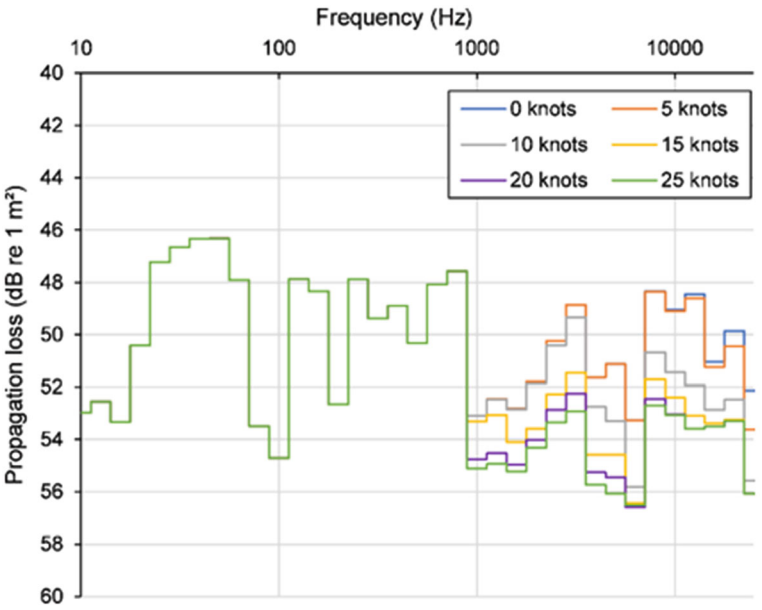



**Fig. 4** Propagation loss between the point source midway down the spar and the HYWIND recorder (880 m distance) for varying wind conditions. The propagation loss for bands below 1 kHz was calculated using the parabolic equation model that does not account for sea surface roughness (i.e., variation with wind speed). (©JASCO Applied Sciences)

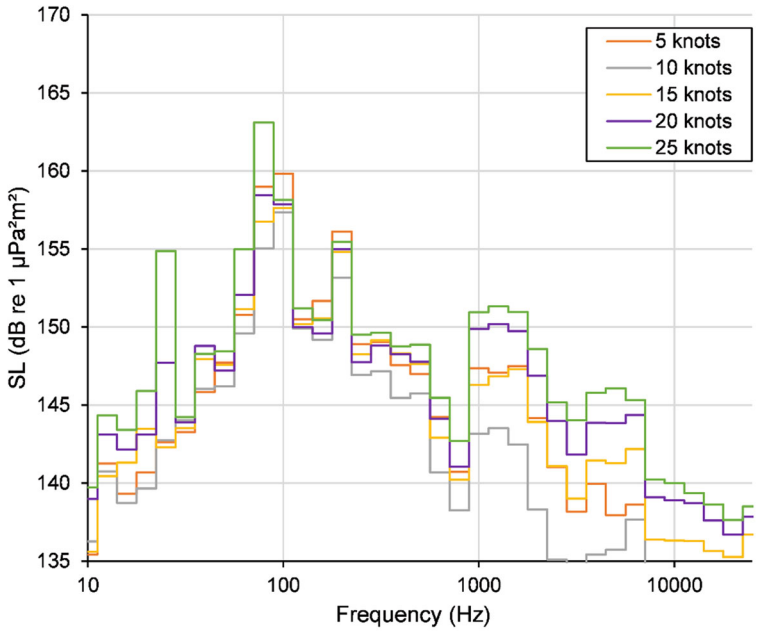



**Fig. 5** Median source levels calculated for turbine HS-1 at different wind speeds

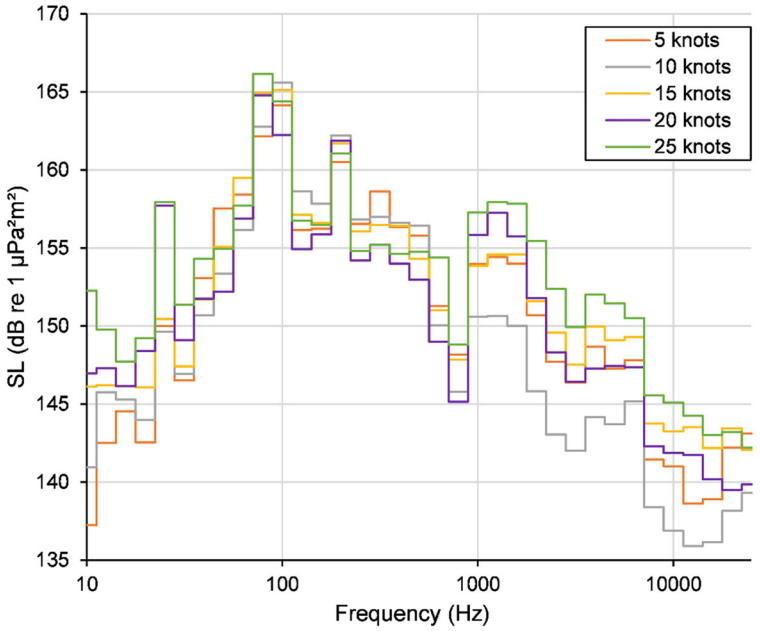



**Fig. 6** 95th percentile source levels calculated for turbine HS-1 at different wind speeds

**Table 3** Back-propagated broadband source levels (unit: dB re 1 $\mu Pa^2m^2$) for the 5th, 25th, 50th, 75th, and 95th percentiles

| | Broadband source level | | | | |
|---|---|---|---|---|---|
| Wind speed (kn) | 5th percentile | 25th percentile | Median percentile | 75th percentile | 95th percentile |
| 5 | 158.9 | 161.5 | 165.1 | 167.2 | 170.5 |
| 10 | 156.7 | 160.1 | 162.5 | 165.6 | 170.8 |
| 15 | 159.6 | 162.0 | 163.9 | 166.5 | 171.4 |
| 20 | 160.4 | 162.8 | 164.8 | 167.2 | 170.6 |
| 25 | 162.1 | 164.9 | 167.2 | 169.3 | 172.0 |

release occurs in many of these noises. The rapid onset combined with the significant intensity of the sound visible in the spectrograms supports this.

In a 20-min segment recorded on 1 January 2021, human analysists manually annotated over 300 detections (Fig. 7). The audible broadband mooring transients shown on the directogram in Fig. 7 are consistently colored blue, pink, or green. This indicates that there were three, discrete, separate sources. The directions of HS-1 (eastward), HS-4 (south-westward), and HS-2 (north-westward) correspond exactly to the direction of the blue, pink, and green sectors on the color wheel. HS-1 and HS-4 are clearly generating less mooring noise than HS-2.

The transients originating from relatively small bearing clusters, aligned with HS-1, HS-2, and HS-4, were each acoustically different and distinct. There were also similar sound pressure levels for each cluster, which indicates a consistent source

generation and distance (Figs. 8, 9, 10, and 11). The same type of mooring movement, such as surfaces rubbing under tension, appears to be generating the sounds. Most transient sounds are a repetition of staccato sounds; however, this mooring noise appears to have unique spectral and temporal identities for each turbine.

The low-frequency "bang" sound (green in Fig. 8) was associated with HS-1. The quietest of the three sounds, the bang was dominated by the lowest frequencies

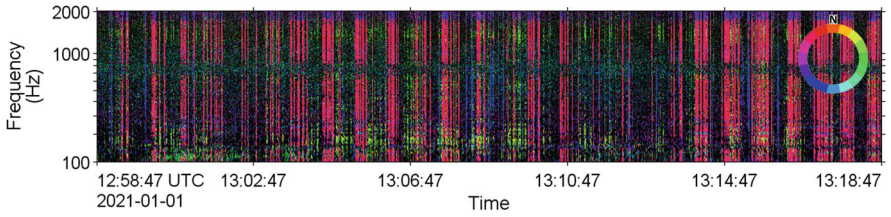



**Fig. 7** Directogram of 20 min of data during winds greater than 20 knots on 1 January 2021

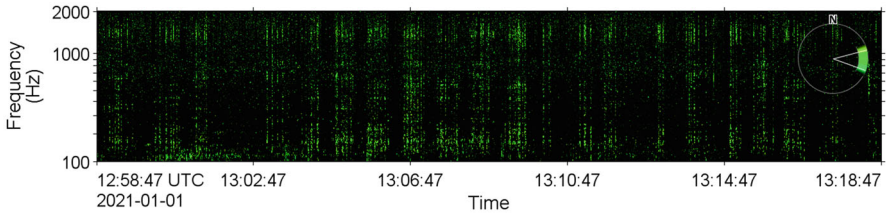



**Fig. 8** Directogram showing only transients received from the direction of HS-1

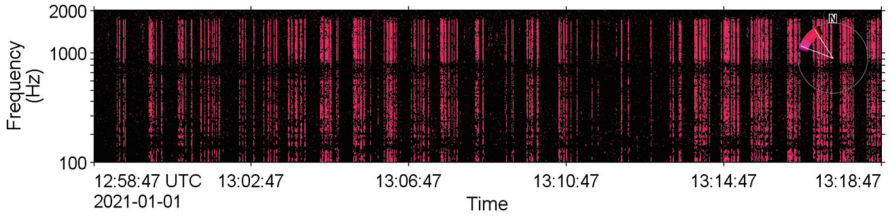



**Fig. 9** Directogram showing only transients received from the direction of HS-2

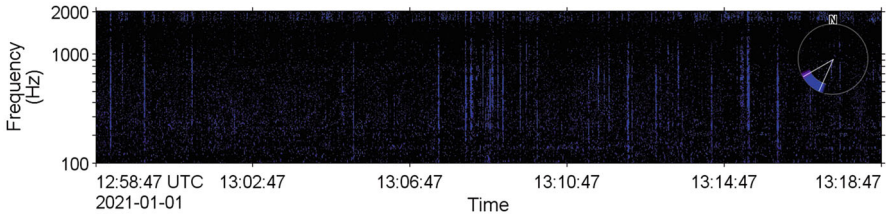



**Fig. 10** Directogram showing only transients received from HS-4 turbine

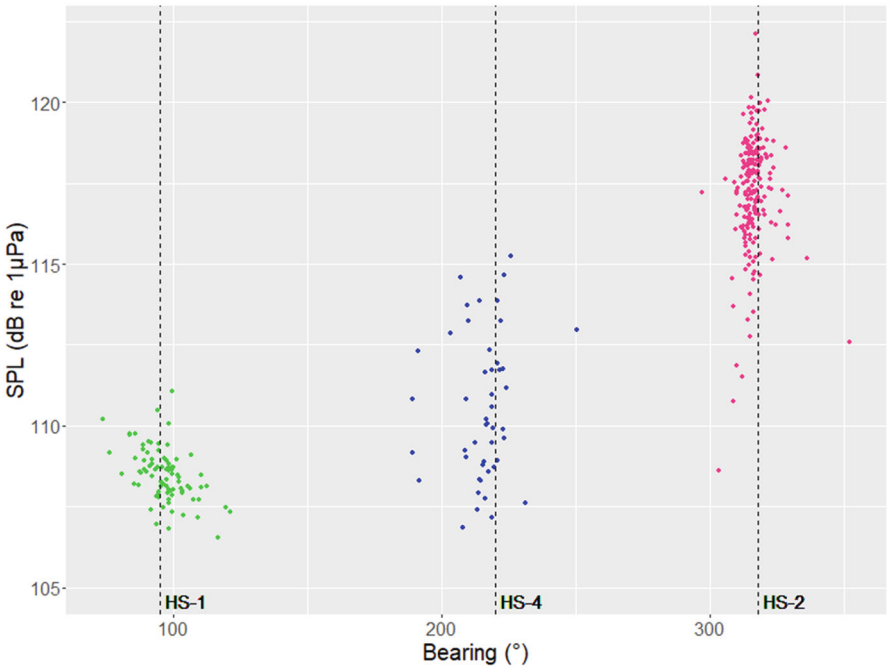



**Fig. 11** Plot of the transient bearing from the AMAR against sound pressure level (SPL) for 20 min on 1 January 2021. HS-1 "bang" transients (green), HS-2 "creak" transients (pink), and HS-4 "rattle" transients (blue). (Burns et al. 2022).

(<250 Hz). While these bangs were usually the shortest-duration sounds, they could, however, last for more than 2.5 s, with occasional instances of prolonged reverberation. This bang was not always audible because of masking by other sounds (or potentially by other ambient sounds).

HS-2 generated a "creak" transient (red in Fig. 9). This had the highest SPL and masked the other two sounds when their occurrence overlapped. The duration of individual sounds was around 0.4 s, with some up to 1.5 s. This sound often had frequencies absent from the 700–900 Hz band.

HS-4 was associated with a "rattle" type sound (blue in Fig. 10). The duration of individual events varied from approximately 0.5 to 1.0 s. It sounded like a repeated tapping noise, with variable numbers of taps. The blue sounds had most of their energy in the 200–800 Hz band.

The variability in bearings versus received SPL for each transient sound type is presented in Fig. 11; this shows that the mooring noise levels differed for each HYWIND turbine and that the sound originated close to the bearing of the pillars. This indicates that the noise was restricted to the mooring components close to or at the pillar, rather than along the mooring chains. It is unclear why one mooring might be noisier than another when both experience the similar dynamic stresses.

The occurrence of mooring transients showed a positive correlation with wind speed but that was non-statistically significant. At higher wind speeds, the amount of mooring noise increased and substantially contributed to the wind farm's overall noise signature.

There was a stronger correlation between the occurrence of transient sounds and significant wave heights. This positive correlation may be due to a greater influence of waves, rather than wind speed, in causing the structure to heave, which therefore caused the mooring system to heave more.

The different transient sounds caused by the mooring were not mutually exclusive, and the data commonly showed combinations of different types of noise. This finding is consistent with the discovery that each HYWIND system seems to produce a unique type of mooring noise.

## Impulsiveness Analysis

A qualitative assessment of the impulses indicated that they tended to be on the order of 1 s long (Ainslie et al. 2021) and that multiple sub-pulses and/or tones were presented within the pulse. The pulses' peak pressures and rise times are clearly identifiable from the background but not as much as usual pile driving or seismic airgun impulses.

The detected impulses had interquartile durations of 1.3–1.7 s, which is longer than the conventional way an "impulse" is defined. As expected, a greater number of impulse detections at higher wind speeds at HYWIND were found by the detector; this was not replicated at the Control site (Fig. 12). As expected, the number of impulse detections at HYWIND decreased with increasing wind speed; above 100 Hz more wind-driven noise is present, which made detecting the impulses a more difficult task. One output of the impulse detector is the per-impulse sound exposure level in decidecade bands, and in Fig. 13 the mean, median, and 90th percentile of these are shown. Figure 14 shows auditory frequency weightings from

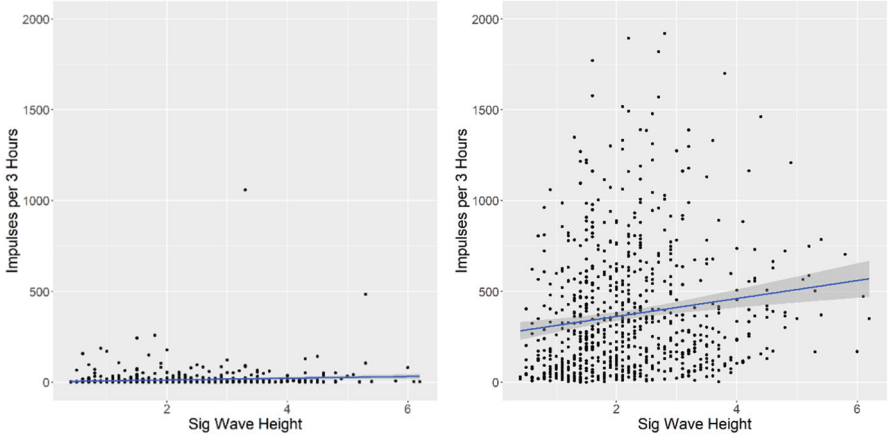



**Fig. 12** Scatterplot of the number of impulse detections per 3 h versus the significant wave heights at (left) the Control site and (right) HYWIND. (Burns et al. 2022)

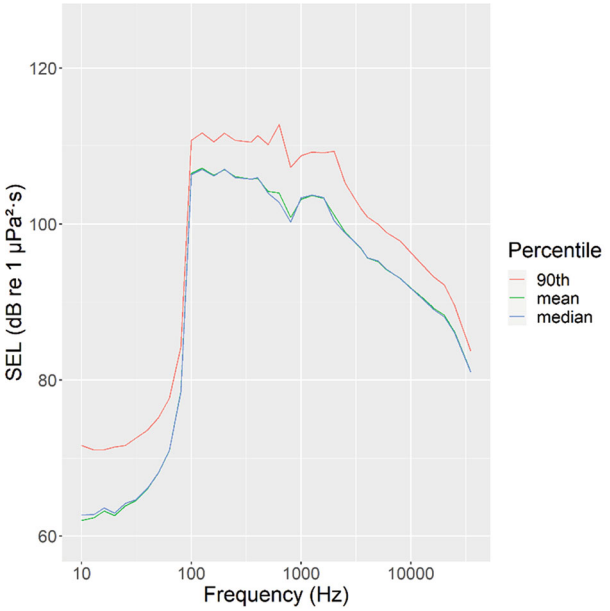



**Fig. 13** The decidecade sound exposure levels (SEL) of the impulses detected at HYWIND in the mean, median, and 90th percentile compared with the decidecade center frequencies

the NMFS (2018) were applied to the decidecade SEL to determine the daily SEL. The impulsive SEL weighted high-frequency cetaceans were mostly 6 dB below the total daily SEL as shown Fig. 15, and they were commonly more than 10 dB below the total. The observation that there were more impulses at the HYWIND site (e.g., Fig. 12) is supported by comparing the kurtosis at HYWIND and the Control site, as shown in Fig. 16. However, highly impulsive minutes ($\beta > 40$) were exceptionally rare at both HYWIND and the Control site. This kind of kurtosis distribution from a sound source is an excellent example of how applying a kurtosis-weighted SEL to predict hearing threshold shifts is valuable for regulatory applications (Zhao et al. 2010, see ▶ Chap. 81, "Impulsive or Non-impulsive: Determining Hearing Loss Thresholds for Marine Mammals," by Zeddies et al.).

Use of the non-impulsive threshold from NMFS (2018) is recommended because minutes with a high kurtosis occurred relatively infrequently, the length of the impulses, and the impulsive SEL being 6 dB or higher below the total SEL. That is, the Temporary Threshold Shift (TTS) for high-frequency cetaceans (porpoise) would be 153 dB re 1 $\mu Pa^2s$.

## Marine Mammal Impact Modeling

Marine mammal impact modeling was performed using the same methods as the back-propagation; however, individual point sources representing the five turbines were modeled over 360° of radial transects. This provided a full three-dimensional

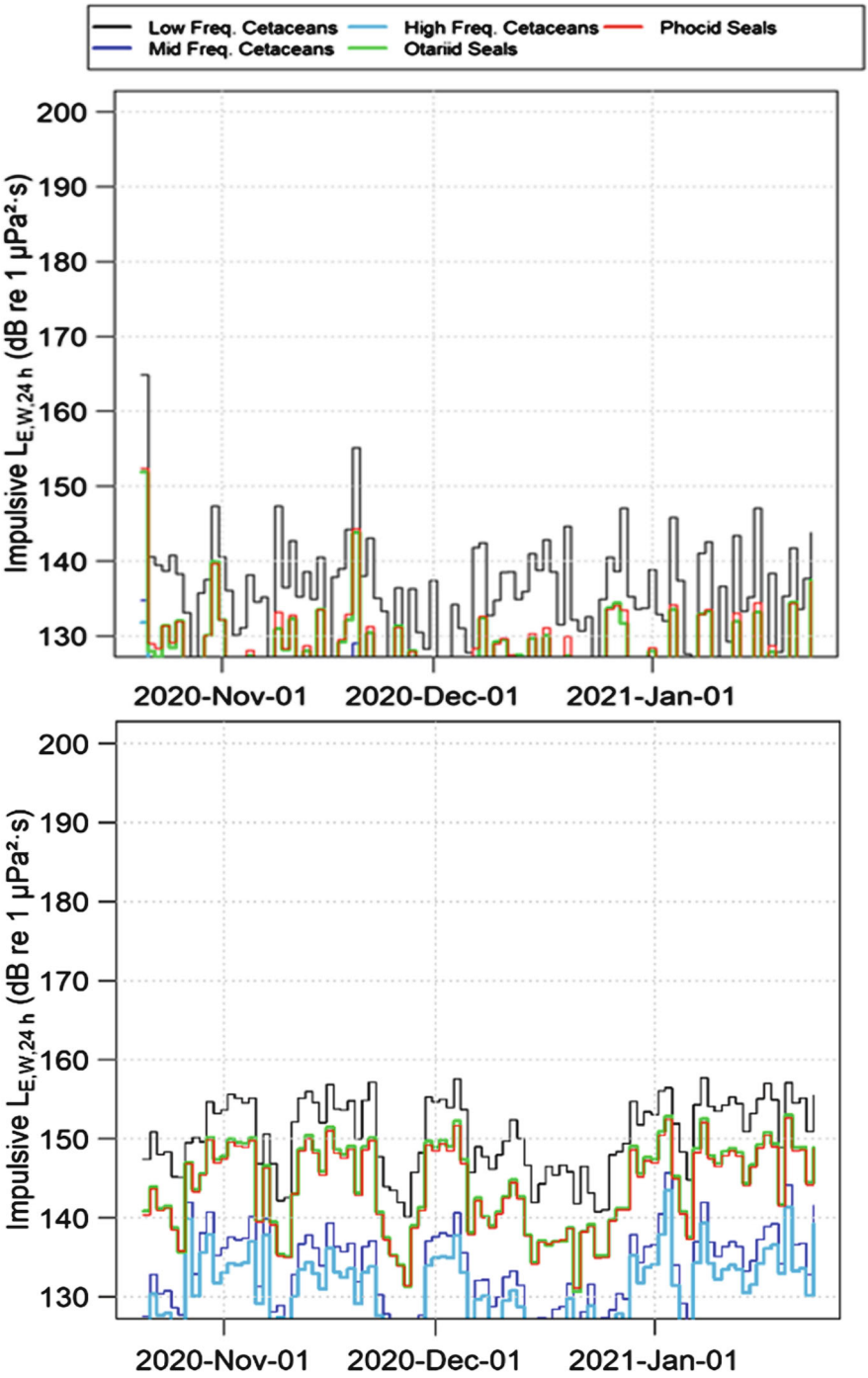



**Fig. 14** Daily auditory frequency weighted sound exposure level (SEL) for the transient events detected at (top) the Control site and (bottom) HYWIND

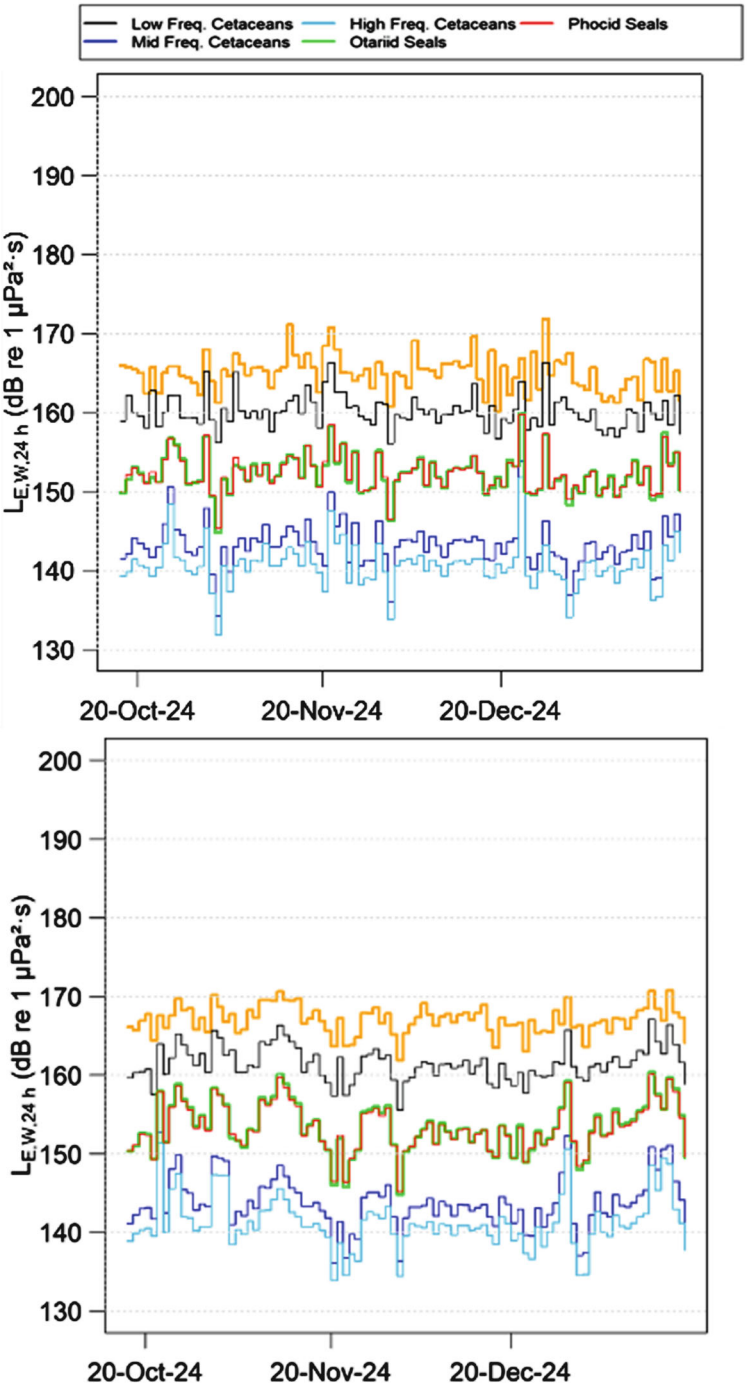



**Fig. 15** Daily auditory frequency weighted sound exposure level (SEL) for full days at (top) the Control site and (bottom) HYWIND

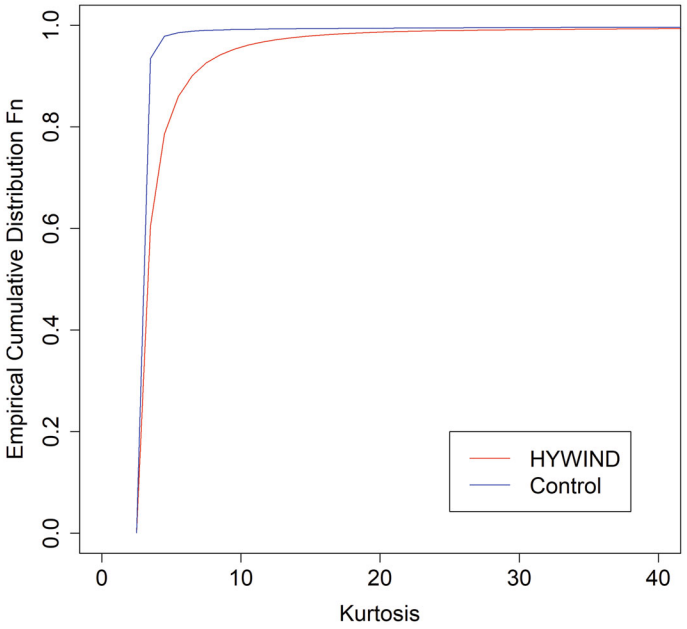



**Fig. 16** Empirical cumulative distribution functions for the 1-min kurtosis at the Control and HYWIND sites

(3-D) sound field. Maps of sound fields indicate the areas exposed to sound at certain levels. Figures 17 and 18 show examples representing the SPL for the 10-knot 50th-percentile and the 25-knot 95th-percentile sound fields, respectively.

The predicted levels for given conditions over the wider area, as well as the Southall et al. (2019) recommended maximum distances to impact thresholds, can be estimated by the model. In Table 4, the modeled weighted received SEL over 24 h for the considered auditory groups are shown, again presuming a constant windspeed of 15 knots, for the median and 75th percentile cases. In Table 5, the maximum modeled distances from the nearest turbine to the calculated isopleths associated with auditory impacts are shown, assuming a 24-h exposure at a 15-knot wind speed and taking the median and 75th percentile cases (i.e., sound levels that are constant over a 24 h).

Over 24 h, sound levels were dominated by the highest levels within the recorded timespan. This means that the modeled, 24-h, 75th percentile sound level matches the recorded daily sound level more closely than the 50th percentile level. The average result skews toward the higher sound levels when sound levels are averaged over a longer timespan (e.g., over 24 h).

The maximum distances to the recommended TTS-onset sound levels for the auditory groups of interest (see Table 5) are those for receivers that are stationary for

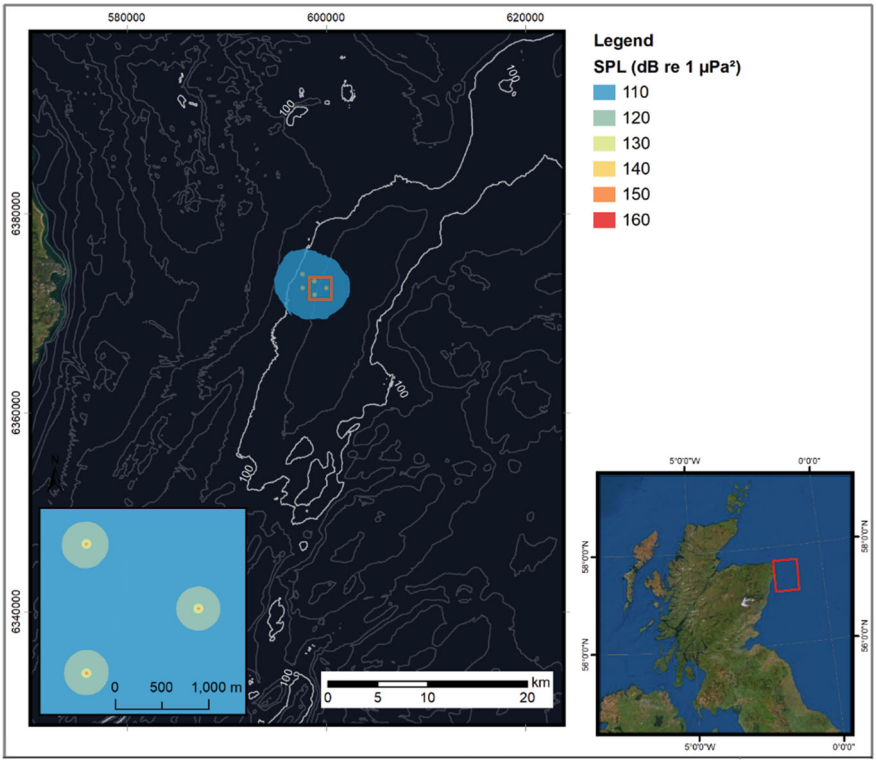



**Fig. 17** The radiated sound field modeled at the HYWIND site. The model assumed a 10-knot wind speed for the 50th percentile source levels. Datum WGS84 projection UTM30N

the exposure duration at the depth at which the sound level is at its loudest. The maximum distance to the TTS isopleth for the cases analyzed, when considering the 75th percentile, is for the very high-frequency cetacean hearing group at an 80-m maximum distance. Sound levels were highly variable, so the exact distances vary.

## Conclusions

The AMAR located at the HYWIND site was approximately equidistant from the three monitored turbines and it was situated to isolate HS-1 to the east. This set up allowed for an uncontaminated directional analysis of one HYWIND Scotland system.

A substantial contribution to the overall noise field from the wind farm was found to be constituted by several transient sounds thought to be associated with the mooring system. Despite the design of the mooring system being identical for

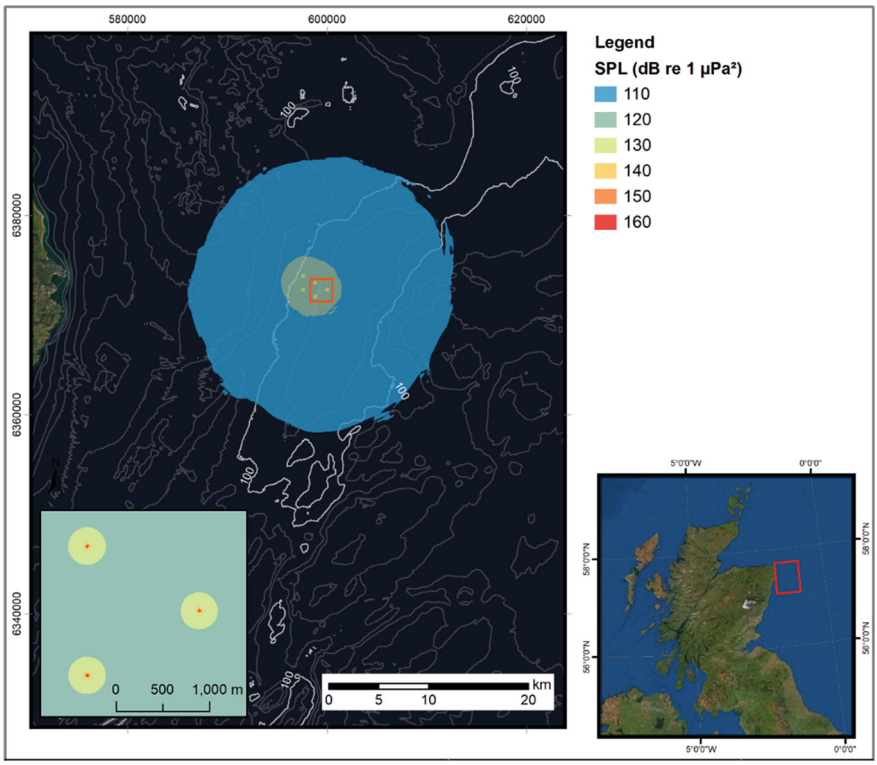



**Fig. 18** The radiated sound field modeled at the HYWIND site. The model assumed a 25-knot wind speed for the 95th percentile source levels. Datum WGS84 projection UTM30N

**Table 4** Modeled median and 75th percentile 24-h SEL ($SEL_{24h;}$ units: dB re 1 $\mu Pa^2m^2s$), assuming a wind speed of 15 knots. The auditory group frequency weightings are from Southall et al. (2019)

| Auditory group | Auditory weighting function | Weighted received $SEL_{24h}$ | |
|---|---|---|---|
| | | 50th percentile | 75th percentile |
| Low-frequency cetaceans | LF | 157.9 | 158.3 |
| High-frequency cetaceans | HF | 139.5 | 141.3 |
| Very high-frequency cetaceans | VHF | 137.2 | 138.8 |
| Phocid carnivores in water | PCW | 148.9 | 150.4 |
| Other marine carnivores in water | OCW | 148.8 | 150.5 |

each monitored turbine, unique signatures were identified for each of the three WTGs characterized. This means that it would be difficult to predict the characteristics of mooring sounds generated in a proposed wind farm installation during the impact assessment stage.

**Table 5** Modeled maximum distances (unit: m) to the temporary threshold shift (TTS; unit: dB re 1 μPa$^2$s) threshold levels from Southall et al. (2019), assuming a wind speed of 15 kn

| Auditory group | Auditory weighting function | TTS onset level | Maximum distance to weighted $SEL_{24h}$ TTS isopleth | |
|---|---|---|---|---|
| | | | 50th percentile | 75th percentile |
| Low-frequency cetaceans | LF | 179 | 40 | 50 |
| High-frequency cetaceans | HF | 178 | 10 | 20 |
| Very high-frequency cetaceans | VHF | 153 | 50 | 80 |
| Phocid carnivores in water | PCW | 181 | 20 | 30 |
| Other marine carnivores in water | OCW | 199 | <10 | <10 |

Unlike the HYWIND demonstration system recordings from 2011, there was little evidence of highly impulsive, "snap" noises (Martin et al. 2011). However, there was a much great amount of transient noise from moorings. This mooring noise had distinct characteristics related to individual HYWIND systems. At HS-1, "bang" was produced, at HS-2 there was a "creak," and at there was HS-4 a "rattle." The directional processing of the AMAR array data showed that each type of sound was clearly connected with each system. Wave height positively correlated with occurrence. As wave height increased, there was a substantial corresponding increase in the contribution of the transient sounds to the overall sound signature of the wind farm.

A quantitative analysis of the impulsiveness of the HYWIND soundscape applied an impulse detector and studied the distribution of the per-minute kurtosis. Summing the SEL of all detected impulses showed that the impulsive SEL was generally less than 6 dB below the daily total SEL. The mean timespan of the impulses was ~1.5 s, longer than the 1.0 s interval typically used to identify impulses to assess the effects of sound on hearing. The soundscape at the Control site had a lower kurtosis than at HYWIND; however, the kurtosis value for HYWIND was too low to be considered impulsive. Therefore, non-impulsive TTS SEL thresholds should be used to assess noise from wind farm.

The impulses per hour rate more positively correlated with wave height than with wind speed. A subjective aural analysis of these noises suggests a slightly lower rate of tension release in this particular mooring design; however, there is currently no quantifiable method with which to confirm this. Directional analysis of noise from the HS1, HS2, and HS4 moorings revealed that the noise came from mooring components at or from somewhere near the pillar itself. No available evidence suggests that the mooring noise was generated from the mooring system away from the bridle connection points on the HYWIND spar. This finding is reasonable since the most substantial movement in the mooring system likely occurs that at the points where the chains are connected around the pillar because anchor chains fall

through a ballasted catenary to the ocean floor and then sinks in the soil a distance away from the anchor. When analyzing mooring noise occurrence with wind speed only a limited correlation was found, but when comparing it with wave height a more direct relationship was identified. This result suggests that the dominant environmental factor influencing movement and associated mooring component friction in the floating structure is wave height.

The NMFS (2018) thresholds for temporary hearing thresholds shifts caused by non-impulsive sounds were applied to evaluate the possible effects on marine mammals by noise at HYWIND. The threshold for TTS was never exceeded and the daily marine mammal weighted SEL at both sites were very similar.

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# Underwater Radiated Noise from Leisure Boats: Measurement Using an Interim Guide

14

Mathias H. Andersson, Emilia Lalander, Peter Sigray, Mikael Svedendahl, and Martin Östberg

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**Abstract**

The knowledge of the noise characteristics stemming from recreational boats is scarce, and the amount of studies is too small to draw any conclusions on impact and recommend any measures. To close this gap, it is necessary to measure the underwater radiated noise (URN) from boats and estimate their source level. This chapter aims to describe the development of a methodology for source-level estimations of recreational boats based on classification society's guidelines and test it in field trials. Three boats with different engine, propeller, and fuel types, including one electric motor, were used. Parameters such as speed, measurement depth, distance, and propagation loss estimations are varied, aiming to study their effect on the estimated source level. By combining the GPS position of the boats and sound data from a hydrophone system, successful measurements of the boats' radiated noise were conducted, and their source levels were estimated. The source spectra of all boats showed clear tones, stemming from the rotational frequencies of the engines and propellers. Finally, a measurement guideline was developed for estimation of the source level from recreational boats that can be used for comparison with published scientific studies and environmental impact studies and as input to source models.

**Keywords**

Recreational boat · URN · Source level · Measurement methodology

## Introduction

Underwater radiated noise (URN) from a recreational boat is generated by the rotation of the propeller and associated cavitation, the firing sequence of the engine(s), other moving parts of the propulsion system, and the release of subsurface exhausts (Matzner et al. 2010; Erbe 2013; Erbe et al. 2016 etc.). Measurements or modeling of the propagation loss makes source-level estimations possible. The magnitude and frequency content of the source level define the acoustic signature, which depend on how the sound is generated, engine type (inboard and outboard), number of cylinders, propeller type and type of craft, and its load (Matzner et al. 2010; Erbe et al. 2016; Parsons et al. 2021). There are numerous variations and combinations of these parameters, making general conclusions of the acoustic signature a challenge.

The large quantity of recreational boats in shallow water archipelagos has the potential to pollute sensitive biological areas with high levels of underwater noise, resulting in an environmental issue that urgently needs to be assessed (Jain-Schlaepfer et al. 2018; Magnhagen et al. 2017; van der Knaap et al. 2022). In contrast to commercial ships, the knowledge of noise characteristics from recreational boats is lacking, and, at present, there is little evidence to draw any firm conclusions on an environmental impact and even less to recommend any measures.

The boat engine noise is commonly characterized by both broadband and narrowband (tonal) components below approximately 500 Hz. These tones are speed dependent and relate to the number of propeller blades, gear design, revolution per minute (RPM), and other engine characteristics (Matzner et al. 2010; Pollara et al. 2017). Propellers are known to produce amplitude-modulated noise through cavitation mostly noted below 500 Hz (Pollara et al. 2017). The cavitation mainly depends on propeller design, speed, and maintenance of the propeller, e.g., a damaged propeller can induce more cavitation than an undamaged one.

Electric motors are considered quieter underwater than gasoline and diesel engines, but at the time of writing, there were no published studies of URN from recreational boats with electric motors. However, for larger commercial vessels, there are measurements showing electrical ships being less noisy than conventional ships (Parsons et al. 2020).

The chapter describes four overall goals: (1) to develop a methodology for source-level estimation of recreational boats based on existing guidelines for commercial ships; (2) to test it in field trials; (3) to study the effect on the source-level estimation of boat speed, measurement depth, direction, and measurement distance; and (4) to develop a measurement guideline for the source-level estimation for recreational boats based on the experiences from the measurements. This chapter is based on Svedendahl et al. (2021), and more details and results can be found therein.

The described approach is based on the guidelines of the classification society Bureau Veritas (2018) with the aim of testing the efficiency of such methods for use on recreational vessels in shallow water. It is recognized that the methods themselves and the meaning of some key terms (e.g., source level, propagation loss, and radiated noise level) may differ from those adopted by recent ISO standards such as ISO 18405 and ISO 17208 parts 1 and 2. However, in the absence of an ISO standard on measuring ship noise in shallow water (ISO 17208-3 being at only Committee Draft stage), and for consistency with the approaches in the classification society guidance, the terminology and basic methodology used in this chapter and indicated in the text where adaptations has been made.

The work described in this chapter was funded by Vinnova, Sweden's Innovation Agency, together with the Swedish Transport Agency, which in turn was supported from the Swedish Agency for Marin and Water Management grant 11.1.

## Standardization of Measurement and Data Analysis

### Current Status of Standardization

Currently, no measurement standard exists for recreational boats. There are several different standards (ANSI 2009; ISO 2016, 2019) and classification societies (Det Norske Veritas AS 2011; Bureau Veritas 2018, etc.) for measurement and estimation of the source level or radiated noise level of commercial ships. The classification societies also have notations to classify a ship as "silent" in various categories

depending on ship class. The guidelines have different kinds of grades, representing different levels of accuracy in terms of the estimated source level (Ainslie et al. 2022). However, none of these guidelines or notations applies to recreational boats. These boats are smaller and have less powerful engines, which most likely result in lower source levels. It is challenging to follow the measurement geometry of the guidelines and class notations. For example, many guidelines limit the minimum measurement distance to a few hundred meters, which could decrease the signal-to-noise ratio of the measurement of a more silent and smaller recreational boat. With shorter measurement distances, the length of the measurement track will be affected, as well as the post-processing of data, as some guidelines divide the measurement into time segments corresponding to a certain distance covered by the boat.

In all above-referenced methodologies for the estimation of a vessel's source level, there are uncertainties at different levels related to the measurements, equipment, and signal processing. The level of uncertainties in a source-level estimation can be divided into different grades. The American National Standards Institute (ANSI) offers three grades of measurement, each with a stated applicability, test methodology, system uncertainty, and repeatability of the measurement. The grades have different requirements on the hydrophone system, ranging accuracy, minimum number of runs per vessel, and methodology to estimate the propagation loss. The propagation loss estimation can range from elaborated numerical modeling to simple laws, e.g., $20 \log_{10}(R)$ (ANSI 2009), where $R$ is the distance between the source and the receiving hydrophone.

## Designing Measurement Geometry and Overview of the Trial Area Environment

To get a rough estimation of the uncertainty of the proposed methodology, two field campaigns were scheduled for two occasions with different sound speed profiles but at the same location. The measurements in this chapter were performed in partial compliance with the shallow water ship URN measurement guidelines of Bureau Veritas (Bureau Veritas 2018), as well as with consideration of the above-mentioned guidelines and notations. Deviations in the measurement setup in terms of measurement depth, distance, and boat speed were introduced to determine a suitable method to estimate the source level from small boats in shallow water.

The trial site Djupviken (N59°3′28.4″, E18°6′52.6″) is located in the southern part of the Stockholm archipelago in eastern Sweden. The area is a shallow water bay (<50 m depth), with muddy sediments, and in general a flat topography in the middle of the bay (Fig. 1, left).

For the sound measurements, a four-channel instrument, an RTsys, SDA014, was used. The system had four hydrophones located at different depths constituting a vertical line array (Fig. 1, right). The array of hydrophones was mounted on a vertical rig with a bottom weight of 20 kg at 30 m depth and a subsurface buoy at 5 m depth in order to keep the array vertical as well as minimizing the influence from surface waves. Data were recorded with a sampling speed of 78,125 Hz, 24-bit, and stored internally on a SD card and a hard drive.

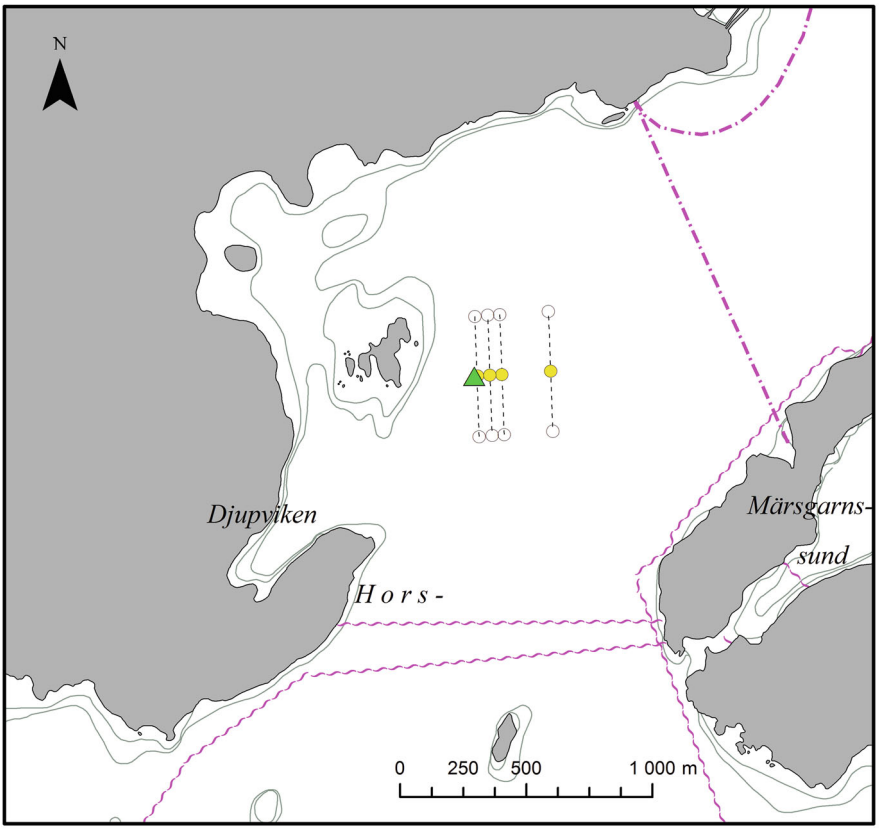



**Fig. 1** Left: The seabed topography of the test site Djupviken on the Swedish east coast south of Stockholm with the test track marked. Right: Sketch of the RTSys array setup and suspension in the water. A 100-m-long cable connected the array to the measurement platform. Hydrophones where (**a**) Reson TC4032, (**b**, **c**) HTI-96-MIN, and (**d**) CO.L.MAR GP090

Salinity, temperature, and sound speed in the water column were measured three times each day with a Sound Velocity Profiler, SWiFT SVP, from Valeport. The sound speed profile of the first measurement campaign 2019-12-04 was almost constant, varying from 1434 m/s at the surface to 1436 m/s at the seabed. During the second campaign 2020-06-24, the situation was different, with a thermocline at 17–20 m depth with speeds spanning from 1440 to 1490 m/s. Propagation loss measurements were only conducted on 2020-06-23, with similar characteristics in the sound speed profile as during the measurement campaign conducted on 2020-06-24.

Propagation loss for the trial area was measured by emitting acoustic pulses at a number of locations in the vicinity to the boat tracks. The receiving equipment was the same as for the sound measurements. The measurements were performed using a workboat, which was set drifting with the current across the boat tracks, while a handheld GPS receiver continuously recorded the positions for each transmission.

The transmitter, of type LL-1424HP, was suspended from a boat at a depth of 5 m. In addition, an active (with internal preamplifier) reference hydrophone of type Neptune D/70/H was positioned at the same depth and 6 m horizontal distance from the transmitter. The complete reference hydrophone system was calibrated a priori down to 100 Hz. Linear frequency-modulated pulses, spanning 100 Hz to 12 kHz, were used during the propagation loss measurements. By match-filtering the received signal with appropriate segments of the emitted signal, the sound pressure level in a given frequency band was obtained. Under the assumption of spherical spreading from the source to the reference hydrophone, the source level was estimated. In other words, the calibrated reference hydrophone was used to determine the SL of the source, and the propagation loss was then calculated using the difference of the SL and the received levels of the RTsys in the specified frequency bands.

Several surface buoys were used to mark the closest point of approach (CPA) to the sensors, indicating the distances 10, 50, 100, and 300 m (Fig. 1). The distance of 100 and 300 m was selected based on the measurement geometry found in the guidelines for commercial ships, but the 10 and 50 m distance was added due to the smaller size of the recreational boats studied here.

The start and end of the track were marked with virtual waypoints in a GPS receiver in order for the boat to keep a straight course. Each track ran in parallel to each other and with the CPA to the RTsys position in the center of the track. The length of the track was 500 meters where the water depth varied between 30 and 40 m.

One run was quantified as a completed passage at a fixed speed between the virtual waypoints marking the start and end, passing the CPA buoys. The boats performed several runs for the speeds 5, 10, and 20 knots. Occasionally a run had to be repeated when other vessels passed the area, increasing the ambient noise above the background level and as a result disturbing the recordings.

Three different kinds of boats were used in this trial, an open boat with a four-stroke outboard petrol engine, Örnvik 540; a day cruiser with an inboard diesel engine, Targa 23, and finally a day cruiser with an electric inboard engine, Eelex 8000, which was a prototype with an experimental hull and propeller (Table 1). The boats were built in 1988, 1997, and 2019, respectively.

The signals from the hydrophones were processed in several sequential steps, based on the guidelines from Bureau Veritas (Bureau Veritas 2018), but with some adjustments to better fit the recreational boat measurements. The guidelines treat the boat in question as an omnidirectional source, i.e., the sound pressure levels measured in different directions from the boat are treated equally and averaged to form the final source level of the boat. The signal processing steps, which are described in detail in Section Measurement Guideline for Recreational Boats (p. 218–223), can be summarized as follows:

1. Combine GPS and hydrophone data.
2. Divide each run into 19 dataset time segments, each of which equivalent to 50 m and thus dependent on the boat speed (see Fig. 2).
3. Calculate PSD from each of the 19 time segments.
4. Convert to 1/3 octave bands (optional).

**Table 1** Overview of the test boats and their technical specifications

| Boat type | Length (m) | Draught (m) | Weight (kg) | Fuel | Engine type | Engine power (hp/kW) | Propulsion |
|---|---|---|---|---|---|---|---|
| Örnvik 540 | 5.4 | 0.4 | 300 + 108 (engine) | Petrol | Outboard, Yamaha, four cylinders (year 1999) | 50 hp/37 kW | Single prop, four blades |
| Targa 23 | 7.6 | 0.9 | 2300 | Diesel | Inboard, Volvo Penta AD31P-A, six cylinders | 150 hp/110 kW | Duo prop, three + four blades |
| Eelex 8000 | 8 | 0.8 | 2600 | Electric | Inboard, electric | 300 hp/225 kW | Single prop, four blades |

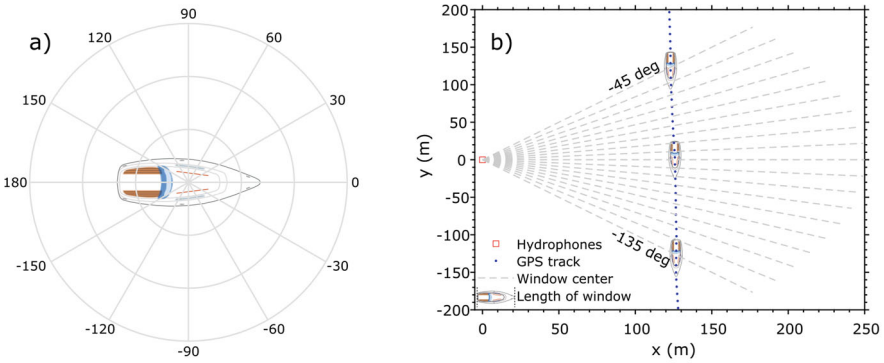



**Fig. 2** The boat angle and data window definitions. (**a**) The boat angle is defined from the source center to the hydrophone(s) and, therefore, (**b**) varies as the boat passes through the measurement area. In this case, a southbound route yields negative angles, while a northbound route results in positive angles. The center of each data window is given by the angles $\pm 45^\circ$ to $\pm 135^\circ$ in $5^\circ$ steps, and the width is 50 m, as indicated by the boat sketch length

5. Correct for background in each window.
6. Correct for propagation loss in each window.
7. Combine the windows into one PSD for the entire run.
8. Combine multiple hydrophone channels (if applicable).
9. Combine multiple runs (if applicable).

## Factors Influencing Estimated Source Levels

Some of the measurement parameters that could influence the estimated source levels include the distance between the boat and the hydrophones, the boat speed, portside/starboard facing the hydrophones or other directional effects, the depth and location of the hydrophones, the PL estimation, and the boat GPS location precision.

### Measurement Depth

The depth position of the hydrophone(s) can lead to a relative change in the measured sound pressure levels but can also make each hydrophone more or less sensitive to other sources. Studying a specific boat run during the measurement campaign, the measured total levels varied from top to bottom with as much as 4 dB, with tonal components varying up to 7 dB. One reason for the observed difference might be the sound speed profile, as the sound speed was causing refraction toward the bottom. The deepest positioned hydrophone also shows higher sound pressure levels.

During the measurement campaigns, it was also noted that a hydrophone closer to the water surface was possibly more influenced by the measurement platform, reflections from floating buoys, waves, and other sound sources located near the

surface. For a hydrophone located near the seabed, other effects can influence the result. It is therefore a good practice not to be closer than 1 m from the seabed (Jensen et al. 2011; Gassmann et al. 2017; ITTC 2017).

## Boat Speed

The boat speed is a parameter that affects the source level. As the speed increases, cavitation may occur at or near the boat propeller leading to increased source levels. A naïve assumption may be that increased boat speed directly leads to increased source levels. However, the noise from the propeller depends largely on the propeller design and condition (see section "Boat-Radiated Noise" for more results).

## Propagation Loss Estimation

One of the largest sources of error in source-level measurements is the propagation loss (PL) of acoustic waves traveling from the source to the receiver. One approach is to measure PL with calibrated sources and receivers. This is, for practical reasons, not always possible. Another approach is to use the boats under investigation to measure and retrieve information on the PL. Another alternative is to numerically model PL based on known parameters of the bottom layer(s) and sound speed profile. Often, the same measurement location is used repeatedly, and knowledge of the expected PL can be gained with time.

Each of the classification society's guidelines for noise measurements of larger vessels has a slightly different approach to handle the estimation of PL, as seen in Table 2.

In short, there are different approaches, each with different pros and cons in terms of time, resources, and accuracy. A measurement will always contain some level of error or uncertainty, but the results may still describe the actual situation better compared to modeling efforts that use some more or less known inputs fottr the final PL estimation. Building knowledge about the measurement area is important, but the underwater acoustic environment can nonetheless change with time.

**Table 2** Propagation loss (PL) estimation methods for different classification guidelines. R is the distance between the source and the receiver

| **Society classification** | Det Norske Veritas Silent-E (2010) | Bureau Veritas URN (2018) | American Bureau of Shipping (2018) | Registro Italiano Navale Dolphin (2016) | Lloyd's Register ShipRight (2018) |
|---|---|---|---|---|---|
| **TL estimation method** | $18log_{10}(R)$ | Acoustic modeling methods | $20\log_{10}(R)$ | $20\log_{10}(R)$ | Measured or modeled TL in shallow waters and $20\log_{10}(R)$ in deep water |

Since modeling PL was not applicable during these measurements, PL measurements were conducted based on two methods: (1) using a known sound source to generate sound at different positions and distances from the receiving hydrophones and (2) to use the boat runs themselves as a source:

1. PL can be measured by drifting through the measurement area with a known calibrated source (as described in section "Designing Measurement Geometry and Overview of the Trial Area Environment"), which creates known sound levels that are measured by the hydrophone(s). The difference between source levels and received levels is then used to estimate PL using GPS location data (Fig. 3). Only data in the far-field regime of the reference hydrophone were evaluated, resulting in a conservative limit of measured data to frequencies above ~500 Hz. Considering all the data points, the average $A$ estimate was 19.8 ± 1.8, 18.4 ± 3.7, 20.0 ± 2.2, and 18.1 ± 3.2 for channels A–D, respectively.
2. PL formulas can also be fitted to measurements of the boats during the noise measurement campaign of the recreational boats. This approach is more uncertain, as the directivity and stability of the source are unknown, but the approach is

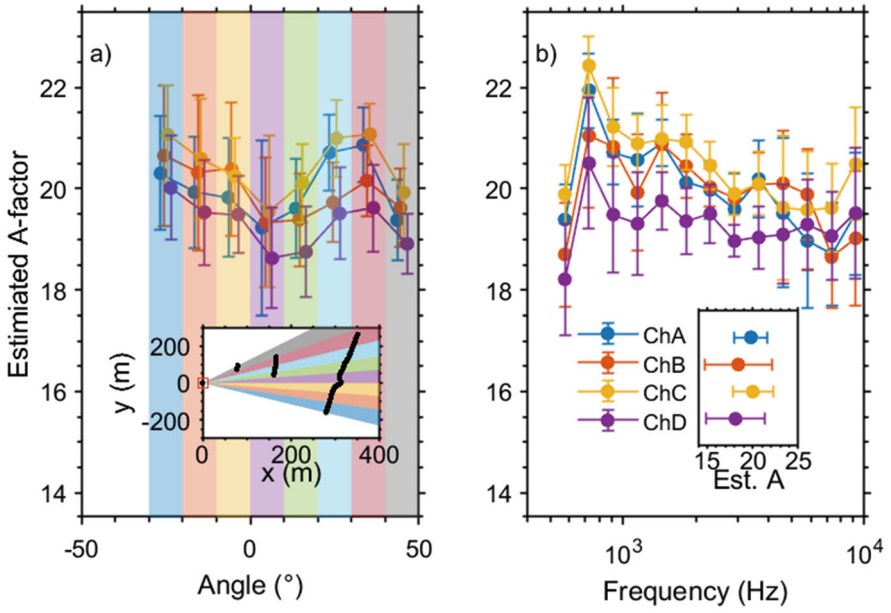



**Fig. 3** Direct measurement results of propagation loss (PL). Using a known source, the PL is calculated and fitted to a $PL = A \cdot log_{10}(R)$ formula in (**a**) an angle range and (**b**) 1/3 octave bands (base 2). The colors of the color-shaded areas in (**a**) correspond to different angle ranges indicated in the inset, where the position of the source throughout the measurement campaign (black) with respect to the hydrophones (red square) define the angle. The inset in (**b**) shows the average estimation of $A$ for the four hydrophone channels, for all source positions and including 1/3 octave bands above 500 Hz

efficient in time and resources. If similar measurement tracks are used during the measurement campaign, the runs can be treated individually and then combined to increase the precision of the track PL estimate (Fig. 4). Different tracks, at different CPA, can yield different PL estimates depending on the bottom layers, water depth, and sound speed profile. In the specific case illustrated in Fig. 4, a simple $PL = A \cdot log_{10}(R)$ formula was used to fit A. The parameter A was found to increase with distance for the four characteristic tracks with CPA 20, 70, 100, and 300 m. Note that the results are vastly different from the measured PL in Fig. 3 which may be due to the nature of the source. The boat was a broadband source and had an unknown three-dimensional directivity. It is possible that tracks with small CPA detect noise that is not detectable at larger distances. Along those lines, it may very well be the data shown in Fig. 3 are not a true PL estimation, but rather an indication of the directivity of the boat itself.

As can be seen in Fig. 5, the distance-dependent PL estimates based on the measurement tracks perform rather well overall, but in some frequency bands, it results in 2–4 dB separation of the source levels from tracks at small and large CPA. Around 20 Hz there is a clear tone in the 310 m CPA data, which is more smeared out in the 120 m CPA case, yielding ~8 dB difference, likely due to other factors than the propagation loss compensation. The PL estimates based on a known source were

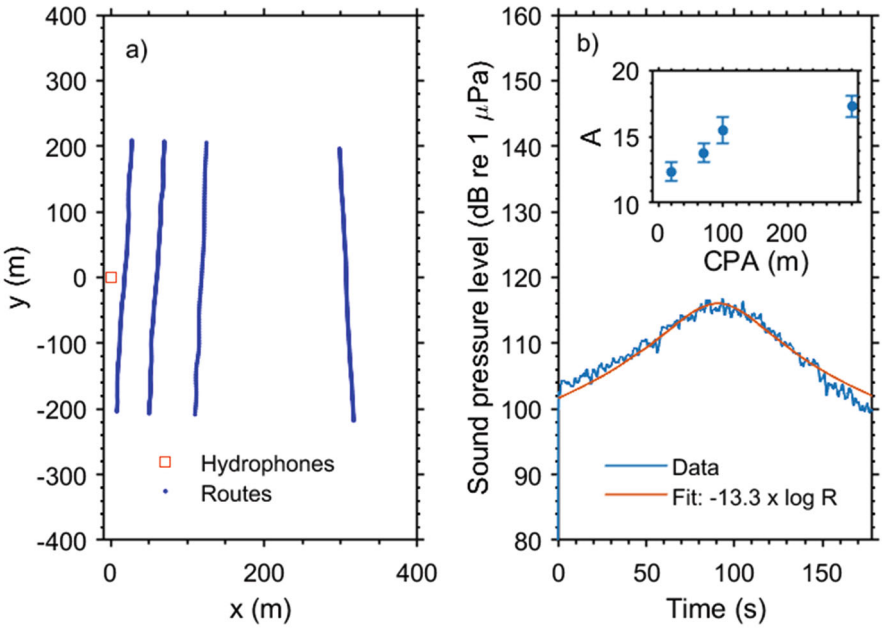



**Fig. 4** CPA-dependent propagation loss (PL) model measurement. (**a**) Runs conducted at different CPA and (**b**) example of the measured sound pressure level (channel A) for one run fitted to a $PL = A \cdot log_{10}(R)$ formula. The inset in (**b**) shows the average values of A, as retrieved from 26 runs for channel A. The error bars correspond to the data included in the analysis only

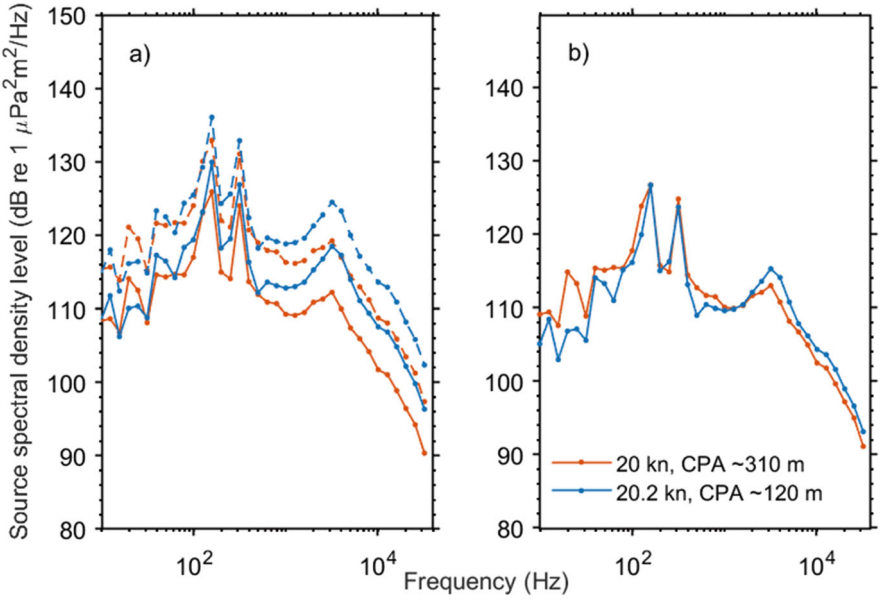



**Fig. 5** Example of propagation loss (PL) implementations. Two runs at similar speeds, but at a CPA of 310 (red) and 120 m (blue) is compared using (**a**) a flat PL = $19.8\log_{10}(R)$ (solid) and PL = $17\log_{10}(R)$ formula (dashed) used throughout the measurement area and (**b**) a distance varying PL model, as described in Fig. 4

performed 1 day before the boat measurements and showed a relatively small distance variation in the fitted PL estimation. The estimates based on a known source varied ±0.8 dB, compared to ±2.5 dB for the measurements based on boat routes. As a conclusion in line with these results, it is important to conduct routes with different CPA to verify the PL model used.

## Distance and GPS Precision

As described above, the amplitude of the sound decreases further from the source, resulting in negligible (relative to the background) sound levels far from the source. It is therefore important to be sure that the signal-to-background level is large enough for the measurement to be successful. For example, the measurements at CPA = 300 m were, on average, 4.5 times more likely to have data points below the background level as compared with CPA = 100 m measurements.

However, there are several factors that come into play when selecting at what distance to undertake the URN measurement. When PL is not very well known, the estimated source-level error becomes larger the further away (see Fig. 2). For example,

assuming $PL = 17 \cdot log_{10}(R)$, when it actually should be $PL = 20 \cdot log_{10}(R)$, results in 6 dB error at 100 m distance.

Finally, the GPS precision may influence the position error. Generally, today's GPS transceivers have a precision of a few meters. Such errors will have a larger impact at close range. At 100 m distance, the effect is significantly smaller compared to the PL-error estimates.

## Starboard/Portside Runs and Directivity

Depending on the design of the hull and propeller(s), the boat constitutes a directional source. For a complete coverage of angles between the hydrophone and the boat, it is important to include both starboard and portside runs. Capturing directivity in a measurement implies that the runs should be long enough such that a large boat-to-hydrophone angle is both obtained and measured accurately.

In other words, the boat should not be too far from the hydrophones both due to signal-to-background limitations but also in order to maintain feasible run times. A 500-m-long track results in a maximum distance of 270 m to the hydrophones and about $\pm 70^\circ$ in angle for 100 m CPA. Increasing the covered angles to $\pm 80^\circ$ would be equivalent to 1200-m-long tracks, that is, more than double the distance for a limited amount of data.

From the directivity in Fig. 6, the sound pressure levels were observed to drop at angles far from the CPA (located at $\pm 90^\circ$). The results suggest that the main angles to take into account indeed are within the $\pm$ (45°–135°) range.

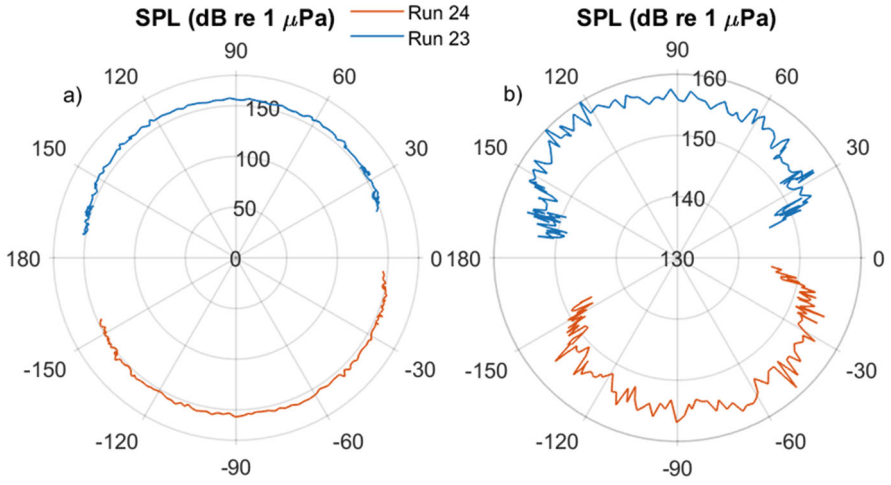



**Fig. 6** Directivity of Eelex 8000, with the origin at (**a**) 0 dB and (**b**) 130 dB. See Fig. 2 for angle definition

## Boat-Radiated Noise

### Source-Level Comparisons of Trial Boats

The three boats constitute a set with different engine, propeller, and fuel types. It is therefore interesting to compare their source levels, as shown in Fig. 7 and Fig. 8. At 5 kn, Örnvik 540 has the highest levels, both in maximum tone levels, found in the narrow band spectrum and in the 1/3 octave spectrum (base 2). In the range 60–200 Hz, this boat is louder than the other two. Furthermore, Eelex 8000 stands out as the quieter at lower frequencies but experiencing higher levels at higher frequencies.

The situation is different at 20 kn, however, where Targa 23 overall seems as the boat with slightly higher levels except for the strong tones near 40 Hz of the Örnvik 540 (Fig. 8). However, the 1/3 octave band (base 2) spectra indicate that the three boats are relatively comparable throughout the measurement frequency range.

At 5 kn, Eelex 8000 has lower levels up to ~600 Hz but was noisier above 4 kHz. The opposite trend was found for 20 kn, where the levels were comparable for the three boats up to 1 kHz, but then Eelex 8000 was quieter, resulting in a lower overall sound pressure level. The calculated source levels were difficult to compare to other studies due to the lack of standardization but are in the same range as other published studies (Erbe et al. 2016; Matzner et al. 2010).

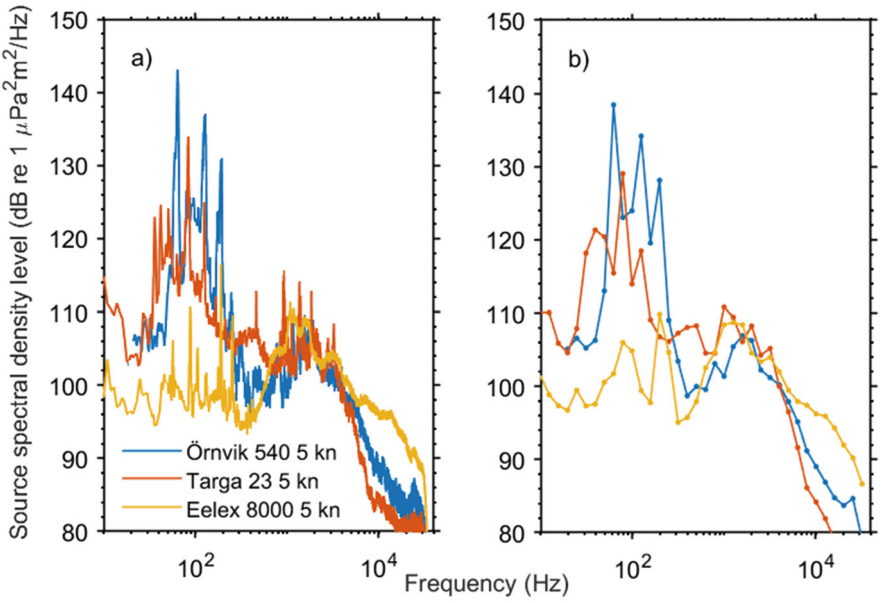



**Fig. 7** Comparison of the source level of three recreational boats at 5 kn. (**a**) Narrow band and (**b**) 1/3 octave band spectra (base 2)

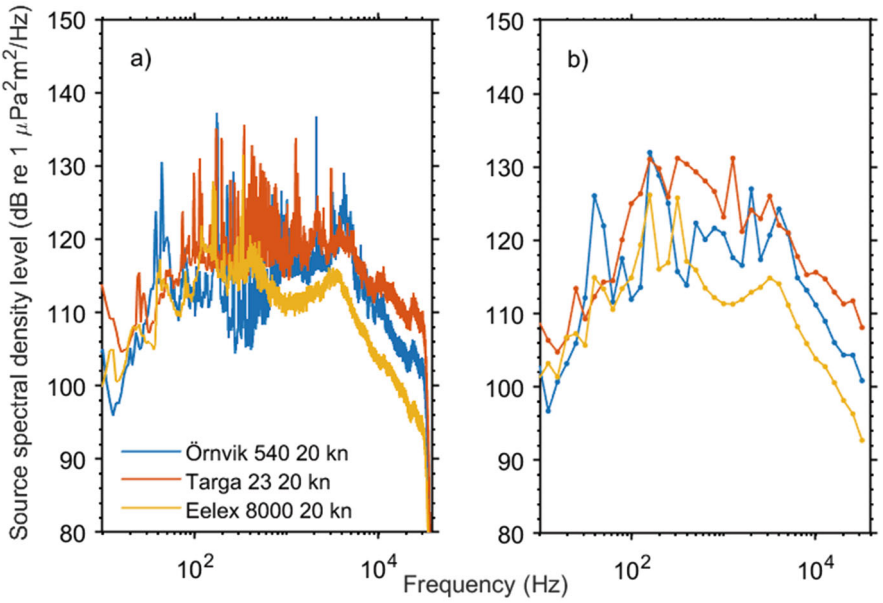



**Fig. 8** Comparison of the source level of three recreational boats at 20 kn. (**a**) Narrow band (1 Hz) and (**b**) 1/3 octave band spectra (base 2)

The broadband source levels for the boats (10 Hz–30 kHz), calculated from narrow band spectra, for 5, respectively, 20 knots were for Örnvik 148 and 158 dB re 1 μPa·m, for Targa 23 145 and 161 dB re 1 μPa·m, and for the Eelex 8000 144 and 153 dB re 1 μPa·m.

## Uncertainties in Source-Level Estimations

As described in section "Source-Level Comparisons of Trial Boats," there are several terms that add up to the measurement uncertainty. The sound pressure level (SPL) measured from an omnidirectional source with source level (SL) is expressed as:

$$SPL = 20\log_{10}(a) - R_r = SL - PL + NL, \tag{1}$$

where $a$ is the measured voltage amplitude of the hydrophone, $R_r$ is the hydrophone sensitivity, $PL$ is the propagation loss, and $NL$ is the background noise level. Assuming that $NL$ is small, eq. (1) is rewritten as

$$SL = 20\log_{10}(a) - R_r + PL. \tag{2}$$

Thus, there are uncertainties introduced in all terms: the measured amplitude, the hydrophone sensitivity calibration, and the propagation loss. Here it is noted that the

post-processing method used herein, specifically the averaging of spectra in different time-segments, treats the boat as an omnidirectional source. This may not be the case, which could introduce additional uncertainty, if the directionality is not fully captured by the measurement geometry. The assumption is commonly used in ship radiation models found in the literature. The close to symmetric curve in the right pane of Fig. 4 suggests that a point source assumption is accurate, at least for the broadband source level.

The measured amplitude error is typically lower than the other terms, and the uncertainty in $R_r$ is most often written in the calibration report, typically around 0.7–0.5 dB, and varies with frequency. The main measurement uncertainty, as mentioned in sections "Propagation Loss Estimation" and "Distance and GPS Precision," is *PL* estimation errors. In this report, this uncertainty has been partly handled by measuring the *PL* using a controlled source. To further estimate *PL* errors, more elaborate methods such as seismic exploration including sediment sampling combined with inverse estimation of the sediment properties could be employed. This is however a laborious task and beyond the scope of this report.

The narrow band *PL* is typically rapidly oscillating function of both frequency and range. For broadband signals, the *PL* can however often be approximated with reasonable accuracy as $PL = A \cdot log_{10}(R)$ where $A$ is constant. Even though there is no formal upper limit on $A$, it typically has a value between 10 and 20. For example, the broadband *PL* estimate in Fig. 3 suggests $A$ is around 19 with a standard deviation of ~3, when combining all channels and source positions in the measurement area. The standard deviation of the $A$ estimation increased with distance. For example, considering distances ranging from 100 to 125 m, from 160 to 225 m, and from 290 to 440 m the standard deviation for channel A was 1.1, 1.33, and 1.9, respectively. Applying the same ranges for channel B, which had the largest standard deviations overall, resulted in standard deviations 1.1, 2.3, and 4.0, respectively. The trend was the same even though only single angle ranges are considered.

The standard deviation of $A$ estimated from fitting *PL* to sound pressure-level measurements, shown in Fig. 4 was found to be 1 or below 1 dB for all tracks.

In the source-level estimations presented in this report, assuming a sensitivity uncertainty of 0.7 dB and an estimated $PL = (15.5 \pm 1)\log_{10}(R)$ for the 100 m tracks (Fig. 4), the total uncertainty adds up to 3 dB.

To study the repeatability of source-level estimates, four runs of Eelex 8000 at 20 kn speed at 100 m CPA were used. The average standard deviation of the 1/3 octave band (base 2) source-level estimation was 1 dB. Comparing the 100 m CPA track source-level estimation with the 300 m CPA track, which was repeated twice, yielded a standard deviation of 0.4 dB.

## Measurement Guideline for Recreational Boats

The experiences and results from the two measurement campaigns conducted here, combined with the Bureau Veritas grade B1 guideline (Bureau Veritas 2018), were used to develop a practical measurement guideline for the estimation of the source levels from recreational boats in shallow waters.

The scope of this guideline is to assess the underwater sound pressure levels normalized at a distance of 1 m from a recreational boat. The measured quantity is the sound pressure level radiated by the recreational boat considered as a monopole source, measured at a certain position in the water. The sound pressure level was normalized to the level at 1 m from the source, using the measured distance from the hydrophone position. The resulting quantity was expressed in terms of the sound pressure spectral density level (in dB re 1 $\mu Pa^2m^2/Hz$).

The following section describes in detail the general recommendations for the test site, the measurement and post-processing procedure, and how to present the results.

By following this guideline, the estimation of a recreational boat's source level can be used for comparison with scientific published studies, environmental impact studies, and mitigation measure calculations and as input to source models.

## Setup and Procedures

### Test Site and Measurement Conditions

The measurement area shall have a low noise level, such that the majority of boat runs have sound levels at least 10 dB above the background level. Note: It is recommended that the area should be at least 3 km away from commercial vessels and other boats. Wave height should be as low as possible, preferable $<0.5$ m, as it affects both the boat's behavior and the background noise levels. Avoid areas with strong currents, which may affect the recording and the position of the hydrophone in the water. Do not perform the measurement in rain as it increases the ambient noise level.

The area should have a size to allow the boat to run in a straight heading at the target speed for at least 500 m.

The seabed should be as smooth and homogeneous as possible to minimize the influence of varying propagation loss.

The water depth should be at least 20 meters. The actual water depth should be measured in the area and especially along the test track and at the hydrophone position.

The sound speed profiles shall be measured before and after the test runs.

The result of the first sound speed profile shall be taken into consideration regarding where in the water column the hydrophone should be placed to have the most predicable propagation loss. Note: The second measurement is used to detect changes in the sound speed profile during the course of the measurement and is used to estimate measurement uncertainties.

The background noise shall be recorded at least 30 min before and after the measurement and 2 min before each run.

The signal stability shall be checked every 15 min in order to ensure the reliability of the measurement.

### Instrumentation

At least one hydrophone shall be used.

The hydrophone should have a known response in the frequency range of 20 Hz–50 kHz.

The hydrophone should be omnidirectional, i.e., have a directivity of <2 dB.

The hydrophone should be calibrated (uncertainty ±2 dB) as well as the entire system (hydrophone, cable, and data collection system) prior to the measurement.

The system should record with at least 50 kHz sampling frequency and have a dynamic range of at least 80 dB.

## Deployment

The hydrophone shall be placed at least 1 m from the seabed and in the lower half of the water column in order to avoid disturbances associated with the sea surface. Note: If the sound speed profile suggests that the sound energy is significantly directed to another depth layer, the hydrophone may be placed accordingly. If not, the hydrophone should be placed below the midst of the water column (e.g., at a depth of 15–29 m if it is 30 m deep at the site).

The hydrophone shall be disconnected from wave motions. The hydrophone can be anchored either on the seabed or suspended from the surface. If the hydrophone is close to the surface, the measurement has a higher risk of being disturbed by wave motion and rig noise.

There should be no air-filled buoy or similar air-filled cavity such as an autonomous logger unit, within 3 m of the hydrophone, as this may interfere with recording.

If the hydrophone is mounted on a rope between the bottom weight and the buoy, it shall be kept vertical in the water and thus not tilted due to currents or waves. Note: If the line is assumed vertical but is not, the hydrophone is not where it is expected to be, which means that distance estimates and depth values are incorrect.

Data recording shall be synchronized with a GPS clock prior to measurements. Note: The data recording can be done either via a cable to land or to a boat or in an autonomous unit in the water.

The position of the surface buoys and the hydrophone positions shall be measured with a GPS receiver.

The error margin of the hydrophone positions should be <5 m.

If a boat is used as measurement platform, it should be anchored at least 100 m from the hydrophone to minimize wave-induced noise when they hit the boat hull.

## Test Track

The start, closest point of approach (CPA) and end point of the course should be marked with surface buoys for easier navigation.

Each run should be at least 500 m long.

Two separate tracks should be employed. The primary tracks CPA should be 100–200 m distant from the deployed hydrophones position.

The CPA of the second track should be 1.5–2.5 times the distance, compared to the primary track. If only one track is used, the measurement uncertainty will increase due to lack of PL estimation verification.

The primary tracks CPA should be 100–200 m away from the deployed hydrophone's position.

### Test Procedure

If a run is disturbed by a boat nearby, the run shall be repeated.

A GPS receiver shall be positioned at the stern of the boat, set to record the position every second to get an accurate recording to the boat's position and speed during the runs.

The boat should have a straight course and even speed/rpm throughout the run.

The designated speed should have been reached before entering the track.

The course shall be repeated at least two times. Note: It is recommended to avoid driving so that the boat hits any wind-driven waves or from the boat's own wake.

The measurement shall be repeated for at least two different speeds: 5 kn and a second speed. Note: Preferable, these speeds should be one without cavitation and one with cavitation. The 5 kn is a common speed into harbor and in sensitive areas. The second speed should be chosen based on a typical speed the boat is designed for. Boats that are driven in a non-optimal way can be very noisy, which results in an uneven comparison if the non-optimal source level is compared to other boats at optimal speeds.

In some areas, other speed limits can occur, and these speeds are worth including as well.

Before testing, engine, propeller type, number of blades and location in the boat, draught, boat size and type should be noted in the protocol.

During the measurement, the speed, power, engine rpm, and run number shall be noted in the protocol.

### Post-Processing

As a first step, GPS tracks are extracted and combined with acoustic data.

The run shall be segmented into 19 segments, centered on a given set of angles between the boat and the hydrophone system. Note: The set of angles is exemplified in Fig. 2 and defined as $\pm 45^\circ$ to $\pm 135^\circ$ in $5^\circ$ increments.

In order for the segments to contain a certain amount of data, the segments should have a length of 50 m.

### Data Segments

For all 19 segments, a PSD shall be calculated using the Welch method, with 1 s time segments and a 50% overlap, yielding a PSD with a bandwidth of 1 Hz.

To calculate the spectrum with 1/3 octave band resolution, the *mean narrow band value* in each band is used.

The calculated spectrum, whether it is narrow band or 1/3 octave band, shall be compared to a corresponding background spectrum:

(a) If the signal-to-background level is above 6 dB, the data point shall be used directly without any modifications.

(b) For a signal-to-background level of 3–6 dB, the background shall be subtracted from the signal.
(c) If the signal-to-background level is less than 3 dB, this data point shall be removed from further analysis.

The propagation loss shall be corrected for in each of the 19 time segments, by taking into account the distance from the hydrophones to the center of each time segment.

The estimation of $A$ should be done using the following order:

1. PL measurement, based on separate calibrated source or boat runs
2. PL modeling
3. Based on previous knowledge of PL in the area
4. Using $\mathrm{PL} = 17 \cdot \log_{10}(\mathrm{R})$.

After the propagation loss correction, all data time segments shall be averaged in the linear domain to create a single spectrum for the run. Thus, the sound pressure spectral density level measured by hydrophone $h$ is

$$L_h = 10 \cdot log_{10} \frac{\sum_t 10^{L_t/10}}{n_t},$$

where $L_t$ and $n_t$ are the sound pressure spectral density level of time segment $t$ and the number of time segments, respectively.

When multiple hydrophone channels are used, their spectra of each run shall be averaged in the linear domain. The measured sound pressure spectral density level of the run is

$$L_r = 10 \cdot log_{10} \frac{\sum_t 10^{L_h/10}}{n_h},$$

where $n_h$ is the number of used hydrophones.

When multiple runs have been conducted with equivalent running conditions, the spectra of the runs shall be averaged in the linear domain. Portside and starboard side passages are then averaged in a similar manner to create a single spectrum for the specific running condition. Therefore, the boat spectrum is

$$L_b = 10 \cdot log_{10} \frac{\sum_t 10^{L_r/10}}{n_r},$$

where $n_r$ is the number of runs.

## Presentation of Results

The protocol shall contain one page identifying the boat to be measured; the location, time, and date of the measurements; and the scientific leader, whom is responsible for the implementation of these guidelines.

The second page shall clearly state the boat characteristics, measurement-specific data, the propagation loss estimation model used, and the sea state. Note: The boat characteristics include engine type, fuel type, propulsion type (e.g., propeller type), and design speed (if known). The measurement data form include the measured boat speeds in knots and the engine rpm for these speeds, the date from the GPS, and the GPS time as well as the recording time.

The measured sound pressure spectrum levels should be presented in 1 Hz spectrum and a 1/3 octave band spectrum, for each running speeds.

The main tones of the narrow band spectrum should be noted in a table and identified.

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# Operational Underwater Noise from Offshore Wind Farms

15

Klaus Betke and Michael A. Bellmann

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**Abstract**

Underwater noise was measured at 25 German offshore wind farms (OWFs) under standardized conditions. The measurements included different turbine types up to 8 megawatt rated power and different foundation types. Water depths ranged from 18 m to 40 m. The two main measurement positions were 100 m from a pre-selected offshore wind turbine (OWT) and in the center of the OWF. Underwater sound was recorded for up to 6 weeks to cover different wind and turbine conditions. The aim was to identify the most relevant influencing parameters of noise radiated by operating OWTs.

The cross-project analysis, based on broadband levels as well as third octave spectra and data, was divided into three wind classes that covered the range from OWT idleness to rated power. Noise from OWTs with gear boxes was slightly higher compared to those with direct drives. No significant differences were found for different foundation types. A trend analysis shows that an increase of radiated noise with increasing turbine size (characterized by its rated power) statistically does not exist. Noise from wind farm-associated vessel traffic was estimated for an exemplary OWF with 87 turbines and was found to be approx. 11 dB lower than the noise radiated by the OWTs.

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**Keywords**

Underwater noise · Offshore wind turbine · Operational noise

## Introduction

The number of offshore wind turbines (OWTs) has strongly increased over the last decade and will increase worldwide significantly over the next decade due to the expansion of renewable energy. Meanwhile, the effect of OWT construction noise (impact pile driving) on marine mammals is well reputed; see, for example, Dähne et al. (2017) and Lucke et al. (2009). The ecological impact of underwater noise from operating OWTs and associated vessel traffic, however, is not yet fully investigated. Several studies indicate that the noise origin are mechanical oscillations of components like the gear box or the generator, which propagate through the foundation structure into the water, and this noise input can dominate the ambient noise measured in the vicinity (Betke et al. 2005; Madsen et al. 2006, Norro and Degraer 2016; Yang et al. 2018). Based on the few available underwater noise studies for operational OWT, Tougaard et al. (2020) and Stöber and Thomsen (2021) tried to extend the current knowledge in order to be able to model future soundscapes. Tougaard et al. (2020) used a general linear model, whereas Stöber and Thomsen (2021) used linear regressions to assess correlations between sound pressure level, measurement distance, turbine size, and wind speed. Both studies used data from smaller and older OWTs, which were gathered in various distances to the turbine and at different wind speeds. Driven by the demand for renewable energy, turbine size has increased significantly over the last 15 years. Upcoming projects will have turbine sizes above 10 megawatt (MW) nominal power. Furthermore, new turbine installation includes direct-driven (gearless) generators. In the North and Baltic Sea, monopile turbines, up to 10 meter (m) diameter, were installed in the latest OWF projects. Therefore, it is uncertain whether the noise from smaller and older OWTs can be extrapolated to the newest generation of OWTs.

In Germany, part of the approval process includes underwater noise measurements prior to construction (background or ambient noise) and during operation, according to a standard procedure developed by the Federal Maritime and Hydrographic Agency (Müller and Zerbs 2011). Currently, about 30 OWFs are in operation, and operational noise measurements were performed in a standardized manner on 25 of them. Post-processed data were uploaded in the MarinEARS database of BSH.

Technical properties of the measured OWTs are diverging: beside different brands with possible subtle design differences, they are erected on different foundation types like monopiles or jacket constructions (Fig. 1), and they have either gear boxes or gearless drives; see Table 1. Although the database of 25 measured OWTs is not extensive, differences in sound radiation that are caused by different turbine parameters should appear in the measurement results. The turbine size has also increased significantly over the years; recent plannings state power ratings of up to 15 MW.



**Fig. 1** Monopile, jacket, and tripod turbine foundations (from left to right). (Photos: itap)

**Table 1** Some details of the 25 tested wind turbine types. Jacket #14 is mounted on so-called suction buckets rather than on piles driven into the ground

| Wind farm no. | Turbine type | Power, MW | Gear box? | Foundation |
|---|---|---|---|---|
| 1, 4 | Siemens SWT 7.0-154 | 7.0 | No | Monopile |
| 2 | MHI Vestas V164-8.0 MW | 8.0 | Yes | Monopile |
| 3 | MHI Vestas V164-8.4 MW | 8.4 | Yes | Monopile |
| 5 | GE Haliade 150-6 | 6.0 | No | Monopile |
| 6, 16 | Adwen AD 5-116 | 5.0 | Yes | Tripod |
| 7 | Senvion 6.3M152 | 6.3 | Yes | Monopile |
| 8, 9, 10, 18 | Siemens SWT 6.0-154 | 6.0 | No | Monopile |
| 11 | Senvion 6.2M126 | 6.2 | Yes | Monopile |
| 12 | Adwen AD5-135 | 5.0 | Yes | Jacket |
| 13 | Siemens SWT 4.0-120 | 4.0 | No | Monopile |
| 14 | Siemens SWT 4.0-120 | 4.0 | No | Jacket |
| 15, 19, 23, 24 | Siemens SWT 3.6-120 | 3.6 | No | Monopile |
| 20 | Siemens SWT 3.6-120 | 3.6 | No | Jacket |
| 22 | Senvion 6.2M126 | 6.2 | Yes | Jacket |
| 16 | Adwen AD5-116 | 5.0 | Yes | Tripod |
| 17 | Siemens SWT 4.0-130 | 4.0 | No | Monopile |
| 25 | Bard 5.0 | 5.0 | Yes | Tripod |

OWTs from 3 to 8 MW are analyzed; hence, some evidence is expected whether larger turbines radiate more noise than smaller ones.

From earlier investigations (Betke 2014; Tougaard et al. 2020), it is known that OWT operational noise occurs mainly at frequencies below 1 kHz, or even below 100, and the acoustical energy is concentrated in a few narrow spectral lines. The strongest of these tones have typically rms levels of 120 dB re 1 µPa at 100 m distance. Mostly, these narrowband sounds recorded underwater are caused by the

tooth mesh frequencies of the gear box and by periodically alternating forces in the generator. In areas with high shipping density (e.g., along main shipping lanes, like most parts of the German Bight of the North and Baltic Sea), operational OWT noise does occur much above the ambient noise level. One important question is if the noise caused by operating OWTs is the only possible impact or if the vessel traffic related to the operation of an OWF will also have an influence on the overall level within as well as in the vicinity of an OWF? Therefore, in some of the German OWF projects measured in 2019 till 2021, the vessel noise caused by the operating OWFs was partly determined, and the underwater noise was also investigated.

## Methods

All operational noise measurements per OWF were conducted within the first years of their operation (new OWTs). The measurements were conducted in the period from 2013 to 2021. The water depth at the measurement locations varied from 18 m to 40 m. The typical setup used throughout the measurements is shown in Fig. 2. The hydrophone was decoupled from the marker buoys; its height above the sea floor was approximately 2 m. The complete setup was designed to minimize self-noise. The devices stayed at their position for 4 to 6 weeks. In general, there were four positions per OWF: (i) 100 m from a pre-selected OWT, preferably one that was located at a corner of the OWF, (ii) in the OWF center, and (iii) 1 km and (iv) 5 km outside the

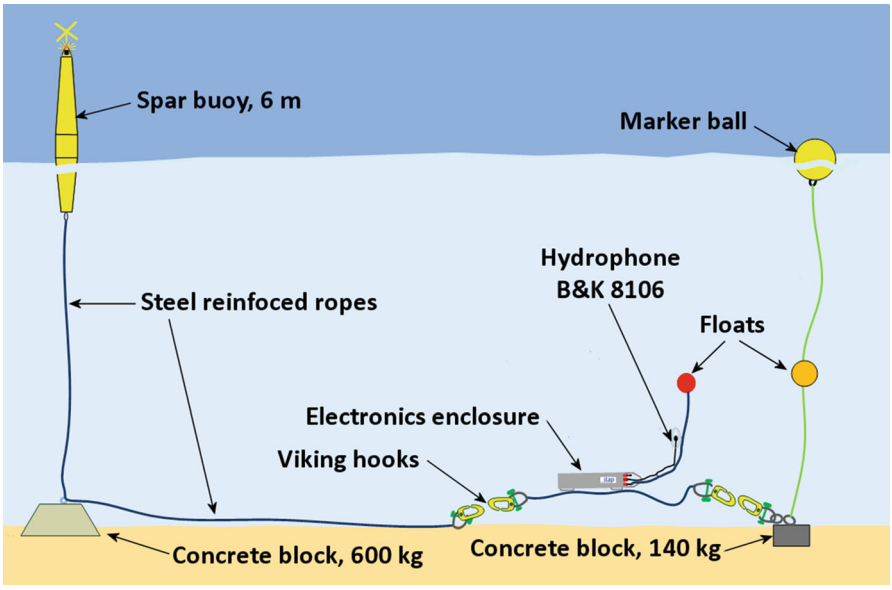



**Fig. 2** Measurement setup with mooring details for operational noise measurements within German OWFs

**Table 2** Definition of wind classes

| Wind class | Wind speed at hub height | Turbine state |
|---|---|---|
| Low | <3.5 m/s | Off or very low power |
| Medium | 7.5–11 m/s | 30–75% rated power |
| High | >11 m/s | >75% rated power |

OWF boundary. For the current evaluation, only the 100 m position was selected, since they promised the highest signal-to-noise ratios between operational noise of the OWTs and the ambient noise.

Sound was recorded intermittently for 10 min every 2 h in an uncompressed data format (wave, 24-bit; $f_s$ 44.1 kHz) due to the fact that the weather conditions, mostly wind and therefore the power generation of OWTs, will mostly change on larger time scales. This scheme proved to provide enough data for the later analysis for different wind conditions. Before deployment and after recovery of the instruments, hydrophones were calibrated with a Brüel and Kjær 4229 calibrator. The actual sound pressure within the calibrator cavity was monitored with a GRAS 46AG condenser microphone; the value went into the calibration. This calibration process is in accordance with ISO 17025 (2022) and given best practice (HELCOM 2021).

The raw data (uncompressed wav files) were evaluated with a validated software package developed at itap (IONIS). In a first step, the equivalent continuous sound pressure level ($L_{eq}$; this quantity equals the rms level) in accordance with the German measurement guideline (Müller and Zerbs 2011) was determined in intervals of 5 s. Based on the 5 s values, the $L_{eq}$ and the exceedance levels $L_{05}$, $L_{50}$ (median), and $L_{90}$ were calculated for both broadband and third octave bands according to the German measurement guideline (Müller and Zerbs 2011). In some cases, and in order to identify tonal noise components radiated by the OWTs, narrowband spectra with 1 or 2 Hz resolution were computed as well. The results were then arranged by three wind classes "low," "medium," and "high"; see Table 2. The wind farm operating companies supplied power and wind versus time data for the relevant OWTs in 10 min steps (mean values). The values in Table 1 were derived from a number of such datasets. The German measurement guideline (Müller and Zerbs 2011) requires at least 3 h of usable sound data per wind class. Despite the intermittent recordings, this was significantly exceeded in all measurements (raw data per wind class ranged from 8 h to more than 50 h).

## Measurement Results

A typical narrowband spectrum of a 6 MW OWT is shown in Fig. 3. The peak at 73 Hz also determines the overall or boradband sound presure (SPL). The spectrum has 1 Hz resolution and 5 minutes averaging time. Most OWTs had only a single prominent spectral peak; there were all found in the frequency range 50 to 200 Hz; however with differing amplitudes.

Figure 4 displays the broadband sound pressure levels of all measured OWTs build-up on different foundation types for the wind class "low" (OWTs off) and

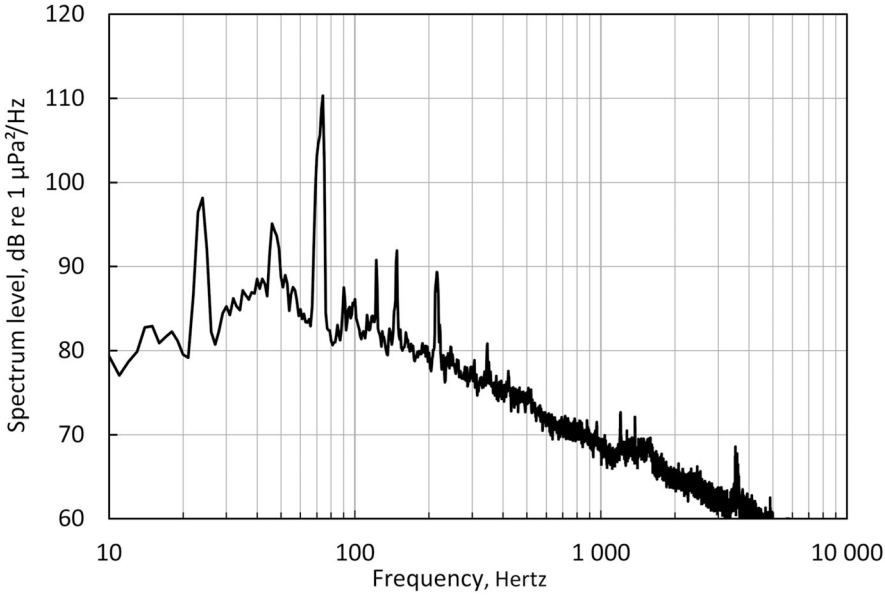



**Fig. 3** Typical narrowband spectrum of the operational noise of a 6 MW measured in a distance of 100 m for the wind condition high (full power)

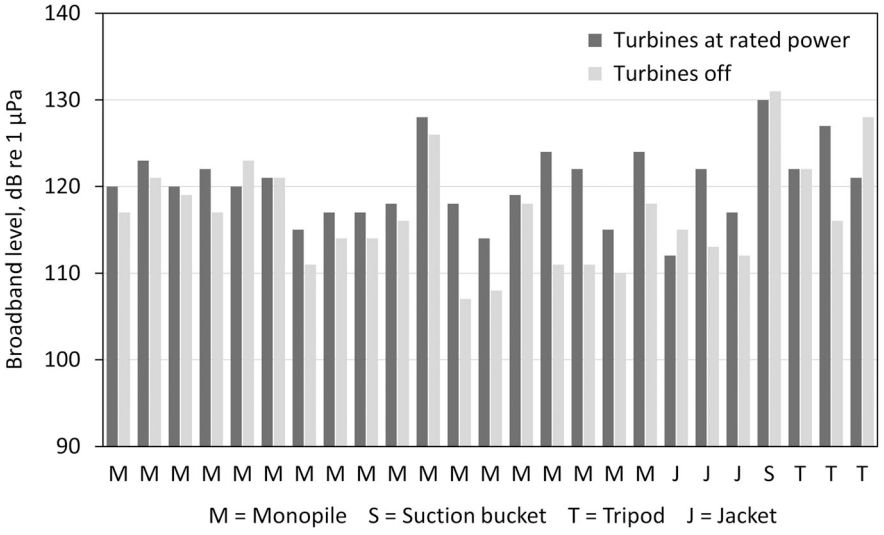



**Fig. 4** Broadband sound pressure levels measured in 100 m distance to the offshore turbines at maximum power and wind class high for different foundation types

"high" (OWTs >75% rated power). In this figure and the following ones, the $L_{50}$ (median value) is shown, which was computed from the whole observation period of 4 to 6 weeks per wind class based on 5-s $L_{eq}$ values. Eighteen of the 25 foundations

were monopiles; the remaining were four-legged jackets and other foundation types. It can be seen that in 22 out of 25 measurements, the sound pressure levels at 100 m distance to the foundation are, as expected, higher in wind class "high" compared to "low" except for the suction bucket, one monopile, and one tripod foundation. The increase between OWTs off ("low") and at rated power ("high") seems to be also depending on the ambient noise. In the case of high ambient noise levels in the vicinity of the OWT, the increase of the broadband level of operating OWTs is limited. Apparently, there is no systematic-level difference between wind classes "low" and "high" for the two foundation groups monopiles and non-monopiles; the actual levels are given in Table 3. Additionally, the differences between wind classes "low" and "high" might be influenced by other parameters like OWT type or gear-/gearless-driven OWTs.

Figure 5 compares broadband averaged sound pressure levels ($L_{50}$) of OWTs with gear box to gearless or direct-driven ones with capacities between 2 and 8 MW. Again, no obvious difference was detected, but Table 3 suggests that gearless OWTs might

**Table 3** Average (median) sound pressure levels measured in 100 m distance from different OWT types operating under full power (ranged between 2 and 8 MW) for the two different foundation types, monopiles and non-monopiles (based on Fig. 4), and separated into with and without gear box (based on Fig. 5)

| Parameter | Average broadband SPL from Fig. 4 and Fig. 5, dB re 1 µPa |
|---|---|
| Monopile foundations | 121.5 |
| Other foundations | 122.0 |
| Drive with gear box | 122.3 |
| Gearless (direct) drive | 120.0 |

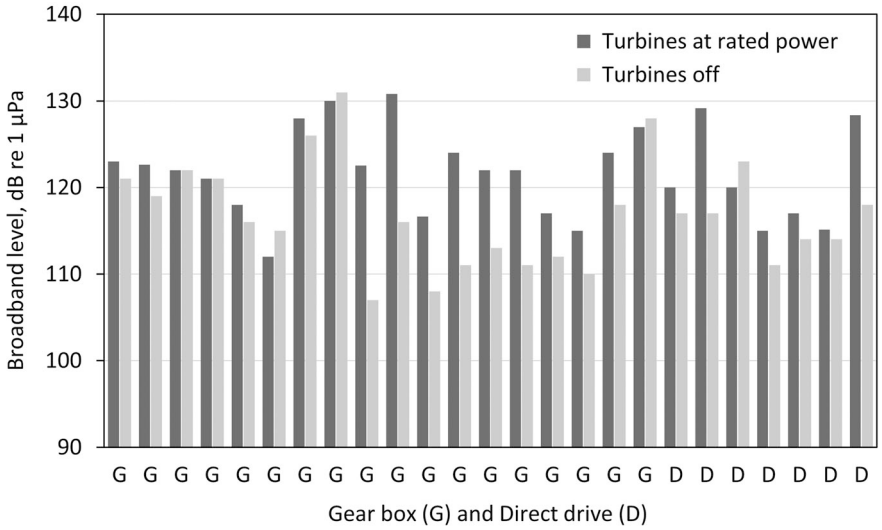



**Fig. 5** Broadband sound pressure levels for turbines with maximum power (ranged between 2 and 8 MW) with gear box and with direct drive measured in 100 m distance

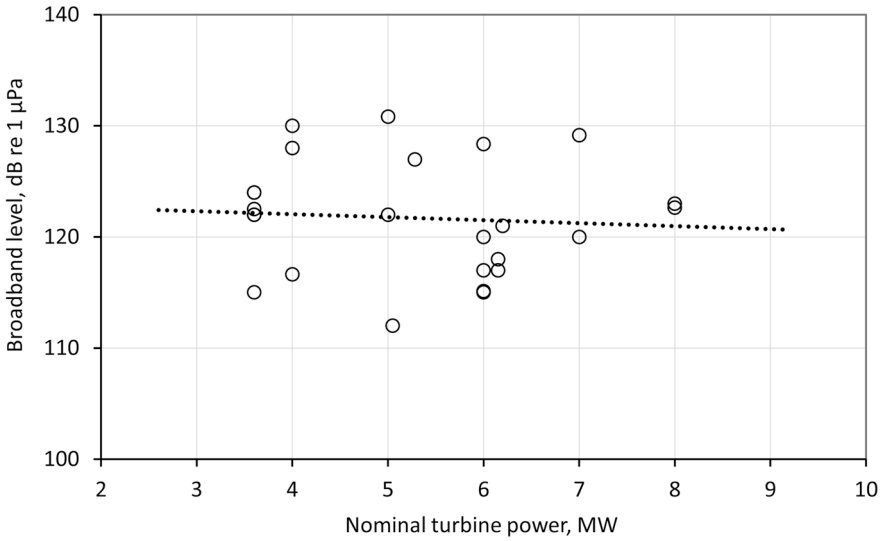



**Fig. 6** Broadband sound pressure level (SPL) measured in 100 m distance versus rated turbine power. The dotted line is a linear regression fit with $L = -0.27\ P + 123$ dB, where P is the rated power and L is the broadband SPL

have 2 dB lower levels in average in the vicinity of the foundation. However, this difference should be treated with caution because of the small sample size, the various combinations of foundation designs and OWT types, as well as the differences in ambient noise levels. Averaged sound pressure levels did not increase with turbine size or turbine power, respectively. In fact, a small negative linear correlation between turbine power and measured sound pressure levels was determined (Fig. 6). With increasing OWT size or increasing power rating, the diameter of the monopile foundation increases significantly as well. Thus, the OWT size is not an independent parameter.

## Wind Farm-Related Shipping Noise

Based on the results in Figs. 4 and 5, the ambient noise levels in the vicinity of OWTs ranged between 107 and 131 dB in wind class low (OWTs off). The only reason for such a high difference in the measured ambient noise can be shipping noise or other anthropogenic sources, like noise from surrounding OWFs in operation. The operational noise measurements in Germany for OWFs grouped in a "cluster" were usually performed in parallel for all OWFs in the cluster. The idea behind this approach was to investigate if operational noise from surrounding OWFs will influence the ambient noise in one specific OWF. These combined measurements point out that in wind class "low," the OWTs are off in all OWFs of the cluster. Therefore, observed differences in ambient noise are probably caused by ship movements related to the OWF maintenance/operation or by public shipping outside the OWF.

A typical situation for an OWF far away from shore is that an offshore service vessel stays in the OWF for 2 to 4 weeks before it returns to the base port for bunkering or crew changes. Exceptions are OWFs which are close to the coast so that service teams can make a round trip within a working day during daylight. Typically, inside the OWF, the service vessel moves between OWTs from time to time to deboard the service teams. Occasionally, crew transfer vessels may be active in the area as well. However, most of the time, the service vessel will be in standby position within the OWF or in the surrounding.

Figure 7 depicts the AIS (automatic identification system; a system by which ships communicate their position and other voyage-related information) track of the offshore service vessel "Bibby WM Horizon" as an example over a period of 52 days in 2021 during the operational noise measurements. The vessel was active in the 2 interlinked OWFs "EnBW Hohe See" and "EnBW Albatros" with 87 turbines of the 7 MW class. During this time interval, the vessel left the area southward toward the coast for its base port only three times for several days (crew changes and bunkering). For about 70 h, the ship was driving inside the OWF with a speed >2 knots (kn); maximum speed inside the OWF was 10 kn. The AIS track points are spaced approx. 30 min; shorter intervals were not available from the Sat-AIS provider.

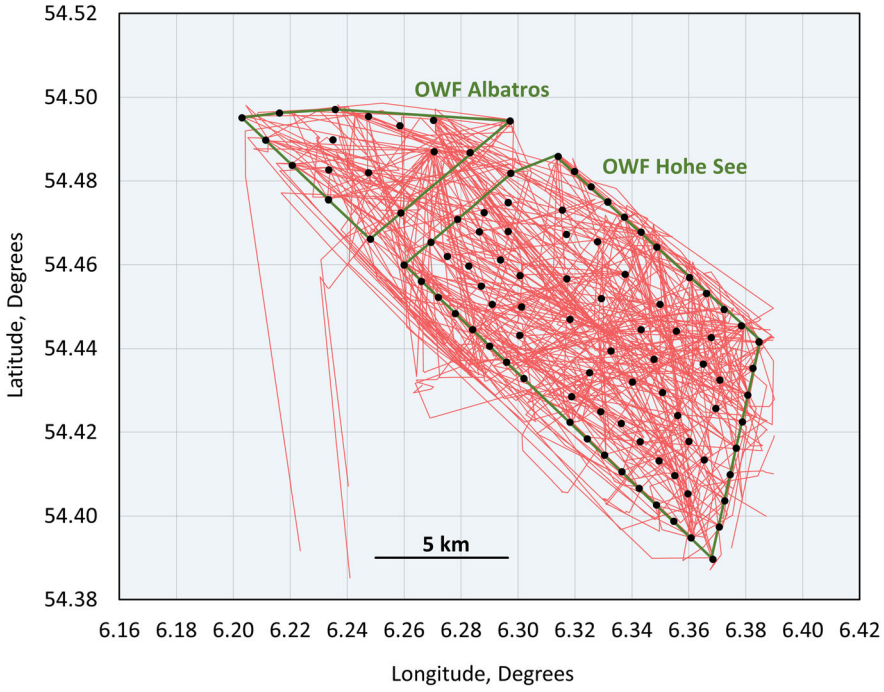



**Fig. 7** Movement of a typical offshore service vessel in and close to an OWF, recorded over a period of 52 days during operational noise measurements

**Table 4** Comparison of the sound energy produced by the OWF and a typical offshore service vessel during stay inside the OWF

| Sound source | Average sound power | | Sound energy emitted into the sea in 50 days | |
|---|---|---|---|---|
| 1 service vessel during transits in OWF (2 to 10 kn) | 0.8 W | 118.8 dB re 1 µW | 3.3 MJ | 125.1 dB re 1 J |
| 87 wind turbines | 10.9 W | 130.4 dB re 1 µW | 47.2 MJ | 136.7 dB re 1 J |

For an estimate of the overall sound energy emitted into the water by both sound sources, (i) OWTs and (ii) service vessel, several assumptions must be made to get a first approximation. First, the service vessel's source level is required. For this specific service vessel, no valid source-level measurement is available, but a similar vessel of this class from itap's measurement archives has a source level of L = 170 dB re 1 µPa @ 1 m. For the noise of a single OWT, a permanent value of L = 120 dB re 1 µPa @ 100 m, based on the measurements of this study, is assumed. Sound propagation was approximated with a 15 $\log_{10}$ (distance ratio) law over the first kilometers. The actual results are listed in Table 4.

With the above assumptions, the OWT cluster emits about 11.5 dB more sound energy within a given time interval than the service vessel in the OWF. The sound pressure level cannot be derived straight from this result since the SPL depends on the location and also on time, because the service vessel is moving. For a long average and at large distances from the OWF, the difference between sound levels caused by all OWTs and the service vessel is 11.5 dB. However, this is a theoretical value, since several kilometers outside the OWF, its noise emission is normally not measurable.

The 11.5 dB difference is mainly caused by the small operation time compared to that of the OWF, which is assumed operational 24 h/day. Additional noise sources like crew transfer vessels or the service vessels' dynamic positioning system might make this difference smaller.

## Discussion

In Holme et al. (2023), a subsample of the measurements of this study (only OWFs Gode Wind 01, Gode Wind 02, and Borkum Riffgrund 02) was used in distances between 63 m from pre-selected OWTs and 5.000 m from the OWF to investigate the influence of turbine size (6.3 and 8.5 MW), wind speed, and foundation types (only monopile and suction bucket) on operational noise. Holme et al. (2023) did not find a correlation between the measured broadband sound pressure level and turbine size or foundation type. They also found out that the noise from OWTs had no significant influence on the broadband ambient noise level at distances of 1 km and more from the OWF boundary. The ambient noise levels inside and outside the observed three OWFs varied without OWTs off (wind class "low") by up to 5 dB, which is a hint that the influence of vessel activities will significantly dominate the ambient noise inside but also outside the OWFs. In the case of operating OWTs in these OWFs, the overall broadband sound pressure level shows partly an increasing, decreasing, or constant

broadband level as function of distance. This finding is in accordance with the findings of this study that the ambient noise outside the OWFs mostly is dominated by other noise sources, most likely noise from OWT service vessels and from commercial ship traffic. For this study, time periods where vessels will be in the vicinity of the measurement positions (<1 km) were excluded to avoid any unwanted effect of crossing vessels. In the study of Holme et al. (2023), the same finding will be reported that nearby crossing vessel showed a fast increase of measured SPL and a fast decrease over several minutes, which is a strong hint that any vessel activity in the vicinity of any OWTs masks or exceeds significantly the overall broadband level generated by an operating OWTs for the short period of crossing nearby.

The measurements indicate that the overall noise emitted into the sea by an OWF (noise from the OWTs plus noise from service vessels) is dominated by OWTs and not by the vessels, since the period of vessel activities within OWFs is limited. Furthermore, based on the rare transits between OWF and base port, the noise from service vessels outside OWFs is limited, and an intermixture with other vessel noise is given. However, the impact of underwater noise from OWFs will also depend on the ambient noise, which is mostly dominated by public shipping noise, especially close to shipping lanes with high vessel traffic.

This study, as well as others (Holme et al. 2023), shows that the most relevant influencing parameters of operating OWTs are currently not identified or well understood.

Within this study, only OWTs during the first years of operation were monitored, but based on experiences from onshore wind farms, a change in the radiated noise from wind turbines running over a long period cannot be excluded. This suggests that modelling the soundscapes of future OWF scenarios might not be possible by interpolating existing measurements from smaller OWT size. In Holme et al. (2023), it was demonstrated that the modelling based on interpolations by Tougaard et al. (2020) will overestimate the noise from OWFs, like Gode Wind 01, Gode Wind 02, and Borkum Riffgrund 01, by up to 8 dB. However, to get statistically valid soundscapes for continuous noise in the water, the often dominating factor shipping noise must be taken into account.

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# Interface Wave Contribution to Acoustic Particle Motion During Offshore Wind Farm Construction

16

Gopu R. Potty, James H. Miller, Ying-Tsong Lin, and Kathleen J. Vigness-Raposa

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**Abstract**

Construction of offshore wind turbines, via impact pile driving, generates and radiates intense acoustic and sediment borne energy through the water column and ocean bottom. The acoustic waves generated by impact pile driving include interface (Scholte) and shear waves in the ocean bottom in addition to compressional waves in the water column and sediments. The particle motion component of acoustic energy is particularly important to fishes and invertebrates. Scholte waves contribute significantly to the particle motion generated in the seabed and have maximum amplitude at the seafloor. Scholte wave contribution to the measured data is identified using Random Decrement technique in this study. Data measured during the construction of the Coastal Virginia Offshore Wind (CVOW) project are analyzed. The CVOW project involved the construction of two 6- megawatt wind turbines located 43 km east of Virginia Beach, Virginia. The turbines were pinned to the seabed using mono piles of diameter 7.8 m. A suite of particle motion sensors, including Ocean Bottom Recorders (OBX),

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was deployed to monitor the acoustic pressure and particle velocity on the seabed during December 2021 to January 2022. To examine the potential effects of sound on fishes and invertebrates, the behavioral thresholds and behavioral audiograms of some of these animals to the particle motion component of sound were compared to the measured data. These measurements were part of the Bureau of Ocean Energy Management (BOEM) sponsored Real-time Opportunity for Development Environmental Observations (RODEO) Program [Work supported by Bureau of Ocean Energy Management (BOEM)].

**Keywords**

Offshore wind farm · Scholte wave · Particle motion · RODEO · Coastal Virginia Offshore Wind

## Introduction

The quest for renewable sources of energy has led to the increase in the development of offshore wind farms over the last decade. Most of the current offshore wind farm developments are in shallow waters (less than approximately 30–50 m) and the preferred method of construction in shallow waters is either using a single cylindrical tower (monopile) or jacket type of structure. Installation of either of these wind turbine substructures requires driving piles into the seabed. Pile driving generates acoustic and seismic waves in the air, water column, and the seabed. Generation of Mach waves during impact hammer pile driving and its propagation characteristics has been well studied and described in Reinhall and Dahl (2011), Potty et al. (2011), Kim et al. (2022), Miller et al. (2016), Wilkes and Gavrilov (2017), and Amaral et al. (2020a). The acoustic energy radiated from the pile directly and reflected from the boundaries of the waveguide contributes significantly to the sound received at any distance from the pile in the water column. The contribution of sound produced during any construction of offshore wind farms can change the underwater soundscape and alter the habitats of marine mammals, fishes, and invertebrates by potentially masking communications or causing mortality, permanent threshold shift (PTS), temporary threshold shift (TTS), or behavioral disturbances for species that rely on sound for mating, navigating, and foraging (Amaral et al. 2020b).

In addition to the sound pressure generated in the water, compressional, shear, and interface waves are generated in the seabed that propagate outward from the pile in all directions (Amaral et al. 2020b). Compressional waves (P-waves) are the fastest traveling waves in the seafloor and are characterized by particle motion that is parallel to the direction of wave propagation, whereas shear waves (S-waves), which are slower than P-waves, have particle motion that is perpendicular to the direction of the propagating wave (Amaral et al. 2020b). Interface waves travel along the water-sediment interface at a speed about 90% of the shear waves with maximum particle motion amplitude at the interface (Kolsky 1963). Interface waves along the water-sea bottom boundary are called Scholte waves. The particle motion amplitude

of Scholte waves will have maximum amplitude at the interface and will decay exponentially away from the interface both in the water column and seafloor. Hence the effect of these types of waves will be felt only within one or two wavelengths from the seafloor. The low-frequency and slow-moving Scholte waves propagate over long distances and can generate large-amplitude oscillations along the water-sediment boundary. These Scholte waves have the potential to affect marine life that are sensitive to this type of disturbance living close to or within the seafloor sediment (Popper and Hawkins 2018).

Research interest in understanding the role of sound in the functioning of aquatic ecosystems, and the ways in which sound affects animal behavior and physiology, has gained increased attention in the recent past. While aquatic mammals use acoustic pressure, many fishes and invertebrates detect and use the acoustic particle motion (Nedelec et al. 2016). Since many aquatic animals sense particle motion, this component of the sound field should be taken into account while assessing the impact of the acoustic field on the environment. Measurements and model predictions of particle motion are few compared to acoustic pressure data, and experimental data on the response of fishes and invertebrates to particle motion are also scarce.

Many invertebrates appear to respond to vibrations (Roberts and Elliot 2017); for example, the bivalve *Scrobicularia plana* (peppery furrow shell) is sensitive to vibration and may therefore react to the foot-fall of its predator *Haematopus ostralegus* (oystercatcher). The anemone *Anthopleura elegantissima* (aggregating anemone) and the sea urchin *Strongylocentrotus purpuratus* (purple sea urchin) can detect vibrations within the frequency range of waves on the shore. Within the seabed, there is evidence to suggest that *Rimicaris exoculate* (deep sea shrimp) detect falling prey using vibration, and flatfish such as *Pleuronectes platessa* (European plaice) and *Solea* spp. may use vibrations to sense predators (Roberts and Elliot 2017). Roberts et al. (2016) conducted a study aimed to determine to what extent the common marine intertidal hermit crab, *Pagurus bernhardus* L. (Family Paguridae) is sensitive to substrate-borne vibration, and to fully define the sensitivity range and behavioral responses in relation to levels produced by anthropogenic activities. They presented results of this study along with published values of behavioral thresholds to vibration (both in water and substrate-borne) for crustaceans for comparison (Roberts et al. 2016).

Acoustic pressure and particle motion measurements were made during the construction phase of the Coastal Virginia Offshore Wind (CVOW) project. The CVOW project, which consists of two 6- megawatt wind turbines, is located 43 km east of Virginia Beach, Virginia. These measurements were made using Ocean Bottom Recorders (OBX) along with many other sensor systems. Four OBXs were deployed as a linear array, and data from one of the OBX are analyzed in the present study. Panel c in Fig. 1 shows the picture of an OBX and the axes definitions. The particle velocities are measured along these axes (x, y, and z directions), and acoustic pressure is measured on a hydrophone co-located with the directional sensors. The OBX sampled the data at 2 kHz during the CVOW deployment. In addition to the acoustic/seismic measurements, the OBX package also has a heading sensor which

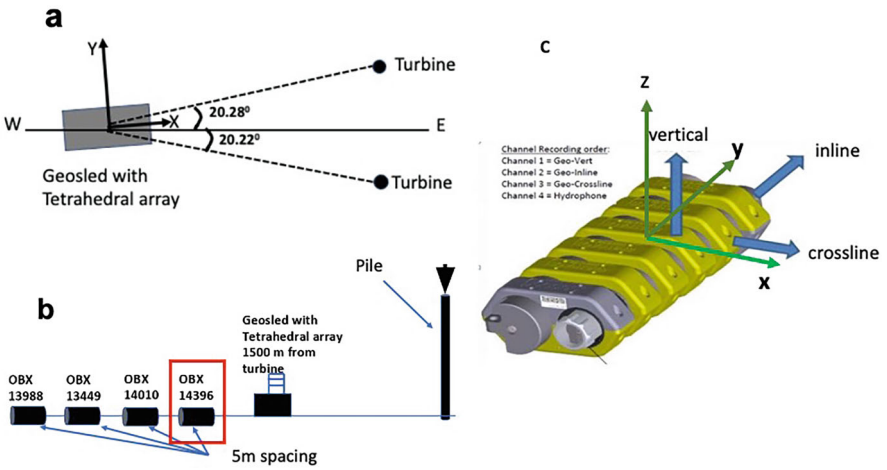



**Fig. 1** (**a**) Position of the OBXs in relation to the two CVOW turbines. The dashed line indicates the propagation path from the turbine to the OBX. (**b**) Geometry of the four OBXs which were attached to a sled. The sled was located at 1500 m from the pile driving locations. The spacing of the OBXs was 5 m. (**c**) Picture of an Ocean Bottom Recorder (OBX) with the axis definitions. Particle velocities are measured along the x, y, and z directions

measures the heading information that is used to properly orient the OBX, in the horizontal plane, in order to calculate the direction of arrival of acoustic signals. The CVOW turbines were installed on monopile type substructure (7.8 m diameter) with piles driven by impact hammer strikes. Locations of the OBX sensor packages in relation to the two turbines are shown in Fig. 1 (panels a and b). No bubble curtains were used during the construction of the north turbine whereas noise mitigation measures (bubble curtain) were used during construction of the south turbine. Based on the triangulated positions of the OBXs, the propagation of acoustic energy from the piles should arrive at 20° angle to the East (as shown in Fig. 1; panel a). The range from the pile to the OBX considered in this study is approximately 1500 m. This range was the closest distance from the turbine where deployments were permitted based on safety considerations.

Figure 2 (left panel) shows particle velocity measured on the first OBX sensor in the array (OBX 14396, highlighted by the red box in Fig. 1; panel b) during pile installation. Data shown in Fig. 2 correspond to the north turbine in which no sound reduction methods (i.e., no bubble curtain) were used during pile driving operations. Figure 2 shows approximately 1 minute of data during which approximately 50 hammer strikes were recorded. Each impulse in the figure correspond to one hammer strike. Upper panel on the left in Fig. 2 shows the vertical component of the particle velocity (in mm/s) whereas the lower panel shows the radial component (along the propagation path in the horizontal plane).

The radial component of velocity is obtained by projecting the X and Y components of the particle velocity components along the propagation path from the turbine to the OBX (shown in Fig. 1; panel a). The radial velocity is approximately

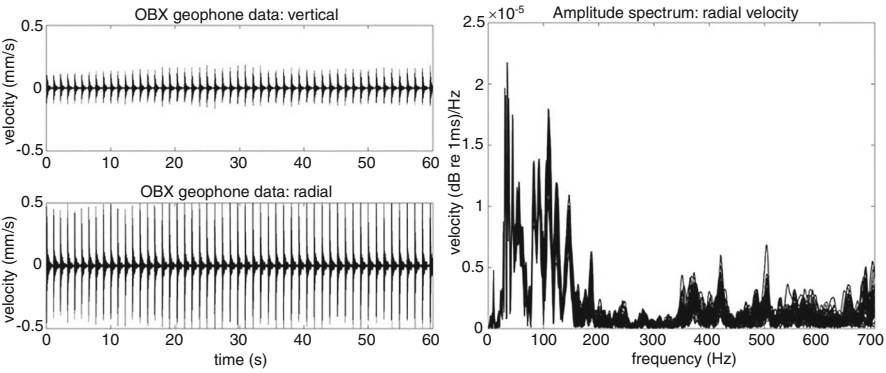



**Fig. 2** Particle velocity measured on the OBX sensor (left panel). Upper panel on the left shows the vertical component of the particle velocity (in mm/s) whereas the lower panel shows the radial component (along the propagation path in the horizontal plane). Right panel shows the amplitude spectrum of the radial velocity component

five times larger than the vertical velocity. Since the OBX is deployed on the seabed, it will measure shear and interface waves in addition to the compressional waves. This makes the OBX data complicated and identification of the contribution of the different wave types will be helpful in interpreting the data. The interface waves will have the maximum amplitude at the seabed; its amplitude decays exponentially away from the interface and cylindrically in range. The right panel in Fig. 2 shows the amplitude spectrum of the radial component of velocity from all 50 events shown on the left, showing the energy distribution as a function of frequency. The frequency content in the different events appears to be very similar showing very little spread in the spectral amplitudes especially at low frequencies. These low frequencies (below 200 Hz) seem to dominate the spectrum.

A technique known as Random Decrement (RayDec) approach is used to identify the Scholte wave arrivals in the data (radial and vertical components shown in Fig. 2). The RayDec approach provides the frequency band in which the Scholte wave arrivals dominate and the direction of arrival (DOA) of these arrivals. The RayDec algorithm calculates the frequencies and DOA of Scholte wave arrivals by searching the data (the vertical and horizontal components of velocity) at various times and frequencies and identifying the 90° phase shift between them (vertical and horizontal components) which is a characteristic feature of the Scholte wave particle motion (Hobiger et al. 2009; Potty et al. 2022).

Figure 3 shows the time-frequency distribution of acoustic energy (top left panel) calculated using wavelet-based analysis and the direction of arrival of Scholte waves (top right panel) calculated using the RayDec algorithm. The portion of the time series (six seconds long with five hammer strikes) which was used to create the time-frequency representations of acoustic energy and azimuth is also shown in bottom right. The time series analyzed corresponds to data from the north turbine (without bubble curtain). Azimuth estimates with a confidence level of 60% and above are only shown in the top right panel. Scholte wave arrivals can be seen in the frequency

band from 30 Hz to 90 Hz. Most of the arrivals, in the band 30 Hz to 70 Hz (indicated by blue color), appear to come at an angle of 20° north of the east-west direction. Arrivals can also be seen at 20 Hz and above 70 Hz which appears to come from different angles. This could indicate a different source (low-frequency micro-seismic energy) or coming through different interfaces/paths. There is also a 10 Hz and 60 Hz spectral component (left panel) which is present throughout and possibly not related to the hammer impact events.

The particle velocity and acceleration amplitudes at the measurement location, 1500 m from the turbines, are significant and have the potential to impact the behavior of fishes and invertebrates. The particle acceleration levels can be compared to the behavioral thresholds to assess the potential for impacts as shown in Table 1. The values shown are acceleration levels in dB re 1 μm/s$^2$ at four different

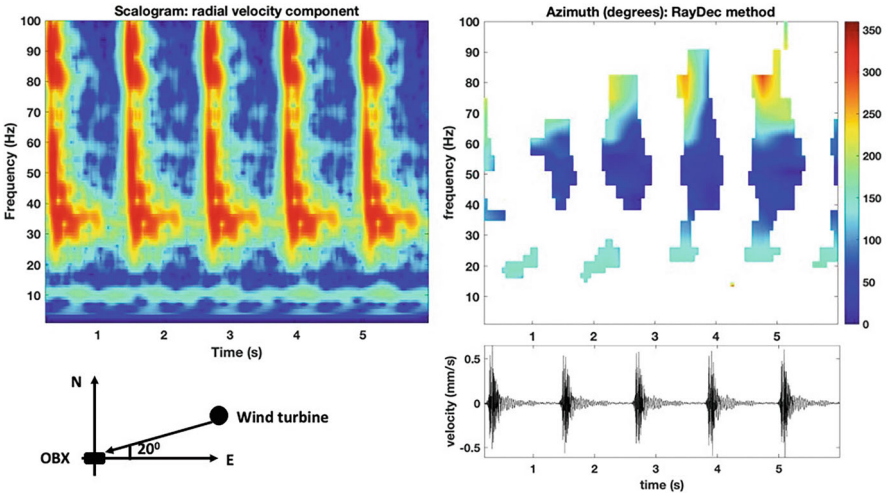



**Fig. 3** Time-frequency distribution of acoustic energy (top left panel) and direction of arrival of Scholte waves (top right) of the section of data (bottom right) analyzed. The propagation path from the north turbine to the OBX is indicated in the bottom left panel

**Table 1** Behavioral audiogram thresholds for fishes and behavioral response thresholds for a crustacean to vibration. Values taken from the literature and compared to CVOW data (Popper et al. 2014; Roberts et al. 2016). Bold numbers indicate threshold values lower than the CVOW levels. The values shown are acceleration levels in dB re 1 μm/s$^2$ at four different frequencies

| | 30 Hz | 35 Hz | 45 Hz | 83 Hz |
|---|---|---|---|---|
| Atlantic salmon | **41.33** | **41.1** | **40.7** | **38.5** |
| Dab | **33.33** | **33.33** | **35.58** | **28.8** |
| Atlantic cod | **31.33** | **28** | – | – |
| Plaice | **28.87** | **27.44** | **25.42** | **22.97** |
| Atlantic sand fiddler | **70.1** | **72.04** | 75.56 | 81.93 |
| *CVOW data* | *71.5* | *73.7* | *73.7* | *76.8* |

frequencies in the band where the acoustic energy is maximum (as shown in Fig. 3). The behavioral audiograms of four fishes (Atlantic salmon, dab, Atlantic dod and plaice) are taken from Popper et al. (2014), whereas the behavioral response threshold values for *Uca pugilator* (Atlantic sand fiddler) is calculated based on values from Roberts et al. (2016). Last row in Table 1 shows the acceleration level measured from the CVOW data as shown in Fig. 2. The threshold values are highlighted in bold numbers if they are lower than the measurement at different frequencies. The measured data are higher than the audiogram threshold values for all of the fishes considered. It indicates they can hear the sound, but whether that will lead to any behavioral changes in fishes is impossible to conclude without behavioral threshold values. The measured levels are slightly higher at two frequencies in the case of *Uca pugilator* (Atlantic sand fiddler) behavioral response threshold.

It should be noted that the results shown in Figs. 2 and 3 and Table 1 correspond to data from the north turbine which did not have a bubble curtain. The bubble curtain did reduce the acoustic energy produced by hammer impacts at the south turbine (with bubble curtain) measured on the OBXs. Figure 4 shows the comparison of the spectra of the data measured on OBX (indicated on Fig. 1 with a red box). As can be seen from Fig. 4, the received particle velocity (both vertical and radial components) levels drops by approximately 10 dB, for frequencies higher than 300 Hz, when bubble curtains were employed. At low frequencies (frequencies less than 300 Hz) the bubble curtain performs not as effectively as at higher frequencies. This frequency dependent performance of the bubble curtain is being investigated in detail currently by analyzing data from other receivers and through modelling.

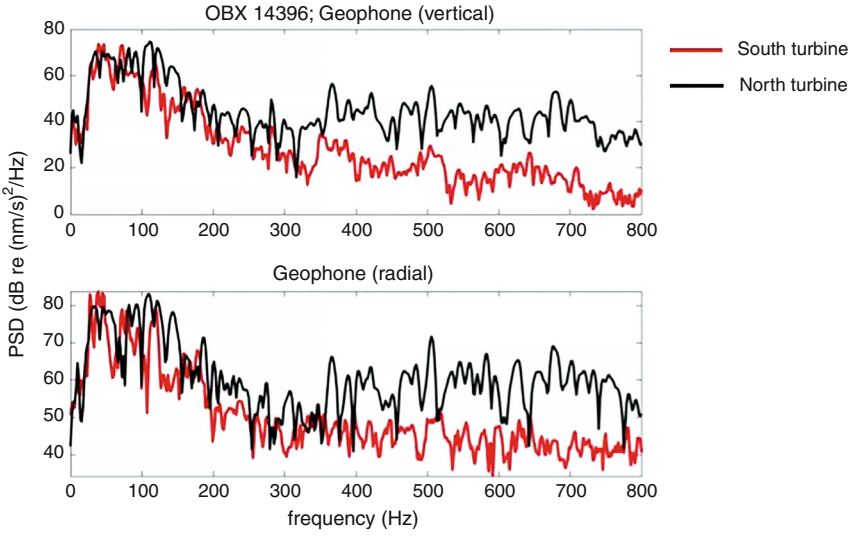



**Fig. 4** Power spectrum of OBX geophone data from the north turbine (black) and south turbine (red). The pile driving for the north turbine was done without bubble curtains whereas the south turbine employed bubble curtain

## Conclusions

The contribution of Scholte waves to the particle motion measured on the seabed at a range of 1500 m from impact pile driving operations was assessed in this study. The frequencies and direction of arrival of Scholte waves were identified using RayDec technique. The acceleration levels measured were also compared to behavioral audiograms and thresholds of some fishes and a crustacean. Preliminary comparisons indicate that the fishes and crustaceans considered in this study could hear or be behaviorally impacted by the pile driving operations at 1500 m.

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# Relation Between Underwater Noise and Operating Offshore Wind Turbines

17

Christian Terkelsen Holme, Matej Simurda, Stephan Gerlach, and Michael A. Bellmann

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**Abstract**

This chapter examines underwater noise measurements recorded within and outside operating offshore wind farms consisting of 6.3 MW and 8.3 MW turbines, respectively. Each wind farm had multiple hydrophones recording simultaneously with the nearest being located 70–100 m from a turbine, while the furthest was positioned 5 km outside the wind farm. Data were collected over 5 weeks to facilitate a statistical examination of how the magnitude of underwater noise changes with turbine activity (power production data) and natural fluctuations (e.g., tides and wind).

The results imply that there is no significant relation between the broadband underwater noise levels and turbine activity for any of the examined wind farms in the monitored distances (up to 70 m). Influence from natural fluctuations was on the other hand evident from the measured noise levels. Moreover, a comparison between recorded noise levels and simulated noise levels from an

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already-established empirical model shows that the model's extrapolated noise levels greatly exceed that of the recordings. This suggests that there are challenges associated with extrapolating aggregated results from smaller turbines to the realm of new, larger turbines. Thus, a new approach or more data are required to predict underwater noise from future operational wind turbines.

**Keywords**

Underwater noise · Offshore wind · Operational wind turbines · Environmental impact assessments · Marine mammals

## Introduction

Earlier studies have been able to trace the vibro-mechanical oscillations from operating offshore wind turbines to changes in the ambient underwater noise levels (Betke et al. 2005; Madsen et al. 2006; Norro and Degraer 2016; Yang et al. 2018). The vibrations are excited by mechanical components such as rotating blades and the turbine's operating gear box which causes vibrations to propagate down through the turbine tower and subsequently manifest as underwater noise radiating away from the foundation. However, while it is evident that turbine activity previously has been detectable in the recorded underwater noise levels, there remains an indefinite understanding of how the underwater noise level will scale with increasingly larger turbines and offshore footprint. For instance, the number of offshore wind turbines will over the coming decades increase exponentially due to an increasing demand of renewable energy resources (COM 2020, 741). Such expansion will cause the spatial coverage of turbines in the ocean to increase accordingly.

Tougaard et al. (2020) and Stöber and Thomsen (2021) set out to address this by aggregating primarily third party's underwater noise measurements with the purpose of establishing an empirical formula that could predict noise levels for larger turbines. Tougaard et al. (2020) used a general linear model to assess the correlations between sound pressure level and measurement distance, turbine size, and wind speed. Stöber and Thomsen (2021) used linear regression to assess the correlation where the distance dependency was removed by assuming that their data sources were subject to same transmission loss of 15 $\log_{10}(r)$ where r is the radial distance from the source. Both studies relied on aggregated data from primarily smaller turbines (0.45–6.15 MW) collected under different environmental conditions and distances to the turbines. Such methodologies are challenged by the difficulty in (1) disentangling the contribution from correlating variables (e.g., wind speed and power production) and (2) estimating the accurate sound source levels as the transmission loss varies across sites.

The motivation of this chapter is to get a better understanding of how underwater noise relates to turbine activity in the vicinity of an operating wind farm. This study has recorded the underwater noise levels in and around ~250, ~330, and ~450 MW offshore wind farms consisting of 6.3 or 8.3 MW turbines, respectively (see Table 1).

**Table 1** Wind farm information

| Wind farm | Nominal power | Wind farm size | Turbine type | Rotor diameter | Foundation type |
|---|---|---|---|---|---|
| Gode Wind 1 | 6.3 MW | ~330 MW | SGRE 6.3–154 (direct-drive) | 154 m | Monopile |
| Gode Wind 2 | 6.3 MW | ~250 MW | SGRE 6.3–154 (direct-drive) | 154 m | Monopile |
| Borkum Riffgrund 2 | 8.3 MW | ~450 MW | V164–8.3 (planetary gear) | 164 m | Monopile and suction-bucket jackets |

The relation between underwater noise and turbine activity is then examined by analyzing the co-variability of underwater noise, wind speed, and power production for multiple turbines sampled in different distances to the source. Data were collected over 5 weeks to facilitate a detailed statistical examination of how the magnitude of underwater noise changes with turbine activity (power production) and natural fluctuations (mainly tides and wind).

## Measurement Campaigns

Noise levels were recorded from three offshore wind farms in the Summer of 2020 (Gode Wind 1 and 2) and Spring/Summer of 2021 (Borkum Riffgrund 2) for five continuous weeks. The Gode Wind turbines have power production capacities of 6.3 MW, while the Borkum Riffgrund 2 turbines have a capacity of 8.3 MW. Each wind farm had four to five hydrophones recording simultaneously with the nearest being located 70–100 m from a turbine, while the furthest was positioned 5 km outside the wind farm. The 5-km position is used as a reference of the ambient noise signal when analyzing the impact from turbine activity on changes in underwater noise levels. The assumption is that there is no influence of turbine noise at 5 km due to the geometrical propagation loss having diminished the signal. All sites have similar water depth of approx. 30 m.

## Hydrophone Setup

Two types of hydrophones were deployed: (1) a completely continuous recording device and (2) a hydrophone that recorded for 10 min every 2 h (see Table 2). The reason for utilizing two different types of hydrophones is that these measurement campaigns were part of the underwater noise measurement requirement for offshore wind farm commissions stated in the German StUK4 (BSH 2013). Thus, in addition to the BSH-requested hydrophones, four additional continuously recording high-resolution hydrophones were deployed to record the underwater noise levels. These devices were positioned as close as safely possible to the pre-selected turbines per wind farm to ensure one could record the actual turbine signal. Furthermore, the

**Table 2** Hydrophone specs: the hydrophones deployed near suction-bucket jackets are marked in the "Position" column

| Wind farm | Measurement name | Recording device type | Hydrophone type | Recording resolution | Position |
|---|---|---|---|---|---|
| GOW01 | MP5 | itap | Bruel & Kjäer 8106 | Uncompressed MP3 format | 70 m to WTG |
| GOW01 | MP7 | Wildlife Acoustics SM2M | Bruel & Kjäer 8106 | Uncompressed MP3 format | 75 m to turbine |
| GOW01 | MP5a | Wildlife Acoustics SM2M | HTI-low noise | 16-bit WAV format | 150 m from turbine |
| GOW01 | MP7a | Wildlife Acoustics SM2M | HTI-low noise | 16-bit WAV format | 70 m from turbine |
| GOW02 | MP6 | itap | Bruel & Kjäer 8106 | Uncompressed MP3 format | Inside OWF (1 km to the nearest WTG) |
| GOW02 | MP8 | itap | Bruel & Kjäer 8106 | Uncompressed MP3 format | Inside OWF (1 km to the nearest WTG) |
| GOW02 | MP9 | itap | Bruel & Kjäer 8106 | Uncompressed MP3 format | Outside OWF (1 km to the nearest WTG) |
| GOW02 | MP4 | itap | Bruel & Kjäer 8106 | Uncompressed MP3 format | Inside OWF (1 km to the nearest WTG) |
| BKR02 | MP8 | itap | Bruel & Kjäer 8106 | Uncompressed MP3 format | Inside OWF (1 km to the nearest WTG) |
| BKR02 | MP4 | itap | Bruel & Kjäer 8106 | Uncompressed MP3 format | 100 m to WTG (suction-bucket jacket) |
| BKR02 | MP4b | Wildlife Acoustics SM2M | HTI-low noise | 16-bit WAV format | 63 m to WTG (suction-bucket jacket) |
| BKR02 | MP5 | itap | Bruel & Kjäer 8106 | Uncompressed MP3 format | 100 m to WTG |
| BKR02 | MP5b | Wildlife Acoustics SM2M | HTI-low noise | 16-bit WAV format | 94 m to WTG |
| BKR02 | MP10 | itap | Bruel & Kjäer 8106 | Uncompressed MP3 format | 5 km outside OWF |
| BKR02 | MP11 | itap | Bruel & Kjäer 8106 | Uncompressed MP3 format | 1 km outside OWF |
| BKR02 | MP12 | itap | Bruel & Kjäer 8106 | Uncompressed MP3 format | 1 km outside OWF |

All other positions are monopile foundations

measurement position close to the turbines were selected in such a way that any disturbing background noise from vessel lanes, other turbines in the vicinity, etc. were minimized.

Each hydrophone had a fixed position at 1.5 m above seabed throughout the campaign and followed the measurement protocols outlined in ISO 18406:2017 and BSH (2011). All hydrophones were calibrated by itap GmbH using a pressure chamber process.

The Gode Wind MP5a data were excluded from the analysis as the data were found to be compromised. This was determined by the deployment of two hydrophones close to each other (MP5 in this case).

## Data Pre-processing

Figures 1 and 2 show that the recorded broadband SPLs oscillate steadily with time (10 min averages of 5 sec SPL measurements). The variation is ranging from 115 to 130 dB, and this steady oscillation is therefore a dominant feature in the noise recordings. The periodicity is first examined through power spectral analysis of the calibrated noise recordings (5 s averages) as seen in Fig. 3. From this, it is evident that all four positions are influenced by the semi-diurnal tidal modes $M_2$, $S_2$, and $N_2$ (~2 cycles/day), while three of the measurement positions are further dominated by the $M_4$ shallow water overtide of the principal lunar (~3.85 cycles/day) (Bartels 1967). The MP7a (GOW02) data don't have the $M_4$ mode even though its water depth of ~33 m is like that of MP5a (GOW01) with ~32 m. This difference might be

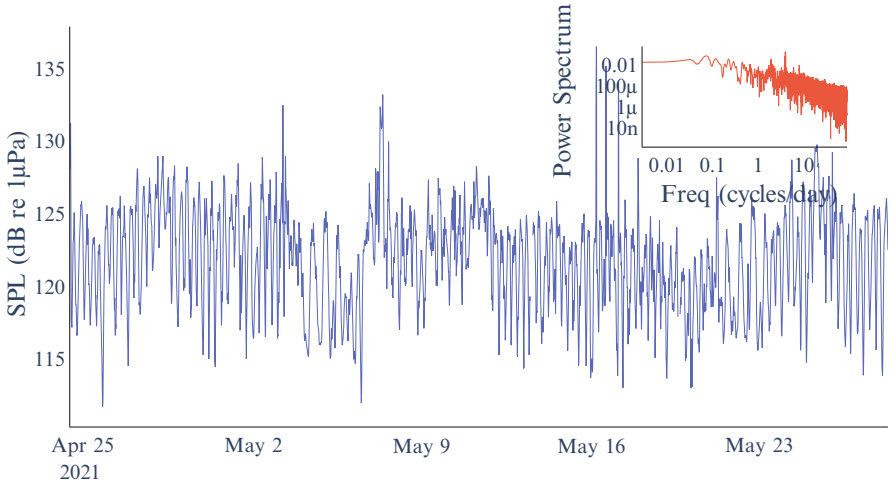



**Fig. 1** Time series of 10-min averaged broadband SPL (BKR02, MP4b, 63 m from a WTG). Top right subplot shows the corresponding power spectral density

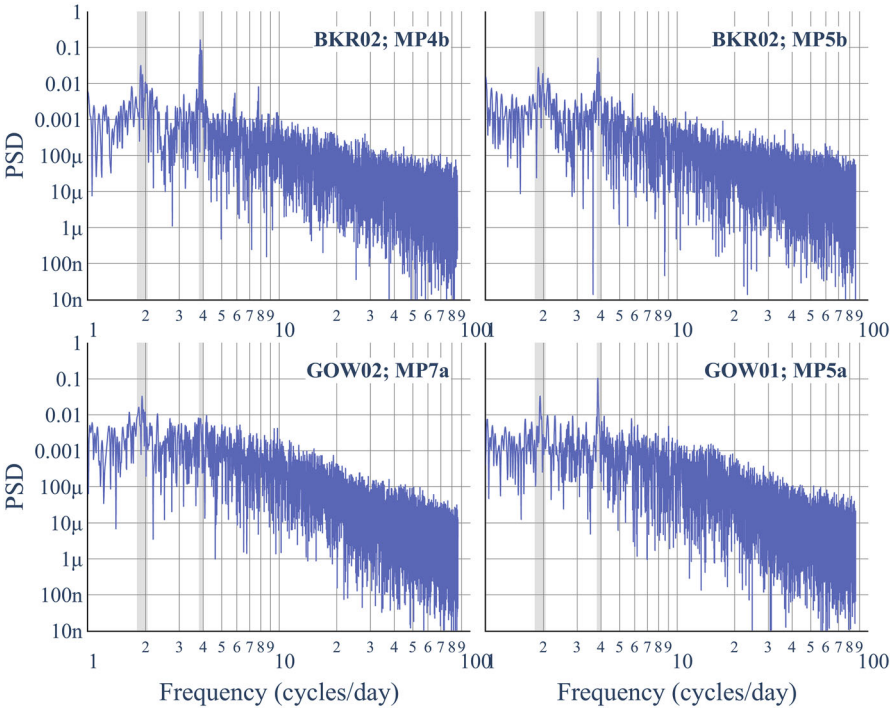



**Fig. 2** Power spectral densities (PSD) of broadband SPL computed through Lomb-Scargle Periodograms. The gray bars denote different observed tidal oscillations (e.g., M2 and M4)

a consequence of some site-specific attribute as the MP7a position is located ~12 km northeast of MP5a.

The power spectral analysis indicates that the changes in SPL predominantly are attributed to tidal variations. Thus, daily averaged SPLs are calculated to remove any bias from the subsequent analyses with wind and power production data. As SPL is a logarithmic metric, the daily averages are calculated through logarithmic means. This is done by (1) taking the anti-log of the SPL series, (2) calculating the mean, and (3) re-applying the decadic logarithmic factor again.

Additionally, in the subsequent analysis, a 30-min moving median filter is applied to the data to remove any noise impacts caused by nearby passing vessels. This is applied prior to calculating the daily averages. The data shows that service vessels visiting the turbines occasionally caused elevated noise when passing the deployed hydrophones. This was visible in the raw broadband SPL data as a sudden and short-duration noise increase of 5–10 dB SPL. Such a change cannot result from turbine-induced noise as it was observed during low winds and since it was characterized by a sudden increase and decrease in noise. Moreover, it takes ~10 min for the turbine to either steadily ramp up or down from full capacity. The sudden noise increase could therefore not be a turbine signal.

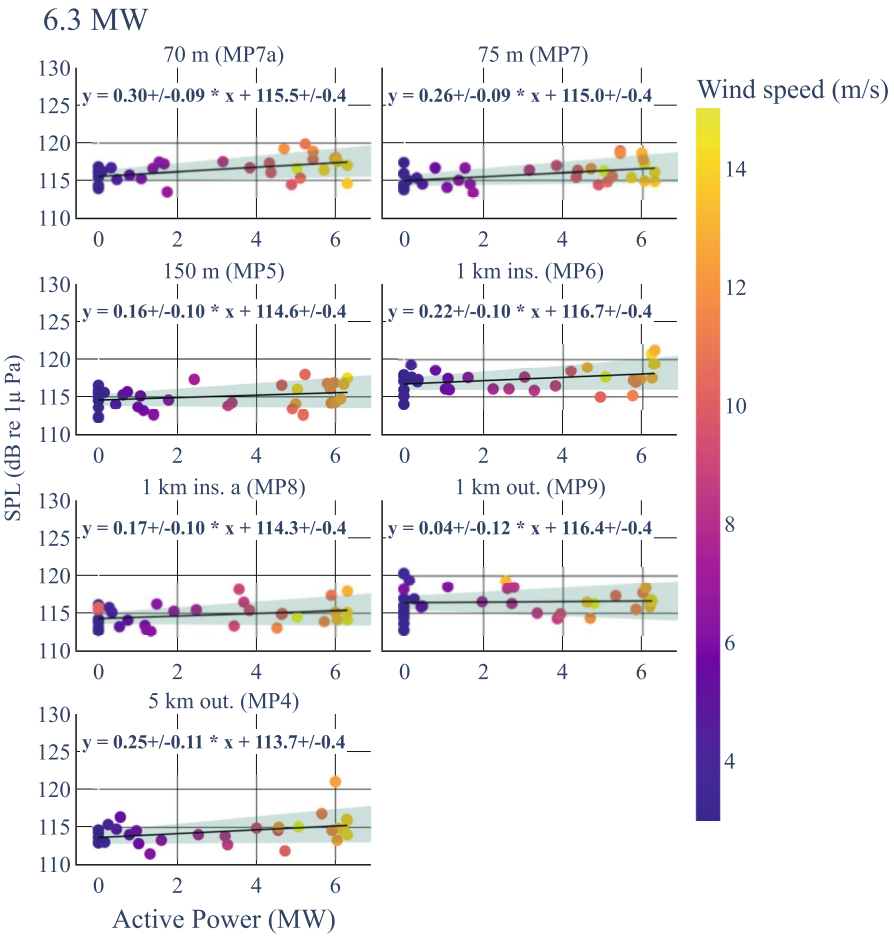



**Fig. 3** GOW01 and GOW02: Daily averaged SPL with respect to power production at the nearest turbine and wind speed (color). Regression coefficients denote the best estimate ± the corresponding standard deviation. Shaded area is 95% confidence intervals based on a two-sided Student *t*-test. Turbines are off at 0 MW, and there is no significant difference in slopes across measurement positions

## Underwater Noise Versus Turbine Activity

The relation between underwater noise and turbine activity is examined by using power production data as a proxy of turbine activity. This metric is directly proportional to the blades' revolution per minute (RPM) and therefore an indicator of the vibrational motion and general activity within the nacelle. In the analysis, 10-min time series of mean active power is converted to daily averages to facilitate a comparison with the daily averaged SPLs. For each hydrophone, the SPL data is

compared with the nearest turbine's power production data. This is because there can be minor differences in the power production data between turbines due to site-specific conditions (e.g., wakes), maintenance, or curtailment requirements. Besides the daily averaged power production data, the daily averaged wind speed measurements recorded at the nacelle are also utilized in the analysis (10 min means converted to daily averages). This level of detailed information enables an examination of how the underwater noise levels change with both turbine activity and naturally caused phenomena (e.g., wind variations).

Theoretically, there exists a log dependency between the displacement velocity (turbine activity) in the structure and underwater SPL. This can be concluded as there exists a log dependence between acoustic pressure in water and SPL. From assuming a linear mechanical system, the displacement velocity in the structure (vibrations) is directly proportional to the RPM (turbine activity). In water, the propagation is in the far field through plane waves where sound pressure is directly proportional to particle velocity. The law of continuity applies for the normal pressure and displacement as the boundary conditions at the pile-water interface. Thus, the particle velocity in water is directly proportional to the displacement velocity in the structure, which implies that the displacement velocity in the structure is directly proportional to acoustic pressure in water. As there is a log dependency between acoustic pressure and SPL, then there also exists a log dependency between the displacement velocity (turbine activity) in the structure and underwater SPL.

The relations between SPL, active power, and wind speed are first examined in Fig. 3 (6.3 MW) and Fig. 4 (8.3 MW). For all measurement positions, it is observed that there is no logarithmic relation between turbine activity and SPL which would be expected. For Gode Wind 1 and 2, some positions exhibit a minor linear SPL increase with power production and wind speed, whereas no relation is evident for Borkum Riffgrund 2. The relations and their significance are first quantified by estimating the linear least-square regression coefficients, their standard deviations, and the corresponding 95% confidence intervals (CI; estimated through a double-sided Student *t*-test).

By comparing the regression coefficients at different positions relative to the turbine and wind farm, it is evident that there is no significant difference in slope (dB/MW) with distance. This is further illustrated in Fig. 5 where the significance of difference in estimated slopes is tested using the 5-km control measurements. This comparison shows that the slopes are not statistically different. Thus, the minor change in elevated broadband SPL values for Gode Wind 1 and 2 must be a wind contribution rather than an influence from turbine activity as a similar relation exists at distances of 5 km away from the wind farm.

The dual influence from the co-varying turbine activity and wind speed on the underwater noise levels is a testimony to the challenges in disentangling the contribution of two statistically dependent variables. Such an effort is therefore only possible if one has multiple hydrophones recording simultaneously in different distances to the turbine.

For Borkum Riffgrund 2 (8.3 MW), there is no impact on daily averaged SPL from neither power production nor wind speed as shown in Fig. 4. This contrasts

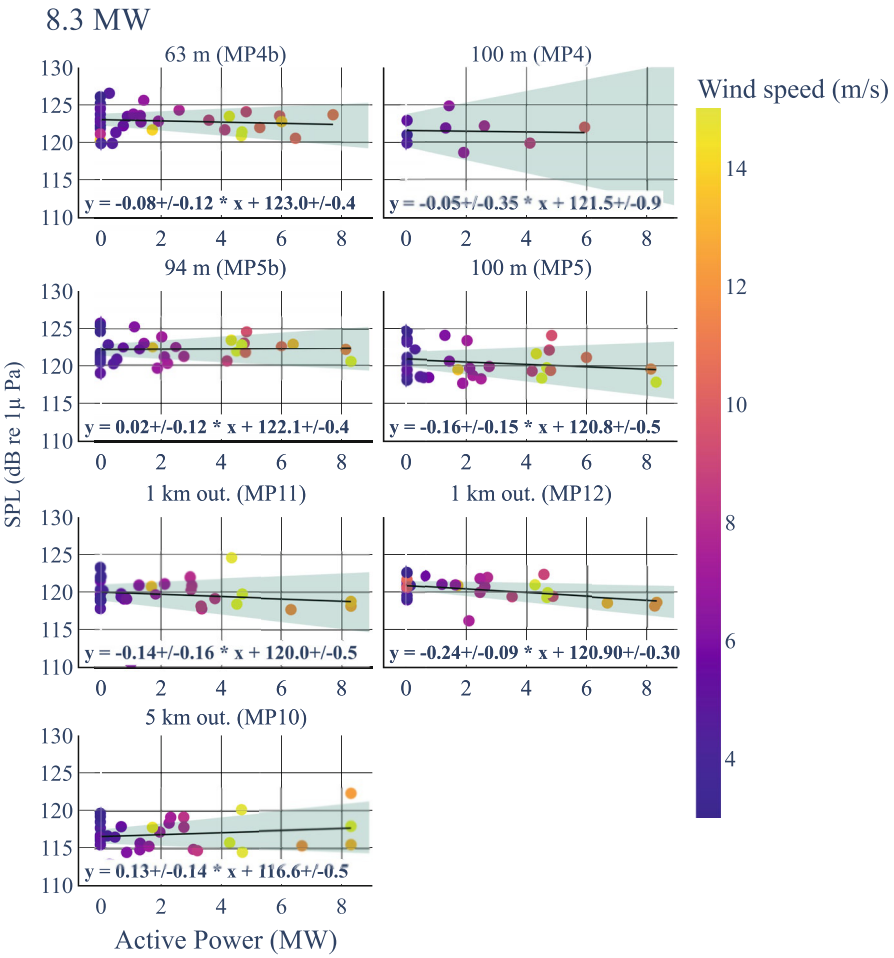



**Fig. 4** BKR02: Daily averaged SPL with respect to power production at the nearest turbine and wind speed (color). Regression coefficients denote the best estimate ± the corresponding standard deviation. Shaded area is 95% confidence intervals based on a two-sided Student *t*-test. Turbines are off at 0 MW, and there is no significant difference in slopes across measurement positions

with the smaller 6.3 MW turbines at Gode Wind where a minor wind dependence might exist (slope is significantly different from 0). The Borkum data were sampled in the Summer of 2021 which generally was a period characterized by low winds. This is also noticeable in the data as a slight non-uniform distribution of SPL data points for different MW. Regardless, it is observed that the variation in SPL during turbine standstill greatly exceeds any potential for turbine-inflicted SPL increases at its maximum power production capacity (8.3 MW). Thus, the great variation in SPL at turbine standstill must indicate that there is some other process that causes the signal to change in low wind conditions (either natural or anthropogenic). Other

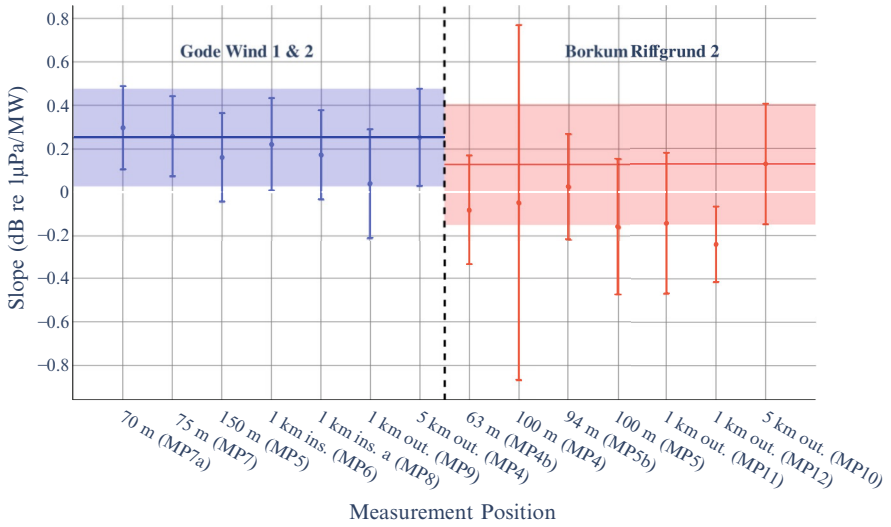



**Fig. 5** Slope (dB/MW) at different positions relative to the turbine and wind farm. Error bars are the 95% CI, while the horizontally oriented bars represent the reference measurements 5 km outside the wind farm with its 95% CI. Left part of the figure is Gode Wind 1 and 2, and the right part of the figure is for Borkum Riffgrund 2

anthropogenic sources could be vessel traffic not identified and removed in the pre-processing (either recreational or transportation), whereas natural influences could be changes in water currents and heavy precipitation which don't average out in daily means.

Moreover, when comparing the average broadband SPL across sites, the values are up to 5 dB greater at Borkum Riffgrund 2 than at Gode Wind despite no turbine-activity dependence. This indicates that there are general differences in ambient noise levels across sites. It can generally be noted that the ambient noise levels are higher across the German Bight than many previously monitored sites (e.g., those presented in Tougaard et al. (2020)). Thus, part of the missing relation between SPL and power production is likely a direct consequence of this area's high ambient noise levels. With that said, there are observed neither single nor cumulating effects from interfering wind turbines on the ambient noise levels.

Even though the analysis shows no link between turbine activity and SPL, it is still clear that the average noise levels throughout the measurement period are lower further away from the turbines. For instance, BKR02 MP9, MP10, and MP11 have lower noise levels than those inside the wind farm. The three measurement positions are located, respectively, 1 and 5 km east of the wind farm. Thus, the reduced noise levels might be a consequence of the hydrophones being located further away from the nearby vessel traffic route (German Bight, Western Approach).

Moreover, both monopile and suction-bucket jacket foundations are examined in the chapter. Most of the foundations are monopiles, but there are recordings for suction-bucket jackets at BKR02 MP4, MP4a. However, the results show that there

are no noticeable differences on the broadband SPL between the two foundation types. This is evident as there are no differences observed neither in the power spectral densities (Fig. 2) nor on the dependence from turbine activity (Fig. 4).

## Comparison Between Data and Model

This section compares the underwater noise measurements with the model predictions of Tougaard et al. (2020). Their model is currently the most comprehensive published study and therefore the best publicly available knowledge and starting point when wanting to predict the underwater noise from future operating wind farms. As noted by Tougaard et al. (2020), this empirical equation has not been tested against independent measurements to validate its predictive power (e.g., cross-validation tests). Thus, the motivation for this chapter is to test the model to benchmark its predictive capabilities as their model primarily was based on data from smaller turbines.

This study only uses Gode Wind in the comparison as the broadband SPL signal had a minor dependence on the wind speed (albeit no dependence on turbine activity). Borkum Riffgrund 2 is deemed unsuitable for this analysis as it is a pure ambient noise signal. The recorded noise levels can therefore not be a representative for an 8.3 MW turbine.

In the model, the Tougaard et al. (2020) equation is used:

$$\mathrm{SPL}_{\mathrm{predictor}} = 109\ \mathrm{dB} - 23.7 \cdot \log_{10}\frac{\mathrm{distance}}{100\ \mathrm{m}} + 18.5 \cdot \log_{10}\frac{\mathrm{wind\ speed}}{10\,\frac{\mathrm{m}}{\mathrm{s}}} + 13.6 \cdot \log_{10}\frac{\mathrm{WTG\ size}}{1\ \mathrm{MW}} \quad (1)$$

For this case, the maximum scenario is analyzed where wind speed is set to 13 m/s, produced power set to 6.3 MW, and the distance as a variable parameter respective to the source. Firstly, all turbines within Gode Wind 1 and 2 (~580 MW) are positioned on a uniform grid based on their coordinates. Secondly, the noise contribution from all 97 sources is calculated based on Eq. 1 for each grid cell. Here the different sources are added together as:

$$\mathrm{SPL}_{\mathrm{sum}} = 10 \cdot \log_{10} \sum_{i}^{N} 10^{\frac{\mathrm{SPL}_{\mathrm{predictor_i}}}{10}} \quad (2)$$

The model prediction is plotted in Fig. 6, and the comparison with the corresponding noise recordings is presented in Table 3. For the measured SPL, only the noise data for the same wind speeds and active powers are used to ensure compatibility with the model estimates. For this, the linear regression coefficients presented in Fig. 3 are used as a representative of the SPL distribution (data-based SPL).

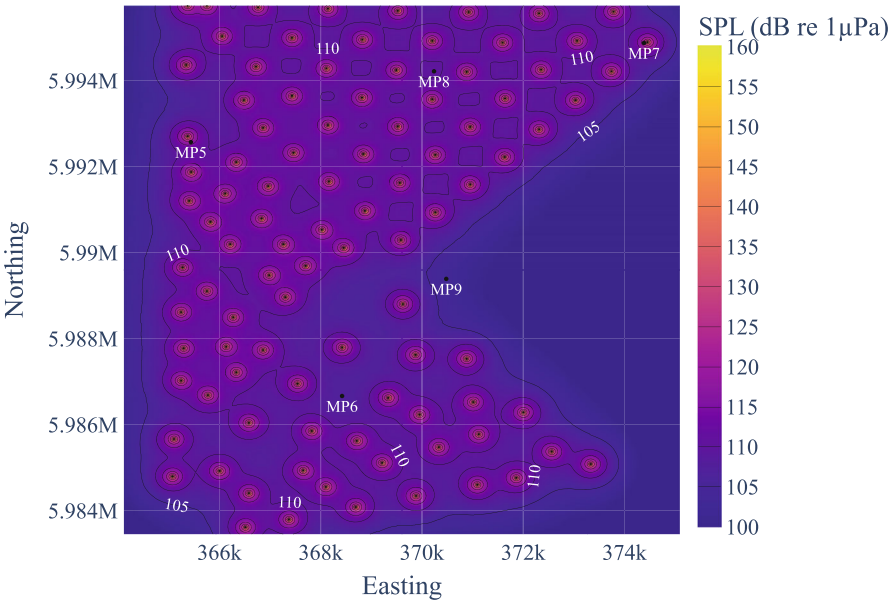



**Fig. 6** Modeled SPL for a ~580 MW OWF based on Tougaard et al. (2020) (assuming 6.3 MW turbines and wind speed of 13 m/s). The *x*-*y* axis contains Easting and Northing values (*m*) with respect to origin

**Table 3** Comparison between model-based and data-based estimates (mean ± 1 standard deviation)

| | Distance | Model-based SPL | Data-based SPL | Model overestimation |
|---|---|---|---|---|
| MP7a | 70 m | 125.4 dB | 117.3 ± 0.9 dB | ~8 dB |
| MP7 | 75 m | 124.7 dB | 116.5 ± 0.9 dB | ~8 dB |
| MP5 | 150 m | 117.9 dB | 115.6 ± 1.0 dB | ~2 dB |

The data-based estimates are calculated from the linear regression coefficients between power and SPL. Only the nearest positions are presented as the model estimates decreases with distance

From Table 3 it is evident that the Tougaard et al. (2020) model overestimates the noise by ~8 dB at the nearest positions of ~70 m. The data-based and model-based results converge as the distance from the turbines increases. This is a result of the model yielding decreasing noise levels with distance due to its transmission loss, whereas the data-based estimates almost remain constant within the recorded distances. Hence, no comparisons were made with the recordings further away as there was no turbine effect on the broadband SPL.

These results can therefore provide a contribution to benchmarking of underwater noise predictions for future studies to avoid under- or overestimations of noise levels. While the results don't show what SPL to expect from these turbine types and sizes, they do provide an upper limit to what the noise levels can be in a certain distance.

## Conclusion

This chapter found no changes to the ambient broadband SPL from neither 6.3 nor 8.3 MW operating wind turbines. While this partly was attributed to the high ambient noise levels of the German Bight, it remained evident that natural effects (e.g., wind speed and tidal changes) were the dominating forces behind changes to the ambient noise levels. Nearby crossing vessels caused short-term elevated noise levels, which were filtered out in the analysis. Furthermore, the results showed that the recorded noise levels were up to 8 dB lower in distances up to ~70 m to the turbines than those predicted through the Tougaard et al. (2020) model. This suggests that there are challenges associated with extrapolating aggregated results from smaller turbines to the realm of new, larger, and technologically different turbines. Such findings have implications to efforts set to predict the noise impact of future offshore wind farms, as broadband SPL does not scale with increasingly larger turbines.

**Acknowledgments** The authors would like to extend their special thanks to the German Bundesamt für Seeschifffahrt und Hydrographie (BSH) for making this study possible.

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# Sound Emissions from Ultrasonic Antifouling Equipment

18

S. Bruce Martin, Alexander O. MacGillivray, Jason D. Wood, Krista B. Trounce, Dominic J. Tollit, and Kanachi Angadi

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**Abstract**

The Vancouver-Fraser Port Authority-led Enhanced Cetacean Habitat and Observation (ECHO) Program has managed underwater listening stations (ULSs) on the approach to the Port of Vancouver since 2015, measuring the sound levels generated by thousands of vessels. Since 2017, these systems have measured at or above a 128 kHz sampling rate.

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Anomalously high sound levels were observed in 212 of the measured ships signatures at frequencies typically associated with navigational, fisheries, and scientific sonars. Sixty-one of these detections were found to originate from a novel continuous sound source in the 20–30 kHz frequency range. During a separate underwater noise monitoring program, a similar high-frequency continuous sound source was identified proximate to a berthed vessel. The vessel engineer identified it as an ultrasonic antifouling system. The sounds from these systems are detectable at 4–6 km from the vessels in deep water.

The measurements indicate that echolocation by lower-frequency delphinids, such as killer whales, may be completely masked when a ship is 3 km away and that porpoises flee from the source at distances of 1.5 km. In shallow waters, a porpoise 70 m from the source is predicted to experience temporary threshold shift (TTS) after 1–2 s, and permanent threshold shift (PTS) after 200 s. It is recommended that use of these systems be restricted or prohibited when there is a possibility of exposing marine mammals to potentially harmful sound levels.

**Keywords**

Antifouling · Vessels ultrasound · Marine mammals · Masking · Hearing threshold shift

## Introduction

It is well known that the underwater radiated noise (URN) from ocean-going vessels is highly variable and depends on factors such as vessel length, type, and speed as well as draft, trim, age, and propulsion system (Wales and Heitmeyer 2002; Simard et al. 2016). Long-term measurements of opportunity are an important means of characterizing the variability of vessel URN. With that in mind, the Vancouver-Fraser Port Authority (VFPA) worked with Ocean Networks Canada (ONC) and JASCO Applied Sciences to use the East Node of the VENUS underwater observatory in the Strait of Georgia as platform for an underwater listening station (ULS). The initiative was part of VFPA's Enhancing Cetacean Habitat and Observation (ECHO) Program. Ships entering or leaving the port were encouraged to participate in the study by traveling past the ULS (Fig. 1). Their automatic identification system (AIS) messages triggered an automated analysis of the vessel's URN. The Strait of Georgia ULS program operated from September 2015 to April 2018 and yielded a total of 7693 vessel URN measurements.

Based on the success of the Strait of Georgia ULS, Transport Canada commissioned JASCO to establish a ULS at Boundary Pass, which is directly on the traffic route in and out of Vancouver (Fig. 1). From August 2018 to May 2020, autonomous acoustic recorders were employed until the cabled ULS hardware was installed at Saturna Island. As an example of the value of large data sets of vessel sound levels, analysis of the combined data from measurements in Strait of Georgia and Boundary Pass and measurements made by the Joint Monitoring Programme for Ambient

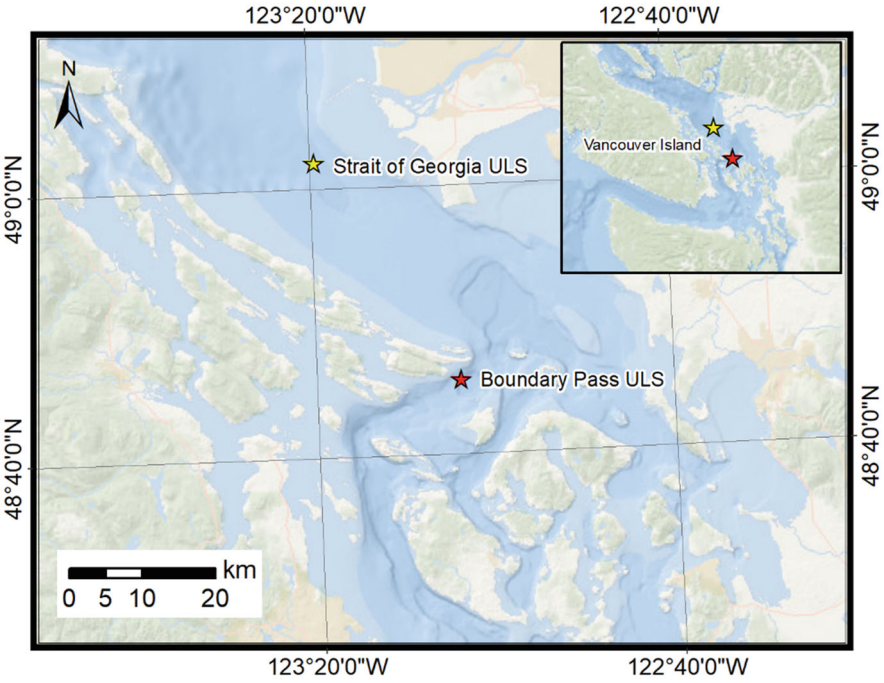



**Fig. 1** Location of the measurement zone for the Strait of Georgia and Boundary Pass Underwater Listening Stations (ULSs)

Noise North Sea (JOMOPANS) resulted in a robust new model for vessel source levels that depends on length, speed, and vessel class (MacGillivray and de Jong 2021). The new model stipulates that the vessel spectra have a 6 dB standard deviation due to the variability in vessel URNs that were measured. The availability of the Boundary Pass ULS has facilitated other URN research, such as evaluating the effects of slowing down vessels (JASCO Applied Sciences and SMRU Consulting 2020), developing methods for estimating the density of marine mammals (Urazghildiiev et al. 2020), and measuring changes in vessel source levels after refitting their propulsion systems.

The sampling rate during the Strait of Georgia Program and early phases of the Boundary Pass project was 128 kHz so that frequencies up to 64 kHz are available for analysis in the data (the Boundary Pass sample rate was increased to 512 kHz when the ULS was deployed in 2020). Active acoustic sources (sonars) such as navigational, fisheries, and scientific echosounders emit sound below 64 kHz and thus should be present in the data if they were in use. An exploratory analysis of the data sets was performed to evaluate the prevalence of sonars in the Strait of Georgia and Boundary Pass data available up to November 2019. Of 15,983 vessel measurements recorded by that time, sonars were detected in 212 measurements. Of those, 110 were single-frequency navigational echosounders and 41 were multifrequency

sonars (likely fisheries and scientific echosounders). The remaining 61 detections were a novel continuous sound source in the 20–30 kHz frequency range. The source was emitted by 14 unique vessels in three vessel classes: 2 tugboats, 2 bulkers, and 10 passenger ships. The received sound levels reached a maximum twice during the passage of larger vessels, suggesting distribution along the vessel hull. The source was detectable at 4–6 km from the vessels, which suggests that the source did not have the downward beam pattern of a typical echosounder.

The ULS systems in fixed locations have proven to be extremely valuable for answering a multitude of research questions; however, in many cases, the measurements must be made at the locations of the sound sources. In 2019, SMRU Consulting conducted an underwater noise monitoring study for VFPA to measure the sounds levels proximate to berthed ships while they were on or off shore power. They also measured the sound in a confined area (Burrard Inlet) from the harbor operations including a Seabus and regular seaplane flights (Angadi et al. 2020). The same high-frequency continuous sound source centered at 23 kHz detected in the ULS data was also identified proximate to a berthed vessel when it was on shore power or its own power. The vessel engineer identified the sound source as an ultrasonic antifouling system. These relatively new devices place ultrasonic transducers along the hull or in onboard water piping and storage equipment to deter biofouling and the transfer of invasive species. These systems are available for full-size commercial vessels and pleasure craft.

The ultrasonic antifouling systems consist of a controller and one or more transducers that are secured to the inside of a ship's hull plates or the outside of pipes and fittings that require protection against the growth of algae, macroalgae, and invertebrates such as barnacles. Accumulations of biofouling on a vessel hull require vessels to consume more fuel and emit more greenhouse gases to achieve the same speed through water. The intention of developing the ultrasonic cleaning systems was to reduce leachates from toxic paints that are an alternate method of reducing biofouling (Legg et al. 2015; Habibi et al. 2016). Developing commercial ultrasonic antifouling systems is the result of research into methods of controlling harmful cyanobacterial blooms (blue-green algae (BGA)). Lee et al. (2001) found that exposure of BGA to ultrasound for 1–5 min caused gas vacuoles inside the cells to collapse, leading to BGA settling to the bottom. This initial work also suggested that the ultrasound affected the ability of the cells to photosynthesize, a result that was subsequently validated (e.g., Zhang et al. 2006). Further work found that ultrasound also affected the ability of barnacles to settle on surfaces, likely by killing the cyprids as a result of cavitating their internal structures (Guo et al. 2011). The best results occurred using an ultrasound frequency of 23 kHz at a pressure of 20 kPa (200 dB re 1 $\mu Pa^2$). These systems have also been shown to cause avoidance in Cuvier's beaked whales (Trickey et al. 2022).

Promotional materials from manufacturers of commercial ultrasonic systems that are now available suggest that the transducers protect a radius of 6–10 m. Each transducer consumes approximately 20 W of electrical power, which, if the transducer were perfectly efficient, would result in an acoustic source level of approximately 187 dB re 1 $\mu Pa^2m^2$. The electrical-to-acoustic conversion efficiency of the

transducer, coupling to the vessel hull, hull material, and number of transducers along the hull all affect the sound levels coupled into the water. The systems are available for commercial-size vessels; however, the primary market for the systems appears to be pleasure craft and yachts.

No trade-off analysis of whether the impacts of sound from these systems may have a greater impact on the environment than the leachates has been performed. Legg et al. (2015) identified effects of noise as a possible environmental effect; however, no conclusions were drawn. The goal of this chapter is to present measurements of sound levels emitted from ultrasonic antifouling equipment that were measured in the real world that can serve as an input to an impact trade-off study. The measurements have been interpreted based on our current knowledge of the thresholds at which sound affects marine mammals. This chapter (1) describes how the measurements were made at the ULSs and Burrard Inlet; (2) describes the source and its radiated sound levels; and (3) provides estimates of the possible effects of these sources on marine mammals in terms of hearing threshold shifts and masking potential.

## Methods

### Measurements

The acoustic data analyzed in this study were collected by various equipment. From September 2015 to April 2018, the Strait of Georgia ULS data were collected by an AMAR G3 in streaming mode (JASCO Applied Sciences, Dartmouth, Canada), connected to M36-V35-100 hydrophones(−164 dBV/μPa nominal sensitivity level; GeoSpectrum Technologies Inc., Dartmouth, Canada). The system was calibrated before deployment with a G.R.A.S. 42 AA pistonphone calibrator (Holte, Demark). During its recording period, system sensitivity was checked daily using an icTalk source (OceanSonics, Truro, Canada). Data from December 2018 to October 2019 were collected using autonomous AMAR G4 recorders suspended above the seabed. These systems were also fit with GeoSpectrum M36-V35–100 hydrophones that were calibrated at deployment and retrieval using the G.R.A.S. 42 AA calibrators. All AMAR data were sampled at 128 kHz. The positions of the hydrophones were determined to within 1 m accuracy by surface localization of the acoustic releases on the moorings. The positions of the ships measured by the AMARs were determined using their AIS tracks. The Strait of Georgia ULS was at a 170 m water depth, and the AMARs at the Boundary Pass location were at 197 m water depth.

The Burrard Inlet data were collected with two Coastal Acoustic Buoys (SMRU Consulting, Friday Harbor, USA). These buoys had two Reson 4014 hydrophones (Teledyne Reson, Slangerup, Denmark), one at mid-water column (10 m) and one just above the seabed (20 m). The hydrophones were sampled at 250 kHz and were calibrated before deployment and after retrieval using a G.R.A.S. 42 AA calibrator. The first Coastal Acoustic Buoy was 73 m from the berthed vessel, and the second was 237 m from it.

## Data Analysis

### Measured Sound Levels

The basic analysis applied to all the data followed the Welch (1967) method: time series audio data were converted to power spectral densities using the Fast Fourier Transform (FFT) with a 50% overlap and a Hann window. A frequency-dependent calibration curve for the hydrophone and recorder was applied to the spectra. The spectra were then summed to generate decidecade sound pressure levels as needed for the analysis (Heaney et al. 2017).

### ULS Vessel URN

The ULS hydrophone data were processed using ShipSound (JASCO Applied Sciences, Victoria, BC) to obtain the URN measurements for each vessel passage. ShipSound tracks vessels broadcasting on AIS as they transit the 1 × 4 km measurement funnel at the measurement sites (Fig. 1). When the transits are complete, the URN measurements are automatically performed. To complete the measurement record, the AIS data are employed to compute the speed over ground of the vessel and identify other nearby vessels. Online web services are queried to obtain the wind and current speeds which are used to compute the speed through water of the vessel.

Source level (SL) was calculated by adding the propagation loss ($N_p$) to the received sound pressure level ($L_p$) at the hydrophones (Eq. 1).

$$L_{\text{URN}} = L_p + N_p \text{ dB} \tag{1}$$

At the ULS, where measurements are collected at distances less than 1 km, propagation loss at the 20–30 kHz operating frequency of the ultrasonic antifouling systems is well approximated by spherical spreading plus seawater absorption:

$$N_p = 20 \log_{10} r + \alpha r \text{ dB} \tag{2}$$

where $r$ is the slant range from the vessel to the hydrophone and $\alpha$ is the frequency dependent absorption of sound by seawater computed using the formula of Ainslie and McColm (1998).

To obtain the URN, analysis was performed for all measurements within a data window defined by a ±30° azimuth angle centered from the closest point of approach (CPA) to the hydrophone. For acoustic data within the measurement window, ShipSound analyzes acoustic data in decidecade frequency bands from 10 to 63,100 Hz. The data are processed using 1-s FFTs with 50% overlap and a power-normalized Hann window which generates the power spectral density (PSD) levels versus time. Since the AIS transmitter/receiver is not at the same location as the vessel's acoustic source, the cepstrogram methods described in Hannay et al. (2016) were employed to find the acoustic CPA. A human analyst then verified, and adjusted if necessary, the automated CPA time. The broadband and decidecade SL values were computed in decibels by taking the linear average of the SL from all 1-second sample locations within the ±30° data window. Background sound

pressure levels (NL) are computed by averaging measured sound levels in the 1-minute intervals just before and just after the vessel was within the measurement. If the difference between the measured SPL and NL was between 3 and 10 dB, the SPL was adjusted in accordance with the ISO 17208-1 standard. When the difference was less than 3 dB, data from those decidecade band levels were not included in the final source level results.

The acoustic CPA identified above was used by ShipSound to analyze the full ship signature, including the ultrasonic antifouling system. Since the ultrasonic antifouling sounds are distributed along the length of the vessel and the source levels computed here assumed a point-source configuration, they are only an approximation of the true source level. However, most results presented are based directly on the SPL data and do not rely on the source level estimates.

### Automated Identification of ULS Vessel Passages Containing Sonar

The ShipSound data were exported from the database to a CSV file with one row per vessel passage. The data were searched to find any passages where any of the decidecade band received levels in the sonar band (20 kHz and higher) were greater than 100 dB, and the level in any one of the bands was at least 3 dB greater than the level in the 10 kHz band. All passages satisfying the criteria were then manually reviewed, and all were found to contain sonar (Precision of 1.0). The recall of this method was not evaluated.

## Results

Figure 2 shows a 30-minute time series and spectrogram during the passage of a vessel with the ultrasonic antifouling system turned on. The AIS data indicated that the vessel was traveling at 14 knots. The primary frequency of the source at 23 kHz was visible about 6 km before CPA and 4 km after CPA, with the first harmonic at 46 kHz also visible near the CPA time. Figure 3 is a 1-second zoom in on the data near CPA, which shows that there is no distinctive structure to the sound; aural listening to the data confirms that it is a slightly raspy tone (when slowed down by a factor of 10 to be audible). The spectrograms of Coastal Acoustic Buoy data from Burrard Inlet shown in Fig. 4 are from the same date, likely the same vessel shown in Figs. 2 and 3. These data were sampled at 250 kHz, and four harmonics are clearly visible. The received SPL from the ultrasonic antifouling system at the Coastal Acoustic Buoy was 156 dB re 1 $\mu Pa^2 \pm 2$ dB throughout the 4-hour measurement. Figure 5 shows the source levels measured for all 61 of the detections of the ultrasonic source.

## Comparison of Sound Levels to Effects Thresholds

To examine the possibility of marine mammals experiencing hearing threshold shifts or disturbance from the passage of an ultrasonic antifouling system, the hearing

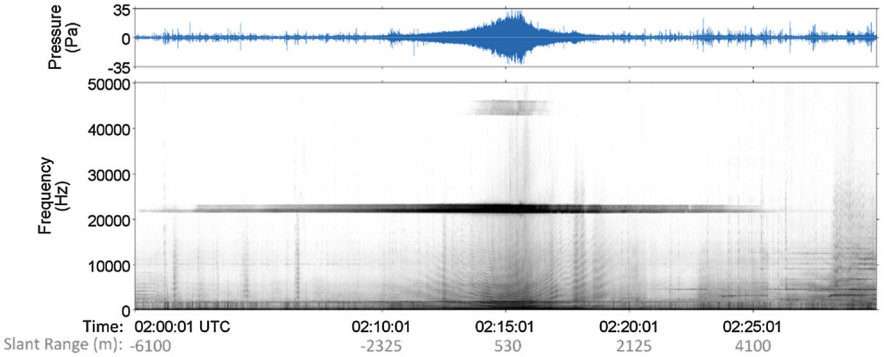



**Fig. 2** Thirty-minute time-series and spectrogram during the passage of a vessel with the ultrasonic system active at the Boundary Pass recorder at a speed of 14 knots at a minimum radial distance of 331 m. The system was detectable for 25 min, 15 min before the vessel closest point of approach (CPA) and 10 min after (4–6 km). These data were high-pass filtered at 100 Hz to remove noise from the time series and enhance the presentation of the vessel passage. Fast-Fourier transform (FFT) duration: 0.25 s, advance 0.125 s, Hann window. The vessel was at a slant range of 331 m at CPA

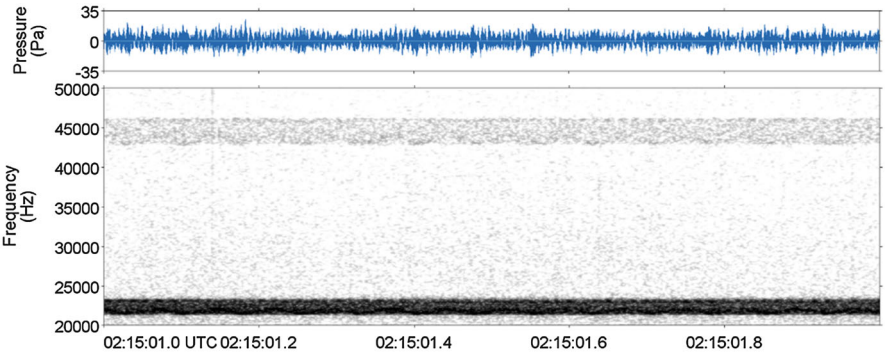



**Fig. 3** One second of data near the closes point of approach (CPA) shown in Fig. 2. Fast-Fourier transform (FFT) duration: 0.01 s, advance 0.001 s, Hann window

weighted SPL and sound exposure levels are shown in Fig. 6 for the same vessel when it passed within 114 m horizontal distance of the ULS (227 m slant range, this passage occurred on a different day than the data in Fig. 2). If we assume a stationary marine mammal, none of the marine mammal groups would have experienced temporary or permanent threshold shift at the location of the ULS sensors (see Southall et al. 2019 for tables of the SEL where temporary or permanent threshold shift begin to occur). It has been shown recently that harbor porpoises have a fleeing reaction when auditory frequency weighted sound pressure levels are 103 dB re 1 μPa$^2$ or higher (DEA 2022). From the current measurements at the ULS, porpoises will flee from vessels with this source at distance less than or equal to 1500 m.

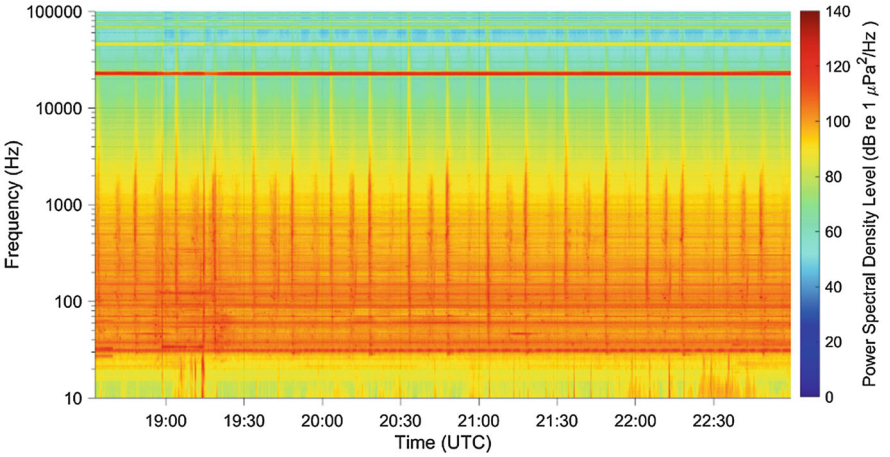



**Fig. 4** Spectrogram from the Coastal Acoustic Buoy measurements at Burrard Inlet measured 73 m from the same vessel as in Figs. 2 and 3. The increases power spectral density levels above 100 Hz occurring approximately every 15 min are from the departure or arrival of Seabuses that passed within 150 m of the buoy

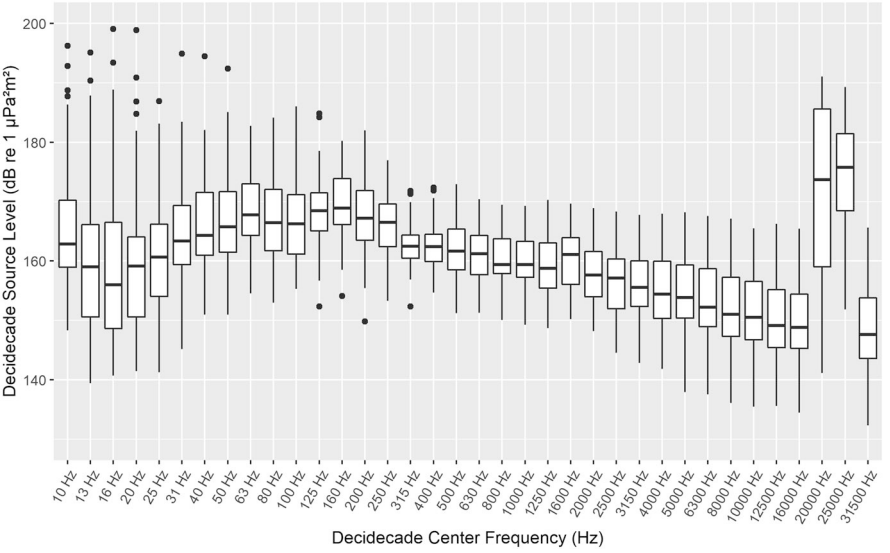



**Fig. 5** Box-and-whisker plot of source levels by decidecade band of the vessels with the ultrasonic antifouling system from all 61 measurements from the ULS systems (September 2015 to October 2019). For each decidecade, the dark horizontal lines are the median values; the top and bottom of the boxes shown the interquartile range; the lines shown the values above and below the quartiles, with dots to indicate outliers. The ultrasonic antifouling systems are in the 20 and 25 kHz decidecade bands

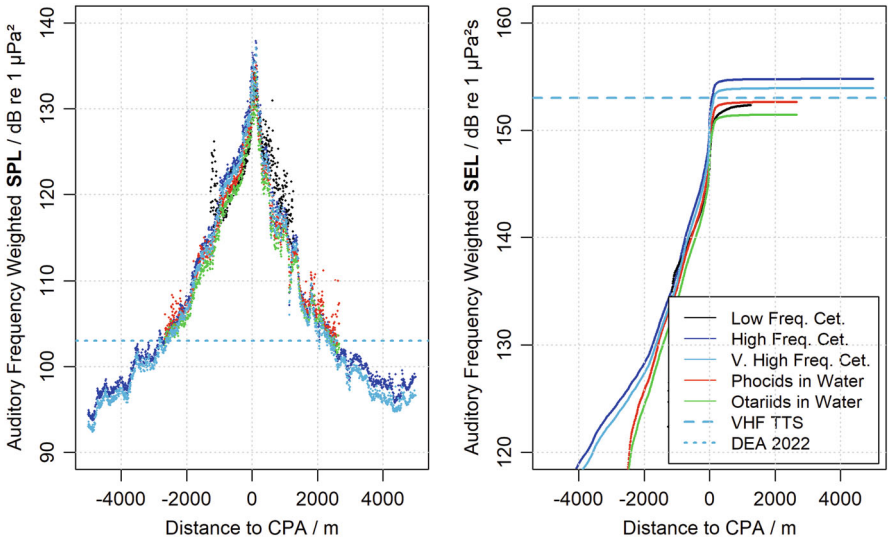



**Fig. 6** Sound levels measured during the passage of the same vessel as the previous figures at a radial closest point of approach (CPA) distance of 114 m. The slant distance between the hydrophone and vessel at CPA was 227 m. (Left) Auditory frequency weighted sound pressure levels (SPL) and (right) cumulative auditory frequency weighted sound exposure levels (SEL). The auditory frequency weightings are defined in Southall et al. (2019); the acronyms are: *LF* Low-Frequency cetaceans (baleen whales), *HF* High-Frequency cetaceans (most dolphins), *VHF* Very High-Frequency cetaceans (porpoises), *PW* Phocid seals in Water, and *OW* Otariid seals in Water

A more general estimate of effects may be found by using the source levels shown in Fig. 5 to simulate the passage of a vessel at different closest point of approach distances and at different speeds (assuming the ultrasonic source level is not correlated with vessel speed). The results shown in Fig. 7 show that at slower speeds there is a greater accumulation of SEL and that the TTS threshold is exceeded at longer distances, up to 450 m for a vessel at 4 m/s, but only 225 m at 10 m/s. The distance to the behavioral disturbance threshold is ~3 km. Note that this analysis assumes the ultrasonic source is a point source since the spatial distribution of sensors along the hull is uncertain. While this is undoubtedly a conservative assumption, the TTS estimates beyond ~100 m range are nonetheless expected to be reasonable given that the results are consistent with the actual measurements shown in Fig. 6.

Due to the measurement occurring in deep water (197 m), the received levels at the ULS were substantially lower than those measured at Burrard Inlet in 20 m water depth. The levels measured in shallow water were ~156 dB re 1 µPa$^2$ (see Fig. 8). The measurements indicate that very high-frequency cetaceans (porpoise) could experience temporary threshold shifts 73 m from the berthed vessel in 1–2 s and permanent hearing threshold shifts in ~200 s. High-frequency cetaceans, such as Southern Resident killer whales, would experience TTS in ~100 s and PTS in 10,000 s (approximately 3 h).

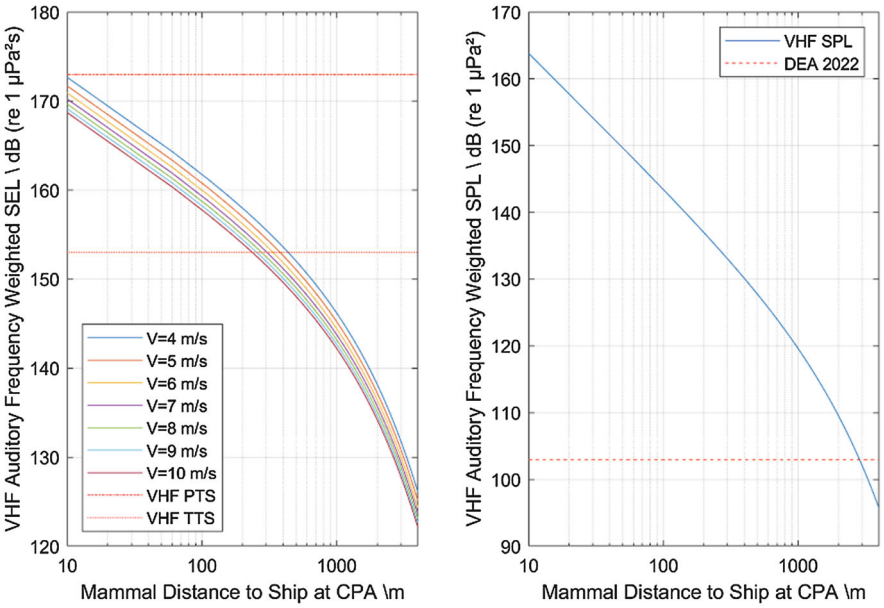



**Fig. 7** Estimates of the (left) very high-frequency (VHF) auditory frequency weighted SEL and (right) SPL using the mean source level from Fig. 5 (182.9 dB re 1 µPa$^2$ in the 20 kHz decidecade band and 180.4 9 dB re 1 µPa$^2$ in the 20 kHz decidecade band, respectively). There is greater accumulation of sound at slower vessels speeds than at higher ones. The Southall et al. (2019) thresholds for PTS and TTS from nonimpulsive sound sources are shown on the left, and the DEA (2022) behavioral disturbance threshold of 103 dB re 1 µPa$^2$ is included on the right

## Masking Effects

It is also possible to estimate auditory masking from the source using the concept of available echolocation range (AER). AER is similar to the concept of listening range reduction (LRR) (Pine et al. 2020), except that the reference value is inverted such that 100% AER means the full listening range is available, which is equivalent to an LRR of 0%. AER is a measure of the percent of the distance from a mammal that is obscured by a sound source compared to what they would normally be able to hear. AER is defined in Eq. 3, where $NL_2$ is the SPL when the masking noise present, MR allows for $NL_2$ to be reduced by a masking release factor, $NL_1$ is the SPL without the masking present, and $N$ is the geometric spreading coefficient of acoustic propagation in the environment.

$$\mathrm{AER} = 100\% \times \left(10^{2\frac{\mathrm{NL}_2 - \mathrm{MR} - \mathrm{NL}_1}{N\ \mathrm{dB}}}\right) \tag{3}$$

The factor of 2 is required since the bidirectional available echolocation range was computed.

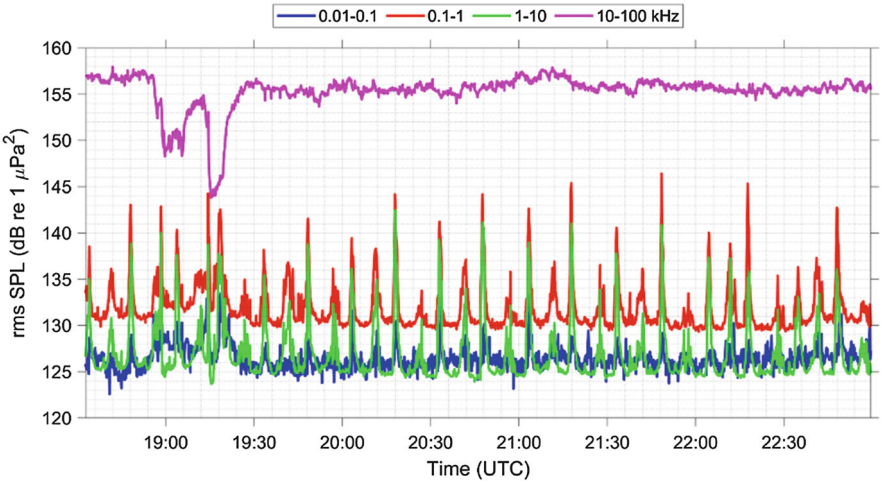



**Fig. 8** Band sound pressure levels (SPL) measured by the Coastal Acoustic Buoy at Burrard Inlet measured 73 m from the same vessel as in Figs. 2, 3, and 4. The reduction in SPL around 19:00 UTC is believed to have been caused by shadowing from passing vessels and/or maintenance activities. The increases in SPL occurring approximately every 15 min are from the departure or arrival of Seabuses that passed within 150 m of the buoy

The masking was analyzed for each decidecade band assuming that this is approximately equivalent to the critical bandwidth of mammalian hearing. The reference noise level, $NL_1$, is taken to be the tenth percentile of the sound pressure level in each decidecade band over a long period (days to months) that includes the period when the masking noise occurred. In cases where $NL_1$ is below the hearing threshold of an animal, the value is replaced by the hearing threshold. The value of $N$ should be representative of the geometric spreading losses in the environment at the frequencies of interest. A value of 20 was employed here. *MR* values of 0 and 10 dB were employed to represent the spatial release from masking that odontocetes can obtain through their directional hearing (Erbe et al. 2016). The value of 0 dB represents zero masking relief if the animal is echolocating toward the source, and 10 dB represents a general case when echolocating away from the source at 20 kHz (Erbe et al. 2016; Accomando et al. 2020).

For high-frequency mammals, such as the SRKW found on the approach to the Port of Vancouver, the peak frequencies of the ultrasonic antifouling source overlap with the peak frequencies of the SRKW echolocation clicks. Figure 7 shows the available listening range for high-frequency cetaceans for the vessel passage shown in Fig. 2. Without spatial release from masking, the AER is near zero when the source is within 4–5 km of the animal. When the animal is able to employ spatial release from masking, the available listening range is reduced to near 0 when the source is 2–3 km from the animal (Fig. 9).

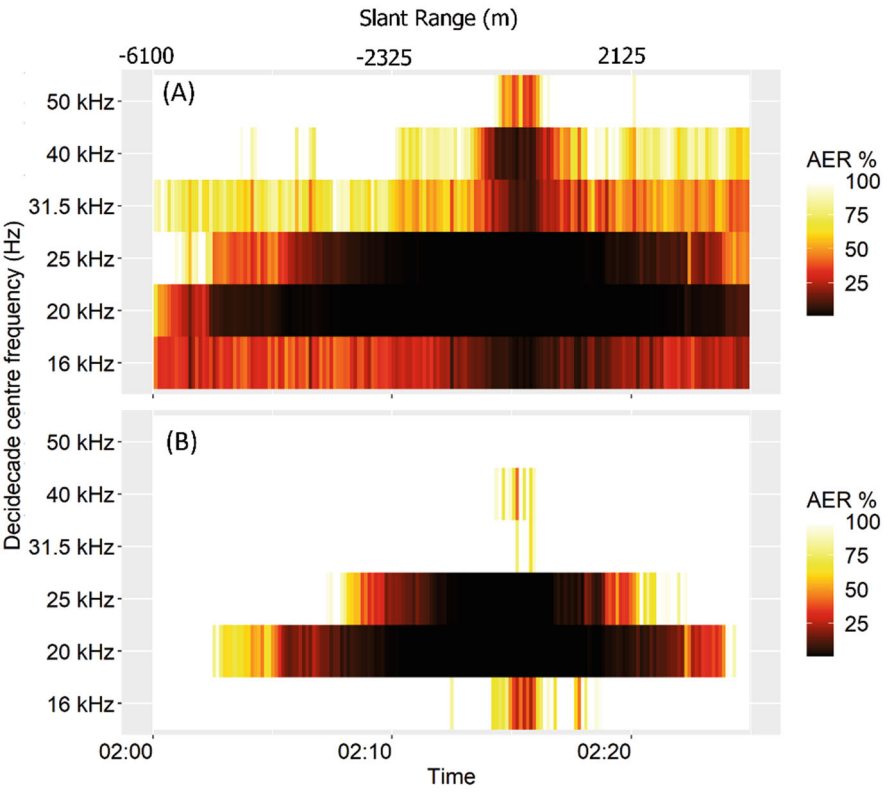



**Fig. 9** Available Echolocation Range (AER) assessment of the passage of the ultrasonic antifouling system for the decidecade bands of 16–50 kHz and the hearing sensitivity of high-frequency cetaceans. Here, a value of 100% is good and 0% indicates completely masked communications. (**a**) The case where an animal is echolocating toward the source, and (**b**) the case of 10 dB spatial release from masking

## Discussion

This chapter reports on two measurement programs that independently identified a novel ultrasonic source present in the areas around the Port of Vancouver. This source was identified as an ultrasonic antifouling system that prevents the build-up of algae, macroalgae, and sessile crustaceans on ships hulls, in pipes, in ballast tanks, etc. These systems work by killing the lower trophic level algae and crustaceans within a short distance (10 m) of the source (Lee et al. 2001; Zhang et al. 2006).

The source was detectable at distances of 4–6 km from the vessels. The analysis suggests that in deep water (200 m) the source will cause avoidance by very high-frequency mammals such as porpoises at a minimum distance of ~1.5 km, perhaps out to 3 km. These mammals could suffer hearing injury at very short distances in

deep water. Measurements were also made in shallow water (20 m) that trapped the sound and minimized attenuation. In these conditions, the measurements suggest permanent hearing injury would occur for very high-frequency cetaceans in less than 2 min if they were closer than 70 m to the source. If the source were turned on while a porpoise was within 25 m of the source, the permanent threshold shift would be even more rapid.

The results shown here are for the systems measured near the Port of Vancouver. The makes, models, number of transducers and electrical drive levels of the systems are unknown. Most of the systems recorded had the same characteristics as were shown here; however, one of them had the frequency-hopping behavior that was also detected in Trickey et al. (2022) To fully understand the risks to marine life posed by the ultrasonic systems, a comprehensive study of the levels as function of the system types, vessel types, and system configurations is required.

The ultrasonic antifouling sources were introduced as a means of keeping vessel hulls clean and preventing the transfer of invasive species. Biofouling of ships hulls increases drag on the vessel and can lead to an increase in fuel consumption (and greenhouse gas production) of up to 55% (Munk et al. 2009; GloFouling Partnerships 2021). Biofouled vessels (and ballast water) are often responsible for transporting invasive species, which provides an additional strong incentive for reducing biofouling. An alternate means of minimizing biofouling is the use of specialized paints that contain copper and zinc or other biocides. The longest acting versions of these paints are self-polishing, which means the flow of water over the hull releases the active biocides for 4–5 years, making the immediate region around the ship's hull toxic to algae and larval crustaceans. The most toxic compounds, such as tributyltin were banned decades ago, which has reduced their accumulation in higher order marine life such as porpoises (Law et al. 2012). There are other coatings that can also be applied that reduce the ability of organisms to adhere to a ship's hull; however, they are significantly more expensive than the biocide paints and are not in wide use. Similarly, a maintenance routine of proactive hull cleaning can help reduce biofouling but at the cost in labour and idle vessel time to perform the cleaning (Swain et al. 2022). A trade-off analysis, bringing in experts from different disciplines, is likely needed to determine which set of impacts is most acceptable: higher costs, acoustic impacts, toxic chemical pollutants, or green-house gas emissions and invasive species.

Following the presentation of the ultrasonic hull cleaner's sound levels (this chapter) and effects on Cuvier's beaked whales (Trickey et al. 2022), many researchers approached the authors to state that they had also encountered this source. In one instance, a researcher reported that they were no longer able to conduct tests near their university because the harbor was so affected by the ultrasonic source. Many others reported that the sources were available for sale at numerous booths at small craft boat shows and vendors had no understanding of how the systems worked or their possible side effects on marine life that is not attempting to attach to a boat's hull. The authors also became aware that the International Maritime Organization may soon endorse ultrasonic hull cleaning as a preferred "green" technology.

It appears there will be a rapid increase in the use of ultrasonic hull cleaners before the scientific community fully understands the hull cleaner's effects on marine mammals. Given the capacity for the ultrasonic systems to impact marine mammals at long distances (kilometers), this suggests that these systems may have greater impacts on marine mammals than the antifouling paints. It is recommended that their use in areas with sensitive or endangered species of marine mammals be restricted. Further study of the risk to marine life as a function of ultrasonic system configurations will be needed to inform the discussions between system developers, users, and international regulators.

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# Modelling Pile-Driving Sound and Mitigation in Realistic Environments

19

Ozkan Sertlek, Yaxi Peng, and Apostolos Tsouvalas

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**Abstract**

Impact pile driving is a transient anthropogenic underwater sound source that can potentially affect marine life. Mathematical modelling tools are essential for predicting sound levels before installing new offshore wind farms. Different modelling approaches are required for modelling the sound generation in proximity to the pile, the mitigation of the noise with the use of air-bubble curtains, and the sound propagation at a larger distance. In addition, the interface and coupling between the different modelling approaches should be carefully considered without losing important details. In this work, a multi-model approach for estimating pile-driving sound in a realistic environment is described. The short-range predictions (up to 750 m) provide detailed spectral and temporal output in various metrics in the water (acoustic pressure, particle velocity) and the seabed (stress and displacement vectors). For the long-range predictions beyond 750 m, only the acoustic pressure metric is calculated, including the range-dependent

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properties of the acoustic environment. Based on the combination of short- and long-range models, sound maps can be created to identify the contribution of the pile driving to the underwater soundscape.

**Keywords**

Sound modelling · Impact · Propagation · Pile driving · Noise mitigation

## Introduction

Anthropogenic offshore activities have raised serious concerns about their impact on marine life in different ways depending on the spatial, temporal, and spectral pattern of sound. One of the main impulsive underwater sound sources of concern is offshore pile driving due to the increasing numbers of offshore wind farms. Furthermore, in the next decade, it is expected that the number of offshore wind farms under construction will accelerate in European countries to provide alternative energy sources. To preserve the marine ecosystem and maintain sustainable development, many countries define noise thresholds at a specific distance in various metrics. Therefore, an environmental impact assessment that includes the prediction of noise levels is required prior to the construction of the installation of piles in most projects. Given the expected noise levels, appropriate noise mitigation systems (NMS), i.e., air-bubble curtains, hydro-sound dampers, or noise mitigation screens, are usually required to reduce the noise. To address this challenge, underwater sound modelling tools can assist in predicting potential sound sources and propagation characteristics.

There is no standard modelling approach for the pile-driving sound. Various computational methods (such as finite element, finite difference, semi-analytic and empirical models) have been used to predict the generated noise during pile driving and propagate the sound field to larger distances where the noise thresholds are defined (Tsouvalas 2020). These models should include the properties of the pile (i.e., length, thickness, hammer properties), the installation tool, and the site-specific environmental parameters (i.e., sediment type, sediment layers, water depth). Although finite element models can provide detailed results, they are computationally slow. In contrast, the fast empirical models are less detailed, especially when noise mitigation systems (NMS) are deployed. Air-bubble curtain systems have been widely used in many offshore projects to mitigate the impact pile-driving noise over the last decade. An accurate description of the acoustic characteristics of the air-water mixture is essential before modelling the noise mitigation mechanism in such cases.

In our work, a computationally efficient and validated pile-driving model is used to predict the generation and propagation of the sound field associated with impact piling at large (from the pile) distances, overcoming the limitations of earlier models (Tsouvalas 2020). In addition, the same model can be easily modified for various system configurations to investigate the water- and soil-borne noise transmission paths in a detailed and systematic way (Peng et al. 2021a, b). Furthermore, the sound

source modelling tool can be combined with different sound propagation modelling approaches to include long-range propagation effects such as the variable bathymetry (Sertlek et al. 2019). The source and propagation modelling approaches are described in the following sections.

## Sound Source Modelling in Impact Pile Driving

During pile driving, the sound propagates through the pile, water, and sediment layers after the source excitation by a hammer and interacts among these different domains (Tsouvalas 2020). In addition, low-frequency sound propagation in shallow waters can propagate long distances in vertical and horizontal directions. Therefore, the modelling approaches should consider the detailed composition of the sediment to capture the sound waves propagating through the multiple sediment layers. To achieve this aim, different modelling approaches have been used to model pile-driving sound such as:

- Finite element models (Reinhall and Dahl 2011; Zampolli et al. 2013)
- Finite difference schemes (MacGillivray 2013, 2015)
- Boundary element methods (Masoumi et al. 2007, 2009; Masoumi and Degrande 2008)
- Semi-analytical solutions by several authors (Tsouvalas and Metrikine 2014; Hall 2013, 2014, 2015; Deng et al. 2016a, b)
- Simplified formulas as damped cylindrical spreading (Duncan and Parsons 2011; Ainslie et al. 2014; Lippert et al. 2018a, b)

Some of these modelling approaches are compared during the COMPILE I Workshop (Lippert et al. 2016) to indicate their differences. In addition to the differences in the modelling of the sound generation, the assumptions related to the sediment play an essential role in the accuracy of model predictions. Some models describe the sediment as an equivalent fluid layer (Dahl and Reinhall 2011; Zampolli et al. 2013). Considering the elastic (Buckingham 2000) or poroelastic (Biot 1956a, b) properties of sediment could improve model accuracy (Tsouvalas 2020).

In this chapter, a semi-analytical solution that could provide a faster solution than the finite element method is used (Tsouvalas et al. 2015). This model assumes cylindrical symmetry for the pile and the surrounding environment. The pile is described as a cylindrical shell in an inviscid compressible fluid medium and horizontally stratified acousto-elastic half-space. The pile properties such as the pile length (L), radius (R), thickness (t), Poisson's ratio (v), and density of the shell ($\rho$) are used as the model inputs. A time-dependent force function is applied to the pile head in the air. The mathematical background is based on the normal modes and branch line integrations for an axisymmetric ring source. The set of equations are implemented in the software package, SILENCE, providing various model options and outputs depending on the requirements of the specific problems. The near-source module calculates the wave field in the close to the monopile by

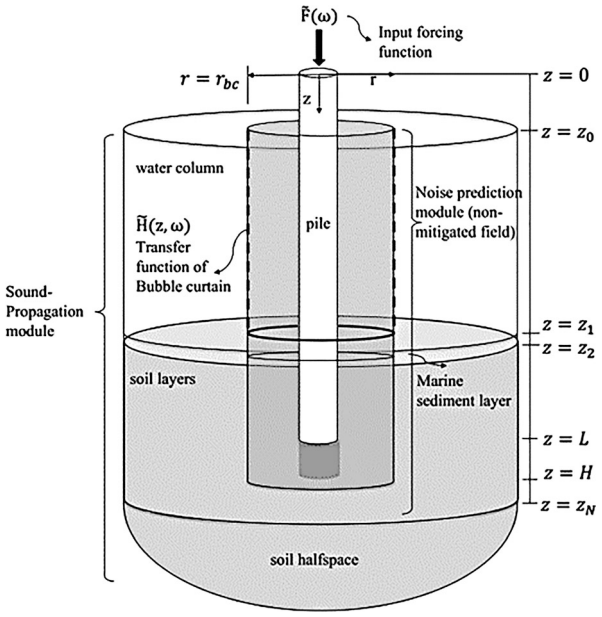



**Fig. 1** The geometry of pile driving sound source model (SILENCE). F(t) is the time-dependent force function due to impact or vibratory hammer. F(w) is the Fourier transform of F(t) (This figure adapted from Peng et al. (2021a))

considering the interactions between the pile, fluid, and soil. Once the wave field at the close range ($r \leq r_0$) is calculated, the far-field module is used to propagate this field to the large distances ($r > r_0$). Since the detailed mathematical description of this model is given by Peng et al. (2021b), it is omitted for the sake of simplicity in this chapter. A basic version of pile-driving sound modelling software (SILENCE) is freely available on the TU Delft Webpage (http://ua.citg.tudelft.nl/) for research and educational purposes (Fig. 1).

In Fig. 2, the amplitude of the particle velocity in the water and sediment layers are shown for a pile with a diameter of 7 m and a length of 78 m (Tsouvalas 2020). Scholte waves propagating along the fluid-solid interface could be noticed in this figure. The vibratory piling can also be modelled with a similar modelling approach based on a vibratory hammer force function. However, since the acoustic energy enters the water column from all vertical angles due to the different temporal structure of the excitation, the coherent Mach cones would not be clearly visible for the vibratory piling.

The calculated sound field close to the pile is propagated over long radial distances with the far-from-source module based on the integral boundary equations considering the reciprocity theorem. The detailed derivation of the far-from-source module is described by Peng et al. (2021b). This model can provide detailed outputs including sound pressure, sound velocity, displacement, and stress in the sediment layers. The modelling of the particle motion in the water and top sediment layer is critical to assess the impact of the pile-driving sounds on fish, which are sensitive to both the pressure and particle motion.

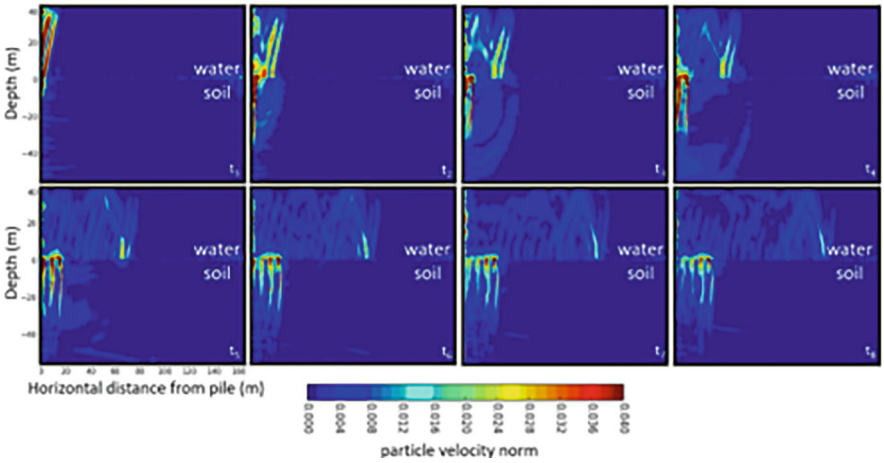



**Fig. 2** The amplitude of the particle velocity after the hammer impact (6 s to 120 s). (This figure is adapted from Tsouvalas (2020))

Once the generated noise during pile driving is modelled, the performance of the noise mitigation systems could be simulated to discover their optimum configuration. Describing the acoustic properties of an air-bubble curtain with effective parameters such as effective wavenumber or sound speed simplifies the problem (Bohne et al. 2019). In our approach, the effective wavenumber formula of Commander and Prosperetti (1989) is applied based on the parameters of the air-bubble curtain (Bohne et al 2020) to estimate the variation of effective local wavenumbers over the entire water depth and frequency band. The bimodal bubble size distribution is introduced by Lethr et al. (2002) with observations of small and large bubbles (Bohne 2020; Milgram 1983). An example of effective sound speed profile evaluation with depth is shown in Fig. 3 for 125 Hz. It can be noticed that the sound speed in the air-bubble curtain is lower than the sound speed in the water. As a result of this difference, a large impedance mismatch at the water and air-bubble interface causes the reflection of the sound wave field back to the source and attenuation as they propagate through the bubble curtain. These transmission coefficients between water and bubble interfaces are calculated and coupled to the free-field noise prediction model through a boundary integral method. The water column and air-bubble curtain are divided into multiple regions along the depth coordinates. At each vertical domain, a one-dimensional problem is considered with an input incident wave and properties of the layers to determine transmission coefficients for depth direction. Once the solution is obtained for the depth-dependent fluid dynamic properties, the local effective wavenumber and the transmission coefficients of the air-bubble curtain are determined. The obtained results with the hydrodynamic model are processed, including the propagation modelling through air-bubble curtains, and the result is translated to the relevant acoustic metrics.

The modelling framework of the noise mitigation with air-bubble curtains should consider a detailed configuration of the mitigation system including the pneumatic

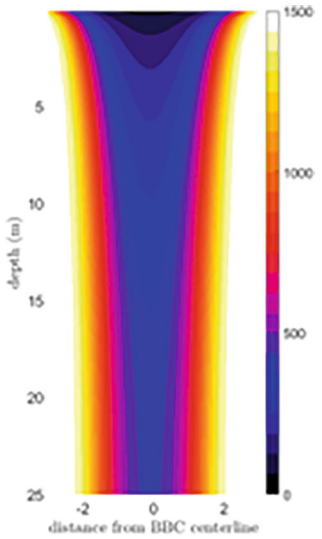



**Fig. 3** The effective sound speed. This result is calculated at 125 Hz for a water depth of 25 m

model for the simulation of the air circulation and transport from the air-supplied vessel to the hose on the seabed. A hydrodynamic model calculates the effective wavenumbers and sound speed through depth and range in the air-bubble curtain. Next, the output of the hydrodynamic model is used to calculate the depth- and frequency-dependent transmission coefficients of the air-bubble curtain. Finally, the mitigated sound levels are calculated as a combination of individual simulation steps and propagated to larger distances by an acoustic modelling approach. More detailed description of the sound mitigation model is given by Peng et al. (2023).

As a summary, the mathematical models require inputs for the pile parameters, environment, and air-bubble curtain configuration. The pile parameters include the length, diameter, thickness, penetration depth, density, Young's modulus, Poisson's ratio, and damping of the pile. The big bubble curtain (BBC) parameters include the distance from the pile, nozzle diameter, nozzle spacing, and airflow rate through the nozzle. The BBC parameters should be calculated by the pneumatic and hydrodynamic models with the feed pressure, hose length and diameter, nozzle diameter, and spacing.

## Sound Propagation in Range-Dependent Media

Long-range sound propagation requires the detailed modelling of the sediment properties, variations in bathymetry, and sound speed. Various modelling approaches (i.e., normal mode method, parabolic equation, ray tracing, energy flux) could be preferred depending on the environmental conditions and frequency range. Sound propagation modelling is usually required in shallow waters for impact pile driving.

Many EU countries (such as the Netherlands and Germany) define the noise criteria at 750 m (Heinis 2015; Heinis et al. 2019a, b; BSH 2011; 2013). Therefore, many modelling approaches ignore the range-dependent properties of waveguides due to the minor environmental changes up to 750 m. Instead, these models focus on the detailed modelling of elastic seabed properties that improve the accuracy of model predictions for pile driving due to the strong interactions between the water and sediment layer, especially at low frequencies. However, when sound propagation modelling is required at long ranges beyond 750 m, omitting the bathymetry changes could result in incorrect results. In this chapter, different modelling strategies are used in order to benefit from the advantages of different models. Up to 750 m, the sound pressure and particle motion are calculated, including the detailed sediment properties based on the range-independent waveguide assumption. Modelling sound propagation, including the detailed elastic properties of sediments, makes it possible to capture the sound energy in the sediment layer, which can tunnel under the bubble curtains and leak back to the water layer. This effect is usually critical for ranges up to 200 m where bubble curtains are typically deployed. Beyond 750 m, a sound propagation model of lower fidelity is used. Different propagation models could be chosen based on environmental conditions and frequency ranges, as shown in Fig. 4.

The parabolic equation model can use a starter field from sound source modelling tools and propagate this to larger distances, including the range-dependent variation in the water depth. Similarly, normal mode models, such as KrakenC (Porter 1991), could be used, including the elastic properties of the sediments and adiabatic approximation for the range dependency. If the elastic properties and layered sediments are ignored at high frequencies, mode-flux theory based on adiabatic approximation is an option. The wavenumber integration method can provide accurate results when the range dependency is weak. Comparisons between different propagation methods are investigated by Sertlek et al. (2019). At the interface of two different models, various approximations could be used (Reinhall and Dahl 2011:

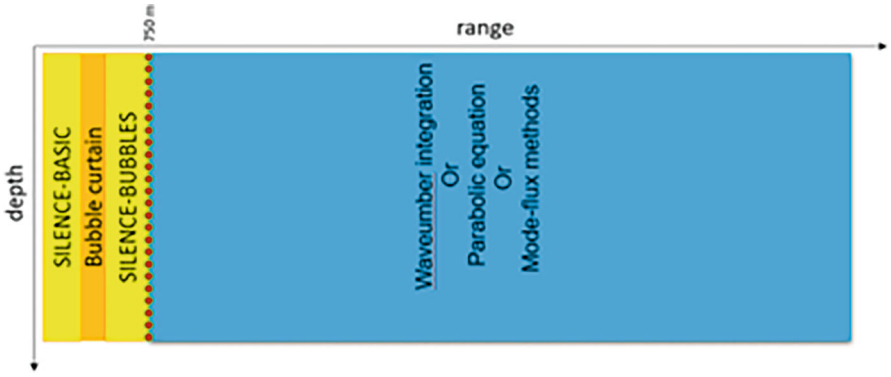



**Fig. 4** Multi-model approach for the simulation of pile-driving noise. The detailed source and propagation model close to the pile (up to 750 m) with the sound pressure, velocity, displacement, and stress outputs. Beyond 750 m, propagation models only with sound pressure outputs are used

TNO report). The mathematical descriptions of these potential approaches are beyond the scope of this chapter. Range-dependent propagation modelling can be used for creating sound maps to provide insight into the sound exposure level (SEL) during the off-shore wind farm constructions. Sound maps can be calculated based on a large number of radial slides between the source location and receiver point as shown in Fig. 5.

As could be noticed from the bathymetry slices, the range-dependent differences are usually minor at 750 m where the range-independent modelling approach is used.

Figure 6 shows a sound map created based on the described steps for a single impact piling strike at 125 Hz. First, the pressure field is modelled with SILENCE up to 750 m. An artificial scenario was created to test the proposed approach in the Baltic Sea. The offshore wind farm location is chosen based on the possible wind farm locations as shown in the EMODNET Human Activities data portal (www.emodnet-humanactivities.eu). The pile parameters are also chosen based on the realistic pile properties from Peng et al. (2021a). The length, diameter, and wall thickness of the pile are chosen as 76.9 m, 8 m, and 90 mm, respectively. The water depth at the piling location is 39.9 m. The maximum blow energy is 1750 kJ. The pressure field at 750 m is calculated for multiple receiver depth locations with 0.1 m resolution. Beyond 750 m, assuming a vertical line source, the sound propagation is calculated using the normal mode method with adiabatic approximation. The sound pressure is averaged over the depth.

Sound mapping helps to predict the maximum impact distances based on the impact thresholds of the different marine animals (Southall et al. 2019). Furthermore, the sound maps could be weighted based on the marine animals' hearing sensitivities and swimming depths over a large frequency band to provide input for the biologists.

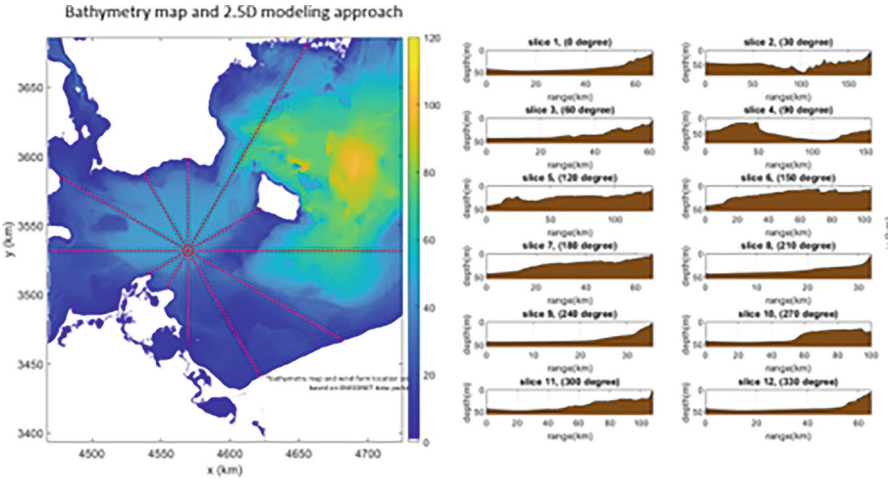



**Fig. 5** The radial slicing of bathymetry as input of sound mapping (on the left panel). Selected water depth profiles from the same area (on the right panel)

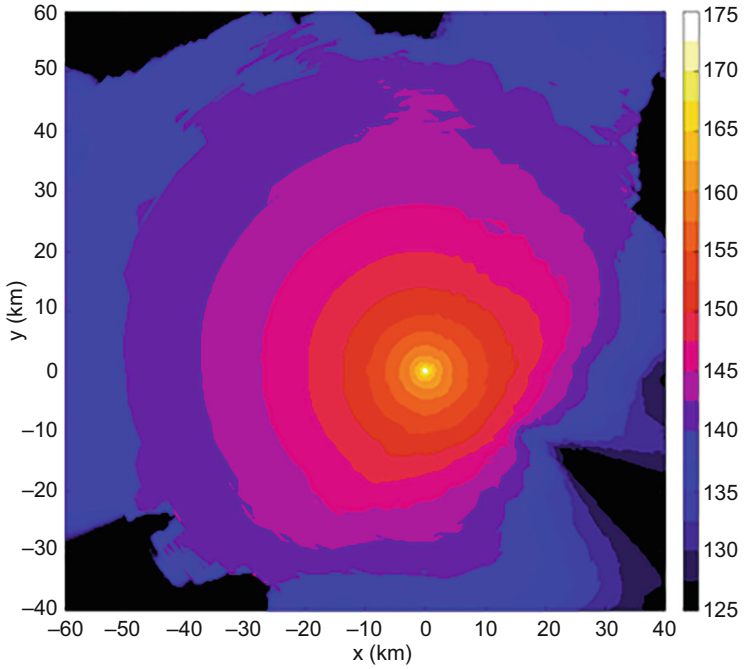



**Fig. 6** Sound mapping based on the multi-model approach for the simulation pile-driving noise. The sound pressure is averaged over the depth. SEL is calculated for a single strike at 125 Hz

## Conclusions

In this chapter, a framework for modelling underwater sound properties due to pile driving is described by considering the detailed environmental inputs, such as the elastic multilayered sediments and range-dependent water depth. As a general summary, the following procedure is followed to predict sound fields during offshore wind farm constructions:

- Modelling underwater sound during an impact pile driving with SILENCE
- Modelling the noise mitigation with a bubble curtain
- Modelling sound propagation up to 750 m with a high-fidelity sound propagation model considering elastic properties (including the particle motion and pressure)
- Modelling far-field propagation with a fast computational model beyond 750 m
- Sound mapping as combination modelling results for the individual radial slices

The proposed approach suggests using a set of various mathematical models at distances that can be accurately and practically implemented. The uncertainty in the environmental and pile parameters could be investigated individually for each region (source, near-field, and far-field). The pile-driving source model can generate outputs

for the sound pressure, velocity, stress, and displacement in the sediment layers. These outputs could be critical for assessing the impact of sound on the fish and invertebrates. Due to the limited availability of the required inputs for the sediment properties and computational expenses, the particle motion, stress, and displacement outputs are only calculated up to 750 m in this work. However, more detailed propagation modelling tools at long distances could be used in future works. Furthermore, the sound propagation modelling approach for the air-bubble curtain could be improved considering the detailed backscattering effects and continuity conditions for the acousto-elastic waveguides.

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# Section III

# Soundscapes

# Ocean Sound: More than Just Amplitude

20

Jennifer L. Miksis-Olds, Dylan C. Wilford, and S. Bruce Martin

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**Abstract**

The Soundscape Code was originally proposed in 2021 to rapidly assess and direct multidimensional comparisons of salient soundscape properties across time and space. The Soundscape Code metrics represent the amplitude, impulsiveness, uniformity, and periodicity properties of an acoustic environment. Here they were applied to recordings from four deep ocean environments along the US Outer Continental Shelf and the shallow water of the Great Barrier Reef. The assessment and comparison of soundscapes recorded in deep sea sand versus deep coral locations and deep versus shallow water coral sites showed the habitat soundscapes exhibited both similarities and unique acoustic properties which were metric dependent. This initial characterization will aid in directing further analyses and guide subsequent assessments to more comprehensively understand soundscape dynamics and the role the local soundscape plays in deep sea ecosystems and coral reef habitats.

**Keywords**

Soundscape · Coral reef · Deep sea · Kurtosis · Uniformity · Amplitude

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## Introduction

The ocean soundscape is a term used to define and describe the acoustic properties of the marine environment at a specific place and time. The underwater soundscape was formally defined by IOS 18405 as the "characterization of ambient sound in terms of its spatial, temporal, frequency attributes, and the types of sources contributing to the sound field" (ISO 2017). Traditionally, variability in amplitude across time, space, and frequency has been the focus of soundscape studies captured by metrics of sound pressure level, power spectral density, sound exposure level, and source-amplitude-related parameters such as number of sources and source classification. Indices such as the Acoustic Complexity Index and Entropy Index have been applied to passive acoustic recordings as a proxy for biodiversity by relating measures of frequency and amplitude over specified time periods (Staaterman et al. 2014; Sueur et al. 2008). Here, consideration of properties that capture elements of a soundscape in addition to the more commonly used amplitude metrics in alignment with the Soundscape Code (SSC) established in Wilford et al. (2021) are advocated. Metrics that define properties of impulsiveness, uniformity, and periodicity were assessed for their ability to facilitate efficient, quantitative, and comprehensive soundscape comparisons across time and space.

Wilford et al. (2021) examined passive acoustic recordings from seven different ocean environments representing global sound sources indicative of natural (biologic and nonbiologic) and anthropogenic activity in the development of the proposed SSC. The best measures were determined to be root-mean-squared sound pressure level (SPL) and peak sound pressure level (PK) for amplitude, kurtosis for impulsiveness, an autocorrelation-based metric for periodicity, and the Dissimilarity Index (Sueur et al. 2008) for uniformity. In this study, the SSC structure was applied to passive acoustic recordings from four locations along the Atlantic US Outer Continental Shelf (OCS) (two sandy bottom locations and two cold water coral locations) and one shallow, warm water coral habitat of the Great Barrier Reef. Ecosystem health and function is linked to local soundscape characteristics in shallow water coral reef habitats (Lamont et al. 2022; McWilliam et al. 2017), but at present it is not known whether this relationship extends to deep water coral reef habitats. The goal was to assess the performance of the SSC in identifying similar and contrasting soundscape characteristics associated with deep sea sandy bottom habitats, deep sea cold water coral reefs, and a warm water coral reef.

## Methods

Five long-term data sets were analyzed using the metrics of the SSC to establish baseline soundscape information, and compare the soundscapes of deep-sea coral and noncoral environments with the soundscape of a shallow coral environment. Recordings were made at four deep-sea sites along the OCS region off the southeastern United States consisting of deep-sea coral (Savannah Deep, SAV; Richardson Hills, RH) and noncoral, sandy bottom (Wilmington, WIL; Blake Escarpment, BLE)

habitats. Recordings from a shallow (6–18 m) coral reef environment part of the Great Barrier Reef (GBR) were examined in contrast to the deepwater sites of the OCS (Fig. 1). The OCS recordings at SAV, WIL, and BLE were made in 2018 as part of the Atlantic Deepwater Ecosystem Observatory Network (ADEON – https://adeon.unh.edu/). The ADEON recordings were made with AMAR G3 recorders (JASCO Applied Sciences, Dartmouth, NS, Canada) sampled at 375 kHz on a 1/20 min duty cycle. The RH recordings were made as part of the DEEP SEARCH program (https://ecogig.org/deepsearch). The DEEP SEARCH recordings were made in 2019 with an icListen Smart Hydrophone (Ocean Sonics, Truro Heights, NS, Canada) sampled at 128 kHz on a 1/30 min duty cycle. Data from the Great Barrier Reef site were recorded in 2013 on an AMAR G3 recorder at a 64 kHz sampling rate with a 13/14 min duty cycle. Recordings from the OCS sites were appropriately downsampled to an upper 32 kHz usuable frequency, so be consistent

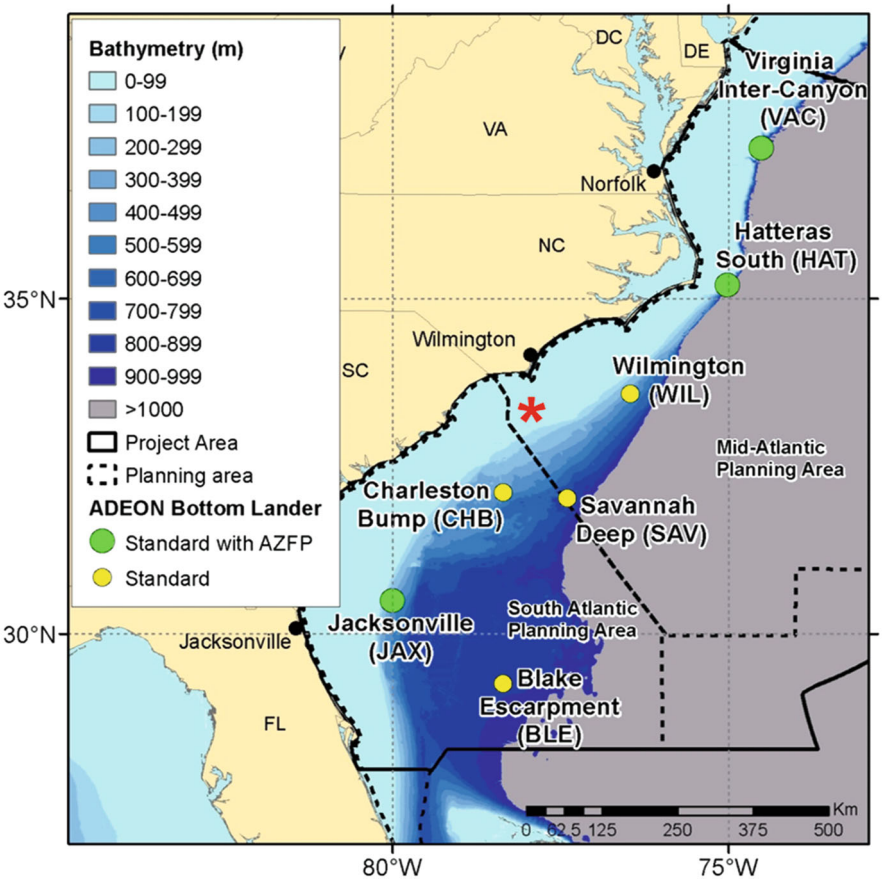



**Fig. 1** Recorder locations and bathymetry for the OCS sites: Wilmington (WIL), Savannah Deep (SAV), Blake Escarpment (BLE), and Richardson Hills (red *)

for comparison with the 32 kHz usable frequency maximum of the GBR recordings. All acoustic data were calibrated using hydrophone sensitivities and identical methodologies so that accurate and meaningful comparisons could be made. A mid-April to mid-May time period was analyzed at the five sites. This analysis period was chosen so that all recordings were related to a similar time period, as recordings from RH and GBR data were the limiting recordings with only April/May data available. Consistency across seasons among the OCS sites was an important factor in the most accurate comparisons. Recognizing the GBR site is in the southern hemisphere and off season compared to the northern hemisphere locations, its proximity to the equator negates the strong influence of seasons observed at the OCS sites which allows for meaningful comparisons given the limiting factor of data availability.

A SSC matrix was computed from the acoustic time series at each of the five sites in four frequency bands: Broadband (BB) (10 Hz- >10 kHz), Low (10–100 Hz), Mid (100 Hz – 1000 Hz), High (1000 Hz – 10 kHz), and Ultra-High (Ult-High) (>10 kHz). The metrics were calculated according to Wilford et al. (2021) for each frequency band. Frequency bands were isolated with Finite Impulse Response (FIR) filters implemented in MATLAB (Mathworks Inc., Natick, MA). The SSCs compare the average level and overall variability of each metric by reporting the median and size of the central 95th percentage (C95) of 1 min metric time series. A color-coding scheme within the SSC matrix reflects the range of metric medians and C95 values across all sites. The overall project range for each metric category was divided into quartiles, and each cell of the Soundscape Code table was colored based on quartile (Q1 blue, Q2 green, Q3 yellow, and Q4 red).

Connections between SSC metric groupings and habitat location/type were explored using k-means clustering computed in JMP Pro™ 14.0.0 (JMP Statistical Discovery LLC, Cary, NC). The number of clusters (k) constraining the analysis relates to the number of sites. For clustering with all the sites, four to six clusters were analyzed, which corresponds to cluster sizes one cluster more ($k = 6$) and one cluster less ($k = 4$) than the five total sites ($k = 5$).

## Results

The SSC output (Fig. 2) provides a high-level, overall comparison of the soundscapes from the five locations. The SSC output provides a large amount of detailed information that can be harvested for detailed comparisons between locations and metrics. Here only a high-level summary of results is captured for comparisons across locations and habitat types (sandy vs live hard-bottom reefs, shallow vs deep reefs).

GBR SPL amplitude medians dominated in the BB, Mid, High, and UH band with PK dominating in all frequency bands. Accompanying the highest median levels at GBR were the lowest variability C95 metrics compared to OCS sites, which report considerable variability in all bands. BLE and WIL had the highest SPL median levels in the Low band. RH had the most variable amplitude metrics.

| BLE | BB | | Low | | Mid | | High | | Ult-High | |
|---|---|---|---|---|---|---|---|---|---|---|
| | med | C95 | med | C95 | med | C95 | med | C95 | med | C95 |
| Amplitude | 103 | 16 | 101 | 18 | 93 | 20 | 94 | 23 | 87 | 12 |
| | 118 | 25 | 114 | 19 | 108 | 26 | 110 | 24 | 105 | 36 |
| Impulsiveness | 3 | 7 | 3 | 1 | 3 | 12 | 3 | 12 | 3 | 1042 |
| Uniformity | 0.015 | 0.041 | 0.022 | 0.029 | 0.015 | 0.038 | 0.012 | 0.019 | 0.012 | 0.025 |
| Periodicity | 0 | 1 | 0 | 1 | 0 | 1 | 0 | 1 | 0 | 1 |

| RH | BB | | Low | | Mid | | High | | Ult-High | |
|---|---|---|---|---|---|---|---|---|---|---|
| | med | C95 | med | C95 | med | C95 | med | C95 | med | C95 |
| Amplitude | 100 | 32 | 94 | 34 | 94 | 26 | 94 | 27 | 87 | 16 |
| | 118 | 38 | 110 | 41 | 111 | 36 | 111 | 36 | 106 | 37 |
| Impulsiveness | 3 | 149 | 3 | 153 | 3 | 188 | 3 | 506 | 3 | 687 |
| Uniformity | 0.015 | 0.082 | 0.025 | 0.091 | 0.015 | 0.095 | 0.011 | 0.034 | 0.011 | 0.032 |
| Periodicity | 0 | 5 | 0 | 5 | 0 | 11 | 0 | 3 | 0 | 1 |

| SAV | BB | | Low | | Mid | | High | | Ult-High | |
|---|---|---|---|---|---|---|---|---|---|---|
| | med | C95 | med | C95 | med | C95 | med | C95 | med | C95 |
| Amplitude | 101 | 33 | 93 | 38 | 94 | 29 | 95 | 26 | 87 | 11 |
| | 117 | 34 | 108 | 41 | 109 | 27 | 110 | 25 | 105 | 25 |
| Impulsiveness | 3 | 7 | 3 | 8 | 3 | 13 | 3 | 11 | 3 | 19 |
| Uniformity | 0.016 | 0.066 | 0.025 | 0.066 | 0.013 | 0.05 | 0.012 | 0.031 | 0.012 | 0.011 |
| Periodicity | 0 | 1 | 0 | 2 | 0 | 1 | 0 | 1 | 0 | 0 |

| GBR | BB | | Low | | Mid | | High | | Ult-High | |
|---|---|---|---|---|---|---|---|---|---|---|
| | med | C95 | med | C95 | med | C95 | med | C95 | med | C95 |
| Amplitude | 116 | 6 | 92 | 15 | 103 | 10 | 113 | 6 | 113 | 7 |
| | 157 | 8 | 114 | 21 | 132 | 16 | 154 | 9 | 153 | 10 |
| Impulsiveness | 391 | 1412 | 11 | 273 | 43 | 459 | 619 | 3154 | 351 | 1028 |
| Uniformity | 0.018 | 0.005 | 0.023 | 0.038 | 0.022 | 0.035 | 0.018 | 0.006 | 0.017 | 0.002 |
| Periodicity | 0 | 1 | 0 | 2 | 0 | 1 | 0 | 2 | 0 | 1 |

| WIL | BB | | Low | | Mid | | High | | Ult-High | |
|---|---|---|---|---|---|---|---|---|---|---|
| | med | C95 | med | C95 | med | C95 | med | C95 | med | C95 |
| Amplitude | 102 | 21 | 97 | 25 | 93 | 25 | 95 | 26 | 88 | 12 |
| | 120 | 26 | 111 | 27 | 109 | 30 | 113 | 27 | 111 | 28 |
| Impulsiveness | 3 | 25 | 3 | 10 | 3 | 134 | 3 | 47 | 3 | 122 |
| Uniformity | 0.016 | 0.047 | 0.031 | 0.05 | 0.014 | 0.057 | 0.012 | 0.03 | 0.012 | 0.012 |
| Periodicity | 0 | 10 | 0 | 10 | 0 | 2 | 0 | 1 | 0 | 1 |

| RANGE | BB | | Low | | Mid | | High | | Ult-High | |
|---|---|---|---|---|---|---|---|---|---|---|
| | med | C95 | med | C95 | med | C95 | med | C95 | med | C95 |
| Amplitude | 100 | 116 | 92 | 101 | 93 | 103 | 94 | 113 | 87 | 113 |
| | 117 | 157 | 108 | 114 | 108 | 132 | 110 | 154 | 105 | 153 |
| Impulsiveness | 3 | 391 | 3 | 11 | 3 | 43 | 3 | 619 | 3 | 351 |
| Uniformity | 0.015 | 0.018 | 0.022 | 0.031 | 0.013 | 0.022 | 0.011 | 0.018 | 0.011 | 0.017 |
| Periodicity | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

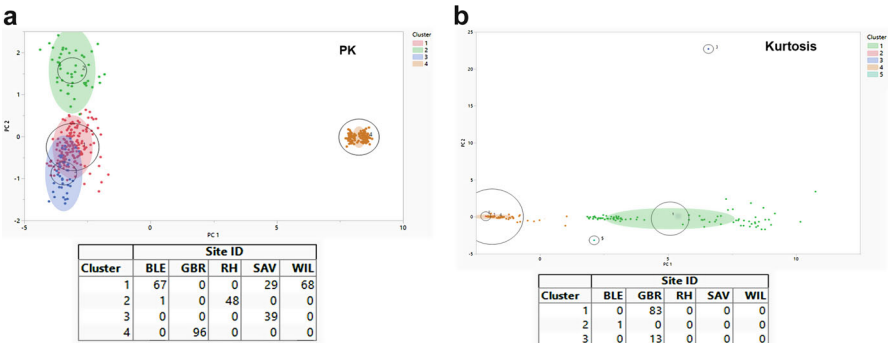
**Fig. 2** Soundscape Code output for the five sites analyzed. The two rows for amplitude reflect the SPL (upper amplitude row) and PK (lower amplitude row) metrics. The ranges reported in the lower right panel indicate the range of SSC medians and C95s across all sites. The total range of the SSC medians and C95s was divided into quartiles, and the cell colors correspond to which quartile the value falls into from low to high: blue, green, yellow, and red

GBR was the most impulsive site in both median and variability parameters. The primary driver for this was related to the presence of snapping shrimp signals visible in the spectrograms. The OCS site impulsiveness (kurtosis) was assessed on the C95 variability as opposed to median levels, as all median kurtosis levels were in the lowest quartile. Among the deep-water OCS sites, RH kurtosis values were the most variable across frequency bands which suggests this site is the most impulsive OCS soundscape. BLE and WIL kurtosis parameters suggest that the soundscapes at these sites contain salient impulsive signals in the Ult-High and Mid frequency bands, respectively. Snapping shrimp signals were not visible in the OCS spectrograms.

GBR reported higher D-index uniformity medians in frequencies above 100 Hz compared to the OCS sites. GBR D-index variability was extremely small compared to OCS sites. For the most part, OCS sites appear acoustically dynamic especially in the Low band. BLE differs in this assessment with the lowest uniformity medians across all frequencies.

While the periodicity medians were all zero, periodic signals appear well represented in all sites, represented by increased variability in isolated frequency bands. WIL and RH were most influenced by periodic signals in the Low frequency band. All sites except SAV showed high levels of periodic variability in the higher frequency bands. The high frequency periodicity was likely related to echolocation signals visible in the spectrograms.

Clustering of SSC metrics revealed trends similar to those observed in the SSC matrix (Figs. 2 and 3). In all SSC metric clusterings, except for periodicity, GBR was clearly separated and identified as different from the OSC sites (Figs. 3 and 4). For some metrics, such as PK, the OCS sites showed distinct clusters for the cold water coral reefs at RH and SAV (Fig. 3). Both the deep water coral sites of RH and SAV clustered without any contributions from the sandy bottom sites of BLE and WIL. The sandy bottom SSC metric clusters from BLE and WIL did have significant overlap with the deep cold water coral reef at SAV, but no overlap with the cold water reef at RH. Clustering for amplitude metric SPL and uniformity metric D-Index (not shown) had similar clustering results with GBR clustering very separately from the




| | Site ID | | | | |
|---|---|---|---|---|---|
| Cluster | BLE | GBR | RH | SAV | WIL |
| 1 | 67 | 0 | 0 | 29 | 68 |
| 2 | 1 | 0 | 48 | 0 | 0 |
| 3 | 0 | 0 | 0 | 39 | 0 |
| 4 | 0 | 96 | 0 | 0 | 0 |

| | Site ID | | | | |
|---|---|---|---|---|---|
| Cluster | BLE | GBR | RH | SAV | WIL |
| 1 | 0 | 83 | 0 | 0 | 0 |
| 2 | 1 | 0 | 0 | 0 | 0 |
| 3 | 0 | 13 | 0 | 0 | 0 |

**Fig. 3** Example cluster analyses for (**a**) PK and (**b**) kurtosis impulsiveness measures of the SSC 1-min values over the duration of the month time period. Shaded regions indicate where 50% of the observations in the cluster fall. Shading is the 50% portion of all points within a specific cluster, whereas the color of shading and points denotes cluster number

OCS sites, and the cold water coral reef soundscapes at RH and SAV clustering separately from each other but with SAV significantly overlapping with the sandy habitats of BLE and WIL.

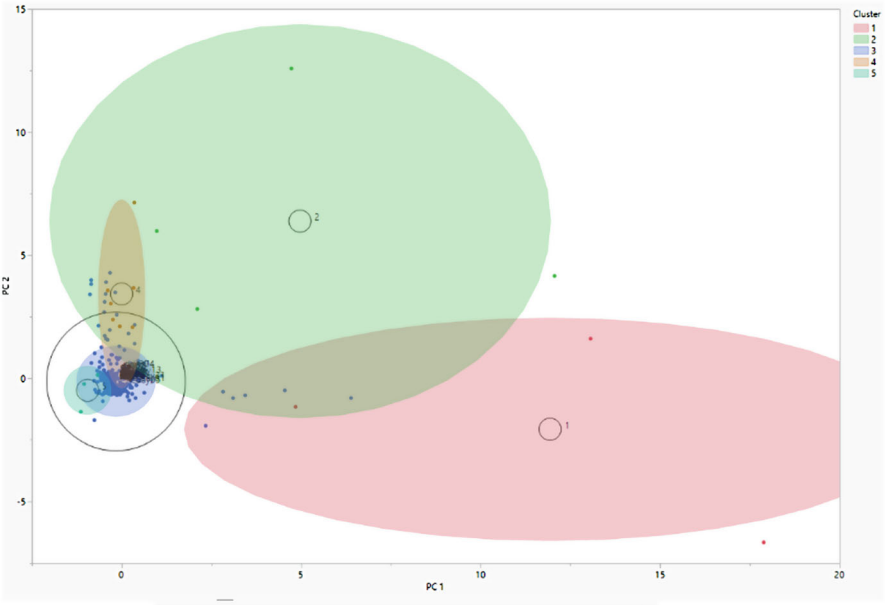
Examination of the clustering for the impulsiveness metric of kurtosis again showed GBR separated out with no overlap with the OCS site soundscapes. However, unlike PK, SPL, and D-Index where there was separation between the two cold water coral reef SSC metrics, there was no separation of any OCS sites by location or habitat (Fig. 3b Cluster 4).

The periodicity SSC metric clustering pattern was unique. GBR did not separate apart from the OCS sites (Fig. 4). Cluster analyses that constrained the number of clusters from 3 to 6 all resulted in a single cluster with contributions from all five soundscape locations. Clusters other than the main periodicity cluster contained only outliers which were defined as including less than seven total data points stemming from one or more locations.

## Conclusion

The application of the SSC proposed by Wilford et al. (2021) to soundscapes recorded in deep and shallow water locations offered the opportunity to determine if soundscape characteristics could be meaningfully compared across habitats with the potential to acoustically inform habitat type and function. The SSC output indicated that the shallow coral reef habitat was the loudest, most impulsive, and least variable (both in terms of metric variability and uniformity). The SSC output for GBR was visibility different in the SSC matrix color presentation and overall visual pattern, identifying it as dissimilar from the OCS locations.

Clustering analysis demonstrated the utility of the SSC metrics in grouping soundscapes with similar characteristics. The GBR soundscape separated from the




| | Site ID | | | | |
|---|---|---|---|---|---|
| Cluster | BLE | GBR | RH | SAV | WIL |
| 1 | 0 | 0 | 1 | 0 | 2 |
| 2 | 0 | 0 | 0 | 0 | 4 |
| 3 | 66 | 96 | 45 | 68 | 56 |
| 4 | 0 | 0 | 1 | 0 | 6 |
| 5 | 2 | 0 | 1 | 0 | 0 |

**Fig. 4** Periodicity measures of the SSC 1-min values over the duration of the month time period. Cluster 3 contains 331 values spread across all sites. The remaining clusters have seven or less outlier points each

OCS sites as distinctly dissimilar in terms of amplitude, impulsiveness, and uniformity, but not for periodicity indicating that the shallow GBR soundscape is likely reflecting an ecosystem that functions differently from the deep-water habitats of the OCS. Clustering of the SSC amplitude and uniformity metrics in the OCS sites further showed separation of the cold water reef habitats compared to the sandy bottom habitats, again suggesting that these two habitats support different ecosystem components and/or function.

As more is learned about how marine life sense and perceive sound, as well as understanding which signal/soundscape characteristics are biologically meaningful to marine life, soundscape characteristics other than traditional amplitude measurements will likely play a larger role. Measures of impulsiveness, uniformity, and periodicity capture different information in the temporal and spectral domains than amplitude. Here the exercise of applying the multidimensional SSC metrics to

different habitats demonstrates that soundscapes can be meaningfully categorized and differentiated on characteristics other than just amplitude, and that the consideration of acoustic metrics in addition to amplitude may noninvasively provide insight into ecosystem changes and function.

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# Analyzing Long-Term Changes in Soundscapes Using Power Spectral Band Sums

21

Joseph J. Luczkovich and Mark W. Sprague

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**Abstract**

Various species of fish make sounds (individually and in choruses) while mating and for aggressive encounters. Recording the sounds of fishes in mixed-species choruses has been done for 26 years in Pamlico Sound, North

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Carolina, USA. But the changing contributions of the various fishes to the soundscape that varies diurnally, seasonally, and over 20 years have not been characterized. Multiple passive acoustic recording systems have been used to make these recordings at a variety of locations. After calibrating these recording systems, an analysis of the soundscape was performed using power spectral band (PSB) sums, which sum the acoustic energy in frequency bands associated with known soniferous fish species. The analysis reveals some interesting patterns related to fish ecology: (1) sound production occurs after sunset with a nightly peak in sound pressure levels; (2) there is a seasonal increase in the power spectral band sums, correlated with increasing water temperatures; and (3) species in the Sciaenidae family have distinct periods of sonic activity. The temporal progression of these species in their spawning areas was plotted using PSB sums. Examples are presented of these soundscapes and descriptions of the changes observed in the soundscapes from samples taken over 20 years (1997–2018).

**Keywords**

Passive acoustic monitoring · Estuary · Spawning · Sciaenidae

## Introduction

Various species of fish make sounds (individually and in choruses) while mating and during aggressive encounters (Mok and Gilmore 1983; Rountree et al. 2006). For more than 20 years, passive acoustic monitoring of the estuarine soundscapes has been used to understand the behavior and seasonal occurrence of fishes that produce sounds (Luczkovich et al. 1999, 2008; Sprague et al. 2000). Sounds that are recorded in estuaries in North Carolina are mainly due to fishes in the drum family (Sciaenidae). These fishes produce nocturnal choruses of varying intensity. The chorusing is associated with spawning activity (they are advertisement calls made by males). But the entirety of the data has not been examined for patterns indicating that the sound intensity measured in the same month or season varies across multiple years.

Therefore, it is the objective in this retrospective to investigate the following questions:

1. How does the chorusing vary geographically in the soundscape (using the focal year 1998)?
2. How does the chorusing change temporally within a year (using the focal year 2006)?
3. How does the chorusing change across years (focal years of 2006, 2017, and 2018)?

## Calibration of Hydrophones and Recording Systems

Three different recording systems were used, and each one is different in hydrophone sensitivity and recording system frequency response (Table 1). They spanned a range of purchase costs: $200 each ($370 corrected for inflation in 2023, using the tool at https://www.usinflationcalculator.com/) for the East Carolina University Sonobuoy with an analog cassette recorder, $5000 each for the Loggerhead LARS digital recorder (an equivalent model called Snap made by Loggerhead, Inc. costs $3995 in 2023), and $10,185 each (purchased in 2015, $ 11,300 in 2023) for the icListen smart digital hydrophone (Ocean Sonics, Inc.). A float collar, 50-m cable, battery pack, and Lucy instrument control software (V4) were also used with the icListen hydrophone, which raised the cost of the system to $22,308 in 2015 ($28,626 in 2023). A description of each system follows.

### ECU Sonobuoys

These sonobuoys were constructed at East Carolina University and consisted of a Gulton GLN-9190 hydrophone (−174 dB re 1 V/μPa sensitivity) connected to a Sony TCM 313 cassette recorder and a timing circuit (Luczkovich et al. 2008). The hydrophone output was recorded on a cassette audiotape (TDK D Type I normal-bias 90 min) for long-term storage and analysis. This recorder system was programmed to be powered on at intervals (30 min or 60 min) selected by the user. After powering up, the cassette recorder made ambient soundscape recordings that were 90 s duration, then shut off until the next power-on interval. A talking clock announced the local time at the start of each recording. Tape recordings with successive intervals lasted for 12–24 h, after which the sonobuoy was recovered and redeployed with new a tape.

### Loggerhead LARS

This recording system has an HTI 96-min hydrophone with a sensitivity of −164 dB re 1 V/μPa and a frequency range of 10 Hz−10 kHz. Digital recordings were made using a Dell Axim and the Loggerhead Pocket PC software application. The calibration procedures for the Dell Axim (LHCal.exe) were followed according to

**Table 1** The passive acoustic hydrophone recording systems and frequency responses of each system used

| Recording system | Hydrophone sensitivity | System frequency response |
|---|---|---|
| ECU Sonobuoy | −174 dB re 1 V/μPa | 30–2000 Hz |
| Loggerhead LARS | −164 dB re 1 V/μPa | 10 Hz–10 kHz |
| Ocean Sonics icListen HF | −171 dB re 1 V/μPa | 10 Hz–200 kHz |

the manufacturer using a signal generator as the input. The HTI 96-min hydrophone sensitivity specification was used (−164 dB re 1 V/μPa). A calibration adjustment was made (+175.5 dB was added to normalized LARS recordings to convert the file sample values to μPa for displays and analysis in Python).

## Ocean Sonics icListen HF

This is a "smart" hydrophone (Ocean Sonics model SB2-ETH) with digital recording and hydrophone combined in a single unit. Calibration was provided by the manufacturer (Calibration certificate C3154). This hydrophone has a sensitivity of −171 dB re 1 V/μPa and a system response frequency of 10 Hz–200 kHz. The float collar was used to suspend the icListen in the water column, which allowed the hydrophone to remain neutrally buoyant 1 m above the bottom. The float collar was attached to an anchored cage where the battery pack was cabled to the hydrophone.

## Calibrations of the Three Hydrophone Systems

Although the three recording systems each had calibrations based on hydrophone sensitivities, the same sounds were recorded with each to make sure they all produced the same levels in recordings. Reference tones and noise (200–10,000 Hz) were played back in a test pool and in the air, while recording the frequency response for each of the hydrophones. Sonobuoy recordings were compared with recordings made simultaneously using a reference hydrophone: Gunnar Rasmussen Hydrophone Model 10CS (−211.7 dB re 1 V/μPa sensitivity, frequency response 0.1–25,000 Hz). The reference hydrophone output voltage was simultaneously recorded to a digital recorder (SONY DAT tape or Zoom Recorder). Sine waves produced by a signal generator with input voltages measured on the oscilloscope were also recorded digitally. The amplitude of the reference hydrophone associated with the input voltages from the signal generator was also recorded for each sine wave test tone. The audiocassette tape in the sonobuoy and the reference hydrophone output were digitized on a Zoom F8 Recorder at 44.1 kHz with a +25 dB gain. The Gulton sonobuoy hydrophone was compared against the reference hydrophone using the known input voltages and digitized audio amplitudes as received at the two hydrophones to develop a calibration relationship. Digitized field tape recordings from the ECU Sonobuoys were recorded on the Zoom F8 multitrack recorder using the same settings.

An inter-hydrophone and recording system comparison test was conducted for the three systems used (ECU sonobuoy, LARS, and icListen) in a freshwater pool and in air to compare the sound pressure levels recorded and the frequency response of each passive acoustic recording system. Such inter-recorder calibration is essential to make comparisons of field recordings by each of the systems over time. To accomplish this inter-hydrophone calibration, comparisons were made of sound recordings from the three recording systems in the pool at distances of 11 m while playing a series of tones and white or pink noise signals through an underwater speaker (Clark Synthesis AQ-339) with a power amplifier (Pyle PMLRA200). For

air intercalibration measurements, an amplified JBL public address system (JBL model EON208P) was used to play the same sounds at a distance of 1 m from the three hydrophone and recording systems.

## Power Spectral Analysis

The procedure in this section was previously published (Luczkovich and Sprague 2022). Power spectral band sums (PSB sums) were computed for specific frequency bands associated with known fish sounds. Table 2 shows the frequency range associated with different fish species. These frequency bands were chosen as indicators of calls by the various species. The bands do not contain all frequencies in the calls. The PSB sum $S_{PSB}$ is the sum of all power spectrum components $P_n$ for frequencies in the band $f_{min} \leq f_n \leq f_{max}$

$$S_{PSB}(f_{min}, f_{max}) = \sum_{n_{min}}^{n_{max}} P_n \Delta f$$

where $n_{min}$ is the index of the smallest frequency component in the band, $n_{max}$ is the index of the largest frequency component in the band, and $\Delta f$ the frequency interval in the power spectrum.

The average PSB sum was computed for each band in Table 2 using 10 s of the recording. The PSB sums were plotted versus time to identify times in the recordings likely to contain fish sounds. One indication of fish activity is when the PSB sum for a frequency band increased with a different pattern than the other bands.

## Study Design

The 1998 ECU sonobuoy survey was used to develop maps of sounds heard at different locations using the Summed Nightly Drumming Index (SNDI), which was scored on 0 (no fish sounds) to 3 (loud chorusing) scale by a human listener for red drum *Sciaenops ocellatus*, spotted seatrout *Cynoscion nebulosus*, weakfish *C. regalis*,

**Table 2** The bands used in the power spectra analysis (power spectral band sums, with frequency max and min) and the fish species that are dominant within each frequency band. A similar table with different band numbers was previously published (Luczkovich and Sprague 2022). Atlantic croaker is assigned to Band IV in this table with a wider frequency range than what was used in the previous study

| Frequency band (Hz) | Common name, species (family) |
|---|---|
| Band I: 100–200 | Red drum, *Sciaenops ocellatus* (Sciaenidae) |
| Band II: 200–300 | Oyster toadfish, *Opsanus tau* (Batrachoididae) |
| Band III: 300–600 | Spotted seatrout, *Cynoscion nebulosus* (Sciaenidae)<br>Weakfish, *Cynoscion regalis* (Sciaenidae) |
| Band IV: 300–1000 | Atlantic croaker, *Micropogonias undulatus* (Sciaenidae) |
| Band V: 600–1500 | Silver perch, *Bairdiella chrysoura* (Sciaenidae) |
| Band VI: 1500–2000 | Striped cusk eel, *Ophidion marginatum* (Ophidiidae) |

and silver perch *Bairdiella chrysoura* (Luczkovich et al. 2008). These sonobuoys were deployed and recovered at random locations (located with GPS coordinates) chosen within two grids (Ocracoke Inlet and Bay River) every month from May through October 1998 (Fig. 1; see the detailed description of the sampling design in the original study Luczkovich et al. 2008). These geolocated passive acoustic recordings were used to produce spatial soundscape analysis via geostatistical kriging interpolation (ArcMap 10.6.1) of the nightly PSB sums for each of the power spectra bands associated with the fish species heard (Table 2) from the digitalized sonobuoy tapes.

The Bay River location was used thereafter for temporal comparisons with the LARS and icListen recording systems in focal years 2006 through 2018 (Fig. 2). Spotted seatrout and red drum are known to spawn in this Bay River grid location. Also, the Bay River grid location is affected by periodic hypoxic conditions and salinity variation, thus temporal changes in the soundscape were associated with these changing environmental conditions. Long-term continuous recordings were made at the Bay River grid fish and at known fish spawning hotspots (Fig. 2). The LARS recorders were used to record the soundscape from May through October 2006 (with a programmed recording duty cycle of 10 s recordings every 15 min). In addition, a water quality sonde (YSI model 6600) was deployed alongside the LARS to monitor temperature (Luczkovich et al. 2013a). The icListen recording system was deployed within the Bay River grid and at a nearby location from August through October in both 2017 and 2018. The icListen duty cycle was programmed to log 5-min recordings at 15 min intervals during this time.

## Spatial Variation in Soundscape

There are differences in the soundscape within the eastern (Ocracoke Inlet) and western (Bay River) sonobuoy grids, with red drum "knocking" sounds and spotted seatrout "grunting" sounds heard and scored using the SNDI index and then interpolated to create soundscape maps (Fig. 3 is the soundscape map for red drum; Fig. 4 is the soundscape map for spotted seatrout). Red drum were highly aggregated and localized in these two grids, occurring more often in deep water (>3 m), whereas spotted seatrout were widespread and occurred over much of the area of both grids; they were heard more often in shallow water (<3 m). The weakfish and silver perch were more dominant in the eastern grid (Ocracoke Inlet) in higher salinity waters (Fig. 5 is the map for weakfish, and Fig. 6 is the map for silver perch).

## Temporal Variation in Soundscape

### Diurnal Variation in the Soundscape

There was a regular nocturnal chorusing of fish on these passive acoustic recordings. Fish made sounds before sunset but PSD band sums peaked after sunset. As an illustrative example, a typical Sciaenidae fish chorus can be observed on 18 Aug

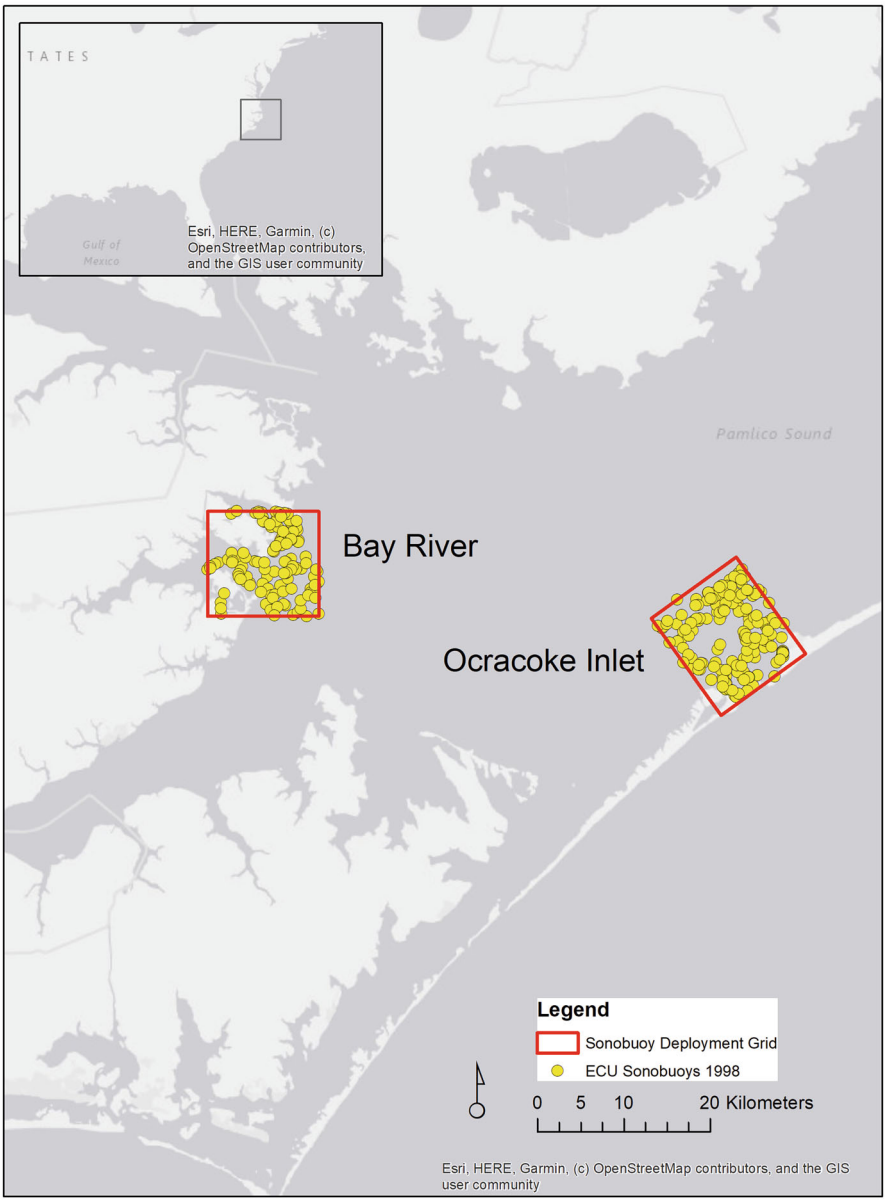



**Fig. 1** Overview map of the Bay River and Ocracoke Inlet study sites in Pamlico Sound North Carolina, USA. ECU Sonobuoys were deployed (yellow circles) May–Oct 1998. Also placed in the Bay River: the LARS recording system was deployed in May–Oct 2006; the icListen hydrophone recording systems was deployed in Aug–Oct 2017 and 2018 (Base map reprinted with permission from Esri, Inc.)

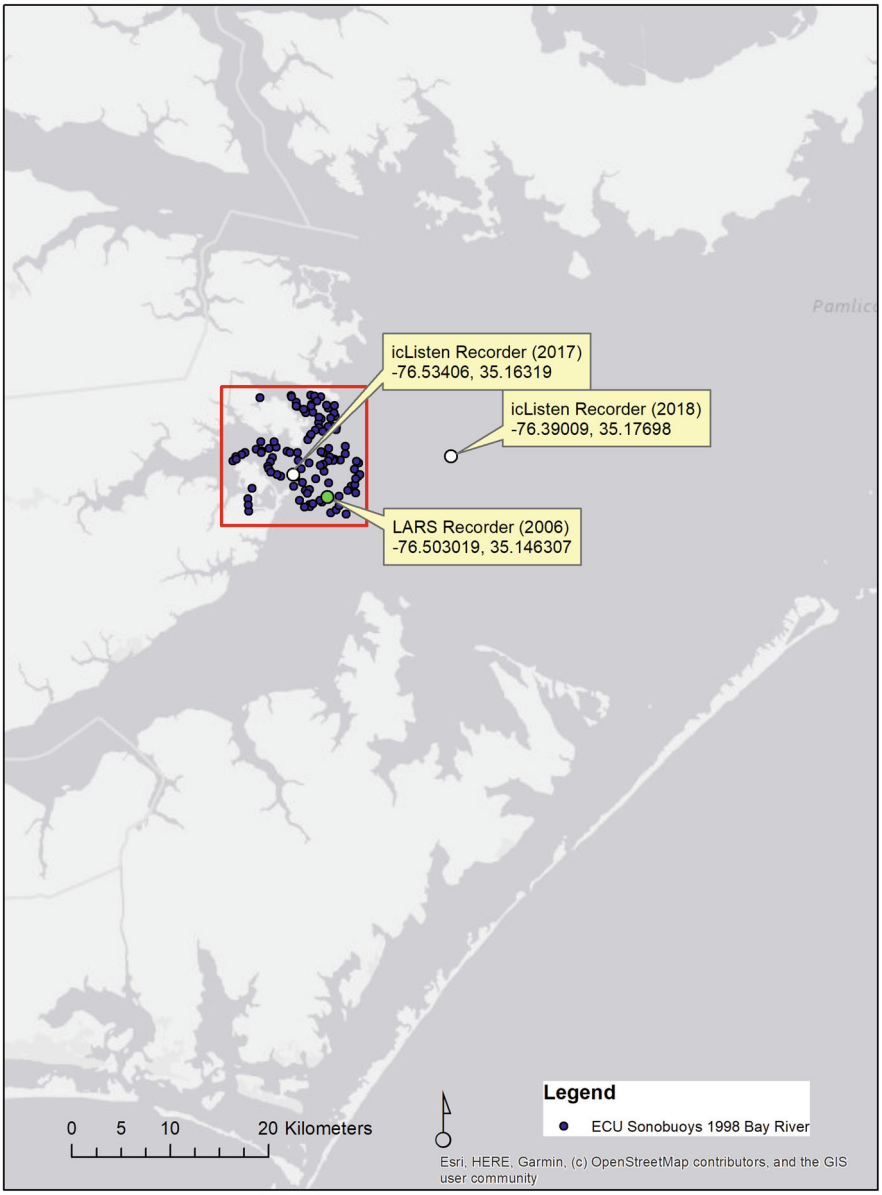



**Fig. 2** Overview map of the Bay River study sites in Pamlico Sound North Carolina, USA. ECU Sonobuoys were deployed at the points indicated during May–Oct 1998. The LARS recording system was deployed in May–Oct 2006 at the position indicated. The icListen hydrophone recording systems was deployed at two locations indicated during Aug–Oct 2017 and again in 2018 (Base map reprinted with permission from Esri, Inc.)

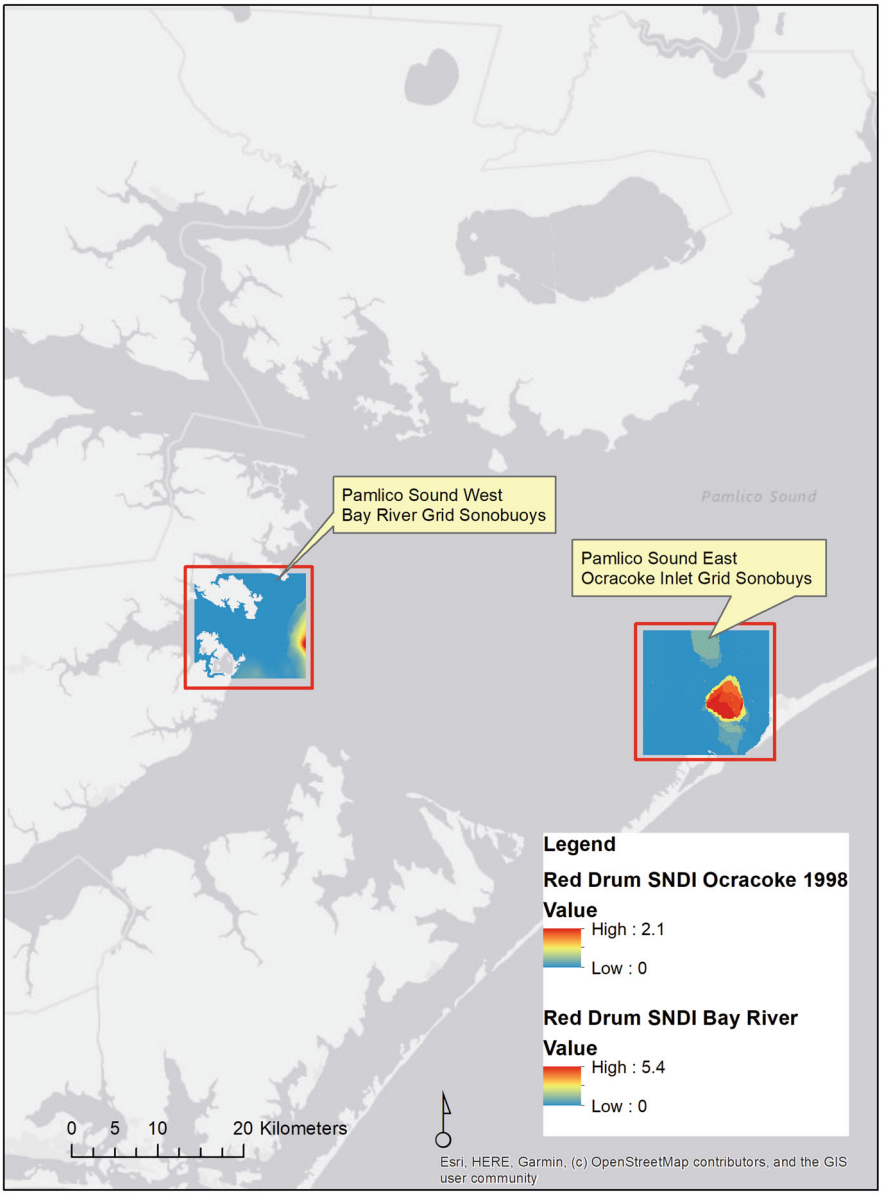



**Fig. 3** The soundscape map for red drum *Sciaenops ocellatus* based on kriging interpolation of Summed Nightly Drumming Index SNDI values from sonobuoy recordings made in 1998 at Bay River and Ocracoke Inlet grids in Pamlico Sound, North Carolina, USA (Base map reprinted with permission from Esri, Inc.)

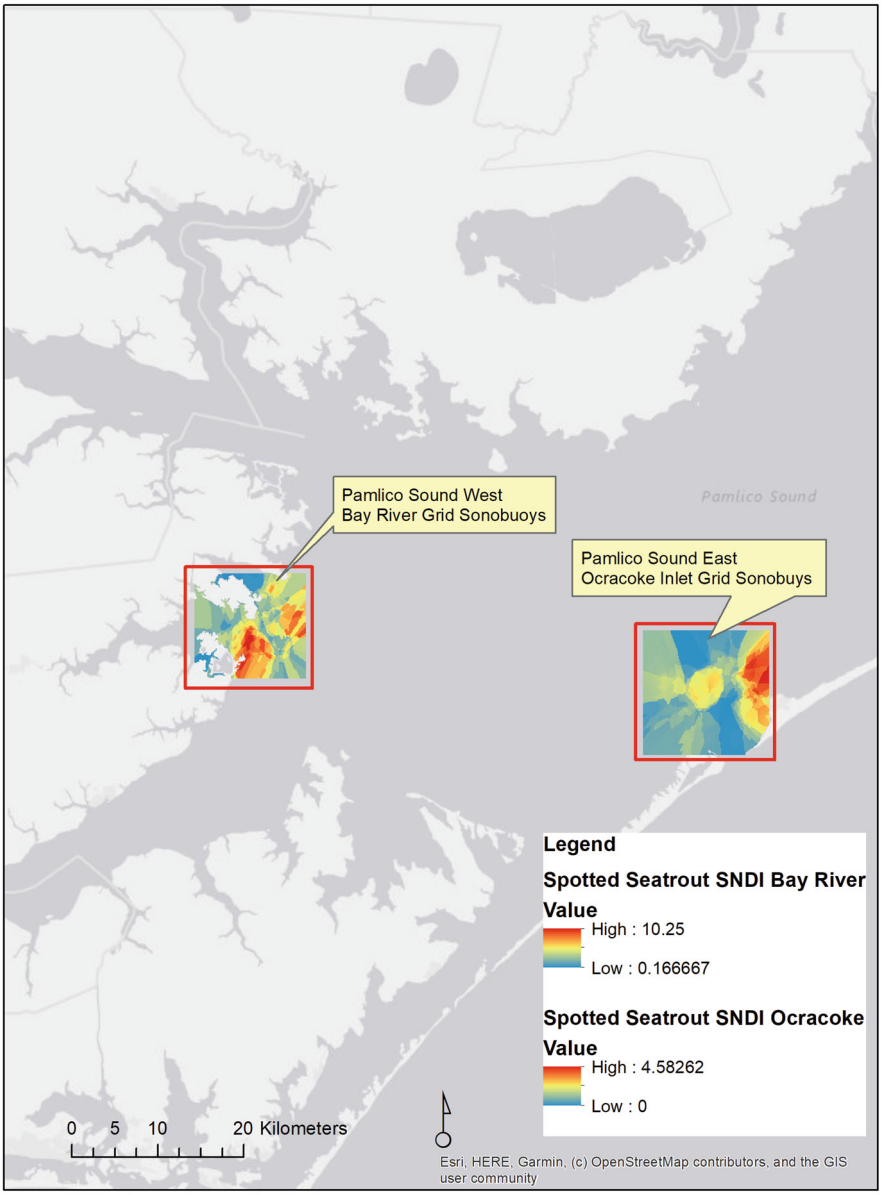



**Fig. 4** The soundscape map for spotted seatrout *Cynoscion nebulosus* based on kriging interpolation of Summed Nightly Drumming Index SNDI values from sonobuoy recordings made in 1998 at Bay River and Ocracoke Inlet grids in Pamlico Sound, North Carolina, USA (Base map reprinted with permission from Esri, Inc.)

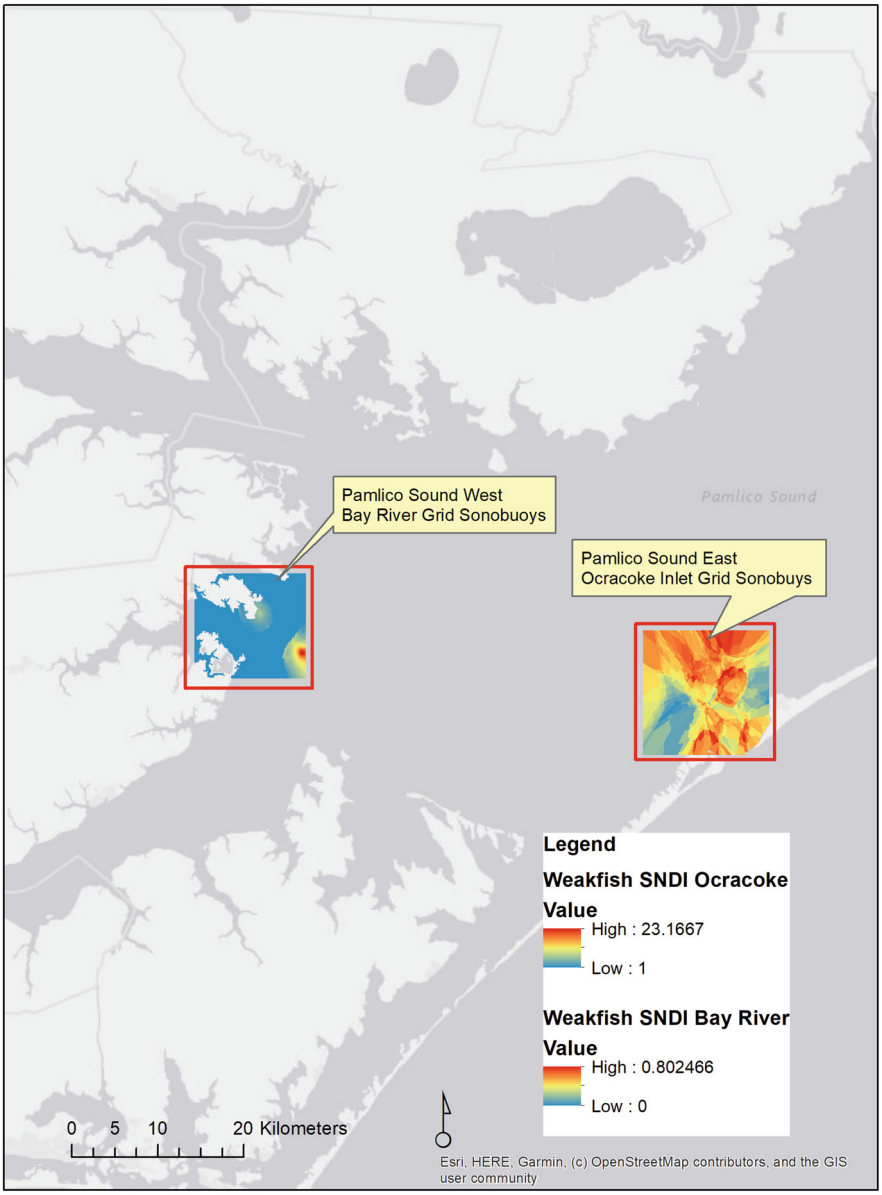



**Fig. 5** The soundscape map for weakfish *Cynoscion regalis* based on kriging interpolation of Summed Nightly Drumming Index SNDI values from sonobuoy recordings made in 1998 at Bay River and Ocracoke Inlet grids in Pamlico Sound, North Carolina, USA (Base map reprinted with permission from Esri, Inc.)

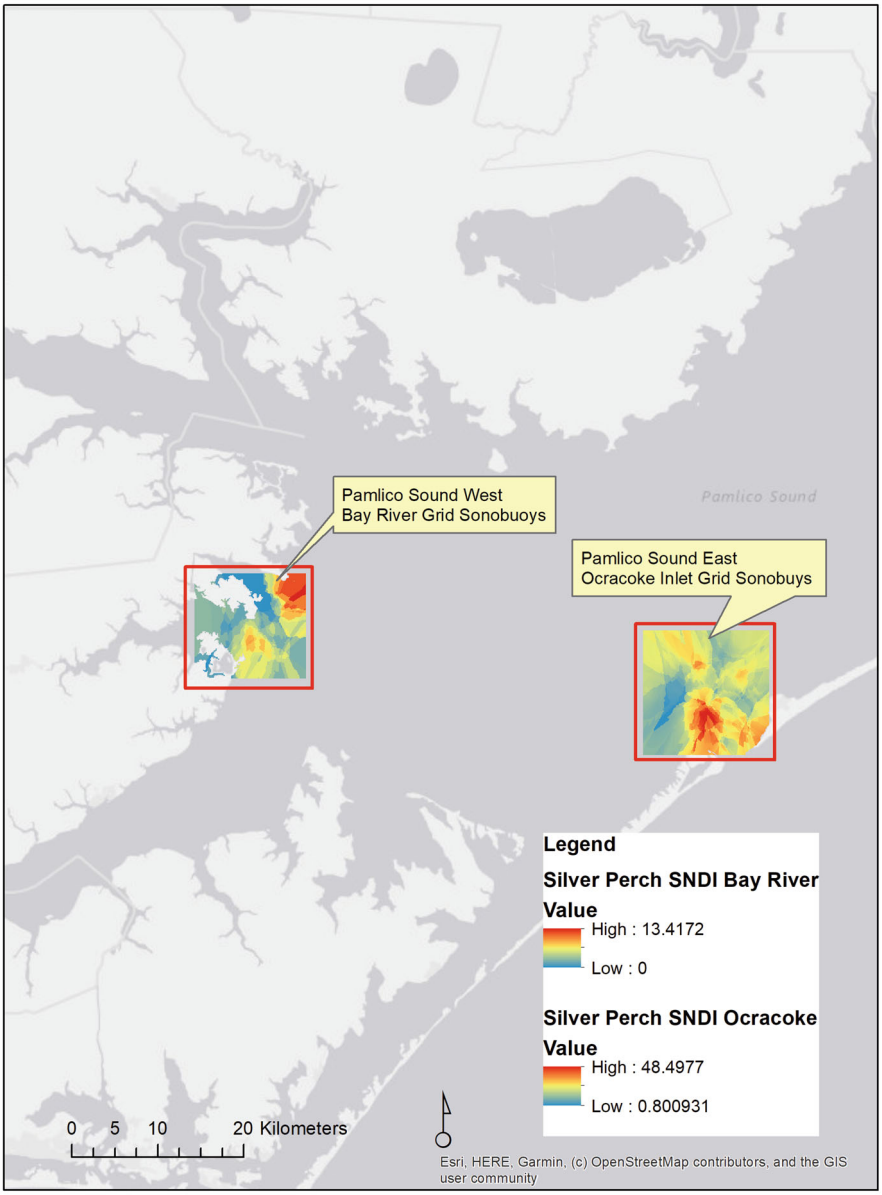



**Fig. 6** The soundscape map for silver perch *Bairdiella chrysoura* based on kriging interpolation of Summed Nightly Drumming Index SNDI values from sonobuoy recordings made in 1998 at Bay River and Ocracoke Inlet grids in Pamlico Sound, North Carolina, USA (Base map reprinted with permission from Esri, Inc.)

2018 in the Bay River grid area (Fig. 7). The PSB sums (Fig. 7a) indicated that the primary contributors early after sunset were red drum and spotted seatrout (Band I 100–200 Hz and Band III 300–600 Hz show peaks on 18 Aug 2018 at approximately 21:00 local time UTC −5 h; the calls of these species were confirmed by listening to

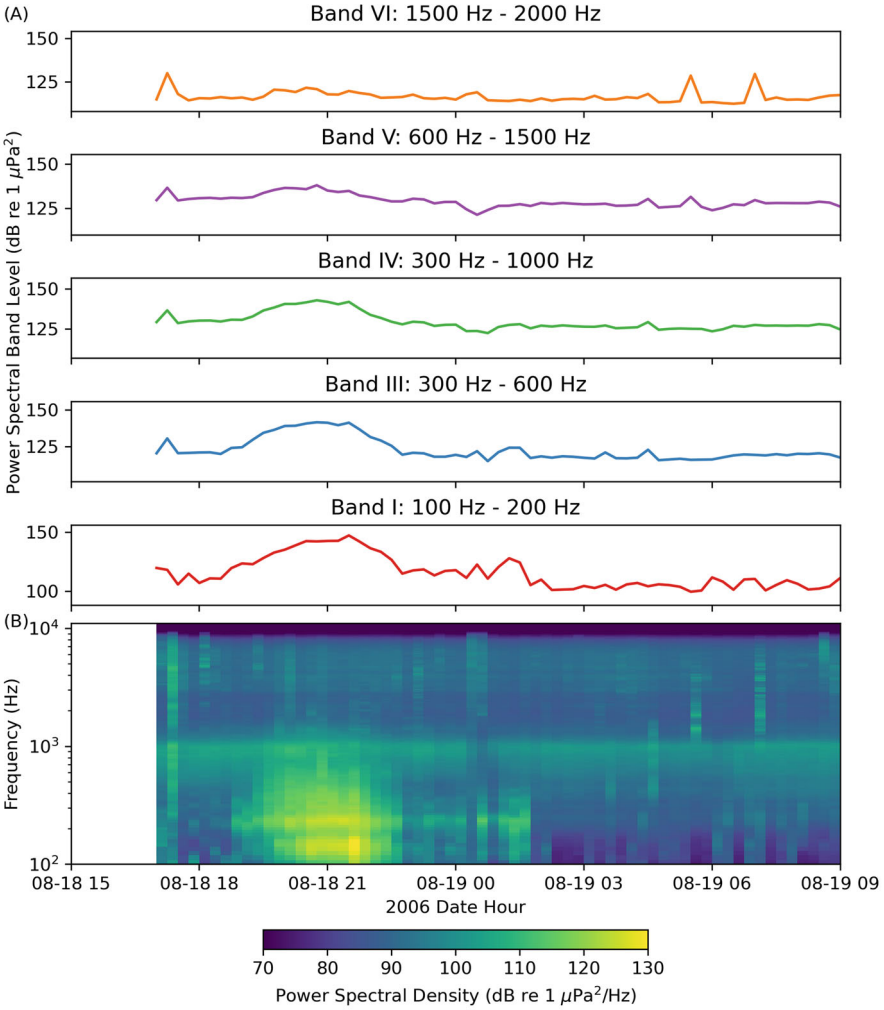



**Fig. 7** The nocturnal choruses of fish the Bay River grid on 18 Aug 2006 1700 through 9 Aug 2006 0900 (local time UTC −5. (**a**) The power spectral band sums (dB re 1 μPa$^2$/Hz) for the frequency bands indicated and associated with fish species are given in Table 2. (**b**) The composite spectrogram of recordings made that night using the LARS recorder system

these recordings). This mixed species sciaenid chorus ended by midnight on 18 Aug (0000 local time UTC −5 h, Fig. 7b). Later in this same set of recordings on 19 Aug between 0400 and 0600 local times (UTC −5 h), a rise was detected in Band VI associated with striped cusk eels *Ophidion marginatum* (Fig. 7a). These are higher frequency (1.5–2.0 kHz) "chattering" calls (Fig. 7b). These sciaenid choruses and individual striped cusk eel chatter sounds became less dominant by daybreak (after 0600 local time) and remained low during each diurnal period, increasing again at sunset the next night. This pattern repeated every night throughout the summer.

## Seasonal Variation in the Soundscape

The soundscape was recorded at a single location (Fig. 2, Latitude 35.146307N and Longitude 76.503019W) in the Bay River grid 2006 from 30 April through 04 October using the LARS passive acoustic recording system. In this long sequence of recordings, using a 10-s recording period with a duty cycle repeating at 600 s intervals (every 10 min), taken over 157 days or more than 5 months, the changing phenology of the soundscape was visualized in a composite spectrogram. The seasonal change in the soundscape reflected different species participating in the chorusing, changing environmental conditions that affected the calling behavior, including warming and cooling water temperatures, lengthening and shortening photoperiods, salinity fluctuations, hypoxic events, multiple species of Sciaenidae (silver perch, weakfish, spotted seatrout, Atlantic croaker *Micropogonias undulatus*, and red drum), Batrachoididae (oyster toadfish *Opsanus tau*), and striped cusk eels performing their nightly choruses with quiet daytime periods intervening were recorded. These increases and decreases in sound amplitude in each of the frequency bands associated with these different fish species (Table 2) are visible as nightly pulsing variations in the PSB sums plots (Fig. 8a 30 April–7 Jun; Fig. 9a 30 June–15 Aug; Fig. 10a, 20 Aug–25 Sep; Fig. 11a, 03 Oct–10 Oct; and Fig. 12a, 28 Sep–04 Nov). Each night, as the fish in the area reached a crescendo of calling activity in a chorus, the PSD sum increased, and in the daytime, as the fish chorusing activity ceased, the PSB sums fell back to a baseline. This pattern was repeated each night for the entire summer, but the choruses' dominant frequency shifted upwards as the environment changed. The water became warmer in May and June, rising from 13 °C on 3 May to 23 °C by 30 May on the water quality sonde deployed along with the LARS (Luczkovich et al. 2013b). As water temperature warmed, the dominant frequency of the fish choruses increased (Fig. 8b) from <200 Hz to 200–250 Hz by 30 May. Water temperatures stabilized in the summer months of July and August (26–28 °C), leading to a regular pattern of seatrout grunting choruses (Fig. 9), and then fell back to lower frequencies in the cooler water in Sep and Oct (Fig. 10, Fig. 11). In Fig. 8b, a frequency band around 200 Hz (Band I, 100–200 Hz) can be observed increasing in frequency each day through the month of May; this activity persisted during the daytime. This frequency band increase was due to an increase in oyster toadfish *Opsanus tau* calling (human listeners could discern individual toadfish calling during daytime on these recordings) and was associated with the warming of the water temperature at the site. Silver perch (PSB Band V

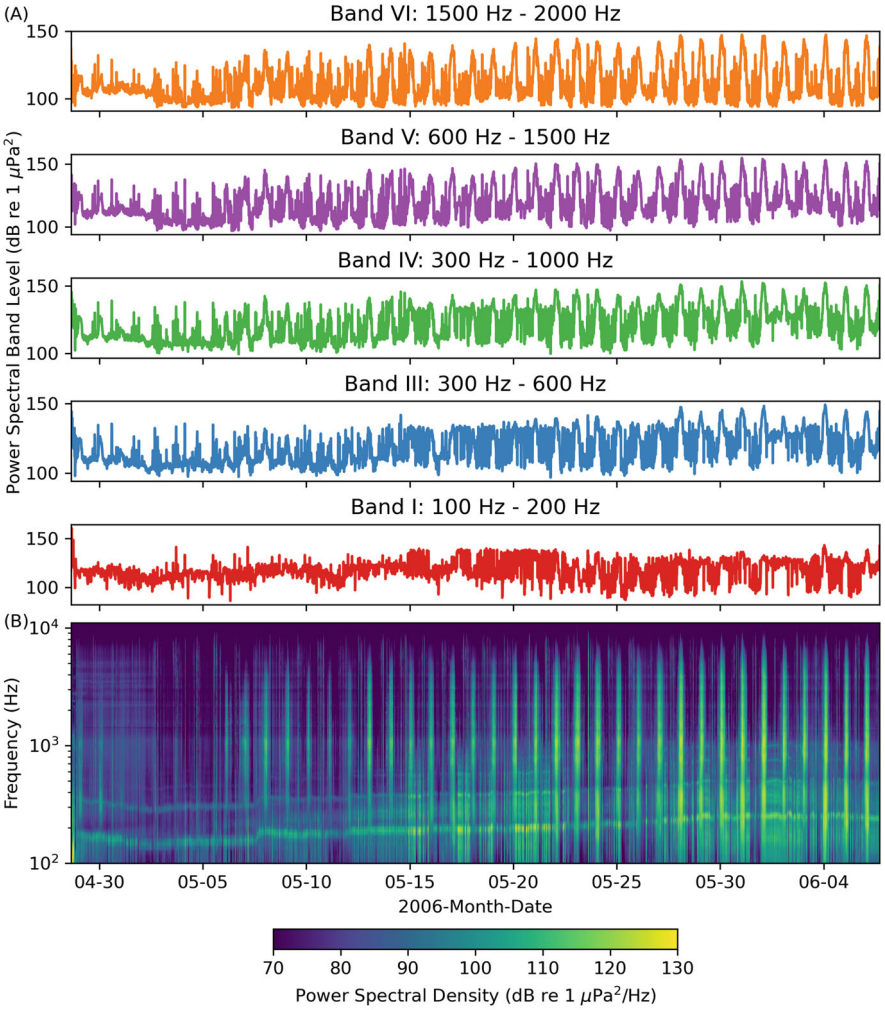



**Fig. 8** The PSB sums and composite spectrograms plotted over time from LARS recordings made between 29 April and 6 June 2006 in the Bay River grid. (**a**) PSB sums for frequency Bands I, III, IV, V, and VI associated with fish species in Table 2. (**b**) The composite spectrogram of the soundscape for this same period with frequency on a log scale (100 Hz–10 kHz) and color ramp scale. All PSD sums and color ramp units are given in dB re 1 μPa$^2$/Hz

600–1500 Hz) frequency band also showed this increase in temperature-associated calling activity, although this species was largely nocturnal in its calling activity. Increased PSB sum with increased water temperature can be seen as increased peaks in the Bands III, IV, and V and as broad spectral pulses (300–1500 Hz) in the composite spectrogram occurring each night (Fig. 8b); these nocturnal choruses became more intense at the end of May and the start of June. During July and August (Fig. 9), the nightly spotted seatrout chorus became less intense and no longer

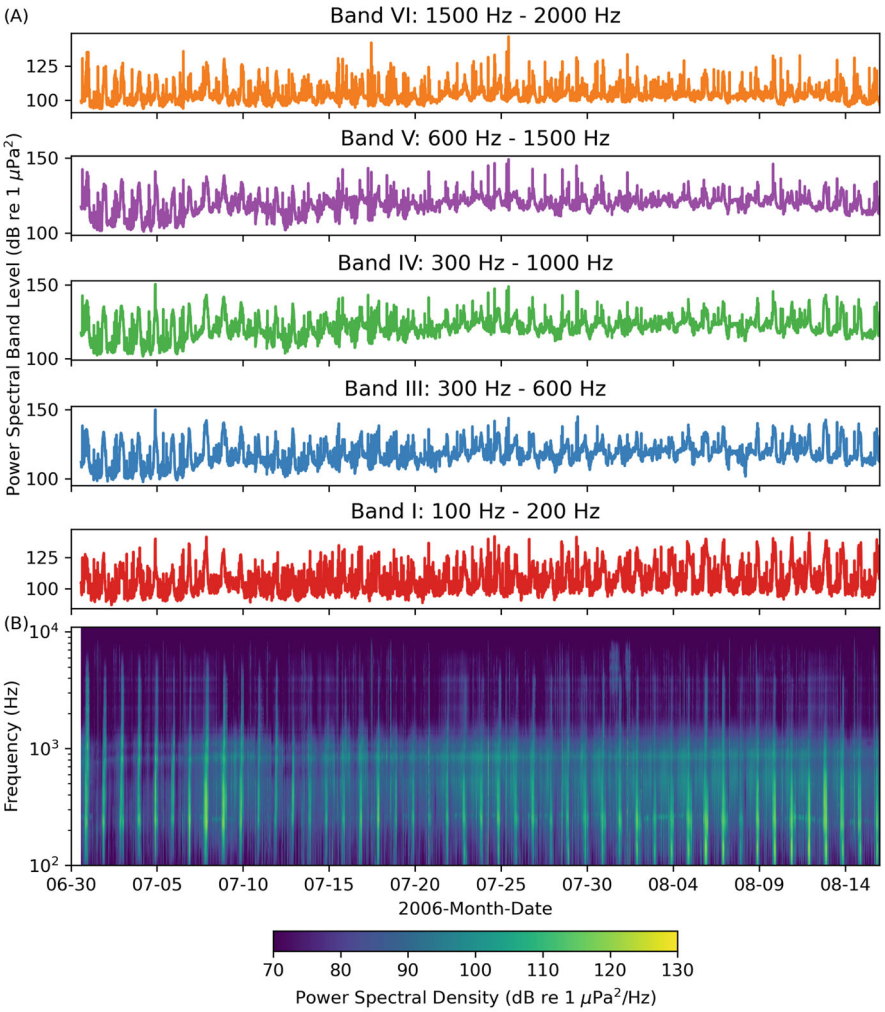



**Fig. 9** The PSB sums and composite spectrograms plotted over time from LARS recordings made between 30 June and 16 Aug 2006 in the Bay River grid. (**a**) PSB sums for frequency Bands I, III, IV, V, and VI associated with fish species in Table 2. (**b**) The composite spectrogram of the soundscape for this same period with frequency on a log scale (100 Hz–10 kHz) and color ramp scale. All PSD sums and color ramp units are given in dB re 1 μPa$^2$/Hz

had the dominant frequency of 200 Hz during the daytime, as the oyster toadfish had ceased calling as much during the day. Instead, during this summer period, spotted seatrout grunts (Band III 300–600 Hz) and red drum knocking (Band I 100–200 Hz) can be heard in late August and September 2006 as part of the nightly mixed sciaenid chorus (Fig. 10). In addition, the nightly chorus at this time was dominated by the grunting of spotted seatrout (Fig. 10a, Band III, 300–600 Hz), which then dominated the soundscape nocturnally. By the end of September (24 Sep 2006), this seatrout

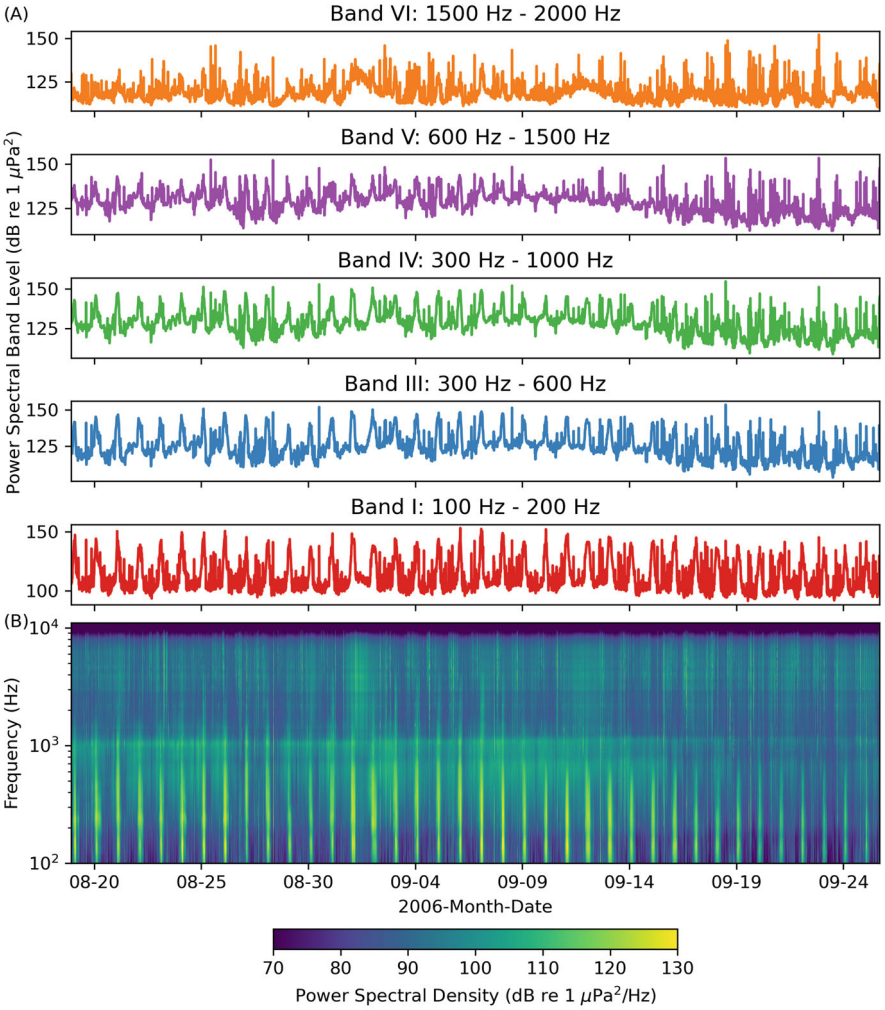



**Fig. 10** The PSB sums and composite spectrograms plotted over time from LARS recordings made between 20 Aug and 25 Sep 2006 in the Bay River grid. (**a**) PSB sums for frequency Bands I, III, IV, V, and VI associated with fish species in Table 2. (**b**) The composite spectrogram of the soundscape for this same period with frequency on a log scale (100 Hz–10 kHz) and color ramp scale. All PSD sums and color ramp units are given in dB re 1 μPa$^2$/Hz

chorus had become less dominant in the soundscape (Fig. 10b). A listener could also hear Atlantic croaker "popping" sounds in this period in Band IV (Fig. 10a), the sound of which overlaps with the *Cynoscion* grunts in Band III but has higher frequency components. In Fig. 11a, b, at the beginning of October, a shorter time window period (03 October through 11 October) was visualized in order to observe the diminished Sciaenidae chorus. (It is still dominated by the species above, spotted seatrout, red drum, and Atlantic croaker, but the chorus is greatly reduced in calling activity at the

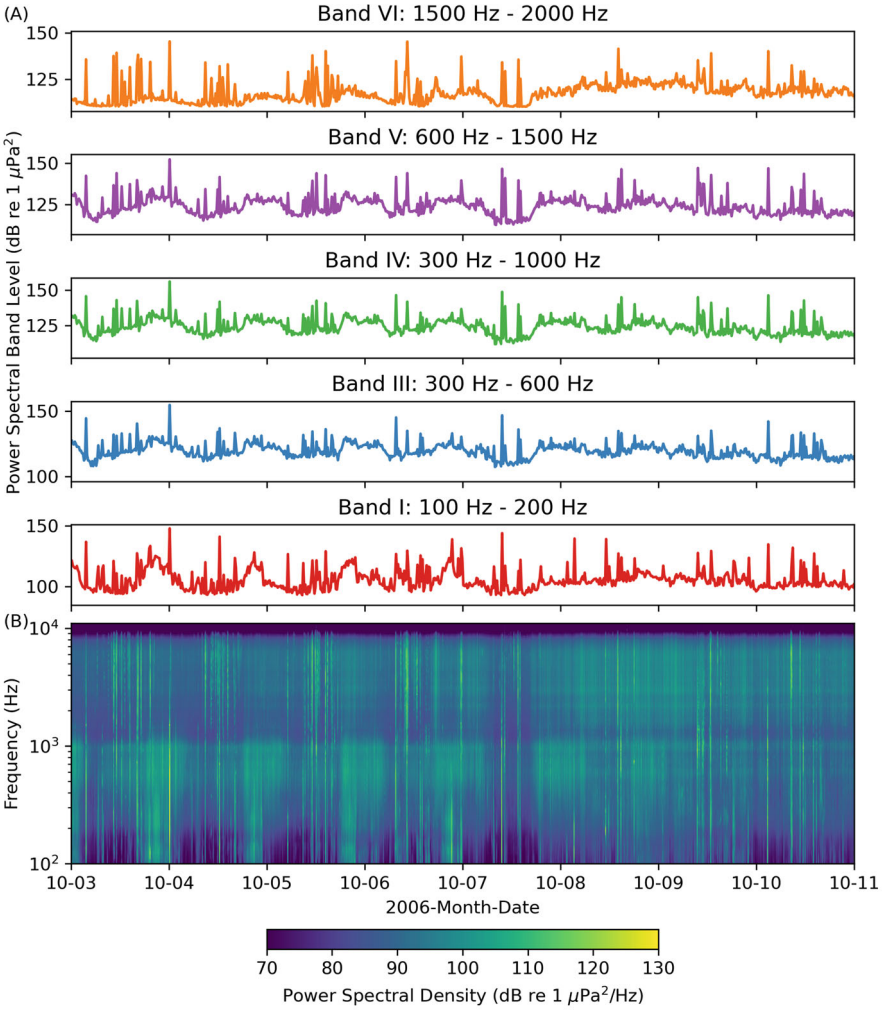



**Fig. 11** The PSB sums and composite spectrograms plotted over a short time window so the diminishing Sciaenidae chorusing can be observed from LARS recordings made between 3 Oct and 11 Oct 2006 in the Bay River grid. (**a**) PSB sums for frequency Bands I, III, IV, V, and VI associated with fish species in Table 2. (**b**) The composite spectrogram of the soundscape for this same period with frequency on a log scale (100 Hz–10 kHz) and color ramp scale. All PSD sums and color ramp units are given in dB re 1 μPa$^2$/Hz

end of this period.) These nightly Sciaenidae choruses persisted throughout October (PSB sums in Fig. 12a and in the spectrogram in Fig. 12b), but at a lower intensity of calling. This is due most likely to the shortening photoperiods, and colder water temperatures (water temperature dropped from 26 °C to 20 °C) at this time of year. By 9 November, the calling activity and fish chorusing is at a minimum, although it still occurs nocturnally, and most spawning activity has ceased (Fig. 12).

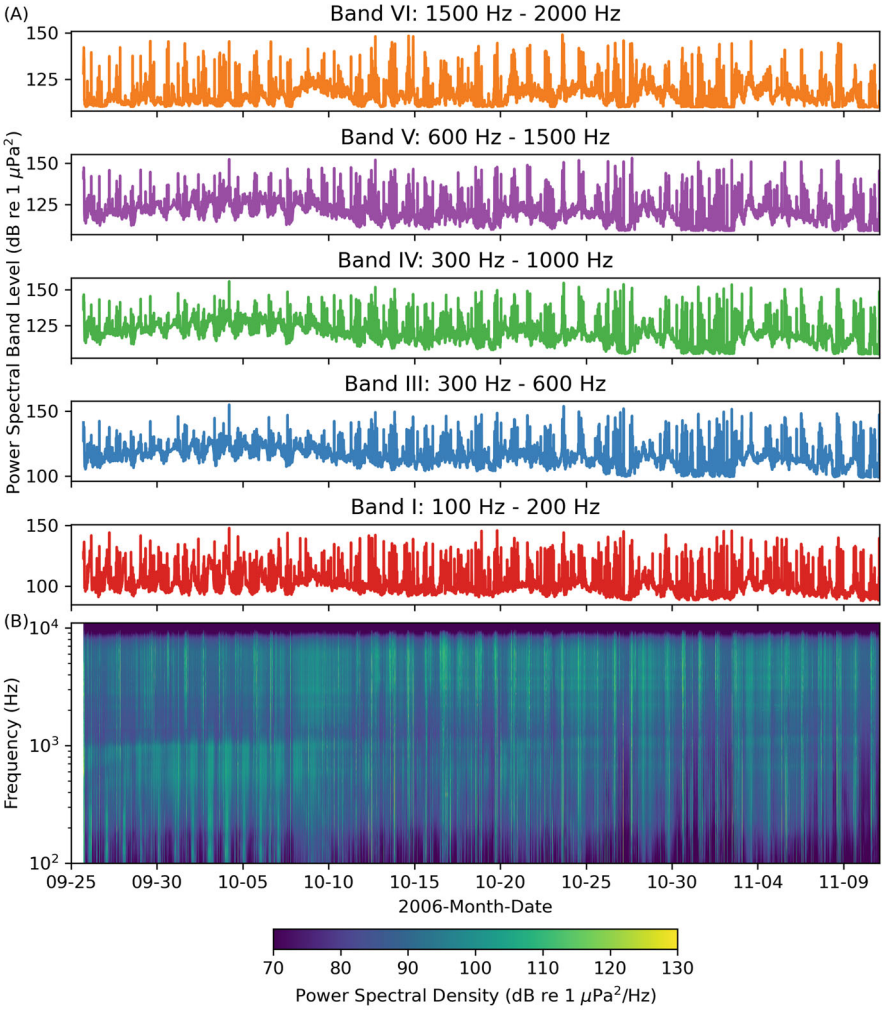



**Fig. 12** The PSB sums and composite spectrograms plotted over time from LARS recordings made between 25 Sep and 11 Nov 2006 in the Bay River grid. (**a**) PSB sums for frequency Bands I, III, IV, V, and VI associated with fish species in Table 2. (**b**) The composite spectrogram of the soundscape for this same period with frequency on a log scale (100 Hz–10 kHz) and color ramp scale. All PSD sums and color ramp units are given in dB re 1 μPa$^2$/Hz

## Interannual Variation in Soundscape

The Band I (red drum 100–200 Hz) PSB sum intensity over the years 2006–2018 at sites in the Bay River grid and on Brant Island Shoal was visualized in a composite spectrogram (Fig. 2). There was no reliable intercalibration possible for the ECU Sonobuoys used in 1998 due to reliability issues with old cassette decks, so these recordings were omitted in the interannual comparisons, limiting this to a 12-year

comparison (2006–2018). These measurements were restricted to the months of Aug–Oct, when red drum are actively calling and reproduction is occurring, effectively excluding any other species besides the red drum when examining Band 1. The icListen recording system was used as the comparative standard and transformed the LARS PSB sums to an equivalent level. Generally, these two recording systems responded similarly when tested in the pool and in air calibration tests to changes in the sound source frequencies and levels. This was especially true in Band 1, but a conversion factor (−0.7 dB LARS to icListen, with the LARS measuring slightly lower than the icListen in this frequency range based on the air and pool calibration tests) needed to be applied to standardize these systems' responses.

The result of the interannual comparison is shown in Fig. 13. There was a significant decline of 27.5 dB in PSB sum values associated with this Band 1 across the years (Table 3; ANOVA $F_{2,24} = 58.7$, $5.72 \times 10^{-10}$). The year with the greatest PSB sum for red drum was 2006, and this has declined over time so 2018 was the lowest year. The difference between 2006 and 2018 was the most significant (Table 3, Tukey's HSD P = 0.000001), although there was a significant decline of 19.4 dB between 2006 and 2017 as well. This suggests that a decline in the calling activity of red drum in the area around the Bay River has occurred over the period 2006–2018. This is a region well known to local fishermen that target the large spawning red drum aggregations for sport fishing. This soundscape measure of fish calling activity could be due to lower red drum population size perhaps due to increased sportfish harvest (prior to this observation period, between 2000 and 2006, red drum sport fish harvest averaged 823,991 fish per year ±18.17 Proportional

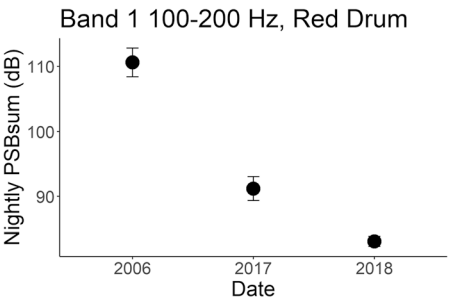



**Fig. 13** A plot of the mean nightly PSB sum during the months of Aug, Sep, and Oct over the years 2006 through 2018 for Band 1 (100–200 Hz), which is associated with red drum Sciaenops ocellatus calling activity. The means are dB ± 1SE (re 1 $\mu Pa^2/Hz$)

**Table 3** Tukey's HSD multiple comparison tests of mean PSB sum means for Band 1 (100–200 Hz), the red drum spectral band

| Comparison years | The difference in PSB means (dB) | Lower bound The difference in means (dB) | Upper bound The difference in means (dB) | P adjusted |
|---|---|---|---|---|
| 2017–2006 | −19.39 | −27.34 | −11.45 | 0.0000079 |
| 2018–2006 | −27.49 | −34.04 | −20.95 | 0.0000000 |
| 2018–2017 | −8.1 | −16.52 | −0.31 | 0.0607722 |

Standard Error (PSE) in North Carolina, whereas in 2017 and 2018, the sportfishing harvest had increased to 2,156,815 ± 18.62 PSE fish per year between 2014 and 2018; NOAA Marine Recreational Information program MRIP statistics, https://www.fisheries.noaa.gov/data-tools/recreational-fisheries-statistics-queries). Another possible cause in the soundscape associated with adult red drum might be due to migration away from this part of the ecosystem to avoid hypoxic stress in the low salinity regions from river runoff and nutrients input causing algal blooms at the river mouth, or some other factors as yet unknown. Water temperature alone is not likely to be responsible for this decline, because if anything, water temperatures have increased over this period.

## Summary

The soundscape in Pamlico Sound has become noticeably quieter in the frequency bands (<1 kHz) used by fishes after accounting for variation in the three recording systems used. There were significant changes in PSB Band I (red drum adults in the Sciaenidae), with a 27.5 dB decrease in PSB sum values. This acoustic soundscape metric (PSB sum) can be used to assess changes in the calling behavior of these fishes, which is closely tied to their spawning and reproductive activity and should be used as a proxy for the health of this Pamlico Sound red drum population. The observed decline in PSB Band I is an indication that the red drum spawning behavior is declining. The methods of soundscape measurements proposed here can be used as a long-term fishery assessment tool.

**Acknowledgments** Krysta Byrd did the digitization of sonobuoy tapes and was supported by a Golden Leaf Summer Internship. In 1998, Stephen Johnson, Chris Pullinger, and Todd Jenkins deployed and recovered the ECU sonobuoy. In 2006, Cecilia Krahforst, JP Walsh, Reide Corbett, and Sophie Dillard deployed and recovered the LARS system. In 2017 and 2018, Eric Diaddorio assisted with icListen deployments, along with the BIOL 5551 (Ichthyology) students at ECU. Grant support was obtained from the East Carolina University Office of Research (REDE), ECU Technology Fund, the National Science Foundation, and the North Carolina Division of Marine Fisheries.

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# Underwater Acoustic Sound Propagations: Climatological Characteristics of Seasonal, Mesoscale, and Submesoscale

22

Sung Yong Kim

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**Abstract**

The underwater acoustic coastal environment is presented in views of seasonal climatology and potential interactions of sound waves with mesoscale and submesoscale processes are examined. The sound speed profiles (SSPs) are computed from two-/three-dimensional underwater acoustic propagation models and observed conductivity-temperature-depth profiles, and submesoscale realistic numerical model outputs. The density profiles, multivariate regressed with a temporal mean, seasonal basis functions, and a linear trend are translated into the propagation and dissipation of the SSP. Moreover, the density profiles in the cross-shore direction are decomposed into a constant density profile in the cross-shore direction, and density profiles having mesoscale and submesoscale features using a band-pass filter in the frequency domain and wavenumber domain. In this chapter, the analysis method to evaluate the influence of submesoscale and mesoscale oceanic variability on the sound propagation is only presented.

**Keywords**

Sound · Mesoscale · Submesoscale · Underwater · Attenuation

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## Introduction

Acoustic waves propagate through the compression and decompression of the medium, and propagation speed depends on the bulk modulus and density of the medium (e.g., Newton-Laplace equation) because wave propagation is directly proportional to stiffness (elasticity) and inversely proportional to the mass of inertia. In particular, the acoustic waves in water have been to study acoustic tomography, marine mammals, underwater communications, defense, and security (e.g., Forget et al. (2015), Johnson and Tyack (2003), and Munk et al. (2009)).

This chapter aims to examine the variability of underwater acoustic environments under seasonal density profiles and the modified acoustic wave propagations associated with interactions between acoustic waves and meso-/submeso-scale eddies, fronts, and internal tides in a coastal environment (e.g., Yoo et al. (2018)). Thus, the climatology of density vertical transects (e.g., time mean, amplitudes and phases at the annual, and semiannual frequencies) from Lee et al. (2019) are adopted and the two-dimensional wave propagation model to derive the underwater acoustic environments is applied. Then, the acoustic wave propagations under scenarios of meso-/submeso-scale eddies, fronts, and internal tides are examined. There are potential deflections (or reflection) of sound speed due to vertically modulated thermoclines associated with temporal variability of three-dimensional density structures in the intra-annual (i.e., seasonal superharmonic frequencies), seasonal, and inter-annual in inter-annual timescales. This work can provide the basic information on the acoustical ocean environments of underwater communication of subsurface oceanographic samplings and explorations (e.g., Send et al. (2012)).

## Methods

The sound speed profiles (SSP) in the cross-shore direction are decomposed into a constant SSP in the cross-shore direction and SSPs having mesoscale and submesoscale features:

$$c(r,z,t) = c_C(z) + c_M(r,z,t;|\sigma| \le \sigma_0) + c_S(r,z,t;|\sigma| > \sigma_0)$$

where a constant SSP ($c_C$) is computed with an average of the sound speed (SS) in time and cross-shore direction at each depth. The mesoscale SSPs ($c_M$) are estimated with first few modes of the spatial time series of SSPs longer than 10 days ($\sigma_0 = 0.1$ cycles per day (cpd)) in the depth and cross-shore direction. A low-pass filter ($|\sigma| \le \sigma_0$) can be applied to the demeaned spatial SSPs first, then the singular value decomposition (SVD) on the spatial time series of SSPs is applied to remove the submesoscale spatial structures. The first few modes accounting for 90% of the sum of entire singular values are only used. The submesoscale SSPs ($c_S$) are estimated with the rest of SSP by subtracting the constant SSP and mesoscale SSPs. Although statistical analysis using SVD may not isolate the dynamics of mesoscale and submesocale or the physical processes in their representing spatial

length scales [$O(100)$ km for mesoscale and $O(1–10)$ km for submesoscale], the low-pass filter takes care of most of decomposition.

As another way to isolate the mesoscale SSPs, a wavenumber-domain band-pass filter of the SSPs with a constant SSP removed at each depth can be applied:

$$c(r,z) = c_C(z) + c_M(r,z) + c_S(r,z)$$

where

$$c_M(r,z) = \sum_{0<k<k_0} \widehat{c}(k,z)e^{ikr}$$

$$c_S(r,z) = \sum_{k\geq k_0} \widehat{c}(k,z)e^{ikr}$$

and $k_0$ denotes a threshold wavenumber to delineate the mesoscale and submesoscale features. The horizontal wavenumber ($k$) is defined as the cross-shore distance at each depth,

$$k(z) = 2\pi\frac{m}{L_z}$$

which may not be identical for all cases because of the nonconstant continental shelf and bottom bathymetry ($m = 1, 2, \ldots M$). Thus, the wavenumber-domain can be defined differently at each depth and may not filter the data in a consistent way. This can lead the vertically discontinued or mismatched SS and generate unnecessary features of SS propagations.

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# Natural Ambient Sounds as Sources of Biologically Relevant Information and Noise for Fishes

23

Sujay Balebail and Joseph A. Sisneros

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**Abstract**

In the field of fish bioacoustics, sounds other than conspecific communication sounds have generally been viewed as background noise interfering with the detection of communication signals. However, many fish species do not vocalize but still possess acute hearing, suggesting that fish hearing may have evolved, in part, to detect and extract information from the "ambient sounds" present in the environment. This chapter highlights some of the major sources of natural ambient sound and discusses how they function as sources of biologically relevant information. Some fishes can extract information about larval settlement sites, spawning events, food sources, and approaching predators from natural ambient sound. Instances where natural ambient sounds function as noise are also discussed. Ambient sound can mask hearing and some vocalizing fishes have been known to either produce louder vocalizations or change vocalization rates to cope with increases in ambient sound levels. Several species living in shallow water habitats vocalize at frequencies that coincide with a dip in acoustic energy in the local ambient sound spectrum. Data on the behavioral and physiological responses of fishes to natural ambient sounds remains scarce. Such information could be important for understanding how anthropogenic sounds affect the survival and reproduction of fishes.

**Keywords**

Soundscape · Masking · Auditory scene analysis · Cues · Anthropogenic sound · Anthropogenic noise · Acoustic daylight · Environmental sound

## Introduction

The International Organization for Standardization, ISO 18405:2017 document defines ambient sound as "sound that would be present in the absence of a specified activity." "Specified activities" include the act of measuring sound, and the radiation of sound from specified sound sources. Ambient sound according to ISO is thus defined based on the perspective of a human experimenter measuring sound, where ambient sound represents the "background sound" from sources other than the one being measured.

Animals produce communication signals to convey information from one animal (sender) to another (receiver) in order to affect the behavior of the recipient (Seyfarth and Cheney 2003). A "signal" in the context of animal communication is defined as an action or behavior produced by a sender that has evolved to convey information and help the receiver make a decision, which, on average, is beneficial to both sender and receiver (Myrberg 1981; Bradbury and Vehrencamp 2011). In contrast, "cues"

also provide information to a receiver, but the content has not been shaped by natural selection in order to benefit both sender and receiver (Bradbury and Vehrencamp 2011). Since it is well established that receivers detect and interpret conspecific acoustic communication signals, any sound in the environment apart from conspecific acoustic signals can be considered as ambient or background sound and a potential source of cues for an animal. In this chapter, ambient sound is thus defined from an animal's perspective as "all the sounds impinging on the animal in a given environment apart from conspecific acoustic communication signals."

Underwater environments are bathed in ambient sound (Hildebrand 2009; Rountree et al. 2020). Before the invention of sound-producing steam ships in the early 1800s (Harris 2022), ambient sound in aquatic environments was predominantly composed of sounds generated by biotic sources such as invertebrates, fishes, and marine mammals (biophony), and abiotic sources such as wind, rainfall, and other environmental phenomena (geophony). Despite recent increases in anthropogenic sound in many aquatic environments, natural sounds still constitute a major component of the underwater ambient sound spectrum (Hildebrand 2009; Frisk 2012).

Actinopterygian or "ray-finned" fishes are the planet's largest and most diverse group of vertebrates (Nelson et al. 2016). At least a thousand species of ray-finned fishes (henceforth called "fishes" for brevity) have been known to actively produce sounds for communication (Looby et al. 2022), and there are likely many more vocal species. The "matched filter hypothesis" proposed by Capranica and Moffat (1983) maintains that the auditory systems of animals are tuned to detect the frequencies present in conspecific acoustic communication signals. According to this hypothesis, any sound other than that produced by a conspecific interferes with the detection of conspecific signals. However, the matched filter hypothesis does not explain why nonvocal fishes have specific tuning characteristics associated with their auditory system. The list of nonvocalizing species includes some species from families such as Cyprinidae (Jacobs and Tavolga 1967) and Mormyridae (Fletcher and Crawford 2001), which possess accessory hearing structures that improve auditory sensitivity and increase the bandwidth of hearing. Additionally, sound production is not an ancestral trait in fishes, evolving independently several times across multiple lineages (Rice et al. 2022). Therefore, since all fishes can hear (Popper et al. 2022), hearing in fishes likely evolved to detect and extract information from ambient sound in the environment.

Buckingham (1999) proposed the concept of "acoustic daylight" wherein similar to how cameras use daylight scattered from objects to form an image of the environment, ambient sound could theoretically be exploited to acoustically image the aquatic environment. Fay (2009) developed this concept in the context of fish hearing and proposed that fishes could potentially use ambient sound as acoustic daylight, extracting information about the surrounding environment. In other words, natural ambient sounds can function as "cues." Thus, natural ambient sounds can potentially provide important behaviorally relevant information that can inform the receiver and guide a particular behavior.

In certain situations, however, natural sources of ambient sound can act as noise, masking the detection of environmental sounds important to the receiver for survival and reproduction such as mating vocalizations, sounds made by approaching predators, etc. *Here, noise is defined as any sound which interferes with the detection and interpretation of biologically relevant signals and cues important for the animal.*

This chapter highlights the major natural sources of ambient sound in the context of how such sources could provide useful information to fishes. Examples from literature on how fishes use natural ambient sounds to extract beneficial information about the environment are summarized. Cases where natural ambient sounds function as noise, interfering with the detection of relevant signals and environmental cues, are also discussed. The chapter ends with a discussion of information gaps in our understanding of how natural ambient sounds affect the physiology and behavior of fishes, and how uncovering this information could provide a more nuanced understanding of the impacts of anthropogenic sounds on fishes.

## Major Sources of Natural Ambient Sounds and their Putative Biological Relevance

Natural ambient sounds can be generated by abiotic sources (geophony) and by animals (biophony). In this section, only those sources of natural ambient sound that fall within the hearing ranges of fishes will be discussed. Fishes are most sensitive to sounds below 1000 Hz, although certain fishes with hearing specializations such as ostariophysians can detect sounds up to 3 kHz (Ladich and Fay 2013), and some alosids can detect ultrasonic frequencies (30–150 kHz) (Plachta and Popper 2003). Prominent sound sources that have been omitted in the section are sounds generated by rainfall, which occur between 10–40 kHz (Scrimger et al. 1987), and sounds generated by the thermal agitation of water molecules (>50 kHz) (Hildebrand 2009). Most studies characterizing natural sources of ambient sound measure sound pressure, whereas fish ears are thought to primarily function as detectors of particle motion (Popper and Hawkins 2018). Therefore, using sound pressure data to infer how fishes extract information from natural ambient sounds likely has limitations.

### Geophony

Geophony that is within the audible range of fishes is primarily generated by: 1) weather events such as wind, 2) geological phenomena such as earthquakes, volcanoes, and release of water and smoke from hydrothermal vents, 3) sea ice, and 4) flowing and falling water (Table 1). To the best of the authors' knowledge, no data exists on the behavioral responses of fishes to geophony, and whether fishes extract information from these sound sources.

**Table 1** Major sources of geophony and hypothesized biologically relevant information contained in them. See the main text for more details

| Sound source | | Frequency content | Hypothesized biologically relevant information contained in these sounds |
|---|---|---|---|
| 1. | Wind-generated waves | ~ 0.5–50 kHz (Hildebrand 2009) | Depth along the water column; and distance from the shoreline. |
| 2. | Flowing water (rivers and brooks) | ~ 10–100 Hz (Vračar and Mijić 2011; Martin and Popper 2016) | Rate of water flow; identity of different habitats such as river stretches and branch points (Velde et al. 2022). |
| 3. | Waterfalls | < 1000 Hz (Lugli and Fine 2003)[a] | Location of jumping sites for migratory fishes such as salmon. |
| 4. | Sea ice | ~ 100–1000 Hz (Miksis-Olds et al. 2013) | – |
| 5. | Underwater earthquakes | ~ 5–35 Hz (Butler 2006) | Occurrence of earthquakes, which have been known to cause mortality in fishes. |
| 6. | Hydrothermal vents | ~ 10–250 Hz (Crone et al. 2006) | Location of hydrothermal vents. |
| 7. | Underwater volcanoes | < ~600 Hz (Chadwick Jr. et al. 2008) | Occurrence of volcanic eruptions, which have been known to cause mortality in fishes. |

[a]The authors characterized acoustic frequencies only up to 1000 Hz;, therefore it is possible that acoustic energy is contained at frequencies >1000 Hz

## Wind-Generated Waves

Breaking waves generated at the water's surface by winds are the primary source of geophony in the oceans, with a frequency range of ~0.5–50 kHz (Hildebrand 2009). There is a strong correlation between windspeed and sound pressure levels in marine environments, even at depths of 1000–2000 m (Kurahashi and Gratta 2008). The cues used by fishes to estimate water depth are not clearly understood, although recent research suggests that hydrostatic pressure may be an important cue (Davis et al. 2021). Since wind-generated sound decays with depth (Short 2005), the presence or absence of this sound might provide fishes with a rough estimate of their vertical position along the water column. This hypothesis requires that fishes possess an internal model of how ambient sound decays with depth. Alternatively, fishes could learn to associate the loudness and spectral content of ambient sound conditions with water depth. No data currently exists, however, to support this hypothesis.

Bardyshev et al. (1973) recorded ambient sound at various distances from the shoreline and found that the ambient sound was more pulsatile closer to the shore, with large increases and decreases in the amplitude of acoustic pressure. The distribution of pressure amplitude significantly differed from a gaussian distribution up to 600 m from the shore. Beyond this distance, the distribution of pressure amplitude of ambient sound resembled that of gaussian white noise, indistinguishable from ambient sound in the open ocean. Therefore, ambient sound from breaking waves could also potentially inform coastal fishes about their distance from the

shoreline. The exact nature of how ambient sound profiles change from the shore to open ocean will depend on conditions such as the slope of the seabed, the nature of the substrate, etc.

### Flowing Water (Rivers and Brooks)

Acoustic energy in rivers appears to be primarily distributed between 10–100 Hz (Vračar and Mijić 2011; Martin and Popper 2016), which falls within the range of hearing of most fishes. Vračar and Mijić (2011) found that, on average, sound pressure levels were higher in faster flowing rivers compared to two slow flowing rivers, suggesting that sound levels tend to increase with higher flow rates. Therefore, in addition to the lateral line system of fishes, which is known to sense flowing water and is important for rheotaxis (Montgomery et al. 2013), fishes might be able to obtain information about water flow rates from the loudness of the ambient sound environment.

A recent study revealed that the spectral signatures of large slow flowing rivers, small slow flowing rivers, and brooks were consistently different in sites across the Netherlands, suggesting that fishes may be able discriminate between these habitats using spectral cues (Velde et al. 2022). The study also revealed that frequency spectra differed between river stretches and branch points, suggesting that acoustic information could potentially be used by migrating fishes to locate branch points along rivers (Velde et al. 2022). Therefore, acoustic characteristics can potentially inform fishes about different habitats in flowing water.

### Waterfalls

Few studies have characterized the ambient sounds produced by waterfalls. Lugli and Fine (2003) measured low frequency sounds (0–1000 Hz) generated by waterfalls in a shallow freshwater habitat (<70 cm deep). Sound pressure levels decayed by 30 dB re 1μPa within two meters from the waterfall, suggesting that waterfalls can only be detected from close range in shallow water environments. Sounds emanating from waterfalls might be especially important for migrating fishes such as salmon, which are known to jump up waterfalls. Migrating sockeye salmon appear to have preferred locations to jump up waterfalls, which is dependent on the depth of the pool below the falls and the height of the falls (Lauritzen et al. 2005). Along with visual and hydrodynamic cues, the acoustic signature of the pool, especially the particle motion characteristics, might be important for selecting suitable locations by particle motion-sensitive salmon for jumping up waterfalls. Studies that measure the acoustic characteristics of jumping and non-jumping sites need to be carried out to determine if the ambient sound is consistently different in these two environments, which could potentially be used by migrating salmon to find suitable jumping sites.

### Sea Ice

In Antarctic and Artic waters, ambient sounds are generated by a combination of geophysical phenomenon such as thermal cracking of ice, glacial calving, and glacial waterfalls (Tegowski et al. 2011). Ambient sounds in habitats occupied by sea ice have been shown to be highly dependent on the amount of ice cover in the area, with

maximum acoustic energy in the 100–1000 Hz spectral band (Miksis-Olds et al. 2013). To the best of the authors' knowledge, no studies have been conducted to determine whether sounds emanating from sea ice provide cues about relevant environmental variables such as the location of feeding and breeding sites to fishes.

### Underwater Earthquakes

Earthquakes generate underwater acoustic waves called T waves, which can propagate for thousands of kilometers (Fox et al. 2001). At the seabed, T waves travel as seismoacoustic polarized interface waves (known as *Ti* waves). Frequencies below ~5 Hz propagate along the sediment bed and frequencies above ~5 Hz propagate as acoustic waves in the water above the seabed. The acoustic aspect of the *Ti* waves contain energy primarily between 5 Hz and 35 Hz (Butler 2006), which falls withing the hearing range of fishes (Sand and Karlsen 1986). Frohlich and Buskirk (1980) calculated the theoretical sound pressure levels of sounds generated by earthquakes and found that the levels are generally above the hearing thresholds of grunts (*Haernulon* species) and Atlantic cod (*Gadus morhua*). This suggests that these, and other fish species, can potentially detect underwater earthquakes. Earthquakes can cause underwater landslides, which can kill fishes and destroy habitat (Zimmerman et al. 2008). Thus, the ability to detect the sounds preceding earthquakes may potentially be adaptive to fishes, allowing them to escape the area before the earthquake strikes. More research is needed to confirm whether fishes detect and behaviorally respond to the low frequency sound waves generated by earthquakes.

### Hydrothermal Vents

Hydrothermal vents produce sounds with acoustic energy primarily distributed between ~0.5 kHz and 3 kHz (Italiano et al. 2011), although majority of the energy appears to be located between ~10 Hz and 250 Hz (Crone et al. 2006). Crone et al. (2006) also found that sound pressure levels in the 0–500 Hz band measured 0.1–0.3 m from the opening of a hydrothermal vent were ~ 10–30 dB re 1μPa louder than the ambient sound in the surrounding water. The authors of this study estimated that sound pressure levels of sounds generated by vents can rise above background sound pressure levels up to ~5–15 m away from the vent opening. They suggest that hydrothermal vent sounds might function as a cue informing marine organisms about the presence of vents. Most vents are greater than 2 km deep (Baker and German 2004). Therefore, deep sea fishes may use sounds emanating from hydrothermal vents to either avoid hot fumes that are extruded from the vents or to locate prey that live close to vents. Further studies are required to explore this question.

### Underwater Volcanoes

Sounds generated from underwater volcanoes are variable in nature. Some eruptions can create harmonic sounds, containing energies at integral multiples of a fundamental frequency. Other eruptions can produce more broadband sounds with acoustic energy distributed across a range of frequencies. Acoustic energy is primarily of low frequency (<~600 Hz) and explosive volcanic bursts can increase local ambient sound pressure levels by ~50 dB re 1μPa, between 10 Hz and 50 Hz (Chadwick

Jr. et al. 2008). Little is known about the responses of fishes to sounds generated by volcanoes. Like earthquakes, volcanic eruptions can cause fish mortality (Caballero et al. 2023). Thus, it might be adaptive for fishes to detect sounds produced by volcanic eruptions, but such a hypothesis remains to be investigated further.

## Biophony

The primary biotic sources of natural ambient sound are invertebrates such as snapping shrimp and hard-shelled invertebrates, fishes, and marine mammals (Table 2). Compared to sources of geophony, more data exists on how fishes extract

**Table 2** Major sources of biophony, evidence for fishes extracting biologically relevant information from these sources of biophony, and additional hypothesized biologically relevant information contained in these sounds. See the main text for more details

| Sound source | | Frequency content | Examples of fishes extracting biologically relevant information | Additional hypothesized biologically-relevant information |
|---|---|---|---|---|
| 1. | **Invertebrates** | ~ 0.04–115 kHz (Duarte et al. 2021) | | |
| a. | Sound produced by snapping shrimp | | Locating coral reefs habitats (Larval fishes) (Leis et al. 2003) | – |
| b. | Sea urchin feeding sounds | | – | Localizing reef habitats (Radford et al. 2008) |
| 2. | **Fishes** | ~ 0.01–5 kHz (Duarte et al. 2021) | | |
| a. | Vocalizations | | Location of coral reefs habitats (Larval fishes) (Tolimieri et al. 2000). | – |
| b. | Passive feeding sounds | | Location of food (Wang et al. 2022). | – |
| c. | Low frequency sounds produced by struggling prey | | – | Location of prey (Richard 1968) |
| d. | Low frequency sounds produced by approaching predators | | – | Presence of predators (Sand et al. 2000; Bui et al. 2013) |
| 3. | **Marine mammals** | ~ 0.01–340 kHz (Duarte et al. 2021) | | |
| a. | Dolphin vocalizations | | Detecting the presence of predatory dolphins (Remage-Healey et al. 2006; Luczkovich et al. 2000). | – |

biologically relevant information from biophony. However, studies characterizing the particle motion aspect of various sources of biophony are still scarce.

### Invertebrates

Many invertebrate species either actively vocalize or produce sounds passively while moving or eating. Sounds generated by invertebrates encompass a wide frequency band, ranging from ~0.04 kHz to 115 kHz (Duarte et al. 2021). Snapping shrimps are usually the dominant biotic source of ambient sound in shallow tropical and subtropical marine environments. They produce broadband clicks with frequencies ranging primarily between 2 kHz and 20 kHz, with peak frequencies falling from 2 kHz to 5 kHz. Snapping shrimp sounds can increase local sound pressure levels by 20 dB re 1μPa, and habitats occupied by them can be as loud as 140–150 dB re 1μPa (Cato and Bell, 1992). However, since the frequencies of best hearing are usually below 1000 Hz in most fishes (Popper et al. 2019), loud sounds made by snapping shrimp are unlikely to cause hearing damage or negatively affect fish behavior. In fact, playbacks of reef sounds dominated by snapping shrimp vocalizations have been demonstrated to attract fish larvae, suggesting that snapping shrimp sounds are important cues for reef localization for larvae of coral-reef fishes (Leis et al. 2003).

Grating sounds made due to the movement and feeding of hard-shelled invertebrates can also contribute to local ambient sound. Sea urchin feeding creates sounds with acoustic energy between 800 Hz and 2800 Hz, in the coastal waters of New Zealand (Radford et al. 2008). These sounds fall within the hearing range of fishes and could possibly be used by predatory fishes to locate urchins and other prey items associated with reefs. Thus, sounds produced by invertebrates could potentially be acoustic cues for locating important habitats such as coral reefs. Whether fishes use sounds from invertebrates for other purposes remains to be further investigated.

### Fishes

Fish choruses are a major source of ambient sound in shallow habitats such as coral reefs (McWilliam et al. 2017), and intertidal habitats (Halliday et al. 2018). The collective ambient sound from chorusing fishes could potentially be used by heterospecific fishes to locate habitats such as reefs and tidal estuaries. Fish vocalizations are generally major components of the playbacks of reef sounds in studies that demonstrate the attraction of fish larvae to reef sounds (e.g., Tolimieri et al. 2004), suggesting that in coral reef environments, fish sounds can serve as important spatial cues to locate settlement sites on reefs.

Besides vocalizations, fishes also extract information about the environment from the sounds produced passively by other fishes while eating and moving. Sounds produced during feeding can inform conspecifics about the presence of food (Wang et al. 2021). Some fishes are repelled by infrasound (<20 Hz) (Sand et al. 2000; Bui et al. 2013), which is thought to be a predator avoidance response since large swimming fishes can generate infrasound (Bleckmann et al. 1991). In contrast, predatory fishes are often attracted to low frequency pulsatile sounds (Richard 1968), which are often associated with wounded or struggling prey. Thus, sounds

produced passively by fishes could inform other fishes about the presence of food, predators, and prey.

### Mammals

Various marine mammal groups such as cetaceans, phocids, and otariids produce underwater vocalizations in the range of 10 Hz to approximately 340 kHz depending on the species (Duarte et al. 2021). Some fishes have evolved behavioral responses to the sounds produced by predatory marine mammals, which likely reduces predation risk. Vocal activity in the Gulf toadfish (*Opsanus beta*) (Remage-Healey et al. 2006) and silver perch (Luczkovich et al. 2000) is greatly reduced when exposed to playbacks of dolphin vocalizations, but not other control sounds. Furthermore, certain members of the order Clupeiformes such as the American shad (*Alosa sapidosomma*) can detect ultrasound and at relatively high intensities can elicit an acoustic startle response by these fishes. Such a response is believed to be an antipredator adaptation to avoid echolocating odontocetes (Plachta and Popper 2003). Thus, there is evidence that fishes can obtain information about the presence of predatory mammals from their vocalizations.

## Natural Ambient Sounds as Sources of Biologically Relevant Information

Few studies have been conducted on whether fishes use ambient natural sounds to extract information useful for survival and reproduction. The goldfish (*Carassius auratus*) has been shown to conduct "auditory scene analysis," which is the ability to identify and segregate the multiples sources of sound that the animal experiences at a given moment This suggests that the goldfish and possibly other fish species may be able to identify and segregate the various sources of sound in the ambient sound spectrum and extract useful information from them. Known cases where fishes extract information about the environment from ambient sound are described in the following sections (summarized in Table 3).

### Location of Larval Settlement Sites

Fish larvae have been shown to be attracted to playbacks of sounds recorded from coral reefs, suggesting that fish larvae use sounds emanating from coral reefs to locate them and choose settlement sites (Montgomery et al. 2006). There is variability among taxa in the spectral components of the ambient sound spectrum preferred for settlement. Most species are preferentially attracted to the component of coral reef ambient sound spectrum primarily generated by invertebrates (>570 Hz), but other fish species demonstrate no preference for the high or low frequency components of the coral reef sound spectrum (Simpson et al. 2005, 2008).

The acoustic characteristics of different coral reef ecosystems appear to show consistent differences, which could potentially be exploited by coral fish larvae to

**Table 3** List of examples of fishes extracting biologically relevant information from natural ambient sounds. Examples where the responses of fishes to acoustic stimuli resembling certain natural ambient sounds suggest that they have the capability to extract biologically relevant information from these natural ambient sounds have been included under "Putative examples." See the main text for more details

| Types of biologically relevant information extracted | | Examples | Putative examples |
|---|---|---|---|
| 1. | Location of larval settlement sites | • Larval fishes attracted to recordings of ambient reef sounds (Simpson et al. 2005, 2008; Montgomery et al. 2006).<br>• Larval fishes attracted to sounds from specific coral reef habitats where they normally settle in but not sounds from other coral reef habitats (Radford et al. 2011).<br>• Some larval species avoid sounds from habitats they do not generally settle in (Radford et al. 2011). | – |
| 2. | Location of spawning conspecifics | Playbacks of sounds made by chubs shaking their tails while spawning attracted conspecifics (Wang et al. 2022) | – |
| 3. | Detection of food sources | Flower fish attracted to playbacks of worm feeding sounds (Wang et al. 2021) | • Cyprinids attracted to playbacks of sounds made while pebbles are dislodged, which may uncover hidden invertebrate prey (Holt and Johnston 2011).<br>• Some predatory fishes attracted to low frequency pulsed sounds, thought to be produced by swimming or struggling prey (Richard 1968). |
| 4. | Detection of predators | Some fishes reduce vocalization rates during playbacks of predatory dolphin vocalizations (Luczkovich et al. 2000; Remage-Healey et al. 2006) | • Certain fishes are repelled by infrasound (<20 Hz), thought to be generated by swimming movements of approaching predators (Sand et al. 2000).<br>• Ultrasound evokes a startle responses in some Clupeiform fishes, which is likely an adaptation to escape predatory odontocetes (Plachta and Popper 2003). |
| 5. | Detection of fishes possessing swim bladders | – | Goldfish (*Carassius auratus*) conditioned to detect sounds resembling scattered ambient sounds by fish swim bladders (Lewis 1994) |

locate preferred habitats (Radford et al. 2010). In fact, larvae of some fish species are known to demonstrate strong preferences for sounds emanating from specific coral reef environments such as lagoons or fringing reefs, suggesting that larvae not only obtain directional information from reef sounds but also use these sounds for habitat identification (Radford et al. 2011). Parmentier et al. (2015) studied the responses of coral fish larvae of 20 fish species belonging to 10 families to ambient sound recorded from different coral reef habitats. The authors observed interspecific differences in larval attraction to playbacks of sounds recorded from the different habitats. Some species were also repulsed by sounds emanating from certain habitats. Thus, in addition to locating favorable environments, coral reef larvae also appear to be able to use ambient sound to avoid settling in specific habitats.

The acoustic environment can vary locally in a coral reef. Kennedy et al. (2010) found that the sound environment was correlated with factors such as the local density and biomass of fishes, as well as the diversity of coral and fish species. It is possible that larval fishes select different microhabitats for settlement based on their acoustic profiles, and further research on this topic is warranted.

The specific acoustic cues used to locate the sources of reef sounds remain poorly understood. Coral fish larvae could be using particle motion vectors as cues to guide them to the reef, similar to how adult female plainfin midshipman (*Porichthys notatus*) localize mating calls produced by males (Zeddies et al. 2010). In Zeddies et al. (2010), female plainfin midshipman localized a sound source ~1 m away from the animal. It has been speculated that larval fishes can localize reefs located at distances of up to 1 km from the animal (Mann et al. 2007). It is unknown if acoustic particle motion fields around coral reefs provide directional information about the location of reefs, which fish larvae could utilize to locate the reef from large distances. Alternatively, larval fishes could be climbing up intensity gradients in sound pressure, or particle motion variables (displacement, velocity, or acceleration; see Sisneros and Rogers (2016)). More research is required on the acoustic cues used by larval fishes, and how acoustic cues could be combined with other cues such as chemical cues (Atema et al. 2002) to locate reefs.

## Location of Spawning Conspecifics

In several freshwater fish species, which spawn in riverbeds, both males and females are known to vigorously shake their tails while mating. This tail-shaking behavior displaces sand and gravel, and helps bury the eggs (Katano 1992b; Chuang et al. 2006). Playbacks of recorded "collision" sounds made by two species of chubs (*Opsariichthys evolans* and *Zacco platypus*) while shaking their tails during spawning, attracted more chubs of both species compared to control speakers either playing music or recorded ambient sound (Wang et al. 2022). Chubs attracted to spawning events generally tend to be satellite males or conspecifics, which cannibalize eggs (Katano 1992a, b). Thus, the presence of collision sounds in the ambient

sound environment appears to provide satellite males and conspecific egg-eating chubs with information about the location of spawning pairs. Future studies should investigate if the attraction to collision sounds is present in other freshwater species which spawn in riverbeds.

## Detection of Food Sources

Sounds produced during feeding can inform conspecifics about the presence of food. A few species in the family Cyprinidae are known to be attracted to playbacks of sounds that are associated with the dislodgement of pebbles during feeding in their native shallow freshwater habitat (Holt and Johnston 2011). Holt and Johnston (2011) suggest that since dislodged pebbles tend to expose hidden invertebrate prey items, the attraction of cyprinid fishes to sounds produced by pebble movement might be an adaptative or learned response for finding prey. Playbacks of sounds produced by flower fish feeding on worms attracted conspecifics (Wang et al. 2021) demonstrating that this species also obtains information about the presence of food from feeding sounds. The ability to locate food by listening to the feeding sounds made by conspecifics/heterospecifics may be present in other species and requires further investigation.

Bleckmann et al. (1991) measured the frequency content of sounds generated by the movement of five fish species: convict cichlid *Cichlasoma nigrofasciatum*, black ghost knifefish *Apteronotus albifrons*, Mexican tetra *Astyanax fasciatus mexican*us, striped panchax *Aplocheilus lineatus*, and rainbow trout *Salmo gairdneri*, when moving freely or startled. The frequencies of sounds produced when these fishes were tethered and struggling to escape the tether were also recorded. A fish struggling to escape from a tether resembles a naturalistic situation when a fish struggles to escape the jaws of a predator. Sounds produced by moving, tethered, and struggling fishes were primarily below 100 Hz. Predatory reef fishes such as groupers (*Epinephelus striatus, Mycteroperca venenosa,* and *Mycteroperca bonaci*), snappers (*Lutjanus analis, and Ocyurus chrysurus*), and the white margate (*Haemulon album*) have been shown to be attracted to playbacks of low frequency (25–50 Hz) pulsatile sounds (Richard 1968). The attraction of predatory reef fishes to these low frequency sounds suggests that predatory fishes can possibly detect and localize the sounds produced by swimming prey, as well as prey that are struggling to escape capture by a predator. Further studies are required to test this hypothesis.

## Detection of Predators

Infrasound (<20 Hz) has been known to trigger an avoidance response in juvenile and adult salmonids, and eels (Sand et al. 2000; Bui et al. 2013). This repulsion of some fish species to infrasound has been proposed to be an adaptation to avoid

predators, as approaching predators likely produce low frequency hydrodynamic sounds while swimming (Bleckmann et al. 1991). The avoidance response of species such as salmon to infrasound occur only when the fish are close to the speakers (< 2 m) (Knudsen et al. 1992), suggesting that animals only detect infrasound from sources a few body lengths away. Therefore, a sudden increase in low frequency sounds in the ambient sound environment could inform smaller prey fishes that a large animal is in close proximity to the individual.

Some vocalizing fish species appear to detect predatory dolphin vocalizations and behaviorally respond by reducing vocal activity, which presumably reduces their chances of being detected by dolphins in the natural environment (Luczkovich et al. 2000; Remage-Healey et al. 2006). Certain species of Clupeiform fishes can detect ultrasound, which is believed to be an adaptation to detect the sounds generated by predatory odontocetes during echolocation (Plachta and Popper 2003). Thus, some fishes have the ability to acquire information about the presence of predators in the environment by detecting their vocalizations in the ambient sound environment.

### Detection of Fishes Possessing a Swim Bladder

Goldfish may be able to sense the presence of other individuals by detecting ambient sound scattered by the swim bladder of a nearby fish. Lewis (1994) simultaneously exposed tethered goldfish to white noise played via a speaker, as well as sound resembling scattered white noise from a fish's swim bladder, generated by a piezoelectric transducer. This experimental setup mimics a situation where another fish with a swim bladder (which scatters sound) is close to the "receiver" goldfish, while the "receiver" is experiencing white noise. Goldfish can learn to recognize the scattered white noise. Thus, the presence of ambient sound may help goldfish and possibly other fishes sense the presence of nearby fishes that possess a swim bladder, or other gas-filled chambers, especially when the fish is in a shoal of fishes that possess swim bladders. It should be noted that in the conditioning experiments carried out by Lewis (1994), the speaker relaying white noise was close to the goldfish (distance to goldfish = 36 cm) and therefore the white noise experienced by the animal was likely very different from ambient sound in a natural setting. The sound levels of ambient sound scattered by swim bladders is likely to be very low in natural environments, and it remains to be seen if goldfish and other fishes could detect the ambient sound scattered by swim bladders in natural settings.

## Natural Ambient Sounds as Noise

Not all natural ambient sounds are relevant to fish survival and reproduction and such sounds are likely to be species-specific and context-specific. Below, we document cases where natural sounds have been shown to function as noise, interfering with the detection of relevant acoustic signals and cues (Table 4).

**Table 4** List of examples where natural ambient sounds function as noise. Examples where the responses of fishes to nonnatural sounds suggest that increased levels of natural ambient sound could function as noise have been included under "Putative examples." Also included under the putative examples column are cases where fishes change vocal activity in responses to events, which are known to increase ambient sound levels (such as increases in wind speeds and storms). See the main text for more details

| The ways in which natural ambient sounds function as noise | | Examples | Putative examples |
|---|---|---|---|
| 1. | Masking | • Hearing thresholds of cods (*Gadus morhua*) for pure tones increased with increased levels of natural ambient sound (Chapman and Hawkins 1973).<br>• Auditory sensitivity to pure tones masked by playbacks of recorded ambient natural sound (Ladich 2013). | – |
| 2. | Vocalization frequencies match "quiet windows" in the ambient sound spectrum | Peak frequency of vocalizations of several shallow water fishes lies close to ~100 Hz. This coincides with a dip in acoustic energy at ~100 Hz in the ambient sound spectra in these habitats (Lugli 2010) | – |
| 3. | Increase in loudness of vocalizations during periods of elevated ambient sound levels | – | Blacktail shiners (*Cyprinella venusta*) increased the loudness of vocalizations of vocalizations under playbacks of white noise (Holt and Johnston 2014). |
| 4. | Changes in vocal activity during periods of elevated ambient sound levels | – | • Brown meagre (*Sciaena umbra*) (Picciulin et al. 2012) and oyster toadfish (Luczkovich et al. 2016) increase and decrease their vocal activity respectively during increased levels of anthropogenic sound.<br>• Pomacentrids increased calling activity during increased wind speeds, known to be correlated with ambient sound levels (Munger et al. 2022).<br>• Calling activity of some fishes increased and others decreased during tropical storms (Locascio and Mann 2005; Boyd et al. 2021). |

## Masking

The presence of ambient natural sound can mask the hearing of pure tones in fishes. Chapman and Hawkins (1973) measured behavioral hearing thresholds of the cod to pure tones in the fish's natural marine environment. Hearing thresholds were dependent on the sound pressure levels of ambient sound, with thresholds increasing with increasing sound pressure levels. During the experimental period, ambient sound was primarily generated by natural sources: wind and rain. This is the only study that measured the effect of natural ambient sound on hearing in a fish's native environment.

Other studies have measured the masking effect of ambient sound on the peripheral auditory system of fishes by using the auditory evoked potential (AEP) technique. Experiments have shown that the sound levels required to generate a detectable auditory evoked potential (auditory thresholds) at the level of the peripheral auditory system for pure tones increase when recorded ambient natural sounds are simultaneously played on underwater speakers (For example, see Amoser and Ladich 2005). AEP studies on freshwater fishes have demonstrated that the physiological auditory sensitivity to pure tones is more masked (resulting in increases in auditory thresholds) in species, which possess accessory hearing structures compared to species that lack them (Ladich 2013). In contrast, the physiological auditory sensitivity (i.e., AEP thresholds) to pure tones was minimally masked in fishes that do not have hearing specializations (Vasconcelos et al. 2007; Codarin et al. 2009). Studies suggest that accessory hearing structures tend to improve physiological auditory sensitivity in fishes, especially at higher frequencies ($>$ 1 kHz) (e.g., see Ladich 2000). Thus, it is not surprising that the generally more sensitive auditory system of fishes with accessory hearing structures appear to be more masked by increased levels of ambient sound. These results suggest that fishes with accessory hearing structures might be more susceptible to underwater noise, masking sounds important of survival (such as the sounds produced by approaching predators).

It has been pointed out that AEP techniques likely only measure the physiological auditory sensitivity of the peripheral auditory system (i.e., hair cells and auditory afferents) (Popper and Hawkins 2021). It is possible that even if sounds do not generate a detectable AEP, central processing in the brain may still allow fishes to hear these sounds. It is not known if the increased masking of the peripheral auditory system by ambient sound in fishes with accessory hearing structures translates to increased masking of hearing. More studies of hearing thresholds measured via behavioral methods under increasing levels of ambient underwater sound (such as in Chapman and Hawkins 1973) need to be performed to address this question.

While masking of pure tones by ambient sound is informative, especially for understanding the auditory physiology of the animal, it is difficult to interpret the ecological significance of these experiments. The frequency content of fish communication sounds can either be broadband, with acoustic power distributed across a large frequency range, or harmonic containing acoustic power at integral multiples of a fundamental frequency (Kaatz et al. 2017). Thus, for acoustically communicating species that use harmonic sounds as part of their communication repertoire,

masking studies of pure tones by natural ambient sound could help determine how detection of conspecific communication sounds is impeded under increased levels of ambient sound. It is harder to interpret the results of masking studies for fishes, which do not communicate acoustically, or which produce more broadband communication sounds. More nuanced studies that investigate how the presence of certain kinds of environmental sounds impede the detection of sounds important for survival and reproduction (such as courtship sounds, sounds made by approaching predators, etc.) are required.

## Vocalization Frequencies Match "Quiet Windows" in the Ambient Sound Spectrum

Acoustic communication in several species living in shallow water environments appear to have evolved to increase signal detectability in the presence of local ambient sound. Lugli (2010) characterized the ambient sound spectra of the habitats occupied by nine species of goby. The habitats were classified into four categories: stony stream, spring, brackish lagoon, and rocky seashore. The peak frequency range and bandwidth of the vocalizations produced by each species coincided with the frequencies of the ambient sound spectrum that contained the least amount of energy ("quiet windows") in the fish's native habitat. This suggests that each species has evolved to maximize acoustic signal transmission in its native acoustic environment. Lugli (2010) also showed that the dominant frequencies of vocalizations produced by several fishes living in nearshore marine habitats, stony rivers and creeks, coincided with the quiet windows in the ambient sound spectra in all three habitats. The quiet window was centered around 100 Hz. Lugli (2010) hypothesized that the major reason for several shallow water fish species evolving low-frequency acoustic communication (~100 Hz) may have been to take advantage of this quiet window in ambient sound.

More studies are required to confirm if acoustic communication frequencies of shallow water species have evolved to take advantage of a quite window in the ambient sound environment, and whether such quite windows exist in different shallow water environments. In Lugli (2010), two arbitrary spectra from two specific locations were used to demonstrate that quiet windows in nearshore marine and stony river/creek habitats occur at around 100 Hz. It remains to be seen if these quiet windows occur in other shallow water locations and if they exist near 100 Hz. Additionally, fish ears are thought to primarily function as detectors of particle motion (Popper and Hawkins 2018). The spectral content of sound pressure and particle motion can differ in aquatic environments, especially in shallow water habitats (Banner 1968; Jesus et al. 2020). Since particle motion was not measured in the study by Lugli (2010), it is unclear if the frequency spectrum of the particle motion aspect of the ambient sound environment in shallow water habitats has a dip in energy close to 100 Hz.

The overlap between the dominant frequencies of fish vocalizations and quite windows in the ambient sound spectrum may not be a universal phenomenon. Coers

et al. (2008) measured the ambient sound in the intertidal habitat of the rock-pool blenny (*Parablennius parvicornis*) and found the ambient sound spectrum to be dominated by frequencies below 250 Hz, which overlaps with the communication calls of the goby. Thus, more research is required to determine if frequencies of fish vocalizations coincide with quiet windows in local ambient sound.

## Increase in Loudness of Vocalizations during Periods of Elevated Ambient Sound Levels

Vocal fish species appear to have evolved behavioral adaptations that can potentially increase the transmission and interpretability of communication signals in noisy environments. The involuntary tendency of a sender to increase vocal effort that can include increases in loudness, frequency, rate, and duration of a signal to compensate for increases in ambient sound levels is termed the Lombard effect or Lombard reflex (Summers et al. 1988). The Lombard effect has been documented in some fishes. Blacktail shiners (*Cyprinella venusta*) demonstrated an increase in the sound pressure levels of vocalizations under playbacks of white noise (Holt and Johnston 2014). Oyster toadfish (*Opsanus tau*) increased sound levels by 7 and 9 dB re 1μPa during and after passages of vessels, which temporarily increased the loudness of ambient sound (Luczkovich et al. 2016). Although fishes were exposed to nonnatural sounds in these experiments (white noise and boat sounds), the fact that fishes demonstrate the Lombard effect when exposed to these sounds suggests that the ability of fish to actively increase the sound pressure levels of their signals may have evolved to increase signal transmission under increased levels of natural ambient sound. It remains to be seen whether fishes can modulate the loudness of vocalizations to improve signal transmission when the ambient sound increases in loudness due to natural causes.

## Changes in Vocal Activity during Periods of Elevated Ambient Sound Levels

Data on fish vocal responses to increases in the loudness of ambient sound due to natural causes is scarce. Boyd et al. (2021) documented that during a tropical storm, underwater ambient sound levels due to abiotic sources such as wind increased at frequencies below 500 Hz. The calling rates of some fish species increased during the storm, whereas the calling rates of other species decreased. Other environmental variables such as water level and temperature also changed during the storm, and therefore it is possible that the change in vocal activity could be driven by these other factors rather than the increase in loudness of ambient sound. Similarly, sound pressure levels of a fish chorus increased during and after a hurricane (Locascio and Mann 2005). The authors attribute the increased sound pressure levels to

increased vocal activity. Munger et al. (2022) found a correlation between vocal activity of pomacentrids and wind speeds. Since ambient sound levels are known to increase with wind speed, it is possible that these fish increased vocal activity in response to increased ambient sound levels.

Fishes also appear to either increase (Picciulin et al. 2012) or reduce (Luczkovich et al. 2016) vocal activity in response to increased levels of anthropogenic sound. Thus, some fishes are capable of modulating their vocal activity to cope with increased levels of ambient sound. This increase and decrease in vocal activity has been shown in two different species, brown meagre (*Sciaena umbra*) (Picciulin et al. 2012), and oyster toadfish (Luczkovich et al. 2016), respectively. However, it is not known whether a given species can both increase or decrease their vocal activity in response to increased levels of ambient sound. Reducing vocal activity may help conserve energy for quieter periods when the probability of masking is lower. Alternatively, increasing vocal activity can likely improve signal detectability. The durations and frequencies of ambient sounds impinging on the animal may determine how fishes modify vocal activity, which needs to be investigated further.

## Information Gaps on the Effect of Ambient Sounds on Fishes

This chapter summarizes the major sources of natural ambient sounds in aquatic environments and discusses how they could provide information about the environment to fishes as well as function as noise, interfering with the detection of relevant signals and cues. Existing studies likely represent a small subset of the ways in which natural ambient sounds affect fishes. Some information gaps on the biological relevance of natural ambient sounds to fishes are highlighted below.

### Defining Ambient Sound from an Animal Perspective

Despite the numerous studies that characterize ambient sound, few studies have defined ambient sound from the perspective of animals. One of the few documented definitions of ambient sound from ISO 18405:2017 defines it from the perspective of a human experimenter measuring sound: any sound in the background other than the sound being quantified is considered ambient sound. However, since it is difficult to pinpoint the sounds an animal is focusing on/paying attention to in the sound environment versus the "background" sound that an animal is not focusing on, translating the ISO definition from an animal's perspective is challenging. The only exception to this is likely conspecific communication signals, which are known to be beneficial to the sender as well as the receiver (Bradbury and Vehrencamp 2011), and therefore by definition the receiver is obligated to "pay attention" to these sounds. Therefore, in this review, all sounds impinging on an animal except conspecific communication sounds have been considered ambient

sound. It is necessary to come to a consensus on a working definition of ambient sound from an animal's perspective to limit confusion about what the term "ambient sound" entails across the bioacoustics community.

## Particle Motion Content of Natural Sound Environments

Most studies that characterize ambient sound environments and the responses of fishes to ambient sound measure sound pressure and not particle motion. Particle motion is considered the primary stimulus for hearing in most fishes (Popper and Hawkins 2021). Using sound pressure to characterize sounds can still inform researchers about the temporal structure of sounds, such as, for example, the calling rates of fishes. Sound pressure data can also be used to determine the bandwidth or frequency content of sounds. However, measuring particle motion is still preferable for estimating the frequency bandwidths of ambient sounds in the context of fish hearing because the frequency content of pressure and particle motion can differ slightly in some underwater environments (For example, see Figs. 13 and 15 in Jesus et al. 2020). Hence, to better understand how fishes extract information from natural ambient sound, ambient sounds need to be characterized via particle motion (displacement, velocity, acceleration, etc.) because this is what the fish inner ear is designed to detect.

## The Use of Natural Ambient Sounds as Cues

Little is known about how natural sounds function as sources of biologically relevant information, with coral fish larvae being attracted to reef sounds being the best documented evidence of fishes using natural sounds as cues. However, there is much still to be uncovered even in this domain such as the acoustic cues used by larvae to localize reefs. To determine if a natural ambient sound source functions as a source of biologically relevant information, a correlation needs to be established between the sound and specific environments or situations. A correlation would demonstrate that the sound provides reliable information about the environment or situation, which could potentially be used by fishes. The next step would be to demonstrate a behavioral response to playbacks of these sounds, demonstrating that fishes use these sounds to guide certain behaviors. Natural sounds should be characterized as completely as possible by measuring sound pressure, particle motion (displacement, velocity, or acceleration), and other measures that define the attributes of natural sounds in a given environment such as skew, kurtosis, etc. (reviewed in detail in Wilford et al. 2021)), to determine the acoustic features of ambient sound that may provide biological relevant information about the environment. Understanding how fishes use natural ambient sounds as cues is required to predict how various sources of anthropogenic sounds such as those produced from shipping, air guns, etc., disrupt the detection of these cues and thereby disrupt behavior important for the long-term survival and reproduction of fishes.

### Natural Ambient Sounds as Noise

Natural ambient sounds can also disrupt the detection of important cues and signals. Little is known about how certain natural ambient sounds can mask hearing of other biologically relevant sounds to fishes, and if fishes adjust their behavior to cope with increased levels of natural ambient sound. This information could be important for identifying anthropogenic sounds that are particularly disruptive to fishes, and the capacity of fishes to cope with anthropogenic sound.

### Fish Responses to Sudden Changes in the Natural Ambient Sound Environment

Few data exist on how fish behavior is modified during periods of increased ambient sound due to natural causes such as storms or other weather-related events. Although calling activity of different fish species have been documented to change during storms (e.g., see Locascio and Mann 2005), it is difficult to rule out the effect of other variables in these studies. More studies are needed to determine if increases in natural ambient sound disrupt important behaviors such as locating mates, finding food, escaping from predators, etc.

### Capacity of Fishes to Learn to Associate Ambient Sounds with Environmental Variables

It is well established that fishes can be trained to associate acoustic stimuli with positive rewards (Yan and Popper 1991) as well as aversive stimuli, such as an electric shock (Tavolga and Wodinsky 1963). The ability of fishes to associate acoustic cues with environmental variables suggests that fishes may possess the ability to learn how to extract information from natural ambient sounds that are useful for survival, such as the sounds produced by predators, sounds associated with feeding grounds, etc. More studies are required to further investigate this area of research.

## Conclusion

The matched filter hypothesis by Capranica and Moffat (1983) maintains that the auditory systems of animals such as ray-finned fishes evolved to detect conspecific communication signals. This hypothesis treats any sound besides a conspecific communication sound as part of the background noise, interfering with the detection of communication sounds. This review focuses on cases where these non-communication sounds or "ambient sounds" have been shown to provide useful biologically relevant information to fishes. Fishes are known to extract information about larval settlement sites, location of spawning conspecifics, food sources, and

approaching predators, from natural ambient sounds. In some instances, however, ambient sounds can also function as noise, interfering with the detection of signals and cues, matching the predictions of Capranica and Moffat (1983). Natural ambient sounds can mask hearing, and certain vocalizing species increase their loudness or change their rate of vocalizations to cope with increases in ambient sound levels. Several species living in shallow water habitats vocalize at frequencies that coincide with a dip in acoustic energy in the local ambient sound spectrum, suggesting that these species have adapted to such environments by increasing the signal to noise ratio of communication sounds. The effect of natural ambient sounds on fishes remains an understudied topic. It is important to determine how ambient sounds function as cues to better understand how anthropogenic sounds could potentially disrupt these cues. Additionally, investigating whether natural ambient sounds function as noise could provide insights into the types of anthropogenic sounds that might be most disruptive to fish survival and reproduction.

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# Soundscape in a Northern Patagonian Fjord with Salmon Farms

24

Elba Avilés-Sanhueza and Iván A. Hinojosa

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**Abstract**

In the Patagonian Fjords (Chile), an explosive development of salmon aquaculture has occurred, making Chile the second largest salmon producer in the world. Several environmental impacts of this industry on marine ecosystems have been identified. However, descriptions of the underwater soundscape near the sea-based salmon installations are scarce. A continuous recording of ~45 h was conducted using a calibrated hydrophone (0.02–60 kHz) anchored at 20 m depth (2 m from bottom) and 250 m from a salmon net pen. Thirteen noises were identified (particle motion metrics were not included). The dominant noise had a

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peak frequency of ~80 Hz, probably due to electric generators ($SPL_{RMS}$ ~ 120 dBre1μPa). The second dominant was ~2500 Hz, probably from automatic feeding systems or well boats in use ($SPL_{RMS}$ ~ 142 dBre1μPa). Outboard motor noises were also present ($SPL_{RMS}$ ~ 130 dBre1μPa). During only 1 h, anthropogenic noises were absent ($SPL_{RMS}$ ~ 100 dBre1μPa), where snapping shrimps and fish sounds were found. Therefore, the soundscape was strongly dominated by anthrophony from the salmon net pen with 97% of the total time and ~ 40 dBre1μPa higher than natural sounds. Moreover, the highest noise might be roughly 175 dBre1μPa at 1 m from the net pen (regarding the simple geometric spreading loss model). Consequently, it is raising the necessity to evaluate the effects of these noises on the marine fauna.

**Keywords**

Aquaculture · Underwater noise · Underwater soundscape · Impact · Salmon farms · Net pen · Acoustic pollution · Well boat

## Introduction

The increasing noise emitted by human activities in the marine environment is generating scientific concern regarding noise impacts on this ecosystem (Ellison et al. 2012). Facing this problem, the use of passive acoustic sensors provides useful information on the composition of the soundscape. The soundscape is the combination of sounds in a particular location and time, and it can be decomposed as biophony, geophony, and anthropophony regarding their source of origin (Farina 2014). In order to control, or to reduce, the impacts of the anthropophony on the marine ecosystem, it is necessary to identify and characterize the sources of noise and its metrics (sound pressure levels, particle motion, and frequency), in conjunction with the identification of its ecological effects on different marine species (Wale et al. 2021). Therefore, in the present study, a description of the soundscape of a fjord from the Northern Chilean Patagonia was conducted, considering the rapid increase of the aquaculture industry (Molinet et al. 2018) and also considering that this is an area with high ecosystem importance for marine mammals (Hucke-Gaete et al. 2004, 2013).

The ocean is an acoustic environment where sound travels about 4.3 times faster than in air (Stojanovic and Preisig 2009). There is a wide range of sounds in the ocean that can be described by their sound pressure level (in pascals or micropascals), particle motion (meter, meter per second, or meter per square second), and their frequency (Hetz). All fishes and several invertebrates detect and use particle motion generated by underwater sounds, but it is still difficult to measure it in the field because of difficulty in obtaining adequate instrumentation (Popper and Hawkins 2019). Herein, we will focus only on pressure levels and frequency. Several marine species from invertebrates to big mammals use acoustic signals for different ecological processes as communication (Whitehead and Weilgart 1991; McCauley and Cato 2000; Wysocki and Ladich 2001), reproduction (Lobel 2002), and defense

(Versluis et al. 2000; Patek and Caldwell 2006; Buscaino et al. 2011), and some fish and invertebrate larvae use acoustics signals as cues during settlement processes (Mann et al. 2007; Lillis et al. 2013; Hinojosa et al. 2016).

Human activities such as shipping, construction, and exploration (e.g., seismic air guns), among other activities, have added several sounds (or noises) into aquatic environments, and consequently this has generated concern regarding their effects on marine life (Popper and Hawkins 2016; Southall et al. 2019; Wale et al. 2021). In order to inform public policies that tackle underwater noise, it is imperative to understand the full range of physiological and behavioral impacts that noise has on marine life (Southall et al. 2019; Popper and Hawkins 2019), but also, as a first step, it is key to identify the anthropogenic noise sources in a particular area.

The Northern Chilean Patagonian Fjords has seen an explosive development of salmon aquaculture (Atlantic salmon-*Salmo salar*, Pacific salmons-*Oncorhynchus* spp., and Rainbow trout-*Oncorhynchus mykiss*) over the last 30 years (Tecklin 2016; FAO 2018). Consequently, scientific concern about the impact that this industry generates in marine ecosystems has received special attention (Poblete et al. 2019). For example, an increase of the nutrients at the seafloor close to fish cages (Carroll et al. 2003), as well as the introduction of chemical substances into the water column (Miranda and Zemelman 2002), has been described. There is also evidence demonstrating the introduction of these farmed exotic species to the natural environment (Correa and Gross 2008; Figueroa-Muñoz et al. 2022), and the introduction of large amounts of floating plastics to this low human populated region (Hinojosa and Thiel 2009; Ahrendt et al. 2021).

However, the description of the underwater soundscape near salmon farm (net pens) installations are scarce. The salmon industry operation at the sea has the potential to introduce sound signals from low to high frequency as a result of its multiple daily functions, such as transportation, electricity production, and feeding processes. Recent descriptions from other parts of the world have shown that salmon net pens are emitting noises of considerable intensity ($SPL_{RMS}$ ~ 110 dB re 1μPa) (Radford and Slater 2019). Additionally, these authors suggest that the noise from higher frequencies (1–2 kHz) come from engines, blowers, and pumps, whereas the noises from lower frequencies (<1 kHz) come from water flows and vibrations. In the Northern Chilean Patagonian Fjords, there is no available noise data from the field regarding salmon farm noises, other than that Vester and Timme (2010) accidentally recorded salmon farm noise in this area as they were recording cetacean sound signals. Therefore, the objective of this chapter is to describe the underwater soundscape of a salmon farm (net pen) in operation in a Northern Chilean Patagonian Fjord.

## Methods

A calibrated hydrophone SoundTrap 300 STD (Oceaninstruments.nz; 0.02–60 kHz) was used to obtain underwater sound recordings. Continuous recordings were made with a sample rate of 96 kHz at the "Comau" Fjord, "Los Lagos" Chilean Region (Fig. 1). The recordings were made from the ~19th hour of November 18 to the

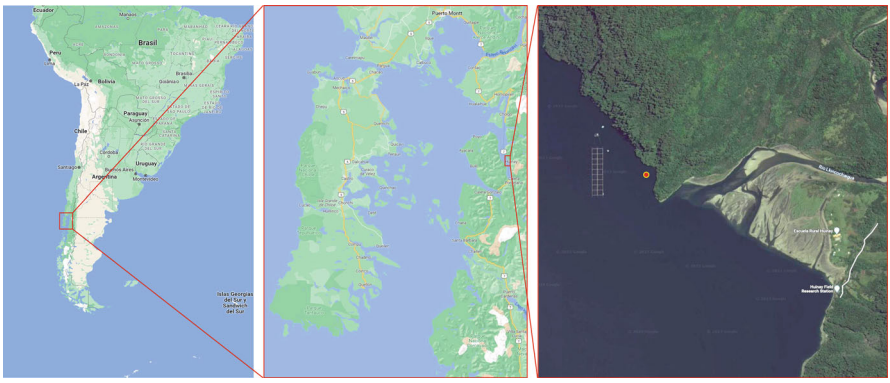


**Fig. 1** Comau Fjord in Los Lagos Region, Chile. Red dot indicates the hydrophone position (42°22′23‘‘S; 72°25′43’‘W), anchored at 20 m depth (2 m from bottom) and 250 m from a salmon farm structure. (Images from Goggle Maps)

~16th hour of November 20 (~45 h). The hydrophone was anchored at 20 m depth (2 m from bottom) and 250 m from the salmon structures (Fig. 1).

The acoustic data analysis was examined using the RavenPro program (V1.5) with a Hamming window, 8.192 samples, 90% overlap, and a sampling frequency of 4.096 Hz. For this investigation, all anthropogenic noises associated with the net pen were characterized in peak frequency (Hz) and duration (h). Anthropogenic noise was defined as any acoustic signal not associated with biological or geophysical sounds. Additionally, the noise metrics as Sound Pressure Levels ($SPL_{RMS}$ and percentiles) and the Cumulative Distribution Function (CDF) were calculated in R with the package PAMguide (Merchant et al. 2015) and using the software dBWav (©Marshall Day Acoustics).

## Results

Total 13 types of anthropogenic noises from variable sources and silence intervals were detected in a ~45-h period of recording (Table 1). For this study, silence was considered as the absence of anthropogenic noise. Additionally, in the absence of anthropogenic noise, sporadic biological sounds such as fish vocalizations and Alpheidae shrimp clicks were identified.

The lowest detected frequency peak was 70.3 Hz, corresponding to type 3 noise associated with a motorboat, while the maximum frequency peak is given by type 11 noise at 2953 Hz from an unknown source. All the noises detected during the ~45 h of recording presented unequal duration. However, there were some types of sounds that were more frequent than others (see Fig. 2).

The noise most dominant in time was type 2, characterized with a low frequency (80 Hz, $SPL_{RMS}$ ~ 120 dB re 1μPa), with a total duration of 23 h and probably associated with noise from an electric generator (Fig. 3). The second most prominent

**Table 1** The different types of noises recorded at the Comau Fjord, Los Lagos Region, Chile

| Noise type | Total duration (h) | Peak (Hz) |
|---|---|---|
| 1 | 12:15:25 | 2718.8 |
| 2 | 23:38:19 | 82.0 |
| 3 | 0:03:40 | 70.3 |
| 4 | 1:09:01 | 97.7 |
| 5 | 0:02:20 | 140.6 |
| 6 | 1:19:07 | 269.3 |
| 7 | 0:34:35 | 82.0 |
| 8 | 2:11:12 | 1066.4 |
| 9 | 0:02:53 | 152.3 |
| 10 | 0:58:20 | 175.8 |
| 11 | 0:03:46 | 2953.1 |
| 12 | 0:51:34 | 2308.6 |
| 13 | 0:14:44 | 316.4 |
| Silence (noise-free environment) | 1:22:02 | NA |

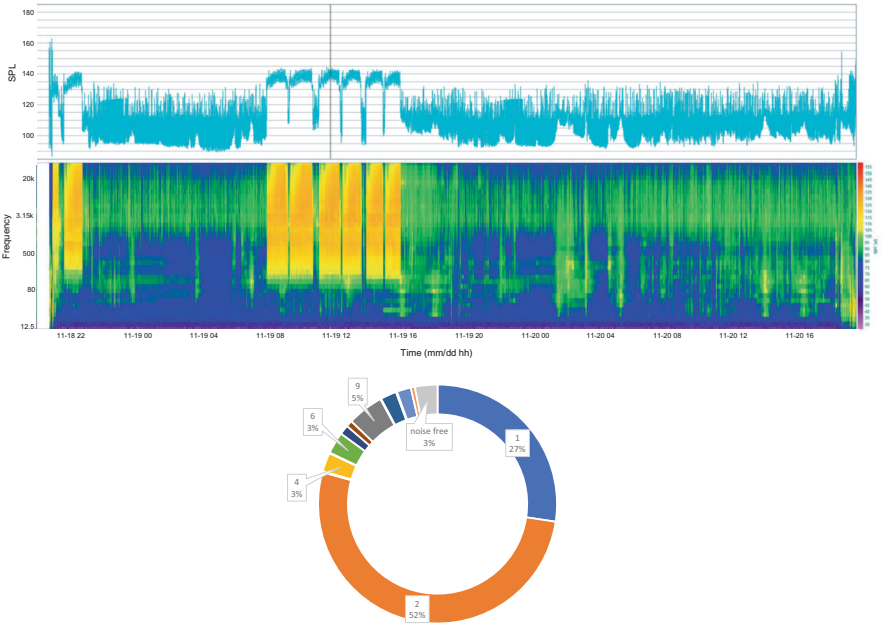



**Fig. 2** Sound Pressure Level ($SPL_{RMS}$, upper graph) and a long-term spectrogram (middle graph) over ~45 h, where the color scale (coldest to hottest) represents the $SPL_{RMS}$. The pie chart at the bottom represents the percentual occurrence of the different types of anthropogenic noise, where only the most common noises were indicated (Type 1, 2, 4, 6, 9, and noise-free). The hydrophone was situated at 250 m from a salmon farm (net pen) in the Comau fjord, Los Lagos Region, Chile

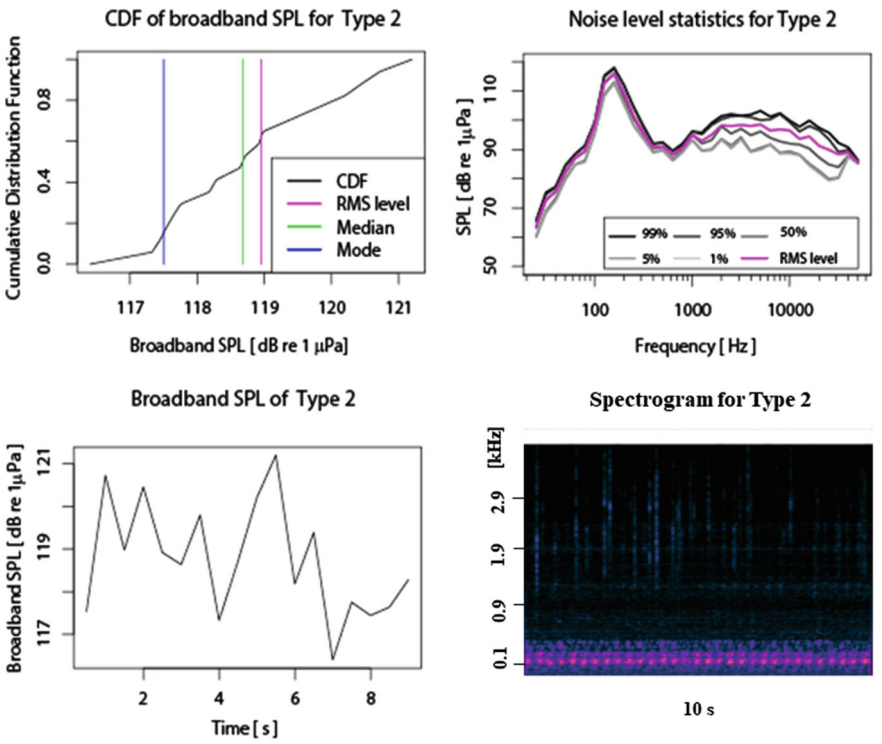



**Fig. 3** Noise metrics from type 2 noise. The graph on the upper left shows the Cumulative Distribution Function (CDF) of broadband SPL, where the RMS levels, the median, and the mode were represented. The upper right graph shows the noise level (SPL) statistics per frequency. The lower left graph shows the broadband SPL variation over time. The graph on the lower right shows the spectrogram where hottest colors represent higher SPL

noise corresponds to type 1. This is relatively high frequency (2500 Hz, $SPL_{RMS}$ ~ 142 dB re 1µPa) and most likely associated with the automatic feeding system or the operation of a well boat (Fig. 4).

During most of the total of recording, the passage of different boats was detected, where there were noises of type 4, 6, and 9, in a frequency range of 70–269 Hz ($SPL_{RMS}$ ~ 110–140 dB re 1µPa), with a total of 3.5 h of presence (Figs. 5, 6, and 7).

Anthropogenic noise-free moments ($SPL_{RMS}$ <100 dB re 1µPa) generally occurred when the most dominant noises (type 1 and type 2) disappeared and, coincidentally, no boats passed by (Fig. 8). In these periods, it is possible to identify the clicks of snapping shrimp and sporadic fish vocalizations. Of the 2 days of recording (~45 h), there were periods free of anthropogenic noise only for approximately 1.5 h.

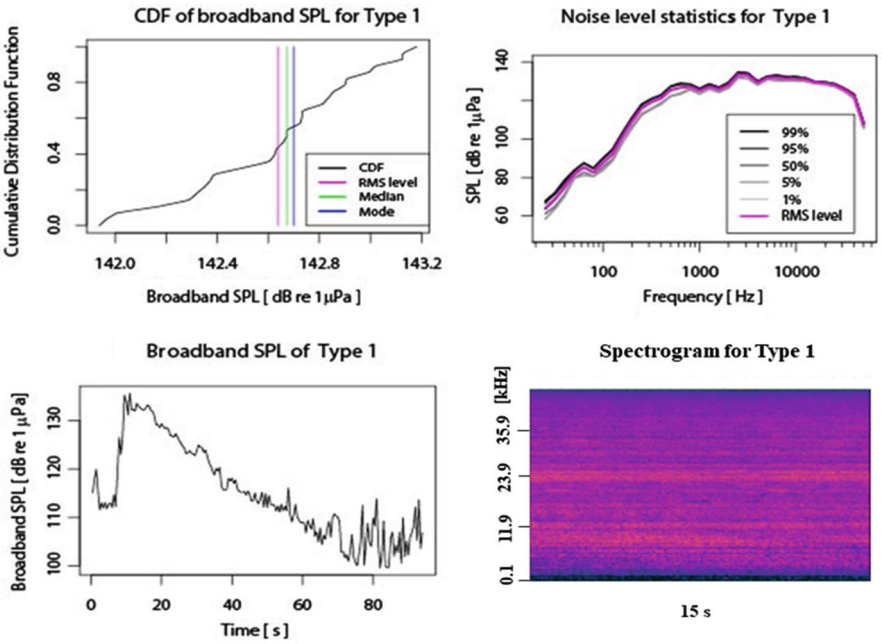



**Fig. 4** Noise metrics from type 1 noise. The graph on the upper left shows the Cumulative Distribution Function (CDF) of broadband SPL, where the RMS levels, the median, and the mode were represented. The upper right graph shows the noise level (SPL) statistics per frequency. The lower left graph shows the broadband SPL variation over time. The graph on the lower right shows the spectrogram where hottest colors represent higher SPL

## Discussion

Although anthropogenic underwater noise has become a global issue (Williams et al. 2015), in Chile it is still considered an emerging problem and even more so, considering it is mostly unknown for the Patagonian Fjords. In general, the variety and intensity of anthropogenic noises in the ocean are related to the associated use, such as marine traffic areas, military training areas, fishing and industrial activities, etc. (Hildebrand 2009). Due to the expansive industrialization in aquatic areas, anthropogenic noise is becoming more prevalent and the natural soundscape is increasingly disrupted. Here, in ~45 h of recording, 13 types of noise from anthropogenic sources were detected in a normal operation of a salmon farm (pen net) in Northern Patagonia. These noises are related to automatic power systems, electricity generators, and traffic boats of different types, the noises from boats being the most diverse in frequency and duration, and notoriously contributing to the composition of the anthropogenic soundscape of a salmon farm.

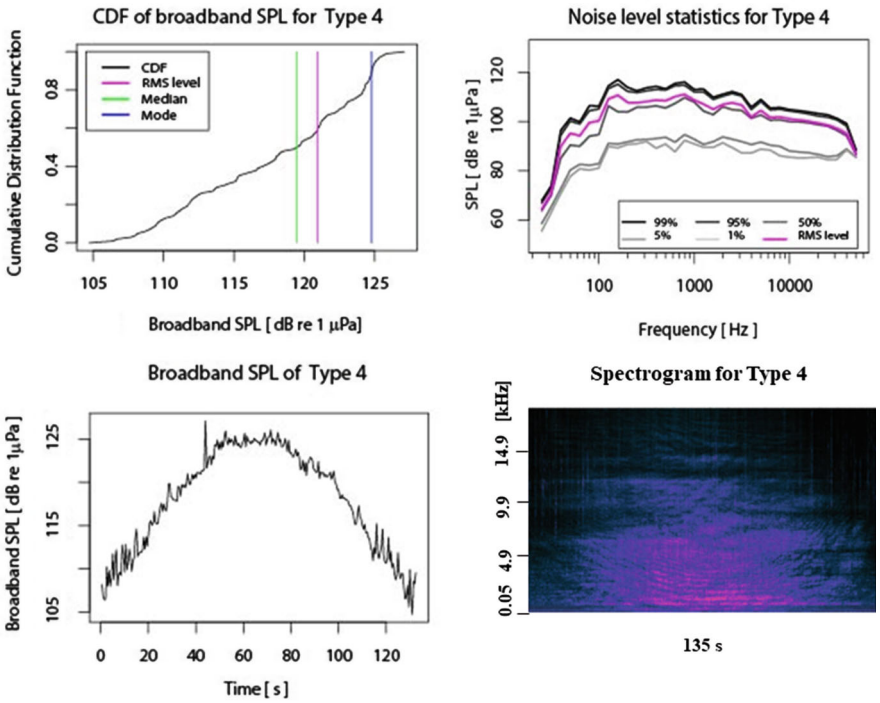



**Fig. 5** Noise metrics from type 4 noise. The graph on the upper left shows the Cumulative Distribution Function (CDF) of broadband SPL, where the RMS levels, the median, and the mode were represented. The upper right graph shows the noise level (SPL) statistics per frequency. The lower left graph shows the broadband SPL variation over time. The graph on the lower right shows the spectrogram where hottest colors represent higher SPL

The masking effect of anthropogenic noise on biological communication by sounds has been extensively investigated in cetaceans (Erbe et al. 2016, 2018; Williams et al. 2020; Branstetter and Sills 2022), and it is probable that vocalizations of these species in the northern Patagonia may be masked by anthropogenic noises from the salmon aquaculture industry (Liu et al. 2013). Our examination of the recorded files revealed the presence of biological sounds (some fish vocalizations and shrimp clicks), so the masking effects generated by net pen noises may also occur over species other than cetacean. Moreover, the spectrum visualization sometimes becomes confusing at low frequencies because the anthropogenic noise was masking all the potential biological sound signals. Type 2 noise was visible in the spectrum for long periods of time but is masked by other higher-frequency noises, so it is possible that it was present in much of the recording except for the rare periods of silence (noise absence). In the case of type 1 noise, which corresponds to the second noise with the highest presence, it is possible that, due to its cyclicity, they may be the feeding periods or another farming process as fish suction for bath cleaning (use of well boats) that occurs regularly and during the daylight. Type

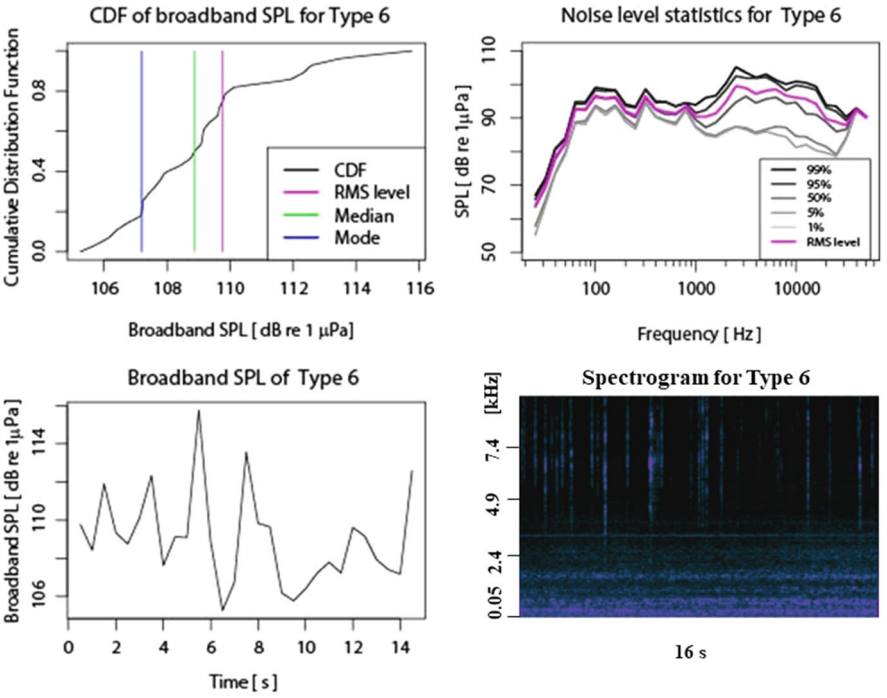



**Fig. 6** Noise metrics from type 6 noise. The graph on the upper left shows the Cumulative Distribution Function (CDF) of broadband SPL, where the RMS levels, the median, and the mode were represented. The upper right graph shows the noise level (SPL) statistics per frequency. The lower left graph shows the broadband SPL variation over time. The graph on the lower right shows the spectrogram where hottest colors represent higher SPL

1 noise was the noise with the higher intensity ($SPL_{RMS}$ ~ 142 re 1 µPa; $SPL_{Peak}$ ~ 170 re 1 µPa; and SEL ~173 dB re 1 $\mu Pa^2s$). Considering a rough and simple spherical transmission loss model, type 1 noise was generated at around 190 dB re 1 µPa at 1 meter from the source which was similar to the noise used for sea lion-deterrent bombs (205 dB re 1 µPa) in some countries (Hildebrand 2009), probably generating some impacts on the fish farmed and also over the surrounding ecosystem (considering a cylindrical transmission loss model, it is 166 dB re 1 µPa at 1 m, and with a combined spreading loss model, it is 176 dB re 1 µPa at 1 meter from de source). Solé et al. (2021) determined that noise at relatively lower frequencies (350 Hz and 500 Hz), and SEL around 190 dB re 1 $\mu Pa^2s$, which is used as acoustic treatment against lice infestation, did not present any risks to the hearing of farmed salmon. However, it is necessary to determine if farmed fish show any behavioral changes when they are exposed to chronic noise of higher intensity as detected in this study. The results of this research indicate that the marine ecosystem surrounding the farm in the Comau Fjord is exposed to anthropogenic noise for around 97% of the examined time, where only 3% of this time was without the presence of any

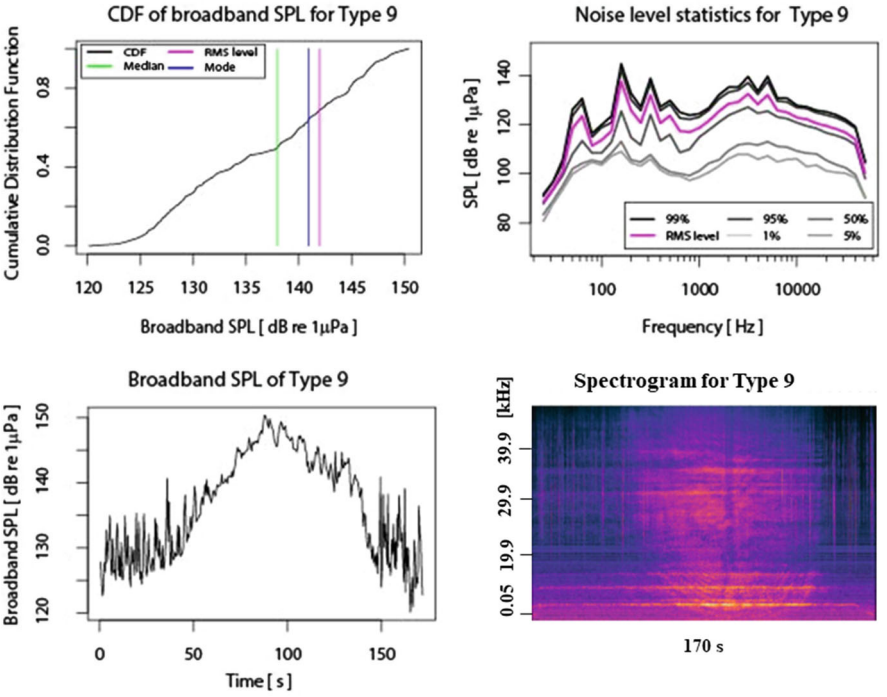



**Fig. 7** Noise metrics from type 9 noise. The graph on the upper left shows the Cumulative Distribution Function (CDF) of broadband SPL, where the RMS levels, the median, and the mode were represented. The upper right graph shows the noise level (SPL) statistics per frequency. The lower left graph shows the broadband SPL variation over time. The graph on the lower right shows the spectrogram where hottest colors represent higher SPL

anthropogenic noise, leaving only few moments for biological acoustic signals which may be important for the ecological processes of a large number of species (Lillis et al. 2013; Hinojosa et al. 2016; Gordon et al. 2019).

Fjords, channels, and estuaries are ecosystems with higher biological productivity and play a very important role in the carbon cycle (Liu et al. 2010). The fjords from the Northern Patagonia are also considered important feeding ground for humpback whales, and feeding and nursery ground for blue whales (Hucke-Gaete et al. 2013; Buchan and Quiñones 2016). The Comau Fjord is considered to be of great importance to shallow cold-water coral formations (Häussermann and Försterra 2007). This area is one of the three fjords in the world that form large banks of the coral species *Desmophyllum dianthus* at shallow depths. In this area, there have been events of massive mortality of this coral due to collateral effects of the microalgal proliferation enhanced by the eutrophication that the salmon farms generate (Försterra et al. 2014). The increasing incorporation of new farming centers may greatly affect the water column (Poblete et al. 2019) and also, as was shown by this study, will introduce different types of noises with higher intensities that may affect

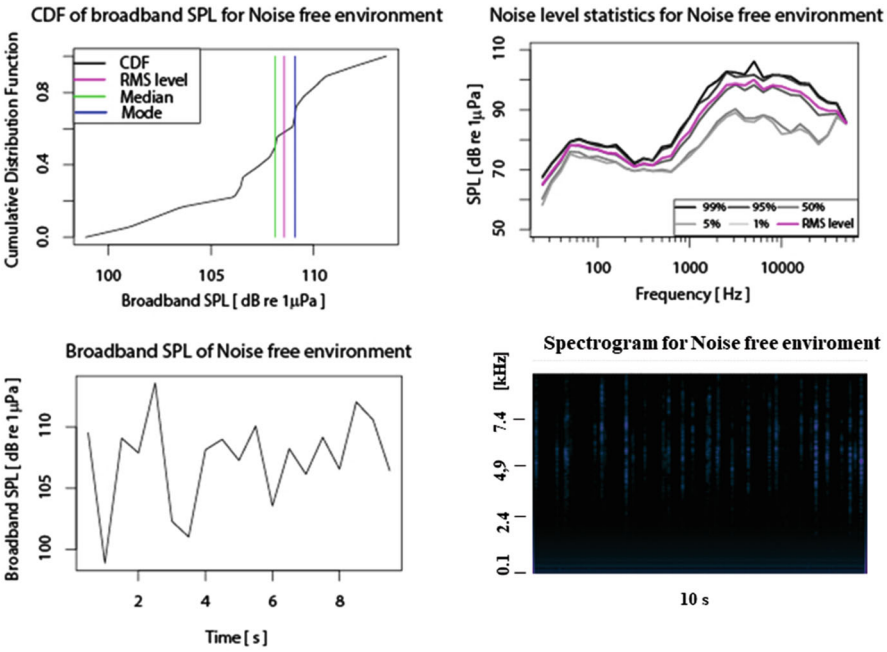



**Fig. 8** Noise metrics from the noise-free environment. The graph on the upper left shows the Cumulative Distribution Function (CDF) of broadband SPL, where the RMS levels, the median, and the mode were represented. The upper right graph shows the noise level (SPL) statistics per frequency. The lower left graph shows the broadband SPL variation over time. The graph on the lower right shows the spectrogram where hottest colors represent higher SPL

the species that inhabit it. Therefore, efforts must be focused on more studies addressing the effects of these noises from the aquaculture industry on the Chilean fjord-land. Therefore, it will help to reformulate environmental regulations when allowing the installation of farming centers in places of ecosystem importance. In addition, detailed environmental impact studies are needed to better understand the biodiversity of these areas, with great population dynamics as they also receive migratory species that would be exposed to chronic anthropogenic noise.

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# Acoustic Profiles of Underwater Soundscapes Affected by Road Traffic

25

Kees te Velde and Hans Slabbekoorn

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**Abstract**

Freshwater ecosystems such as rivers and lakes are heavily threatened by a multitude of stressors, including noise pollution by human activities. Noisy activities on and in the water, such as the prominent use of motorized boats, may alter natural underwater soundscapes. Traffic noise and sound from other human activities on land may also penetrate into the water, especially through bridges. Natural soundscapes vary with hydro-morphological variation among river types and sections and with the heterogeneity in local diversity of sound-producing animals and plants. Natural soundscapes may play an important role in orientation and navigation, in particular for migratory fishes, but still very little is known. It is clear that anthropogenic noise may deter, disturb, distract, and mask natural soundscapes, but few studies have addressed the extent of potential masking impact in freshwater systems. Many rivers flow through urban areas, where typically many bridges provide crossing opportunity to noisy road traffic. Weeklong underwater recordings reveal diurnal and weekly fluctuations of sound pressure levels (SPLs) in underwater soundscapes, likely related to road traffic over urban bridges, in a Dutch lowland river system. Weekdays and weekends

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can acoustically be distinguished in these underwater soundscape profiles. These results provide insight into the potential for noise pollution impact on migratory fishes and other aquatic animals.

**Keywords**

Anthropogenic noise · Diurnal cycle · Masking · Migratory fish · Urban

## Introduction

Underwater soundscapes reflect underwater habitat features and locally audible biodiversity (Alcocer et al. 2022; Desjonqueres et al. 2015; Gottesman et al. 2020). A soundscape refers to all sounds audible at a particular location and is determined by the presence of sound sources and propagation properties of the waterbody (Mullet et al. 2017). Sound sources include natural sources of abiotic (wind, waves, water currents and turbulence, running sediment), biotic (animals, plants) origin (Hildebrand 2009; Fine and Parmentier 2015; Geay et al. 2017; Freeman et al. 2018), and anthropogenic sources of human activities making sound purposefully (sonar, airguns) or as a by-product (vessel, construction, and piling noise, and sounds from weirs, locks, and pumping stations). These environmental sounds are also referred to as geophony, biophony, and anthrophony (Mullet et al. 2017; Pijanowski et al. 2011). Sounds attenuate over a distance because of spherical spread, and high frequencies fade at a faster rate than low frequencies, except in shallow waters, in which wave length and water column height determine a low-frequency cut-off for wave lengths that do not fit the depth (Forrest et al. 1993; Larsen and Radford 2018; Rogers and Cox 1988; Urick 1975).

Habitat-associated soundscapes determined by the acoustic presence of aquatic animals have often been reported for marine environments (Kaplan and Solomon 2016; Kennedy et al. 2010; Lin et al. 2021). Furthermore, acoustic monitoring has been successfully applied to assess fluctuations in habitat quality, for example, for tropical and temperate reefs (Bertucci et al. 2016; Harris et al. 2016). Also, freshwater soundscapes consist of a wide variety and potentially habitat-dependent sound sources with diverse spectral characteristics (Desjonqueres et al. 2015; Gottesman et al. 2020; Martin and Popper 2016). Biotic sounds include fish calls (Colleye et al. 2013; Cott et al. 2014; Parmentier and Fine 2016), aquatic invertebrate chorusing (Desjonqueres 2016; Greenhalgh et al. 2021), amphibian and bird calls (Narins et al. 2007; te Velde et al., n.d.), and even aquatic plant respiration (Kratochvil and Pollirer 2017; van der Lee et al. 2020). These biotic sound sources typically co-occur with abiotic sound sources, and the mixture of sounds is filtered by environment-specific propagation features, which can all provide critical information about the environment to aquatic animals (Fay 2009; Slabbekoorn et al. 2010).

Habitat-associated soundscapes determined by hydro-morphological features have been investigated in particular for freshwater systems. Wysocki et al. (2007) and Amoser and Ladich (2010) revealed, for example, habitat-associated soundscape diversity as well as year-round soundscape diversity in Austria. Most pronounced

were the differences between quiet stagnant waters and loud fast-flowing waters (Amoser and Ladich 2010; Wysocki et al. 2007). Tonolla et al. (2010, 2011) studied the relationship between physical river features with soundscapes in more detail for high-elevation fast-flowing rivers in Switzerland (Tonolla et al. 2010, 2011). He found considerable temporal variation in spectral profiles, but this was exceeded by spatial variation associated with distinct river sections, such as pool, riffle, and running sediment. Kacem et al. (2020) had similar correlative results in Canadian rivers and additionally found brook trout (*Salvelinus fontinalis*) densities associated with both physical features of river sections and variation in sound levels therein (Kacem et al. 2020). Different sections of a relatively slow-flowing Dutch river system, with relatively low-amplitude SPLs, also varied spectrally with several hydro-morphological features (te Velde et al. unpublished manuscript).

Human activities on, in, and near waterbodies introduce anthropogenic noise in natural soundscapes. Habitat-dependent soundscapes that may provide cues to aquatic animals may thereby be disturbed or masked (Duarte et al. 2021; Mickle and Higgs 2018; Popper and Hawkins 2019; Slabbekoorn et al. 2010). Boat noise is generally composed of a broadband component and a set of tonal sounds that are generally high in amplitude compared to natural soundscapes (Hildebrand 2009; Jansen and De Jong 2017). Traffic noise outside the water, on nearby roads or bridges, can also affect underwater soundscapes, especially for the low-frequency ranges (Holt and Johnston 2015; Martin and Popper 2016; te Velde et al. unpublished manuscript; Zang et al. 2019). However, still only a few studies report spatial and temporal variation in underwater sounds that originate from human activities on land (Martin and Popper 2016; Rountree et al. 2020). Furthermore, while there is increasing attention toward anthropogenic noise and soundscape deterioration in marine systems (Havlik et al. 2022), river systems remain largely overlooked.

In the current study, temporal patterns are investigated in a weeklong underwater recordings at urban bridges and at more quiet rural areas, with more and less nearby road traffic, respectively, in a Dutch lowland river system. With the aim to answer the following questions: Are diurnal and weekly fluctuations of road traffic density reflected in patterns of sound pressure levels (SPLs) of underwater soundscapes? Are urban bridge sites more noisy in overall underwater SPLs than rural quiet sites, and are relatively quiet, nighttime periods shorter within than outside the city? Is there significant variation among different days of the week, and can weekdays and weekends acoustically be distinguished in underwater soundscape profiles? Answers to these questions should provide insight into the potential for noise pollution impact on migratory fishes and other aquatic animals that rely on natural soundscapes for daily survival and reaching reproduction sites and life stages.

## Methods

Continuous one-week recordings were made synchronously at four bridges twice at two cities, at high- and low traffic-density sites, in the Valleikanaal-Eem river system in the Netherlands (te Velde et al. unpublished manuscript). As a reference for traffic density, sound level values were extracted from an in-air traffic noise map made by

the "RIVM atlas voor de leefomgeving" (RIVM 2021) (Fig. 1). In this map, sound level was predicted from data on hourly motorized vehicle passages, and is expressed in Lden (Level Day-Evening-Night), which yields a mean dB re 20 uPa level per 24 hours, where evening and night have a higher weighting than daytime levels. Samples were taken in the relatively small, slow-flowing river Valleikanaal, within and outside the city of Veenendaal, and in the relatively large, slow-flowing river Eem, within and outside the city of Amersfoort. In March 2022, four Soundtraps were deployed to get a weeklong, continuous recordings, simultaneously at two high traffic-density sites (under bridges) and two low traffic-density sites (rural area). This was carried out in the cities of Veenendaal (8–16 March 2022) and Amersfoort (22–31 March 2022) (Fig. 1). This resulted in 4 one-week recordings at high traffic-density sites and 4 one-week recordings at low traffic-density sites.

Four Soundtrap 300 STD hydrophones (Ocean Instruments, NZ, Acoustic Monitoring Systems) were used, suspended between an anchor and a sub-surface buoy to prevent surface motion, in the middle of the river, 30 cm above the riverbed and 1.50 m below the water surface. The soundtraps were set at a sampling frequency of 96 kHz, high PreAmp gain, and with the high-pass filter turned off. Spectrograms were generated of the weeklong recordings with a temporal resolution of 1 minute,

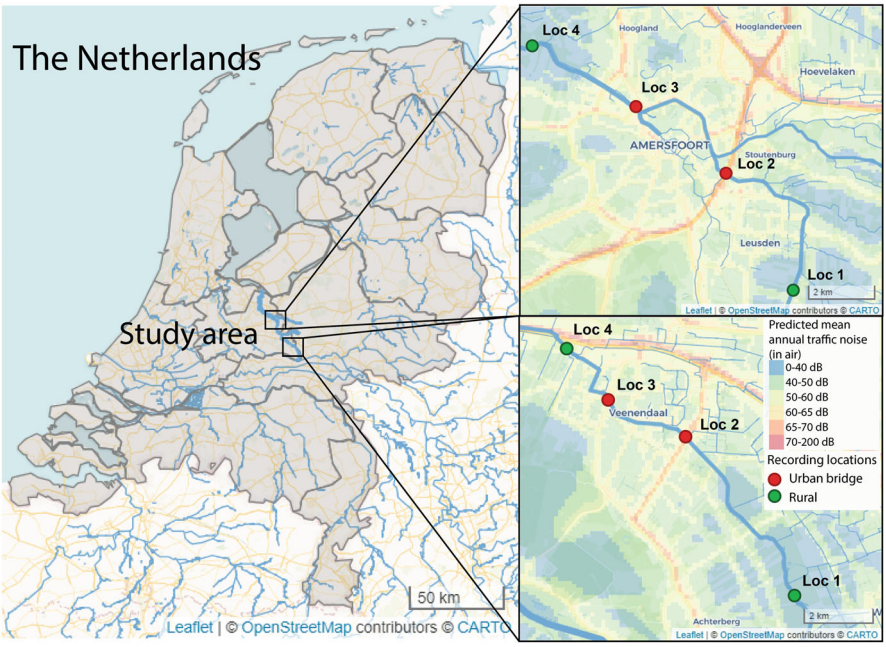



**Fig. 1** Recording locations in the Valleikanaal-Eem river system in and around the cities of Amersfoort and Veenendaal with predicted in air sound map from traffic noise. The traffic noise map was adapted from the RIVM atlas leefomgeving. Sound in the map was predicted from data on hourly motorized vehicle passages, and is expressed in Lden (Level Day-Evening-Night) which is a mean dB re 20 uPa per 24 hours, where evening and night have a higher weighting than day levels

and a frequency resolution of 1 Hz, to explore temporal and spectral patterns in the river soundscapes (te Velde et al. unpublished manuscript) (see Fig. 2 for example spectrograms of two locations). Any patterns that stood out from the long-term spectrogram were inspected further through visual inspection of 2-hour

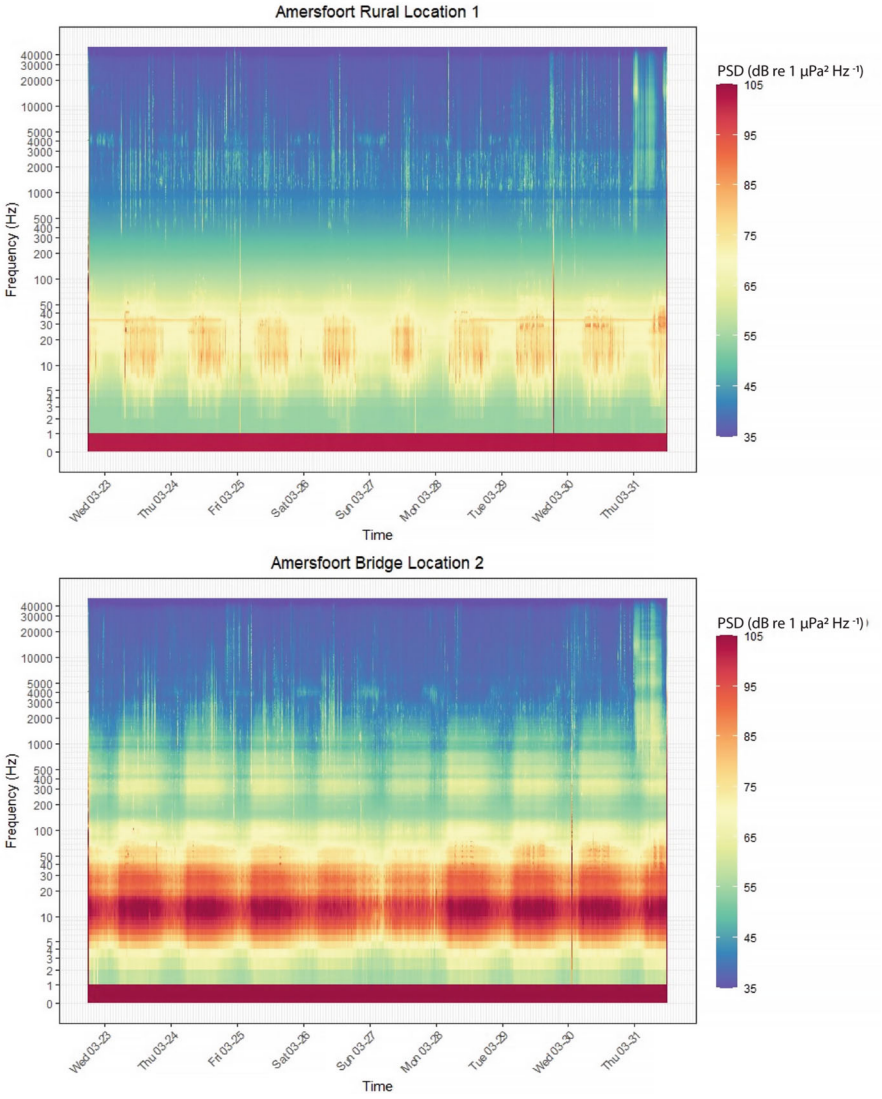



**Fig. 2** One-week spectrogram examples of one of the rural and urban Bridge recording locations in and around Amersfoort. Sound energy between 10 and 50 HZ is likely largely attributable to daily fluctuations in traffic noise levels. At 4000 Hz, repeated presence of water boatmen in the evenings is visible for both sites. Also present on the last day for both sites: rain sound across the full bandwidth, especially visible in the highest frequencies (300–40,000 Hz)

spectrograms with a higher temporal resolution in the audio program Audacity™. The window length settings were often adjusted to achieve a desired temporal and spectral resolution depending on recording conditions.

Temporal variation in sound levels was investigated, over the day and throughout the week, by depicting sound pressure level (SPL, in dB re 1 μPa), per hour, separately for each day of the week. Sound energy for the 10–50 Hz frequency band was selected, which should capture road traffic related noise presence well (te Velde et al. unpublished manuscript; Holt and Johnston 2015; Zang et al. 2019; Song et al. 2020; note that soundtraps only gradually provide accurate data in the range up to 20 Hz, which calls for some caution in comparing absolute levels among studies, but the frequency range of 10–50 Hz provides well comparable level within this study). Whether there was significant variation in SPL between urban bridge locations and rural quiet sites, and whether there was significant variation among days was tested. This was done using two-way repeated measures ANOVA, followed by a post-hoc pairwise comparison using the Estimated Marginal Means (EMMs) with Tukey adjustment. The period of relatively quiet night time (between 19:00 and 7:00) was measured, at the arbitrary threshold of 75 dB re 1 μPa, and again tested whether there was significant variation in quiet night duration between urban bridge locations and rural quiet sites, and whether there was significant variation among days. This was also done using two-way repeated measures ANOVA, followed by a post-hoc pairwise comparison using the Estimated Marginal Means (EMMs) with Tukey adjustment.

## Results

All amplitude patterns showed a distinct day-night pattern over the day (Fig. 3), with higher daytime than nighttime SPLs. Typically, sound levels rose sharply between 5:00 and 7:00 in the morning, to reach a plateau for the day, to level off gradually again over a long period, between 15:00 in the afternoon and 3:00 the next morning. The rural nights had a longer quiet night period than the urban bridge sites, roughly centered between 23:00 and 4:00. The urban bridge sites had weekday plateaus at 88 and 90 dB re 1 μPa in Veenendaal and 90 and 95 dB re 1 μPa in Amersfoort. The rural quiet sites had weekday plateaus at 68–70 and 75 dB re 1 μPa in Veenendaal and 73–75 and 72–74 dB re 1 μPa in Amersfoort. The urban bridge sites were significantly more noisy than the quiet rural sites ($F(1, 6) = 44.43$, $p < 0.001$). The difference between daytime high and nighttime low ranged from 15 to 35 dB at urban bridges, while this was around 5 dB for the rural sites. There also was significant variation among days ($F(6, 36) = 22.993$, $p < 0.001$), with weekend days in particular being more quiet than weekdays at the urban bridge sites (see Fig. 4 for pairwise significance).

The duration of relatively quiet nighttime periods (Fig. 5) varied significantly between urban bridge and rural quiet sites ($F(1, 6) = 103.2$, $p < 0.00001$), there was no significant difference in quiet hours among nights. The sound level did not get

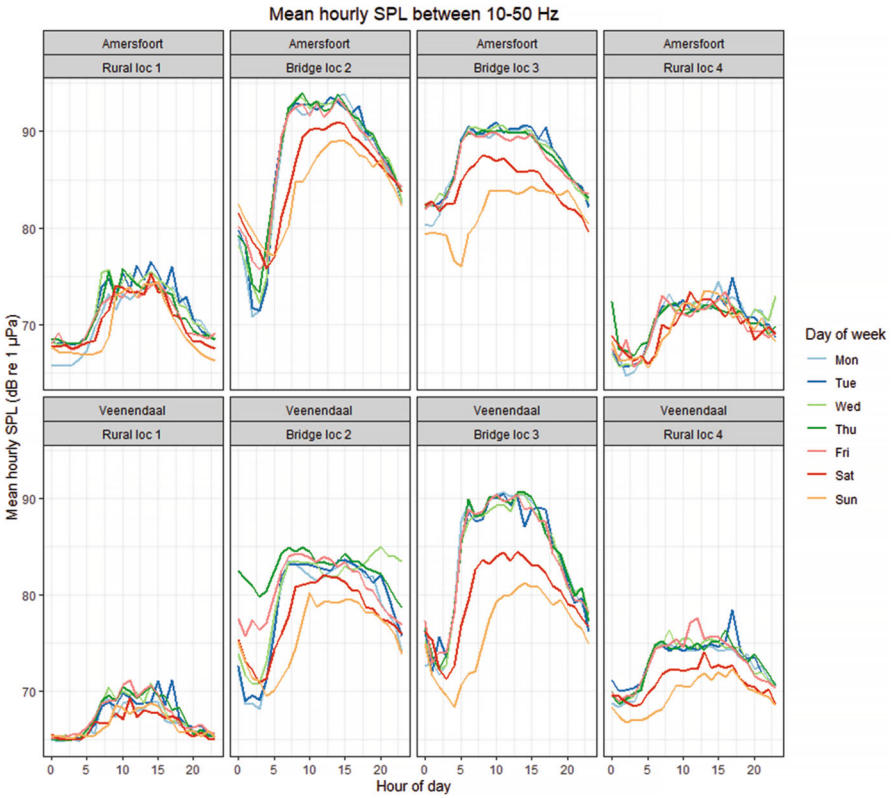



**Fig. 3** Mean hourly Sound Pressure Levels (SPL) between 10 and 50 Hz at each recording location, colored lines indicate the days of the week

above the arbitrary threshold of 75 dB re 1 µPa at rural sites 1 and 4 of Veenendaal, and not below at the urban bridge site 3 in Amersfoort.

## Discussion

Diurnal and weekly fluctuations of road traffic density were found, reflected in patterns of sound pressure levels (SPLs) of underwater soundscapes from these Dutch lowland river sites. Urban bridge sites were noisier in underwater SPLs than rural quiet sites, and relatively quiet, nighttime periods were significantly shorter within than outside the city. Also, significant variation was found among different days of the week, and weekdays could acoustically be distinguished from Saturday to Sunday, in particular for the urban bridge sites of both Veenendaal and Amersfoort. How these findings provide insight into the potential for noise pollution impact on migratory fishes and other aquatic animals will be addressed below.

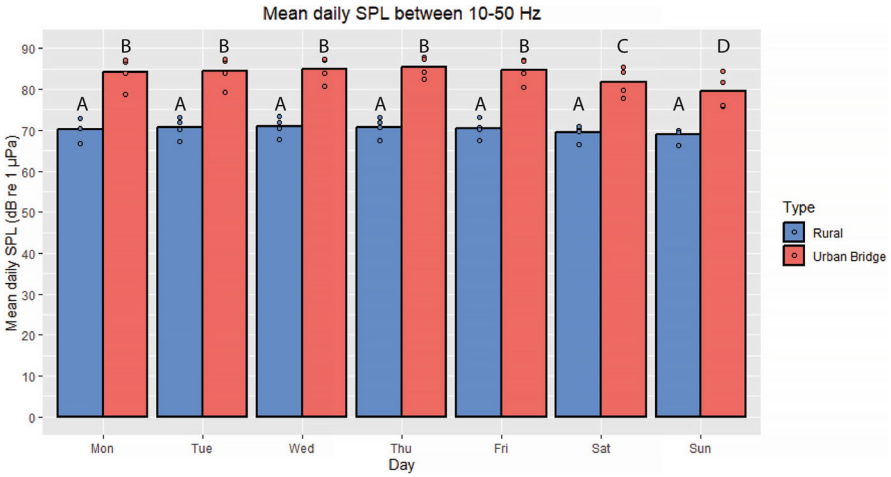



**Fig. 4** Mean daily SPL between 10 and 50 Hz of rural and urban bridge recording sites. Dots represent the mean daily SPL of each separate recording location. Letters indicate pairwise significance, showing that rural is significantly different from urban bridge, and Saturday and Sunday in the city are significantly different from other days in the city

## Human Presence in Underwater Soundscapes

The patterns in underwater sound pressure level results suggest large differences between day and night and between urban bridges and rural sites. Also at the rural sites, differences were seen between day and night, but also sound traces on the sonograms that were very similar in frequency, shape, and pattern to the road traffic related sounds for the urban bridge sites (also see Holt and Johnston 2015; Martin and Popper 2016; Zang et al. 2019; Song et al. 2020; te Velde et al. unpublished manuscript)). So, it is not certain whether any of the diurnal fluctuations are related to natural fluctuations at the rural sites, but for the urban bridges it is clear that traffic noise causes strong elevations of daytime sound pressure levels and shortening or complete absence of relatively quiet nighttime periods. Martin and Popper (2016) also found SPL differences under a large bridge over the Hudson River (USA) between weekends and weekdays, summer and fall, and peaks during rush hours. The variation in underwater sound levels between weekdays and weekends implies that impact could also have a weekly cycle. Such variable impact was found for free-ranging mulloways (*Argyrosomus japonicus*) related to higher boat activity in weekends (Payne et al. 2014). Individual fish were less active and inhabited deeper waters during the weekend, and they had less full stomachs and fewer fish in their diet over the weekend.

The road traffic related elevation of sound pressure levels is most prominent up to 100 Hz but is visible on the sonograms up to 3000 Hz and is typically at its high end from 7:00 to 15:00, after which sound levels only gradually decline throughout the afternoon and evening. In contrast with Martin and Popper (2016), the current study does not show significant SPL peaks at rush hours. Fish and other aquatic animals

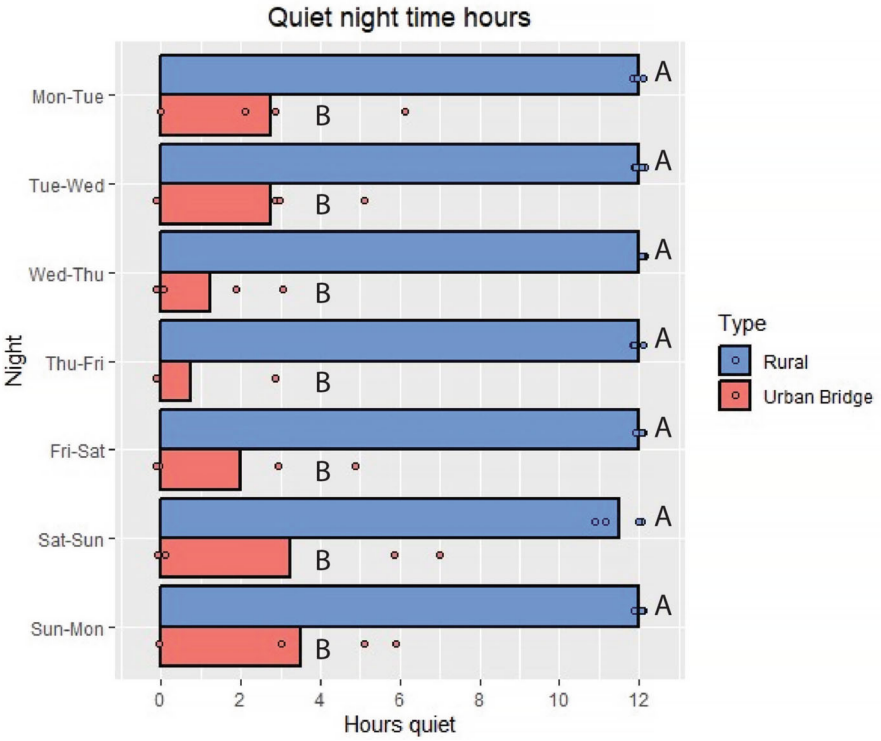



**Fig. 5** Number of "quiet" hours per night (between 19.00 and 7.00), in which SPL between 10 and 50 Hz is below 75 dB re 1 µPa. Dots represent hours per day of each separate recording location. Letters indicate pairwise significance

are expected to be able to detect particle motion, and some fish and invertebrate species can detect sound pressure, due to the presence of a gas-filled bladder or bubble, depending on its size and distance from the ear (Amorim et al. 2015; Fine and Parmentier 2015; Ladich 2015; Popper et al. 2019). Particle motion levels are not the same as sound pressure levels, especially not close to water-air or water-substrate boundaries. The frequency range of most interest for communication and environmental cues for fish and aquatic animals is typically only up to a few hundred Hz or maximally up to a few thousand Hz, depending on the species. Little is known about diurnal activity patterns or fluctuations in relative importance of audibility of the natural soundscape, but many fish species migrate or spawn in the evening and at night (Brönmark et al. 2014; Lucas and Baras 2008; Northcote 1984), invertebrate species are also known to chorus in the evening and at night (te Velde et al. unpublished manuscript), and light conditions obviously make vision less useful overnight. The presence of anthropogenic noise in the Dutch river system, especially in the urban environments at bridges, seems therefore likely a potentially detrimental masker for relevant sounds and can limit animal access to important acoustic information (Codarin et al. 2009; Popper and Hawkins 2019).

## Conservation Implications

Freshwater ecosystems are among the most vulnerable in the world (Reid et al. 2013; Deinet et al. 2020; Tickner et al. 2020), typically threatened by a multitude of stressors (Parrish et al. 1998). These stressors include: blockage of routes for migratory species, overfishing, water quality and quantity changes, and riverbed deterioration (Allan et al. 2005; Belletti et al. 2020; Brevé et al. 2014; Forseth et al. 2017; Grill et al. 2019; Tamario et al. 2019). Anthropogenic noise may add up to these stressors, through acoustic deterrence, disturbance, distraction, and masking. Although the surface has just been scratched about variation in natural soundscapes, the importance of sound to aquatic animals has become sufficiently clear, as is the associated threat by noise pollution (Cox et al. 2016; Duarte et al. 2021; Fay 2009; Slabbekoorn et al. 2010; Slabbekoorn and Bouton 2008). A few examples of anthropogenic noise-related activity level, swimming patterns, and foraging efficiency, already indicate that sound may indeed have fitness consequences for fish (Payne et al. 2014; van der Knaap et al. 2021).

Noise pollution may especially affect migratory fish. Migratory fish are constantly faced with challenges on their way and continuously need to take decisions based on the environmental information available to them. They thereby depend on external stimuli with critical information about their surroundings, such as habitat suitability, predator presence, potential mate quality, and food availability. External stimuli can be detected through a variety of senses: taste/smell, sight, touch, and hearing (Lucas and Baras 2008), but the latter is so far little investigated and underrated or overlooked. Several studies have shown that fish species use sound in orientation and habitat selection, especially in low-visibility conditions of murky waters and during dark nights (Gordon et al. 2018, 2019; Holles et al. 2013; Huijbers et al. 2012; Montgomery et al. 2006; Radford et al. 2011; Slabbekoorn and Bouton 2008). This supports the idea that migratory fish may use the spatial heterogeneity of sounds to gain information about their surroundings. This also implies that masking by anthropogenic noise of the natural soundscapes and acoustic cues in lowland rivers may mean trouble for the animals in this already heavily impacted ecosystem.

## Conclusions

Rivers of high and low elevation, fast- and slow-flowing vary in absolute underwater sound levels. All river types and sections have more or less locally distinct soundscapes related to hydro-morphological features and the local sound-producing biodiversity. Traffic noise in urban areas, over bridges, and even on roads near waterbodies transmits diurnal and weekly activity cycles acoustically to the underwater world. The disparity between the relatively quiet natural soundscapes and the relatively loud presence of anthropogenic noise for large parts of the day suggests another stressor of serious concern for this already threatened ecosystem. Still, little is known about the importance of natural soundscapes for aquatic animals and migratory fish in particular. Therefore, more studies are needed that explore spatial and temporal

variation in soundscapes and the impact of these soundscapes on the distribution and welfare of aquatic animals. Special focus should be put toward the acoustic ecology of migratory fish and to what extent their natural auditorily driven behavior is affected by anthropogenic noise from boats on the water and road traffic near the water.

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# Baseline Assessment of Underwater Ambient Sound in the Northern Adriatic Sea

26

Antonio Petrizzo, Michol Ghezzo, Hrvoje Mihanović, Marta Picciulin, Jarosław Tęgowski, Predrag Vukadin, Iole Leonori, and Fantina Madricardo

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**Abstract**

Baseline information on underwater ambient sound levels is poorly available for the Mediterranean Sea. The SOUNDSCAPE project filled this knowledge gap, developing a basin-scale, cross-national acoustic monitoring in the Northern Adriatic Sea. Data were collected from March 2020 to June 2021, including the full lockdown period related to the COVID-19 pandemic, at nine stations covering different natural conditions and anthropogenic pressures. Soundscape was monitored continuously 24 h a day by using calibrated stationary recorders (44.1 kHz sampling rate, 16-bit resolution). The entire dataset was analyzed to 20-second

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averaged sound pressure levels (SPLs) in all the one-third octave frequency bands (base 10). Noise levels for the 63 Hz and 125 Hz one-third octave bans were described, in compliance with the descriptor 11.2 of the EU-Marine Strategy Framework Directive. A difference of about 30–33 dB was found in these frequencies between the quietest and the noisiest monitoring stations, indicating a high spatial variability. Higher SPL median levels were observed where shipping activity was dominant. The SOUNDSCAPE dataset establishes baseline levels for the assessment of future trends in the Northern Adriatic Sea.

**Keywords**

Underwater noise monitoring · SOUNDSCAPE · Mediterranean Sea

## Introduction

Since the nineteenth century, marine soundscapes have been progressively altered by the sound generated by human activities, such as maritime traffic, coastal construction, seismic surveys, etc. Nowadays, underwater sound has been demonstrated as a potential threat to marine life and as an emerging pollutant (Duarte et al. 2021). To determine the degree of threat requires monitoring and mitigation measures. Adopted by the European Union in 2008, the Marine Strategy Framework Directive (MSFD) is currently the only one example of legislation which directly addresses this source of pollution (Merchant et al. 2022). In a larger context, the MSFD asks Member States of the European Union to attain a good environmental status (GES) by defining eleven qualitative descriptors. Descriptor 11 focuses on introduction of energy (including underwater noise) into the marine environment. Descriptor 11 refers both to impulsive sound in water (Descriptor 11.C1) and continuous low-frequency sound in water (Descriptor 11.C2). Originally D11.C2 implied the monitoring of trends in continuous underwater sound in marine waters but more recently, "levels of anthropogenic continuous low-frequency sound that do not adversely affect population of marine animals" have been requested by the European Commission Decision of 2017 (Comm. Dec. 2017).

Following MSFD guidance, several joint international UN monitoring programs have been developed in European Seas (reviewed by Merchant et al. 2022). In this framework, the Italy-Croatia SOUNDSCAPE project, funded by EU within the Interreg (European Territorial Cooperation) instrument, developed the only basin-scale, cross-national underwater acoustic monitoring run so far in the Mediterranean Sea, with special regard to the Northern Adriatic Sea.

The Adriatic Sea is the northern part of the Mediterranean Sea (Fig. 1), extending as far north as 45°47' N. It is an elongated, landlocked narrow basin that covers an area of about 138,000 km$^2$. The North Adriatic Sea (NAS) includes the northern part, from the Gulf of Trieste to the Middle Adriatic Pit; a transect southeast from Split, Croatia (MS8), to northwest from Ancona, Italy (MS3), was considered for defining the SOUNDSCAPE study area. NAS is characterized by

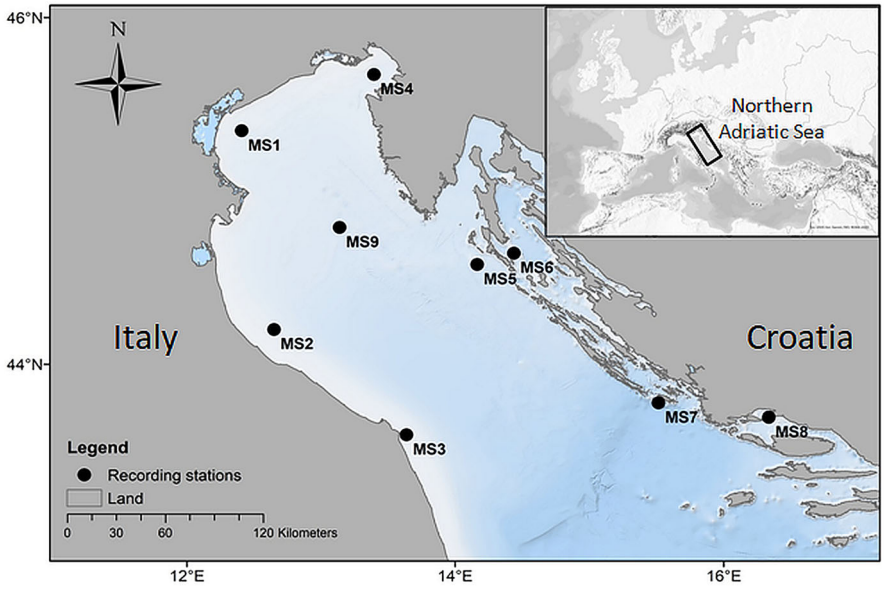



**Fig. 1** Hydrophones positioning in North Adriatic Sea; MS1, Venice (Italy); MS2, Rimini (Italy); MS3, Ancona (Italy); MS4, Trieste (Italy); MS5, Susak Lošinj (Croatia); MS6, Lošinj (Croatia); MS7, Žirje (Croatia); MS8, Split (Croatia); MS9, Ivana D (Croatia). Maps were created using ArcGIS software by Esri. ArcGIS and ArcMap™ are the intellectual property of Esri and are used herein under license. Copyright © Esri. All rights reserved. For more information about Esri software, please visit www.esri.com

relatively shallow waters, starting with a depth of 15 m along the Venice-Trieste coastline, increasing slowly southward, and then sharply reaching about 270 m in the basin central part. The northern and western coasts are generally sandy-muddy; in contrast, the eastern coast is mountainous, including numerous islands, bays, and coves (Artegiani et al. 1997; Marini et al. 2008). NAS presents strong seasonal variability in vertical temperature and salinity profiles, which could possibly be reflected in seasonal variability in the underwater noise levels. Surface temperature ranges from 4 °C (in extreme winters in the shallow NAS) to 29 °C (during warm and dry summers). Salinity values range from near zero near the mouths of rivers to a maximum of about 39.2 psu (practical salinity unit) on the east part of the area (Mihanović et al. 2013, 2021).

Further, while having a very vulnerable biodiversity including endangered and threatened marine species (Coll et al. 2010), NAS is highly impacted by human-generated activities due to a high concentration of harbors and by many anthropogenic activities including fisheries, mussel, and fish farms, as well as commercial, urban, and touristic activities (Maglio et al. 2016).

The heavy shipping activity that characterizes the NAS, combined with the shallow and stratified waters and the strong seasonality typical for the area, create a unique acoustic environment for a case study.

SOUNDSCAPE started in February 2020. This project created a scientific and institutional cooperation between eight institutional partners, leading to: (i) monitoring the underwater soundscape; (ii) evaluating possible impact on marine biological resources; and (iii) developing and implementing a planning tool for straightforward management. This chapter specifically focuses on the underwater ambient sound monitoring results.

## Underwater Ambient Sound Monitoring in the Northern Adriatic Sea

Nine acoustic monitoring stations were implemented along the Italian and Croatian shallow waters (Fig. 1). Their locations were determined after analysis, considering expected anthropogenic noise pressures, deployment and servicing complexity, soundscape modeling requirements, and cost. All the stations were provided with a calibrated autonomous passive underwater acoustic recorder (hereafter APUAR) Sono.Vault by Develogic Subsea Systems GmbH (Hamburg, Germany), featured with an omnidirectional broadband Neptune Sonar D60 Hydrophone(−192.7 dB re 1 V/μPa sensitivity; flat frequency response: 10 Hz–20 kHz ± 3 dB). Further, the same mooring setup was used in all the monitoring stations. This allowed the data collected to be compatible and comparable between stations. Image and scheme of the Sono.Vault APUAR equipment and accessories are presented in Fig. 2 APUARS were set to record continuously at a sampling rate of 44.1 kHz, providing a recording bandwidth of 22 kHz (16-bit resolution).

Data were collected from March 2020 to June 2021, including the full lockdown period related to the COVID-19 pandemic (i.e., March–April 2020). A subset of data encompassing one calendar year (1/4/2020–31/3/2021) was analyzed in detail.

Processing and visualizing tools based on Python code were specifically developed for SOUNDSCAPE by the University of Gdansk and CNR-ISMAR in accord to Betke et al. (2015). The Jupyter Notebook interactive document for data post-processing is now freely available in ROHub, the research object management platform (Petrizzo et al. 2022a). Metrics used are sound pressure levels (SPLs) in one-third octave frequency bands (hereafter TOBs) (base 10) in units of dB re 1 $\mu Pa_{rms}$. During data analysis, a 20 second temporal observation window was applied. Percentiles values (10th, 50th, and 90th percentiles) obtained per each TOB have been here used as metric, employing the median rather than the arithmetic mean as measure of tendency in the dataset. This is in accord with the most recent recommendations from the Technical Group on Underwater Noise (Sigray et al. 2021). The complete dataset of 20 seconds averaged SPLs output files was recently released by Petrizzo et al. (2022b, 2023).

The SOUNDSCAPE project dealt specifically with monitoring continuous low-frequency sound pressure as requested by the MSFD Descriptor 11.C2. Considering the sound emitted by shipping and the hearing ability of different marine species, frequencies suggested for this descriptor are one-third octave bands centered at 63 and 125 Hz. Therefore, TOB's centered frequencies of 63 and 125 Hz (re 1 $\mu Pa_{rms}$) were the primary focus of the data processing within SOUNDSCAPE, although the sound

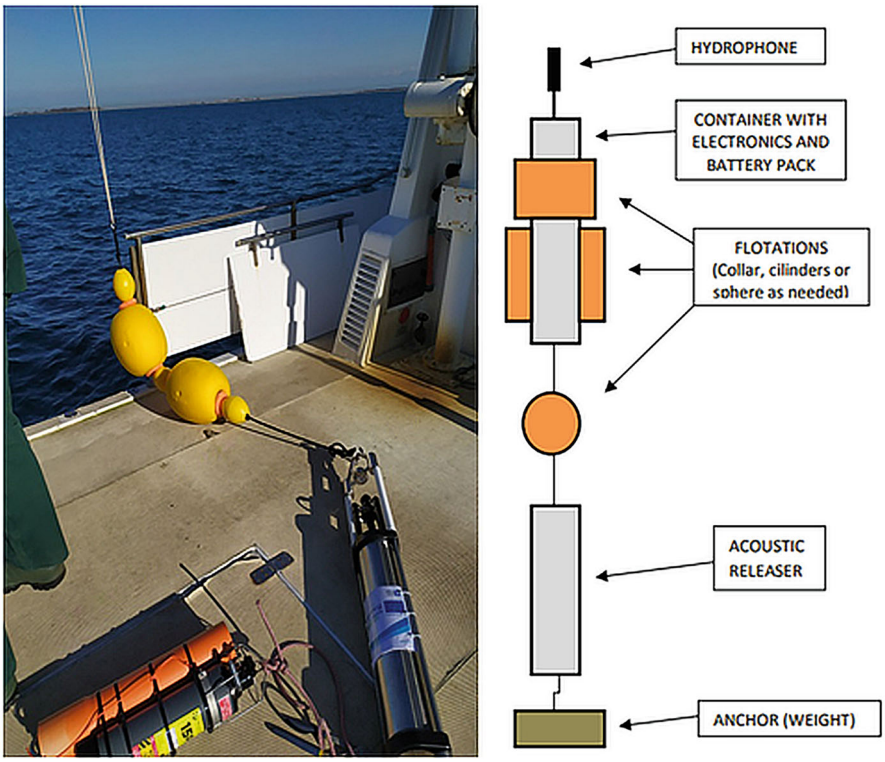



**Fig. 2** Equipment picture (left) and scheme of the autonomous passive underwater acoustic recorder (APUAR) mooring (right); the rig design includes an anchor, an acoustic releaser, the APUAR itself is secured with ropes, and flotations that assure a positive buoyancy at about three meters above the sea bottom. For a detailed APUAR description, please refer to Petrizzo et al. (2023)

measurements themselves covered a wider frequency range (10 Hz–20 kHz). Interestingly, these bands already proved to be predictive of local predominant ship type over time (Picciulin et al. 2015). Two other bands were specifically defined within the project: the 250 Hz and the 4 kHz TOBs, based on two target species identified by the SOUNDSCAPE project. The 250 Hz TOB has been chosen because it falls in the acoustic sensitivity of the target species loggerhead sea turtle (*Caretta caretta*): Its best hearing range falls between 100 and 400 Hz (Martin et al. 2012). The 250 Hz band also falls in the hearing sensitivity of range of most fish species (Ladich and Fay 2013); consequently, it was considered a good proxy to evaluate possible impact on non-mammalian marine species that are relevant for fisheries in NAS. Finally, 250 Hz TOB was found to contain a greater amount of the noise from small vessels, that are highly diffusive in coastal areas, being, at the same time, responsive to noise exposure from large vessels as the 125 Hz band (Merchant et al. 2014).

The bottlenose dolphin (*Tursiops truncatus*) was the other target species identified by the SOUNDSCAPE project. Its hearing sensitivity peaks between 15 and

130 kHz but has a lower but still present acoustic sensitivity between 1 and 10 kHz (Lawson et al. 2001). In the context of SOUNDSCAPE, 4 kHz TOB frequency was chosen as proxy for possible human-generated sound effect on the species. This accords with Jensen et al. (2009) that estimated the potential masking impacts on bottlenose dolphin communication by considering five TOBs with centered frequencies from 4–10 kHz.

Yearly SPL values collected at basin scale are presented here in compliance with the requests of the EU-MSFD. Figure 3 represents the nine NA monitoring stations as circles with different diameter, sized according to their median value (50th percentile) calculated for the 63 Hz, 125 Hz, 250 Hz, and 4 kHz TOBs. These maps graphically indicate those values that are exceeded for the 50% of the recording time.

Overall, for the MSFD frequencies (63 Hz and 125 Hz TOBs), the stations MS3 (Ancona), MS6 (Losinj), and MS1 (Venice) show lower median annual SPLs (63 Hz and 125 Hz TOBs) compared to the other stations, whereas MS2 (Rimini), MS4 (Trieste), MS8 (Split), and MS9 (Ivana) were the noisiest locations. A difference of about 30 decibels is found in the medians between the quietest and the noisiest monitoring stations; in its turn, this confirms the spatial variability in the NAS underwater sound levels.

Figures 4 and 5 show the 10th and 90th percentiles SPLs: for the 10% of time underwater noise values NAS recordings reach values up to 112–115 dB re 1 μPa, mostly accounting the human-made noise contributes for the total noise budget in these frequencies. If 250 Hz TOB median SPLs distribution largely reflects the one

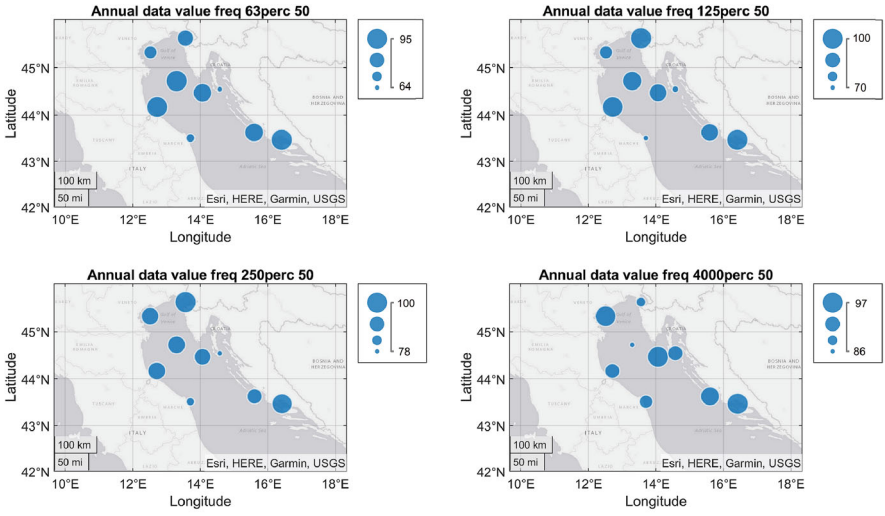



**Fig. 3** Representation of nine NA monitoring stations as circles with different diameters that are sized according to their total 50th percentile (dB re 1 μPa) calculated for the TOBs centered at 63 Hz and 125 Hz over one year of continuous recordings

observed for the case of 63 Hz and 125 Hz TOBs, the same is not true for the 4 kHz TOB, whose highest SPLs are found in MS8 (Split) but also in MS1 (Venice) and MS5 (Susak-Losinj¸ Fig. 4).

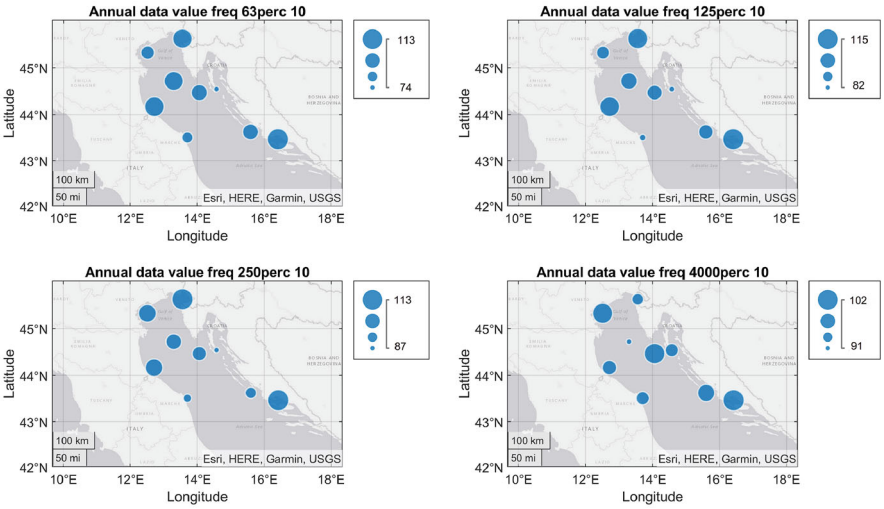



**Fig. 4** Representation of nine NA monitoring stations as circle with different diameter, sized according to their total 10th percentile (dB re 1 μPa) calculated for TOBs centered at 63 Hz and 125 Hz on one year of continuous recordings

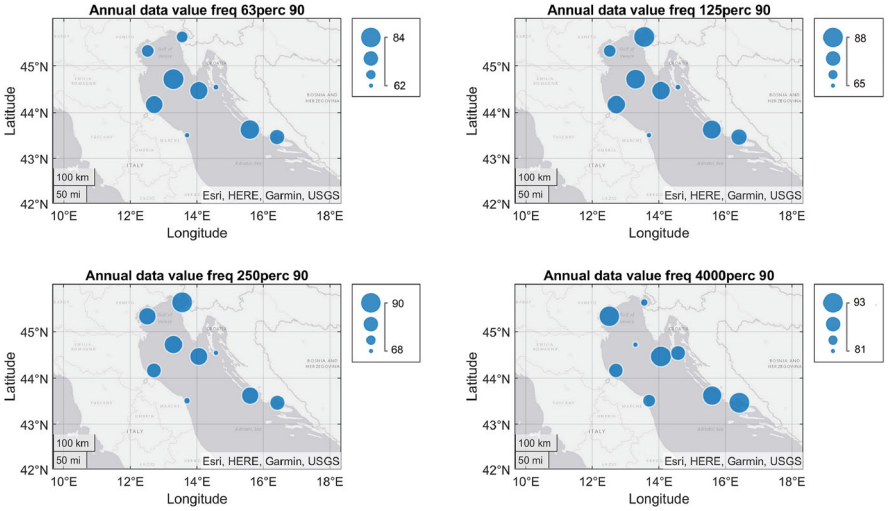



**Fig. 5** Representation of nine NA monitoring stations as circle with different diameter, sized according to their total 90th percentile (dB re 1 μPa) calculated for TOBs centered at 63 Hz and 125 Hz on one year of continuous recordings

## Conclusions

SOUNDSCAPE Interreg project developed an underwater sound monitoring project run at the basin scale in the Northern Adriatic Sea. The acoustic evaluation was expanded into uncovered areas by including the development of sound modeled maps (see Ghezzo et al. present issue). Higher levels in the 63 and 125 Hz TOBs were found where shipping activity was dominant, such as close to ports for MS4 (Trieste) and MS8 (Split) and along shipping lines, as MS9 (Ivana). This suggests that the observed heterogeneity was mainly driven by identifiable sources of human activity. Such a conclusion is further confirmed by the maximum values in the 10th percentile, being lower for the 4 kHz than the 63 and 125 Hz TOBs, the latter having been chosen as proxy of shipping activities. The SOUNDSCAPE dataset establishes baseline levels for the assessment of future trends in the Northern Adriatic Sea, which is particularly relevant since the commercial marine traffic of NAS ports is expected to grow in the next decade (Kocian 2014). The data here presented suggests for the need of mitigation measures in the NAS areas: These areas are relevant for environmental and species conservation and the underwater noise can potentially harm marine organisms in a variety of ways (Duarte et al. 2021).

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# Soundscape of the Northeast Pacific Ocean Revisited

27

Michael A. Ainslie, Rex K. Andrew, Peter L. Tyack, Michele B. Halvorsen, Justin M. Eickmeier, Alexander O. MacGillivray, Sophie L. Nedelec, and Stephen P. Robinson

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**Abstract**

The measured changes in northeast (NE) Pacific Ocean ambient sound levels in the 63–125 Hz bands are explained by the contribution from shipping to the sound energy budget using a globally averaged sound energy model. The energy model fails to identify the dominant source of sound in the 32 and 40 Hz bands because its estimates deviate from the measured levels. More research is required to resolve this discrepancy. Ships in cold deep water contribute more to ambient sound than in warm shallow water. This suggests a potential mitigation action for the NE Pacific Ocean soundscape that would involve avoiding cold deep routes where a warmer or shallower route is available.

**Keywords**

Ambient sound · Baleen whale calls · Global average spectrum · Global shipping fleet · Sound energy · Sound power · Sound pressure · Source level

## Introduction

### Eightfold Increase in Global Fleet Tonnage: 1964–2017

The amount of sound energy in the ocean associated with a given type of source is proportional to the number of such sources and the mean sound power radiated by each. While one can assign a value, $M$, to the number of ships and estimate their power based on existing measurements, it is problematic to do the same for other man-made sources (e.g., airgun arrays) and natural sources (e.g., marine animals, breaking waves, seismic activity). To analyze non-shipping contributions, empirical ambient sound measurements can be considered instead.

Apart from a small decline in the 1980s, the global commercial shipping fleet size increased monotonically between 1955 and 2017 (Fig. 1). During the 1964–2017 period, the number of vessels increased by a factor 3; the average tonnage of each ship also increased, such that the global tonnage increased by a factor of 8 in the same period (Ainslie et al. 2021; Table 1).

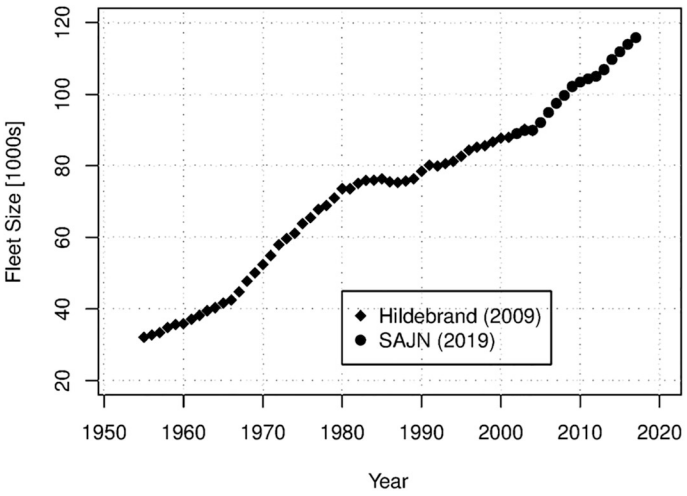



**Fig. 1** Based on Hildebrand (2009) and Shipbuilders' Association of Japan (SAJN) (2019), the global fleet size nearly quadrupled between 1955 and 2017 (Source: Ainslie et al. (2021))

**Table 1** Properties of global shipping fleet since 1964 (see section "Global Average Sound Field") (Based on Ainslie et al. (2021))

| Input | 1964 | 1998 | 2017 |
|---|---|---|---|
| Number of vessels $(M_0)/10^3$ | 40 | 86 | 116 |
| Source depth $(d)$/m | 4.24 | 5.00 | 6.05 |
| Total gross tonnage $(G)/10^9$ | 0.15 | 0.53 | 1.29 |
| Average gross tonnage per ship $(g)/10^3$ | 3.71 | 6.19 | 11.14 |
| Low-frequency level increase since 1964, in decibels, estimated as $10\log_{10}\frac{M_0\ d^2}{(M_0\ d^2)_{1964}}$ | 0 | 4.7 | 7.7 |
| $10\log_{10}\frac{G\ g^{-1/3}}{(G\ g^{-1/3})_{1964}}$ | 0 | 4.7 | 7.8 |
| $10\log_{10}\frac{G}{(G)_{1964}}$ | 0 | 5.5 | 9.3 |

## Observed 3–12 dB Increase in Sound Pressure Level: 1964–1998

Measurements at Point Sur in the northeast (NE) Pacific Ocean (Andrew et al. 2002) show an increase in sound pressure level (SPL) of 3–12 dB in the 10–400 Hz frequency range between the mid-1960s and late 1990s. The largest increase (8–12 dB, corresponding to a rate of 0.24–0.35 dB/a, where "a" is the international standard symbol for year (ISO 80000-3)) occurred in the 10–50 Hz frequency range. Similar increases in the second half of the twentieth century are reported by Ross (2005), McDonald et al. (2006), Andrew et al. (2011), and Chapman and Price (2011). Andrew et al. (2002) present measurements for four sites in the 16–100 Hz decidecade frequency bands (Fig. 2). (Here and throughout, decidecade bands are identified by their nominal center frequencies, in this case 16, 20, 32, 40, 50, 63,

80, and 100 Hz.) The increase in the power-averaged level over the four sites is 2–9 dB (Fig. 3).

Ships are known to radiate sound (Ross 1976), and according to Wenz (1962), the 10–100 Hz "ambient noise" frequency band is dominated by "oceanic traffic."

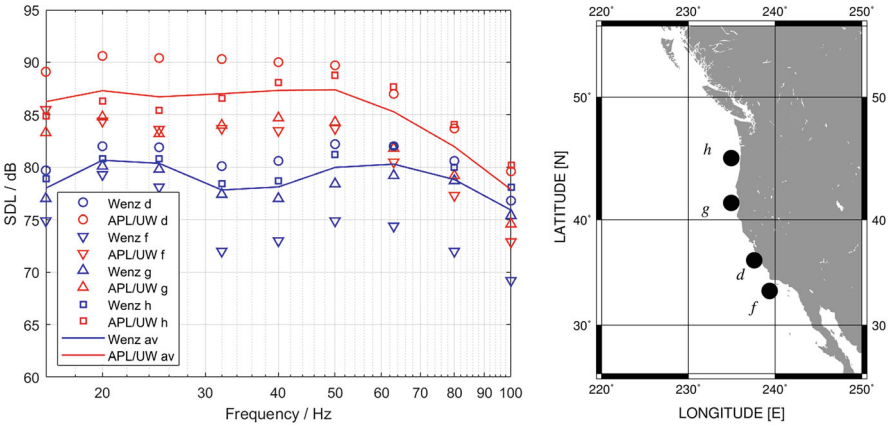



**Fig. 2** Left: Decidecade band power spectral density level (SDL; re 1 μPa$^2$/Hz) versus decidecade band center frequency for 1963–1965 ("Wenz") and 1994–2001 ("APL/UW") averaged over four measurement stations (d: Point Sur, California; f: San Nicolas Island, California; g: southern Oregon; h: southern Washington) in the NE Pacific Ocean. Right: Location of the measurement stations (Right-hand figure reproduced from Andrew et al. (2011) with permission)

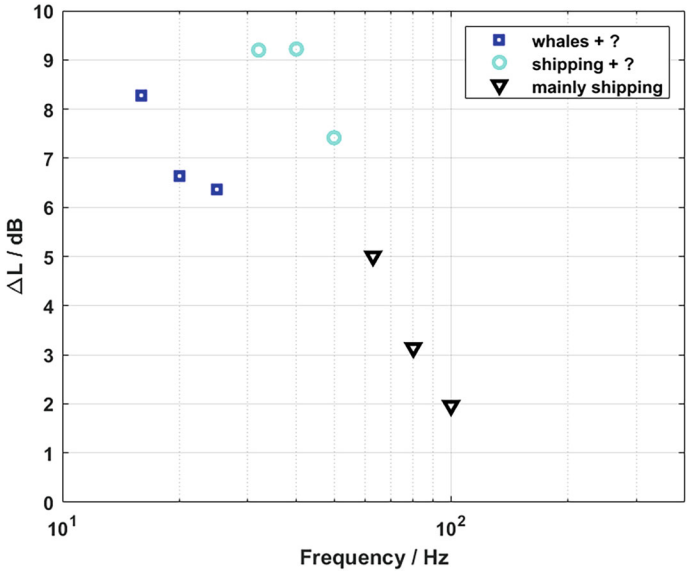



**Fig. 3** Increase in the level versus decidecade band center frequency from 1963–1965 ("Wenz") to 1994–2001 ("APL/UW") averaged over the four sites (d, f, g, and h)

Perhaps for this reason, increases in the 10–100 Hz frequency range are sometimes attributed to shipping traffic (Ross 1993; Erbe et al. 2019). However, while there is strong evidence that the 63–125 Hz decidecade bands are dominated by sound from ships (Ainslie et al. 2021), baleen whale calls make important contributions to other decidecade bands, especially those centered at 16 and 20 Hz (Curtis et al. 1999; Burtenshaw et al. 2004; McDonald et al. 2006). Because whales are known to dominate ocean sound at frequencies between 15 and 25 Hz during some seasons in the NE Pacific Ocean, these frequencies are labeled "whales + ?" in Fig. 3.

## Claimed 15–20 dB Increase in the Sound Pressure Level

According to Hildebrand (2009), available data "provide evidence for an overall increase of at least 20 dB in low-frequency ambient noise from preindustrial conditions to the present," while Frisk (2012) and Duarte et al. (2021) claim increases of 19 dB (1950–2007) and 15 dB (in 50 years), respectively.

The statements of Hildebrand (2009) and Frisk (2012) are partly based on Ross (2005), who demonstrated an increasing trend at 50 Hz of about 11 dB in a 20-year period between the 1950s and 1970s. More precisely, Ross's Fig. 5 provides evidence of an average increase of 0.55 dB/a from 1954 to 1974. On this basis, it seems reasonable to assume a 7.7 dB average increase from 1950 to 1964 at 50 Hz. At the same frequency, there was a subsequent increase of 7.3 dB from 1964 to 1998 (Fig. 3). The total is 15.0 dB in the 48 years from 1950 to 1998, corresponding to an average rate of 0.31 dB/a at 50 Hz. There is also an 11 dB increase at San Nicolas at 16 Hz (Andrew et al. 2011), a band contributed to by baleen whale calls (Table 2, Fig. 5). Finally, Fig. 3 shows an average increase of 2–5 dB in 34 years (1964–1998) in the frequency range corresponding to the 63–100 Hz decidecade bands.

The possibility of larger increases cannot be ruled out, but a careful search revealed no evidence to support claims of an average increase exceeding 15 dB over any time period. The observations in the Pacific Ocean are consistent with an

**Table 2** Changes in SPL in the 16–125 Hz decidecade bands during a period of approximately 34 years (nominally 1964–1998) in the coastal NE Pacific Ocean

| Frequency range/Hz | ΔL/dB | Rate of increase/(dB $a^{-1}$) | Notes |
|---|---|---|---|
| 16–25 | 6.4–8.3 | 0.19–0.24 | Baleen whale calls (Curtis et al. 1999; Burtenshaw et al. 2004; McDonald et al. 2006; Andrew et al. 2011); possibly some shipping |
| 32–50 | 7.4–9.2 | 0.22–0.27 | Some shipping (Andrew et al. 2011); some baleen whale calls (Burtenshaw et al. 2004; McDonald et al. 2006; Andrew et al. 2011);<br>the increases were highest in the 32 and 40 Hz bands, for which the level is overestimated by the global energy model with the Wales-Heitmeyer source level (Ainslie et al. 2021) |
| 63–125 | 2.0–5.0 | 0.06–0.15 | Mainly shipping (Ainslie et al. 2021) |

increase of 0.2–0.3 dB/a in decidecade bands centered between 16 and 50 Hz (Table 2). They do not support a claim of an average increase exceeding 15 dB increase in 50 years and are also not compatible with large increases across the 10–100 Hz frequency range (increases exceeding 5 dB have only been reported for frequencies up to 50 Hz).

## Contribution to Ambient Sound from Ships

Shipping provides a ubiquitous contribution to the soundscape in the NE Pacific (Curtis et al. 1999), with ambient sound levels varying as a functional of seasonal changes in sound propagation. In the 80 Hz decidecade band, the levels reach a maximum in March (Fig. 4), and SPL tracks sea surface temperature (SST) in a mechanism closely related to the afternoon effect (Ainslie et al. 2021). The pattern is repeated over 12 years, especially at site h, and in all bands between 63 and 125 Hz, consistent with a continuously radiating source of sound close to the sea surface (a shipping lane), indicating that ships dominate the coastal NE Pacific Ocean soundscape in this frequency range (Ainslie et al. 2021).

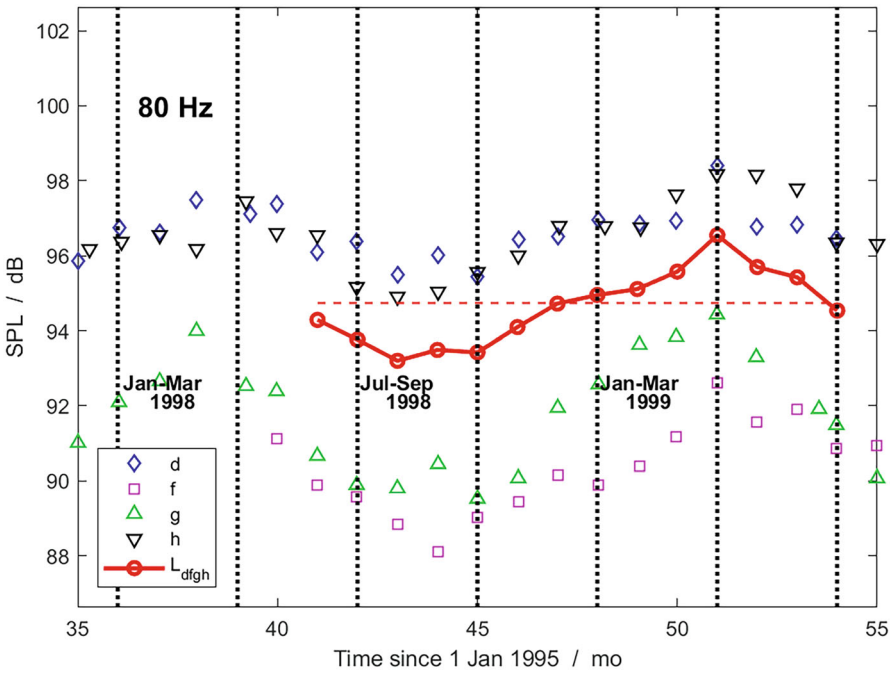



**Fig. 4** Time series of monthly medians of decidecade band SPL (re 1 μPa$^2$) for each of the four sites (d, f, g, and h) versus time in the 80 Hz decidecade band. The red curve is averaged over all four sites. This plot shows 14 months from May 1998 to June 1999, inclusive (Reproduced from Ainslie et al. (2021) with permission)

## Contribution to Ambient Sound from Baleen Whales

McDonald et al. (2009) and Burtenshaw et al. (2004) point out that during the period covered in Fig. 5, fin and blue whales in the NE Pacific Ocean had a spectral peak at 16–18 Hz with blue whale harmonics in the 32, 50, and 63 Hz decidecade bands. McDonald et al. (2009) point out the spectral peak from blue whales decreased from around 22.5 Hz in 1964–1965 to 16 Hz by the early 2000s. Some of the sound in the 16 Hz decidecade band (Fig. 5) undoubtedly originates from whales, and perhaps most of it, but the possibility of a contribution from some other (natural or anthropogenic) source or local effect on sound propagation that also peaks in August–December cannot be ruled out.

Fin whales are present year round at coastal sites in the NE Pacific Ocean (Curtis et al. 1999; Burtenshaw et al. 2004). They transmit a pulse at 15–20 Hz, with low energy in harmonics. Other fin whale calls also have energy in the 40 and 80 Hz bands (Buchan et al. 2019; Wiggins and Hildebrand 2020). Thus, baleen whales contribute to all decidecade bands from 16 to 80 Hz.

Blue whale calls are recorded at the four recording sites mainly between August and December (Curtis et al. 1999; Burtenshaw et al. 2004; McDonald et al. 2006).

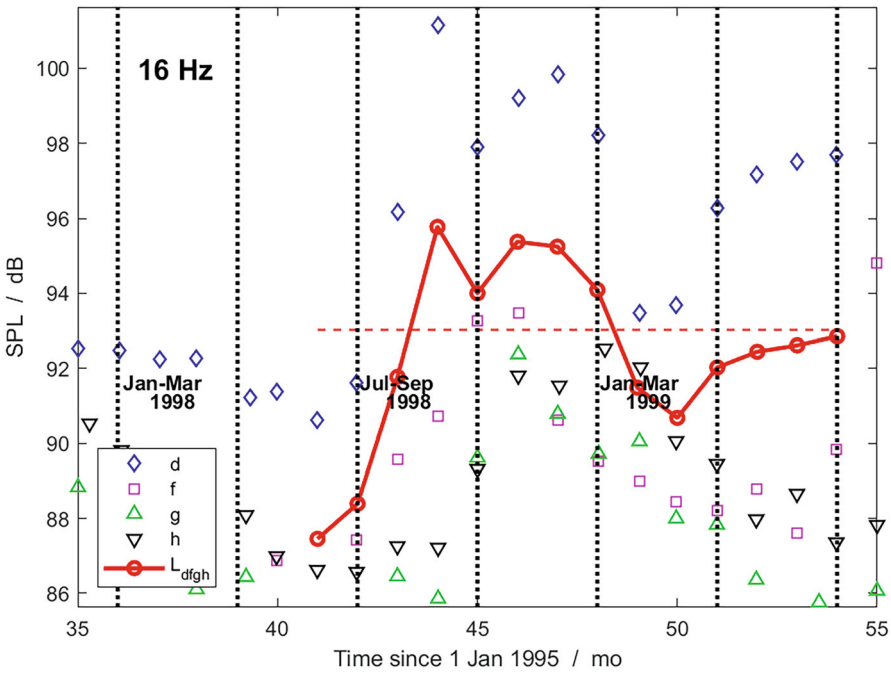



**Fig. 5** Time series of monthly medians of decidecade band SPL (re 1 μPa$^2$) for each of the four sites (d, f, g, and h) versus time in the 16 Hz decidecade band. The red curve is averaged over all four sites. This plot shows 14 months from May 1998 to June 1999, inclusive (Reproduced from Ainslie et al. (2021) with permission)

The 16 Hz decidecade band, which corresponds to a dominant frequency of blue and fin whale calls during the recording period and locations covered in Fig. 5, also has a strong seasonal peak (roughly from August to December; Fig. 5), corresponding to the blue whale calling season off the Californian coast (Curtis et al. 1999; Burtenshaw et al. 2004). The blue whale signal is present at all four sites and is most prominent at sites d and f (Ainslie et al. 2021). The whales arrive later in a northern habitat, and the calling season extends from October to February off the Washington coast (Burtenshaw et al. 2004).

## Motivation for a Global Sound Energy Budget

Managing the risk of underwater sound to wildlife requires understanding what doses of sound cause adverse responses, and measurements of where sound levels may cause adverse effects in areas where sensitive species are found (Tyack et al. 2022) or to which access by sensitive species is prevented. Once adverse effects are observed or expected, mitigation requires understanding of which sound sources contribute to the problem. Management actions to reduce sound enough to reduce the risk require information on how much the main human sources contribute to causing these effects.

Unweighted sound exposure (a physical quantity, the definition of which is independent of any biological receiver) is proportional to average sound energy density. The source contributing the highest average amount of energy to the sound field in each frequency band therefore contributes the largest sound exposure in that band (Ainslie and Dekeling 2011). Thus, a ranking of source types from highest to the lowest sound energy (in joules) facilitates appropriate mitigation measures by targeting those sources making the largest contribution to the sound exposure. Successful risk management and mitigation rely on identifying the source correctly (Hatch et al. 2008).

Shipping noise has the potential to disrupt marine life in multiple ways. The contribution from shipping to the sound energy budget is estimated in the 30–500 Hz frequency range, and much concern about effects of shipping noise focuses on baleen whales, which specialize in low-frequency hearing and communication. Around September each year, North Atlantic right whales transition from their foraging season and prepare for migration. Therefore, the expectation is for stress hormone levels to be elevated by late September. However, after the shipping noise decreased in the North Atlantic following attacks on 11 September 2001 (“9/11”), samples taken from North Atlantic right whales after 9/11 unexpectedly showed *lower* levels of stress hormones relative to earlier in that year. The decrease in stress hormones during the period when noise levels decreased suggests that normal shipping noise levels could be chronically elevating stress in this species (Rolland et al. 2012). Several modeling efforts suggest that shipping noise decreases communication range in baleen whales because of the overlap in the frequency range of the whale calls and shipping noise (e.g., Hatch et al. 2012; Putland et al. 2018; Cholewiak et al. 2018). Baleen whales have mechanisms to compensate for the risk that noise will mask signals, including calling louder, shifting call frequencies out of the noise range, and repeating or lengthening calls (Tyack 2008). Models of

masking that do not account for these compensation mechanisms will overestimate masking effects (Tennessen and Parks 2016). However, there are limitations to these adaptations as they reach a biological limit, e.g., in how loudly an animal can call.

Vessel noise can also harm fishes and marine invertebrates. Reduced reproductive output in a species of coral reef fish was observed at SPL (re 1 $\mu Pa^2$; in the frequency band 108–2046 Hz) of 123 dB or sound particle acceleration level (PAL re 1 $\mu m^2/s^4$, in the same frequency band) of 93 dB (Nedelec et al. 2017), and in a marine invertebrate at SPL of 90 dB or PAL of 73 dB (Nedelec et al. 2014). Vasconcelos et al. (2007), Slabbekoorn et al. (2010), Stanley et al. (2017), and Putland et al. (2018) suggest that shipping noise decreases communication range in fishes because of the overlap in the frequency range of the fish hearing and shipping noise. Rogers et al. (2021) point out that masking can occur when anthropogenic noise overlaps with signals in time, frequency, and direction. The global extent of shipping activity is such that most species of marine fishes and invertebrates that can detect sound are likely to be able to detect ship sound.

According to Erbe et al. (2019), increased shipping is "responsible for the steady rise in ambient noise at low frequencies (10–100 Hz) in many ocean regions." However, large increases occur in bands containing clearly identifiable contributions from baleen whale vocalizations (Curtis et al. 1999; McDonald et al. 2006; Andrew et al. 2011), especially the 16 and 20 Hz decidecade bands (Fig. 3, Table 2), implying that part of the increase in these bands might be caused by increasing baleen whale populations. If this is true, then incorrect identification of the origin of, for example, the observed 8–9 dB increase in the 16–50 Hz decidecade bands may lead to poor management decisions and unexpected results. If the increase in one or more of these bands is caused primarily by an increase in the baleen whale population, then reduction of shipping noise may have less of a quieting effect than expected.

## Sound Energy Budget

Wherever there is sound, there is sound energy. For a medium of density $\rho$ and sound speed $c$, the energy density, $H_V$, is proportional to the mean-square sound pressure, $\overline{p^2}$,

$$H_V = \frac{1}{\rho c^2}\overline{p^2} \tag{1}$$

or mean-square sound particle velocity, $\overline{u^2}$,

$$H_V = \rho\overline{u^2}. \tag{2}$$

Integrating the energy density over all space results in the total sound energy (cf. Eq. 12),

$$H_{\text{tot}}(f) = \int H_V \, \mathrm{d}V, \tag{3}$$

which is related to the globally averaged mean-square sound pressure (Ainslie et al. 2021).

## Purpose and Scope

The purpose of this chapter is to establish a firm foundation for further research by quantifying the contribution from shipping to the global sound energy spectrum in the 30–500 Hz frequency range (delimited by validity of Wales and Heitmeyer (2002) at low frequency and the presence of wind-generated sound at high frequency) and correcting misconceptions arising from observed increases in the level in the frequency range 10–100 Hz.

## Terminology

Acoustical terminology follows ISO 18405:2017. Standard one-tenth decade (i.e., decidecade) frequency bands (IEC 61260-1:2014) are identified by their nominal center frequencies. International standard unit symbols are used for decibel (dB), neper (Np), and year (a) (ISO 80000).

# Methods

## Modeling

### Transfer Function

The quantity of interest (the modeling output) is the total sound energy in the sea (symbol $H_{\text{tot}}$), which is related to the total source power (symbol $W_{\text{tot}}$) up to the sea surface critical angle (symbol $\psi$). The role of the surface critical angle is to determine the amount of sound energy trapped in the water column (steeper paths lose energy on reflection from the seabed and do not contribute to the global sound budget in the frequency range of interest). The transfer function $\tau$ is defined as the ratio of output to input, i.e.,

$$\tau(f) \equiv \frac{H_{\text{tot}}(f)}{W_{\text{tot}}(f)}. \tag{4}$$

the total energy is therefore

$$H_{\text{tot}}(f) = \tau(f) W_{\text{tot}}(f), \tag{5}$$

where for $M_i$ sources of type $i$, each radiating power $W_i$

$$W_{\text{tot}}(f) = \sum M_i W_i(f) \tag{6}$$

and for sources of effective depth $d_i$ and source factor $S_i$ (Ainslie et al. 2021)

$$W_i(f) = \frac{4\pi S_i(f)\sin\psi}{\rho c}\left(1 - \frac{\sin 2k(f)d_i\sin\psi}{2k(f)d_i\sin\psi}\right) \tag{7}$$

where $k$ is the acoustic wavenumber $2\pi f/c$. The transfer function is inversely proportional to the absorption coefficient $\alpha$ (Ainslie et al. 2021)

$$\tau(f) = \frac{1}{2\alpha(f)c}. \tag{8}$$

## Global Average Sound Field

It follows from Eqs. 1 and 5 (using Eqs. 4 and 7) that

$$\langle P_f \rangle = \frac{2\pi}{V_{\text{tot}}}\sum\frac{M_i S_{f,i}(f)\sin\psi}{\alpha(f)}\left(1 - \frac{\sin 2k(f)d_i\sin\psi}{2k(f)d_i\sin\psi}\right), \tag{9}$$

where $V_{\text{tot}}$ is the total volume of all the world's oceans; $P_f$ and $S_f$ are the spectral densities of the mean-square sound pressure, $\overline{p^2}$, and source factor, $S$, respectively; and $<>$ indicates a global average. Equation 9 is a function of six variables, three of which ($S_f$, $kd$, $M$) are properties of the sources (ships) and the other three ($V_{\text{tot}}$, $\psi$, $\alpha$) are properties of the propagation medium. At low frequency ($2kd\sin\psi \ll 1$), it simplifies to

$$\langle P_f \rangle = \frac{4\pi}{3V_{\text{tot}}}\sum M_i \cdot (kd_i)^2 \cdot \sin^3\psi \cdot \frac{S_{f,i}(f)}{\alpha(f)}. \tag{10}$$

The low-frequency dependence on tonnage (Table 1) arises from Eq. 10 by noting that the factor $Md^2$ is approximately proportional to $G/g^{1/3}$. As observed by McDonald et al. (2006), the use of $10\log_{10}G$ would predict an increase of about 6 dB in the last three decennia of the twentieth century, similar to the 5 dB increase predicted by $10\log_{10}G\ g^{-1/3}$ (Ainslie et al. 2021). By contrast, the $10\log_{10}G^2$ dependence assumed by Frisk (2012) would overestimate the observed increase in 63–125 Hz decidecade bands (dominated by shipping) by about 8 dB.

The proportionality with $\sin^3\psi$ in Eq. 10 leads to a seasonal dependence because the critical angle decreases monotonically with increasing SST. A 4 °C temperature difference between summer and winter thereby leads to 2.4 dB higher SPL in the winter for the same shipping traffic (Ainslie et al. 2021). The same proportionality leads to a dependence on water depth because the critical angle increases with increasing water depth in deep water.

## Source Factor and Absorption Coefficient

Following Ainslie et al. (2021), the Wales-Heitmeyer (Wales and Heitmeyer 2002) formula is used for $S_f$ (Fig. 6) (increased by between 1.1 and 3.2 dB, depending on frequency, to account for the difference between the mean value of a level and the level of a mean value), and assume the absorption coefficient $\alpha(f)$ takes the form

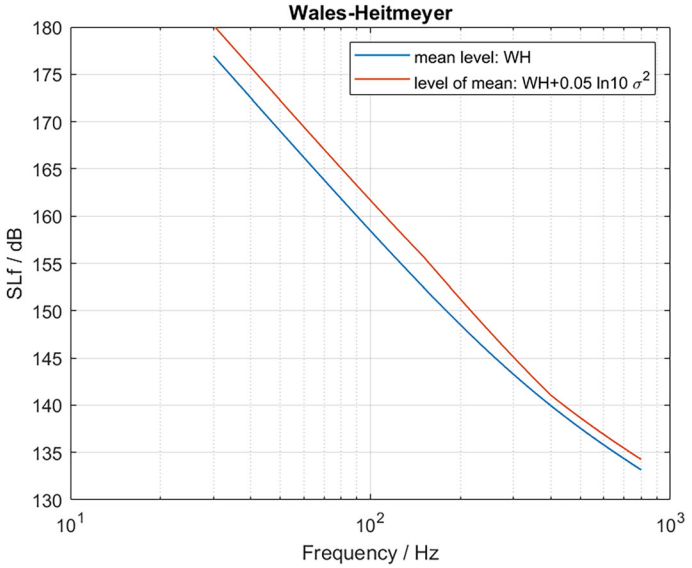



**Fig. 6** Wales-Heitmeyer source spectral density level (re 1 μPa$^2$ m$^2$/Hz) versus frequency calculated as mean level (blue) and level of mean source spectral density (red)

$$\alpha(f) = \alpha_{\min} + \alpha_0(f), \tag{11}$$

where $\alpha_0$ is the contribution from molecular relaxations (mainly boron) and $\alpha_{\min}$ is an empirical constant to be determined, possibly in the 0.2–0.4 Np/Mm range for the North Pacific Ocean (Kibblewhite 1989).

## Measurement

The model described in section "Modeling" (Eq. 9) predicts the globally averaged mean-square sound pressure

$$\langle P_f \rangle = \frac{1}{V_{\text{tot}}} \int P_f \mathrm{d}V. \tag{12}$$

To test the prediction, one needs a measurement of this global average. The ideal solution would be to employ a global network of hydrophones, for example, using the Comprehensive Nuclear-Test-Ban Treaty Organization network (Robinson 2021; Gibbons 2022) or Argo floats (Gould 2005). However, in practice this work is limited by the data available to the authors at the time of writing, at four sites in the NE Pacific Ocean (Andrew et al. 2011). Following Ainslie et al. (2021), an average over these four stations is used as a proxy for a global average.

## Results

The global energy expression (Eq. 10) requires an average low-frequency absorption coefficient. Predicted SPL for multiple values of the low-frequency absorption coefficient (0.05 Np/Mm $< \alpha_{\min} <$ 8 Np/Mm) is compared with measurements in Fig. 7. For each $\alpha_{\min}$, the proportion of the global shipping fleet traversing over deep ocean (denoted $x$) is adjusted to match the measurement at 80 Hz. Different spectral slopes are associated with different low-frequency absorptions $\alpha_{\min}$ (the lower $\alpha_{\min}$, the greater the slope). The value of $\alpha_{\min}$ that best matches the slope in the range 50–100 Hz is 0.05 Np/Mm, but this value is outside the expected range of 0.2–0.4 Np/Mm. Additional hindcasts are made for the years 1964 and 2017 using $\alpha_{\min} = 0.2$ Np/Mm and $x = 0.31$ (corresponding to 31 % of the global fleet operating in deep water and $x/\alpha_{\min} = 1.6$ Mm/Np; Table 3) in Fig. 8.

Figures 7 and 8 demonstrate that the level and spectral slope in the 63–125 Hz decidecade bands (and the 1964–1998 increase in level) are explained by Eq. 10 with the Wales-Heitmeyer source-level spectrum, strengthening further the case for identifying these bands as dominated by shipping (Ainslie et al. 2021).

The theory also explains the 50 Hz decidecade band level, but it overestimates the 32 and 40 Hz band levels and underestimates the level increase at 32 and 40 Hz. Thus, the dominant source type for these decidecade bands remains unidentified.

The same information can be converted back to energy (Fig. 8 right) using

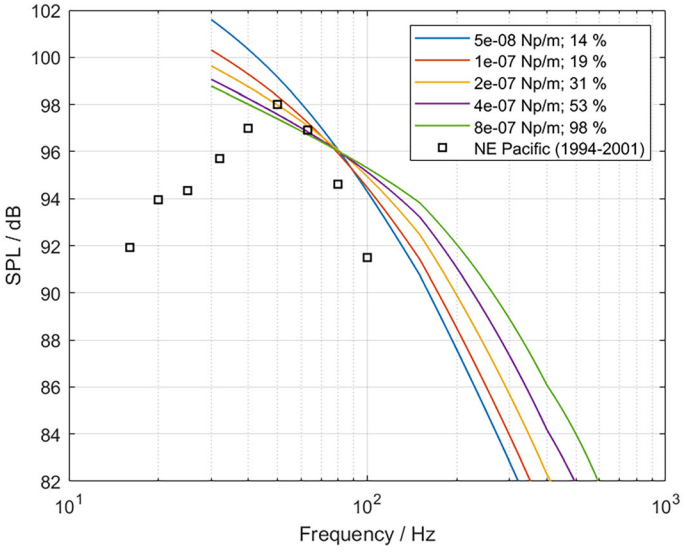



**Fig. 7** Globally averaged decidecade band SPL (re 1 μPa$^2$) versus decidecade band center frequency, scaled to match the 80 Hz measurement for 1994–2001 (0.05 Np/Mm $< \alpha_{\min} <$ 0.8 Np/Mm)

**Table 3** Combinations of $\alpha_{\min}$ and $x$ consistent with late 1990s measurements at 80 Hz in the NE Pacific Ocean

| $\alpha_{\min}$/(Np Mm$^{-1}$) | $x$ | (x/$\alpha_{\min}$)/(Mm Np$^{-1}$) | $\tau_{\max}$/(s Np$^{-1}$) | Minimum decay rate/(Np ks$^{-1}$) |
|---|---|---|---|---|
| 0.05 | 0.14 | 2.8 | 6710 | 0.15 |
| 0.10 | 0.19 | 1.9 | 3360 | 0.30 |
| 0.20 | 0.31 | 1.6 | 1680 | 0.60 |
| 0.40 | 0.53 | 1.3 | 840 | 1.19 |
| 0.80 | 0.98 | 1.2 | 420 | 2.38 |

$$H_{\text{tot}} = \frac{V_{\text{tot}}}{\rho c^2} \langle P_f \rangle. \tag{13}$$

Figure 8 is seen as the first step in constructing a global sound energy budget for both man-made (primarily ships and airgun arrays) and natural sources (primarily wind and whales) in the 10–1000 Hz frequency range (Sertlek et al. 2019).

These measurements are compared with model predictions of the globally averaged spectra, calculated using the method of section "Modeling." The number of ships $M_0$ and their source depth $d$ are obtained from Table 1. In Fig. 7 for 1998, minimum absorption coefficient $\alpha_{\min}$ is varied between 0.05 and 0.80 Np/Mm, and the value of $x$ is adjusted accordingly to match 96 dB in the 80 Hz decidecade band (1 dB higher than the measured value to account for the expected difference between mean and median (Dekeling et al. 2014)). Also needed are total ocean volume $V_{\text{tot}}$ (1.33 Mm$^3$) and sea surface critical angle $\psi$ (0.2), from Ainslie et al. (2021).

The transfer function $\tau$ is equal to the time it takes the amplitude of low-frequency sound to decay by 1 Np (it is the reciprocal of the decay rate). The value of $\tau$ corresponding to 0.2 Np/Mm is approximately 1700 s/Np (Table 3).

## Potential Mitigation Measures

Ships in cold deep water contribute more to ambient sound than those in warm shallow water. This suggests a possible mitigation action involving avoiding cold deep routes where a warmer or shallower route is available, for example, by avoiding the coldest months of the year and giving preference to shallow water routes. Such options should be considered before adopting mitigation measures.

## Conclusions

During a 34-year period in the twentieth century (about 1964–1998), for four sites in the NE Pacific Ocean, the four-site average:

- Increased by 2–5 dB (about 0.1 dB/a) in the 63–125 Hz decidecade bands (mainly shipping sound)

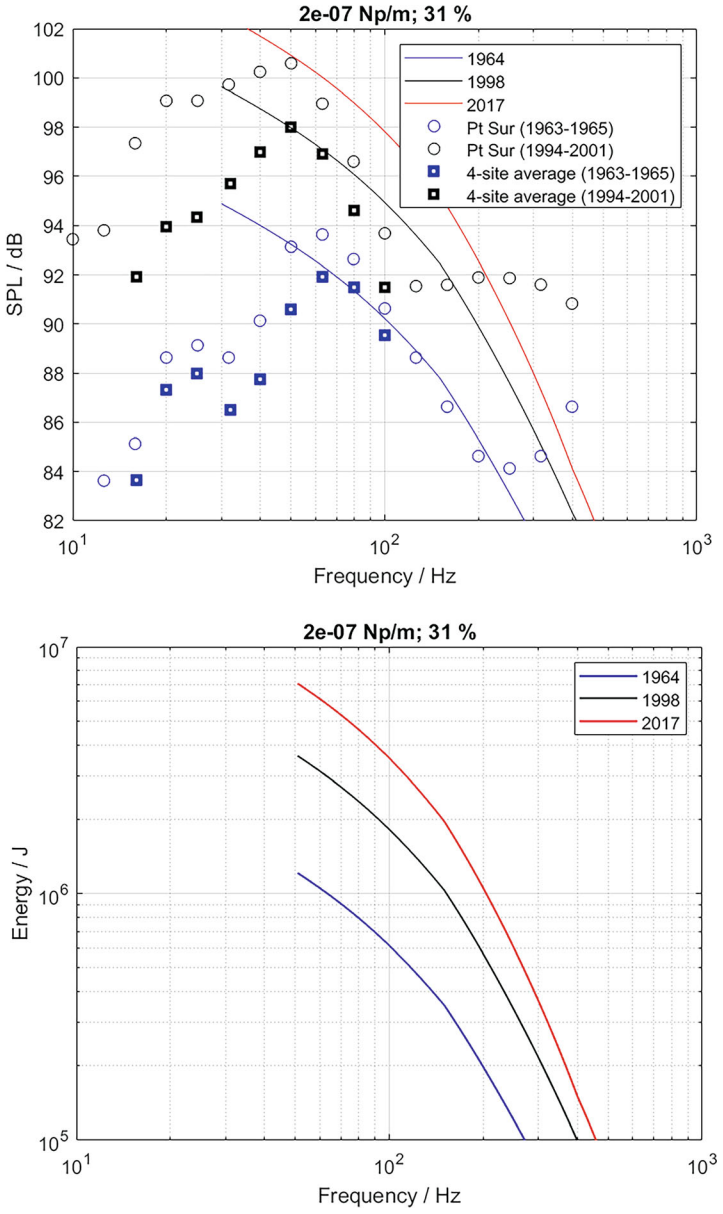



**Fig. 8** Decidecade band hindcasts for 1964, 1998, and 2017 (0.2 Np/Mm). Upper: SPL (re 1 $\mu Pa^2$) versus decidecade band center frequency. Lower: Decidecade band total sound energy versus decidecade band center frequency

- Increased by 6–8 dB (about 0.2 dB/a) in the 16–25 Hz decidecade bands (containing baleen whale calls)
- Increased by 7–9 dB (about 0.25 dB/a) in the 32–50 Hz decidecade bands

At site f (San Nicolas Island), the level increased by 11 dB at 16 Hz (containing baleen whale calls) and by 12 dB in 34 years in the 32 Hz decidecade band. At the same site, increases of up to 12 dB in the 30–50 Hz frequency range were observed in 38 years.

At 50 Hz there is evidence of a 7–8 dB increase between 1950 and 1964, implying a 15 dB increase at this frequency from 1950 to 1998.

No evidence was found at a fixed location of any increase exceeding 12 dB (1964–1998). Without ruling out the possibility of larger increases, no evidence was found of a spatially averaged increase exceeding 15 dB in any time period.

The contribution to ambient sound from ships and its potential to disrupt marine life have long been recognized. Effective management of ship noise would be facilitated by a deeper understanding of the relative contributions from ships and baleen whale calls to the 10–100 Hz ambient sound spectrum. While the ~5 dB increase observed in the NE Pacific Ocean in the 63–125 Hz decidecade bands is fully explained by changes in the number and size of ships between 1964 and 1998 (the power spectral density level varies with global tonnage $G$ and tonnage per ship $g$ as $10\log_{10} G\ g^{-1/3}$), the cause of the ~10 dB increase in the 32 and 40 Hz decidecade bands remains a mystery.

## Research Questions

Research questions for future consideration include the following:

- How much sound energy do ships contribute to the global energy spectrum up to 50 Hz?
- How much sound energy do man-made sources other than ships (e.g., airgun arrays) contribute to the global energy spectrum in the 10–100 Hz frequency range?
- How much sound energy do natural sources (e.g., wind, whales, fishes) contribute to the global energy spectrum in the 10–100 Hz frequency range?
- What causes the larger-than-predicted-level increase in the 32 and 40 Hz bands?
- What infrastructure (e.g., CTBTO, Argo) is best suited to construct an effective, global, hydrophone network to monitor the global soundscape?

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# Influence of Tugboats on the Underwater Soundscape Along the West Coast of Canada

28

Kelsie A. Murchy, Svein Vagle, and Francis Juanes

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**Abstract**

Ambient underwater noise levels have been increasing, leading to concern about the impacts to marine species. Extensive research has been conducted on the changes to ambient noise levels of large commercial vessels and potential mitigation measures. However, little is known about smaller commercial vessels (e.g., tugboats) moving throughout coastal waters and what can be done to decrease their influence on marine species. To understand changes to the ambient noise levels (20–24,000 Hz) when tugboats are present, stationary hydrophones were deployed between August and October at two locations: Cowichan Bay, British Columbia (BC) (2019/2020) and Argonaut Wharf, Campbell River, BC (2021). Log transportation tugboats were assessed in Cowichan Bay and the results show

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significantly elevated median sound pressure levels (10 dB) when log transport tugs were detected near the estuary where log storage occurs. Even though Argonaut Wharf was used as a site without log transportation tugboats, tugboats were present throughout the study period. Median sound pressure levels were also significantly elevated (6 dB) when tugboats were present in Argonaut Wharf. The results demonstrate the impact that tugboat activities have on underwater soundscapes and are the first step in understanding the impact these activities have on marine organisms and important ecosystems.

**Keywords**

Sound · Acoustic disturbance · Shipping impacts

## Introduction

Anthropogenic noise has been modifying the global soundscape causing ambient levels to increase (reviewed in Hildebrand 2009). One example is seen in the Northeast Pacific Ocean, where noise levels have increased by around 3 dB/decade for decades leading up to 2000 (McDonald et al. 2006). One main contributor to this increase is the shipping industry, which has been growing for many decades (Tournadre 2014). Shipping noise dominates low-frequency noise (40–1000 Hz), which has increased by at least 10–12 dB since the 1950s in some areas (Ross 1976; Ainslie et al. 2021). Noise produced by large commercial vessels has been extensively documented in the literature (e.g., McKenna et al. 2012, 2013; Gassmann et al. 2017), and mitigation methods are being developed to reduce their impact on the underwater soundscape (MacGillivray et al. 2019; Williams et al. 2019; Burnham et al. 2021).

One shipping noise source that has only been minimally accounted for in the literature is noise produced by tugboats. Tugboats are used in coastal regions to push or tow equipment and to guide larger ships. Previous research demonstrated tugboats produce noise below 3 kHz (frequency range: 170–2629 Hz) with a source level of 119 dB re 1 μPa (predicted) when transiting (not pulling or pushing) using a stationary hydrophone (Amron et al. 2021). When pushing a barge, Sprague et al. (2016) found that a tugboat produced noise at sound pressure level (peak) of 124 dB re 1 μPa (frequency range: 10–22,050 Hz) and a sound exposure level of 143 dB re 1 μPa (time = 0.125 s) at roughly 600 m away from a stationary hydrophone. They also documented that the frequencies most impacted by tugboat sound are frequencies below 1000 Hz. Vessel size has also been shown to have an impact on the noise produced, with tugboats over 50 m having a mean broadband radiated noise level (RNL; 10–64,000 Hz) of 186.2 dB re 1 μPa @ 1 m compared to tugs less than 50 m that have a mean RNL of 180.5 dB re 1 μPa @ 1 m (Houweling et al. 2022).

Around Vancouver Island, British Columbia, tugboats are operating in every month of the year (Houweling et al. 2022) and have the highest vessel hours detected compared to all other automatic identification system (AIS)-equipped vessels in the

area due to high volume of activity (MacGillivray et al. 2016). In addition to traditional roles, tugboats around Vancouver Island are also used to move logs into bays and estuaries where logs are stored prior to processing at pulp and sawmills. Logging booms have been documented to negatively impact estuarine habitats (Power 1987; Levy et al. 1990; Nahirnick et al. 2020), but the additional noise generated by these activities has not been quantified. The goal of this chapter was to examine the underwater noise produced by tugboat activities in two different locations on Vancouver Island, British Columbia, to document differences in sound pressure levels and frequency composition under the different activities that tugboats are used for.

## Methods

### Study Locations

Cowichan Bay, British Columbia, is an approximately 800 ha industrialized bay located at the mouth of the Cowichan River near Duncan, British Columbia on Vancouver Island (48° 44.975' N, 123° 35.922' W) (Fig. 1). Cowichan Bay has a high level of anthropogenic noise that originates from recreational boats entering and leaving two marinas and from commercial activity (e.g., tugboats and bulk carriers).

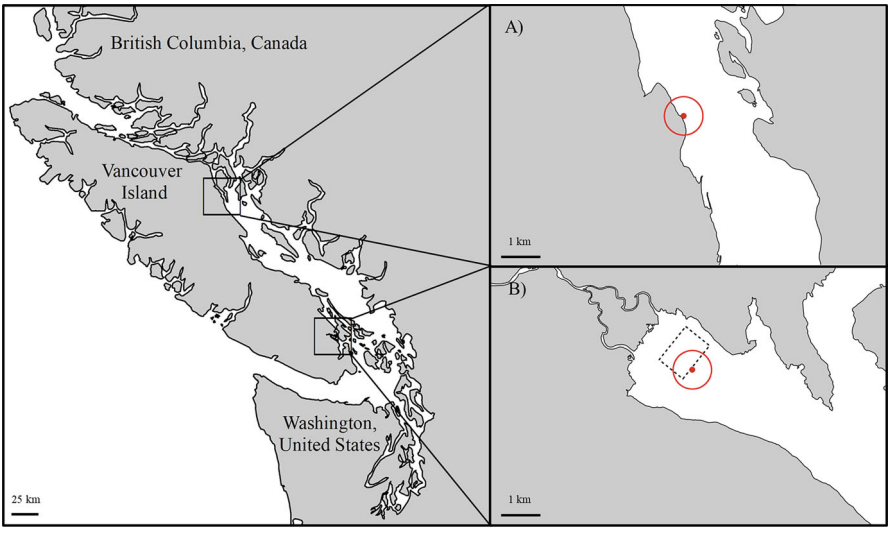



**Fig. 1** Map of Vancouver Island, British Columbia. (**a**) Argonaut Wharf in Campbell River, BC. Red dot shows SoundTrap hydrophone mooring location. (**b**) Cowichan Bay, BC. Red dot shows SoundTrap hydrophone mooring location, and black dashed box denotes the log boom sorting area in the Cowichan River estuary. Map shapefiles were obtained from the Natural Earth and BC Provincial data catalog

Cowichan Bay also has a sawmill that stores log booms in the Cowichan estuary before being brought up to the mill to be processed.

Argonaut Wharf is located at the mouth of the Campbell River, in Campbell River, British Columbia, and extends ~60 m into Discovery Passage, on the east coast of Vancouver Island (50° 2.543' N, 125° 14.834' W). Argonaut Wharf also is in an area of high anthropogenic activity including a ferry that runs every 30 min to and from nearby Quadra Island, commercial vessels (e.g., tugboats, cruise ships) transiting through, and recreational and fishing vessels entering and exiting a marina located 1 km away. Unlike Cowichan Bay, there is no sawmill near Argonaut Wharf.

## Recordings

A SoundTrap (300 STD, Ocean Instruments, New Zealand) internally recording hydrophone system was deployed from August 15, 2019, to October 31, 2019, in Cowichan Bay and then redeployed from August 11, 2020, to October 22, 2020, in the same location (Fig. 1b). The SoundTrap hydrophone was set to a sampling rate of 48 kHz at 16 bits and recorded for 15 min every hour. The SoundTrap was deployed in Cowichan Bay following the methods used in Murchy et al. (2022), on a polysteel line 1 m off the bottom (45 m water depth). Retrieval of the hydrophone was done using an acoustic release.

In 2021, the location was changed to Argonaut Wharf and deployed from August 13, 2021, to September 27, 2021 (Fig. 1a). The SoundTrap was set to the same sampling rate (48 kHz) but recorded continuously throughout the deployment. The deployment method was also modified, and the SoundTrap was secured to an underwater piling using cable ties, at a depth of 5 m off bottom (10 m water depth). Calibration of the SoundTraps was conducted by the manufacturer at 250 Hz with a source level of 120 dB re 1 μPa (B & K 2236 pistonphone).

## Analysis

Data were stored as compressed sud files (.SUD) on the internal memory of individual SoundTraps until retrieval at the end of each deployment. All following data analysis was done using code written in Python (version 3.7). Sound pressure levels were examined in a time series for the full broadband frequency range of the recorder (20–24,000 Hz). The soundscape at both locations was described using 5-min averaged data.

Automatic identification system (AIS) data from equipped vessels in a 5 km radius around Cowichan Bay was obtained for August through October of 2019 and 2020 in 5-min bins and 1-min bins for a 10 km radius around Argonaut Wharf for August and September 2021. AIS data obtained were accurate to ±1 m for the vessel's GPS location. All tugboat data found within the AIS data set were extracted and tracks were generated in ArcMap 10.8 (ESRI, California, USA). Distance between each tugboat's GPS location and the SoundTrap location was generated

using the NEAR function to obtain the closest point of approach for each tugboat, and only tugs that were within 500 m of the SoundTraps were used in the analysis. Additionally, for the Argonaut Wharf data set, tugboats needed to be within 500 m for at least 2 min (half of the 5-min bins used for SPL calculations) to be included.

SPL measurements were binned into two equal time periods (with and without a tugboat present). Depending on the length of the tugboats' presence on a given day, the "no tugboat" time period was selected to match. Power spectral density (PSD) plots were also generated for both time periods for the entire frequency range and compared. Sound pressure measurements between time points with and without tugboats were normally distributed, so *t*-tests were used for analysis, and were performed in RStudio version 4.1.2.

## Results

### Tugboat Logging Activities (Cowichan Bay)

Log transport activities did not occur during 2019 due to a strike at the Western Forest Mill located in Cowichan Bay, but restarted once the strike ended in early 2020. During August and September of 2020, six tugboats pulling log booms were detected within Cowichan Bay using AIS; however, only four times did the tugboat get within 500 m of the hydrophone while it was recording.

Tugboats pulling logs near the estuary in Cowichan Bay significantly changed the soundscape for the full broadband range collected (20–24,000 Hz; Fig. 2a). Median

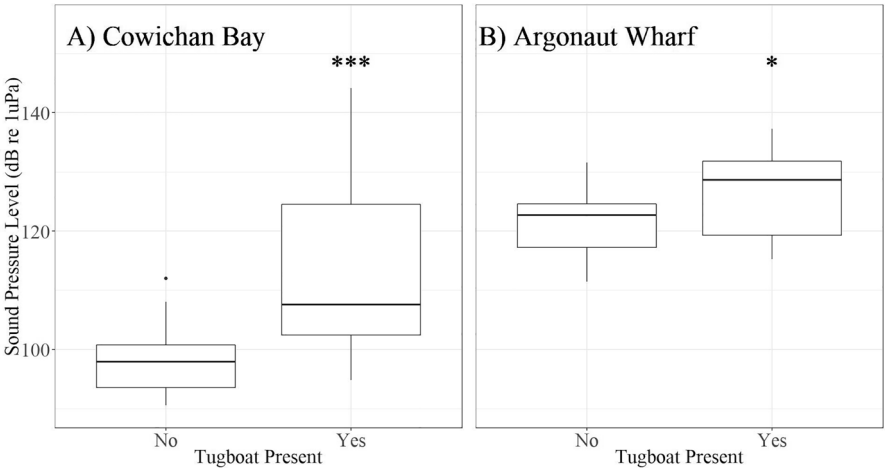



**Fig. 2** Boxplots of sound pressure levels (broadband: 20–24,000 Hz) in (**a**) Cowichan Bay and (**b**) Argonaut Wharf without and with a tugboat within 500 m of hydrophone. Asterisks (***) indicate significantly different mean sound pressure levels at $p < 0.0001$, and a single asterisk (*) indicates significance at $p = 0.02$. Significance was tested using paired *t*-tests

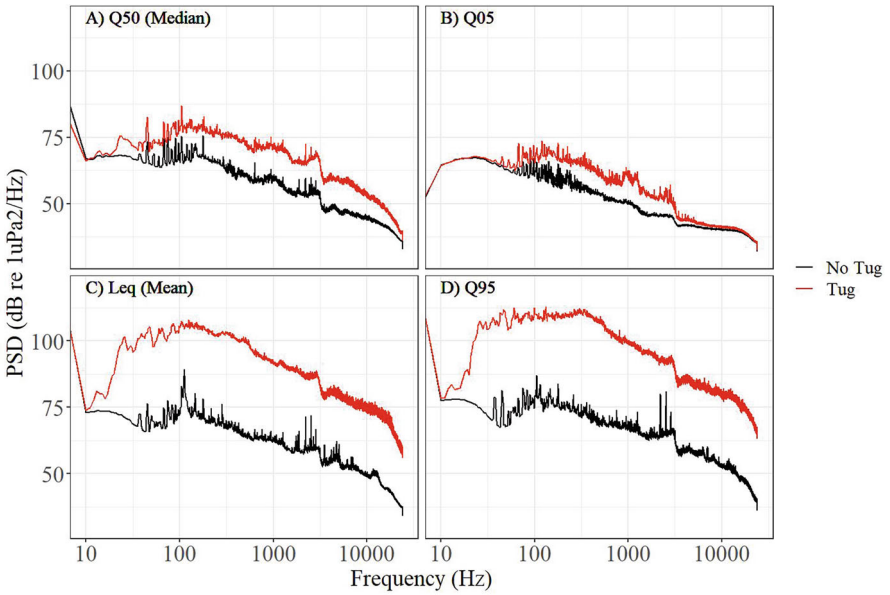



**Fig. 3** Power spectral density (PSD) plots of logging tugboats in Cowichan Bay (**a**) median values (50th percentile), (**b**) fifth percentile, (**c**) mean values (Leq), and (**d**) 95th percentile with no tugboats (black) and while tugboats were moving logs into the estuary (red)

(5%, 95% CI) sound pressure levels near the estuary were 97.8 dB re 1 μPa (91.3; 106.4 dB re 1 μPa) when no tugboats were present. Once a tugboat with logs was within 500 m of the hydrophone, median sound pressure levels were significantly elevated to 107.6 dB re 1 μPa (98.0; 138.3 dB re 1 μPa) (*t*-test, $p < 0.0001$, df = 60, $t = -7.3$).

Tugboats pulling log booms into Cowichan Bay substantially changed the frequency composition near the estuary across the entire frequency range collected (10–24,000 Hz), at all percentiles (Fig. 3a–d). Median increase in sound pressure level at different frequencies varied between 0.5 dB re 1 $\mu Pa^2$/Hz (11 Hz) and 17.2 dB re 1 $\mu Pa^2$/Hz (2216 Hz) when a log transport tugboat was present. However, the largest change was observed at 47 Hz with a 43.6 dB re 1 $\mu Pa^2$/Hz increase in the 95th percentile.

## Other Tugboat Activities (Argonaut Wharf)

Argonaut Wharf had more tugboat activities compared to Cowichan Bay, with 15 tugboats detected within 500 m of the SoundTrap hydrophone in 2021. However, only five tugs were detected within 500 m for longer than 2 min, and only a single tugboat was considered towing, as obtained from the AIS data (September 16). The other four tugboats could not be confirmed to be towing, but it also could not be determined that they were not.

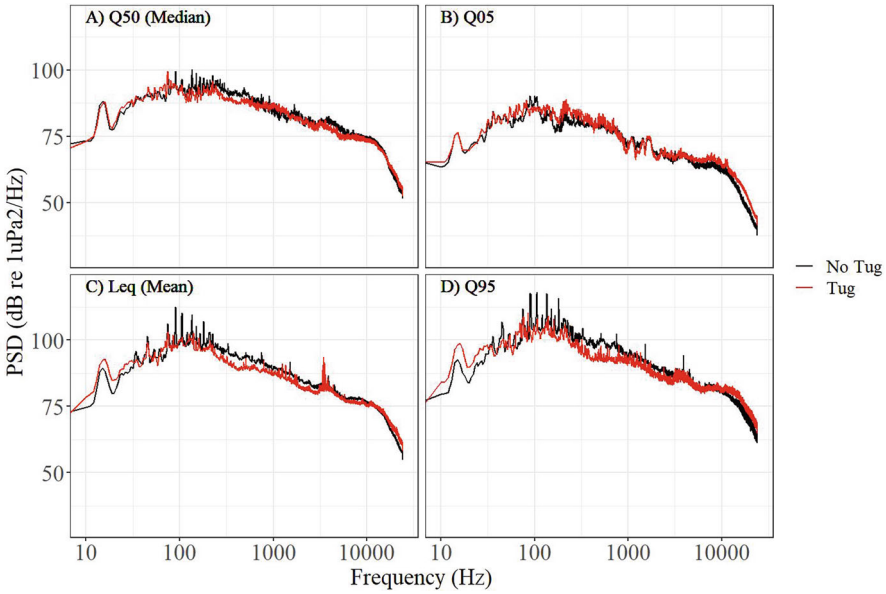



**Fig. 4** Power spectral density (PSD) plots of tugboats in Argonaut Wharf (**a**) median values (50th percentile), (**b**) fifth percentile, (**c**) mean values (Leq), and (**d**) 95th percentile with no tugboats (black) and with tugboats within 500 m of hydrophone (red)

Sound pressure levels (20–24,000 Hz) were higher in Argonaut Wharf compared to Cowichan Bay, but tugboats still had a significant impact on the underwater soundscape (Fig. 2b). Median (5%, 95% CI) sound pressure levels were 122.7 dB re 1 μPa (91.3; 106.4 dB re 1 μPa) when no tugboats were present. Once a tugboat was within 500 m of the hydrophone, median sound pressure levels were significantly elevated to 128.6 dB re 1 μPa (98.0; 138.3 dB re 1 μPa) ($t$-test, $p = 0.02$, df = 33, $t = -2.3$).

Tugboats present in Argonaut Wharf only slightly altered the frequency composition at any of the percentiles examined, with only approximately 40% of frequencies elevated (Fig. 4a–d). Median increase in sound pressure level at different frequencies varied between 0.0002 dB re 1 $\mu Pa^2$/Hz (12,004 Hz) and 6.1 dB re 1 $\mu Pa^2$/Hz (1155 Hz) with a tugboat present. However, the largest change was observed at 7 Hz with a 12.6 dB re 1 $\mu Pa^2$/Hz increase in the 95th percentile.

## Discussion

The presence of tugboats had a significant impact on the underwater soundscape in both locations considered. In Cowichan Bay, tugboats used for log transport activities caused short-term changes to the soundscape by elevating the sound pressure level near the estuary, across all frequencies collected. Similarly in Argonaut Wharf,

sound pressure levels were increased, but frequency composition was similar between time points. The differences observed between the two locations indicate that the impact of tugboats on the underwater soundscape is context specific and varies between locations.

Log transport tugboats present in Cowichan Bay increased the soundscape noise levels near the estuary adding an additional impact to critical habitat already degraded by log storage activities. Currently in British Columbia, timber is stored as rafts in estuaries prior to processing at mills, which can lead to changes to benthic communities through bark being deposited on the ocean floor, and compaction of the sediment (Toews and Brownlee 1981). Bark deposition has also been linked to reduced zooplankton abundance and dissolved oxygen (Power 1987). Additionally, the lower oxygen levels near log storage areas diminished the number of juvenile salmon in those areas, potentially displacing them (Levy et al. 1990). Results from this chapter demonstrate an additional stressor, underwater noise, that requires additional research into its impact to the marine animals that live in the estuary.

Argonaut Wharf had higher sound pressure levels compared to Cowichan Bay when no tugboats were present. This difference is likely due to the high presence of anthropogenic activities in the Strait of Georgia (Erbe et al. 2012). One noise source not found in Cowichan Bay that was present in Argonaut Wharf was passenger ferries. A car and passenger ferry runs from Campbell River to Quadra Island twice an hour transiting between 1.7 and 2.0 km away from the hydrophone location. When examining the contribution (i.e., number of vessels, time present, and source level) of ferries to the underwater soundscape in the Strait of Georgia, passenger ferries account for a high relative noise contribution (approximately 70–88%), but the exact noise contribution varies depending on the time of year (MacGillivray et al. 2016). In comparison, tugboats in the Strait of Georgia account for roughly 7–22% of the relative noise contribution to the area (MacGillivray et al. 2016).

Significant impacts to the underwater soundscape were observed in the study presented in this chapter, but the study was not without its limitations. The study was only conducted over short time periods between August and October. During these time windows, Chinook salmon (*Oncorhynchus tshawytscha*) are returning to spawn in both locations. Cowichan Bay has a First Nation subsistence fishery, while Argonaut Wharf has an open recreational fishery directly in front of the deployed hydrophone, resulting in higher amounts of recreational vessel activities in this area that could be minimizing the observed impacts. Additionally, due to the local geography of the two sites, the data collected for this chapter could only be used to examine the impact of tugboats on underwater soundscape on limited spatial scales (500 m). Larger-scale spatial impacts could not be evaluated. However, estimates of long-distance impacts can be generated using the sonar equation to calculate where the noise produced by tugboats would equal background noise levels. Source levels of tugboats collected in Cowichan Bay and at Argonaut Wharf were back-calculated from the median sound pressure levels when tugboats were within 500 m and were 137 dB re 1 μPa @ 1 m and 155 dB re 1 μPa @ 1 m, respectively. In Cowichan Bay, impacts of tugboat noise were estimated to be above

median ambient sound levels (97.8 dB re 1 μPa) until approximately 10 km away from the hydrophone. Argonaut Wharf had higher median ambient sound levels (122.7 dB re 1 μPa), which resulted in a shorter distance (2 km) at which the tugboat noise was above ambient noise levels. Overall, tugboat noise could be altering the underwater soundscape between 2 and 10 km away from the vessel. Future work should evaluate the impact of tugboats on the underwater soundscape on seasonal scales and a range of spatial scales to account for seasonal variation in vessel activities and oceanographic conditions.

Both locations used in this chapter represent critical habitat for many marine species, but the influence of the noise produced by tugboats on these species has not been evaluated. Resident killer whales (*Orcinus orca*) display avoidance (horizontal and vertical: Williams et al. 2002, 2009) and change foraging behaviors (Lusseau et al. 2009) around vessels. Similarly, Pacific herring (*Clupea pallasii*) and juvenile salmon (*Oncorhynchus* spp.) have behavioral responses to elevated sound pressure levels produced from vessels (van der Knaap et al. 2022). Recent meta-analyses have also documented widespread impact of shipping noise on fish and invertebrates (Cox et al. 2018; Murchy et al. 2020), indicating broad implications for elevated noise levels due to shipping activities. Future research should continue to evaluate the impacts of underwater noise on marine species, and tugboats need to be represented in these studies.

## Conclusions

Large commercial vessels have been documented to increase underwater noise levels, with mitigation methods already being implemented (MacGillivray et al. 2019; Williams et al. 2019). In British Columbia, tugboats represent a significant contributor to local underwater noise, with the highest total vessel hours for all AIS vessel categories (MacGillivray et al. 2016). The work presented in this chapter demonstrates that tugboats significantly elevate ambient underwater sound levels when present, representing an understudied contributor to the underwater soundscape. Future work should continue to examine the impact of smaller commercial vessels, like tugboats, to the underwater soundscape and marine species to help develop mitigation measures to reduce their impacts.

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# Small-Scale Variation in the Soundscapes of Coral Reefs

29

Ashlee Lillis, Amy Apprill, Maickel Armenteros, and T. Aran Mooney

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**Abstract**

Coral reefs are complex habitats that emanate an array of sounds. These acoustic cues are integral to animal behaviors and are increasingly leveraged to discern ecological information. Reef soundscapes are often measured with an overarching view to characterize broad differences between reef sites. Further, there is a

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limited understanding of soundscape variation within particular reefs. Here, three acoustic recording methods (benthic recorders, vertical arrays, drifters) were used to address if and how soundscapes vary across coral reefs. Research took place at eight sites in the highly protected region of Jardines de la Reina, Cuba. Sound levels and frequency composition varied across small horizontal and vertical spatial scales. Horizontal differences were notable within reefs over meters, particularly when comparing recordings from adjacent dead and live massive coral. Vertically, sound levels and variation changed with distance from the benthos, with greater variation at lower frequencies. The drifter revealed sound levels that varied substantially relative to bottom type and nearby biota. Overall, the coral reef soundscape, like its biodiversity, was not uniform in distribution. Acoustic patterns varied substantially in ways that likely offer fine-scale acoustic cues to fishes and invertebrates. Moreover, this chapter highlights that small-scale spatial variation must be considered when implementing broad scale eco-acoustic monitoring of coral reef habitats.

**Keywords**

Acoustic environment · Acoustic ecology · Noise · Biodiversity · Bioacoustics · Recruitment · Cuba · Caribbean · Fish · Coral disease

## Introduction

Marine animals move through a sea of chemical, visual, and acoustic cues as they select habitats, avoid predators, forage, and enact other vital behaviors. While some of these cues have long been studied, there is a growing understanding of the ecological importance of underwater acoustic cues and signals. Sounds can provide habitat and situational information to a diverse and increasingly realized number of animal taxa including many invertebrates, fish with and without auditory specializations, seabirds, and marine mammals (Buxton et al. 2017; Ellison et al. 2012; Fay 2009; Myrberg 1986). Soundscape data are increasingly being measured for the potential to discern broad-scale differences between habitats such as coral reefs of different quality or health, reefs of different coral cover, or reefs versus adjacent sandy and seagrass habitats (Kaplan et al. 2015; Radford et al. 2014). Yet most reef and coastal animals enact behaviors that likely require sensory information on smaller scales. For example, acoustic communication in fishes is generally considered to be limited to several body lengths or several meters (Coombs et al. 1992; Myrberg 1997; Zeddies et al. 2012). Coral and fish larvae must use vertical water column cues to address depth, vertical distance from a reef, and whether the reef is a proper location to settle (Leis and Carson-Ewart 1999; Leis 2004). Cues in the horizontal plane may offer information such as predator presence, distance of a predator, conspecific territoriality distance, and location of attractive mates (Hanlon and Budelmann 1987; Sweatman 1983). Temporally varying choruses may offer information regarding spawning, help induce physiological or behavioral responses,

coordinate activities, and provide cues to potential settlement locations (Kaplan and Mooney 2016; Mann and Grothues 2009; Parsons et al. 2016).

Despite these ecological benefits, there is limited information regarding the spatial or temporal variation of acoustic cues on smaller scales that likely more relevant to a particular animal. However, initial work suggests such variation exists. An acoustic hydrophone array deployed in the Northwest Hawaiian Islands found sound pressure directionality and hotspots of snapping shrimp sound levels (Freeman et al. 2015). Multiple recorders deployed on reefs in the US Virgin Islands showed small, but significant differences between sites within reefs (Kaplan et al. 2015). The biological relevance or importance of these differences was not considered. Surface drifting hydrophones have detected significant changes in soundscape characteristics across meters within a reefscape, related to benthic habitat structure differences (Lillis et al. 2018). Further, modeled water columns around reefs reflect what are likely distinct vertical gradients of sound, based on frequency and propagation conditions from the reef (Salas et al. 2022). Given this growing suggestive evidence, there is a need to collect empirical data addressing the prevalence of these acoustic hotspots (or cue spaces), and evaluations of their biological relevance.

The reefs of Jardines de la Reina ("Gardens of the Queen") Cuba are an ideal environment to examine soundscapes. The reefs host some of the highest coral cover and fish abundance in the Caribbean (González-Díaz et al. 2018; Navarro-Martínez et al. 2022). This high habitat quality is a result of many factors including their inclusion within one of the largest Caribbean marine protected areas, their large distance from shore, and their strong enforcement of no-take fishing areas. The Jardines de la Reina (JR) reefs are therefore a unique place to study Caribbean reef sounds due to the abundant and diverse reef life as well as limited exposures to boat noise, which can alter animal behaviors (Simpson et al. 2016).

A 2017 cruise of opportunity to the shallow forereefs in the Jardines de la Reina system was used to address fundamental questions in reef soundscapes. Are there within-reef acoustic hotspots? Are there habitat-scale differences within a reef? How do soundscape cues vary in space and time on a reef? The effort sought to more generally address whether there are potential acoustically mediated differences within a reef that may be relevant for fundamental behaviors such as communication, predator detection, settlement, and within-reef habitat selection.

## Methods

Work was conducted from small boat operations based off the *M/V Alucia* scientific vessel. The research cruise took place from November 4–20, 2017. Measurements were made exclusively within Jardines de la Reina National Park, Cuba (permitting noted below). Observations were made opportunistically during the expedition with the overall goal of surveying the reef system using multiple acoustic platforms and examining multiple reefs across multiple days. Long-term deployments in a single location were not possible due to the cruise schedule.

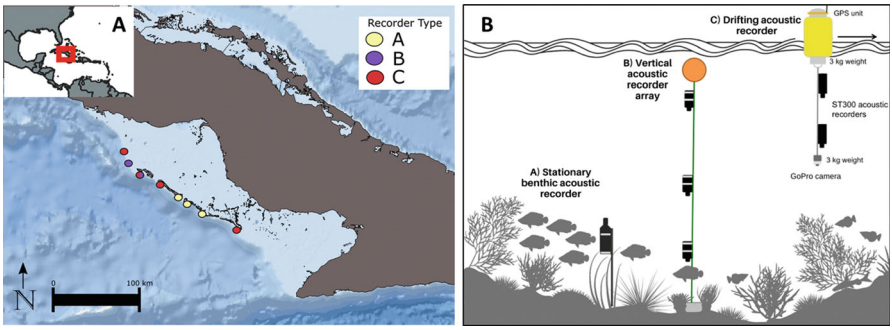



**Fig. 1** (**A**) Map of the study sites off the south shore of Cuba. (**B**) Schematic depicting the three acoustic recorder configurations used to examine scales of soundscape variation on coral reefs

Passive acoustic work was conducted at eight reef sites across the archipelago (Fig. 1a). Data were collected using three complementary recorder configurations with the goal of investigating the small-scale soundscape variation within and above reef mosaics (Fig. 1b). Recorders utilized were either one-channel SoundTrap 300 s or 4-channel SoundTrap 4300 s [ST-300/4300; Ocean Instruments NZ (Ferguson et al. 2022)].

The first configuration consisted of stationary benthic recorders placed at multiple adjacent locations within a reef allowing for comparisons of simultaneous recordings across the horizontal plane of the reef, targeting small-scale differences (tens of meters) in habitat predicted to affect biological sound production (e.g., coral type, bathymetry, fish aggregation distribution). During the study, the set of recorders were able to be placed at three reef sites for 48–72-h periods, capturing 1-min recordings every 5 min.

Second, to examine small-scale depth variation in soundscapes above reef habitat we positioned acoustic recorders on a vertical array. The array was moored in one location and had a subsurface float and from that float, a surface buoy. The latter allowed the array to be sighted from the surface for retrieval while the subsurface float enabled the array to suspend vertically in the water column. On each vertical array, recorders were placed at 1, 5, and 10 m from the seafloor, and sites were selected where water depth was between 12 and 15 m. These systems were deployed at dusk for overnight recordings at 3 reef sites during the study period. Each deployment captured 3–12 h of continuous recording at 96 kHz.

The third recording configuration utilized a drifting hydrophone setup described previously (Lillis et al. 2018). Briefly, a GPS-outfitted surface buoy relayed location to a handheld GPS unit allowing the recording system to be deployed and left to drift on its own, facilitating later retrieval. Below the float was a line with a 3 kg weight at the bottom, and two acoustic recorders were attached vertically along the line. The recorders were located 1 and 2 m from the surface. A GoPro camera facing downwards recorded the benthic composition during the drift track. The drifter allowed for mobile recordings while minimizing the interfering noise of autonomous

vehicles. Surface current and wind direction were visually observed just before deployment to facilitate predicting a general drift direction. The drifter was then typically deployed "up reef" (windward side) and allowed to drift over the reef and observation area, and later retrieved via the GPS location. Drift trajectories, distance, and length depended on currents, wind, and conditions, and were typically 1–2 h in duration, and covered ~0.5–1 km distance. This sampling was conducted at 4 JR reefs, with 2–4 drifts achieved at each site.

### Acoustic Analyses

Acoustic data were analyzed in MATLAB using custom-written code. The files were initially examined for potential transient or anthropogenic (e.g., boat, diver) noise and any such files were removed. Comparisons were then made using acoustic power spectra. Sound pressure levels (in rms – root-mean-squared) were calculated for low (0.1–2 kHz) and high (2–23 to 48 kHz) frequency bands which generally constitute the frequency ranges of fish and snapping shrimp sounds, respectively (Au and Banks 1998; Urick 1983). Where appropriate, the median and interquartile range of spectrum and sound pressure levels were calculated from the analysis of non-overlapping 1-s duration windows spanning the 1-min recordings, which could then be used for plotting and further statistical analysis where appropriate.

## Results

Approximately 16,000 1-min audio files were successfully recorded between the different configurations applied across 8 reef sites in the Jardines de la Reina (Fig. 1) during the research cruise. Below are results describing findings that exemplify the types of small-scale soundscape variation observed using this dataset.

### Horizontal Environment

At site JR10, a relatively high number of large, endangered pillar coral (*Dendrogyra cylindrus*) colonies were observed, of varying size and condition. As these large corals are known to create distinct three-dimensional structure and provide habitat hotspots for reef diversity, the distinct reef features were targeted for acoustic sampling and provide an ideal example for the examination of within-reef soundscape patterns related to coral type and health. Three SoundTrap 300 (ST-300) acoustic recorders were placed at *D. cylindrus* pillar coral towers, and three in nearby soft coral patches for a 72-h period. Recorders were all deployed within a small study area ~100 m × 20 m reef segment. The colonies denoted as DCYL 1 and 2 were live, but the DCYL 3 structure was dead coral skeleton with minimal living tissue (Fig. 2). The spectra of all three sites can roughly be divided into two

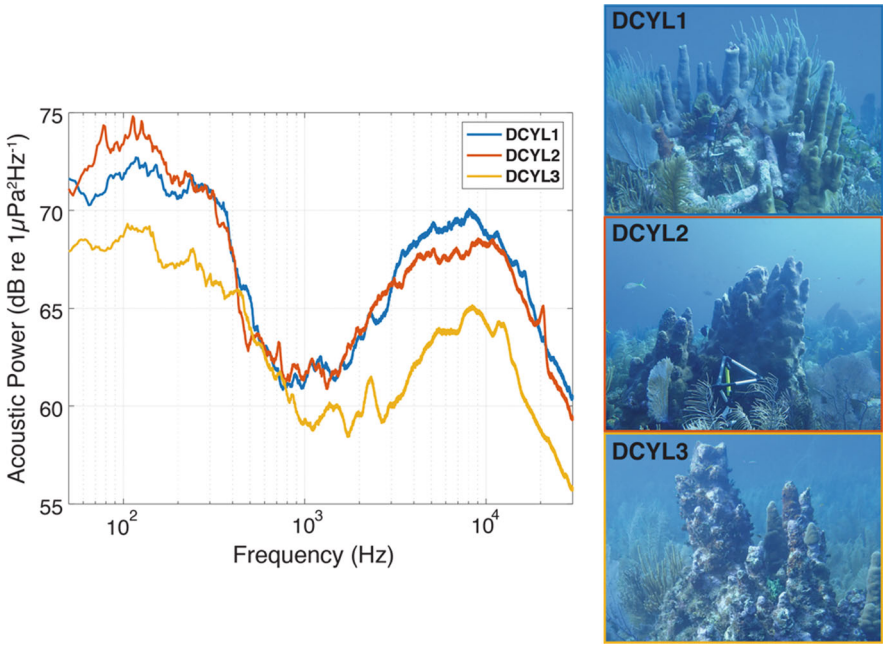



**Fig. 2** Full band acoustic power at two healthy (DCYL1 and 2) and one dead (DCYL3) *Dendrogyra cylindrus* colonies. Note the overall lower sound levels at DCYL3 suggesting a diminished community of soniferous fish and invertebrates

components, below 1500 Hz and above 2000 Hz. Fish sounds are generally found in the former, low-frequency band, while snapping shrimp sounds are typically higher in frequency (>2000 Hz) (Kaplan et al. 2015). All three DCYL spectra showed peaks in the low and high frequency bands, near 100–200 Hz in the low frequency and 7–9 kHz in the high frequency. The sites differed in the amplitude of these peaks; DCYL 1 and 2 were similar in spectral characteristics, and while the spectral shape of DCYL 3 followed the same pattern, median power was ~4–7 dB lower in both the low and high-frequency ranges compared to the healthy DCYLs (Figs. 2 and 3). In comparing the median SPLs by site for the two frequency bands, significant differences existed among nearly all recording sites (Fig. 3b; Kruskal-Wallis test: low frequency $\chi^2 = 977$, $p < 0.001$, high frequency $\chi^2 = 2414$, $p < 0.001$) underscoring the high small-scale inter-reef variability in soundscape. Low frequency sound levels were significantly lower at DCYL 3 compared to DCYL 1 and 2; however, it is important to note that one of the octocoral sites exhibited similar SPL levels to the healthy DCYL in the low frequency band (OCTO 2; Fig. 3b(i).

Spectrograms demonstrate the influence of time of day on these within-reef soundscape patterns and highlight differences among the nearby recording sites (Fig. 3a). At lower frequencies (<2 kHz), diel rhythms were notable at all recording sites with a consistent peak in SPL just after sunset each night (Fig. 3a). DCYL 1 and

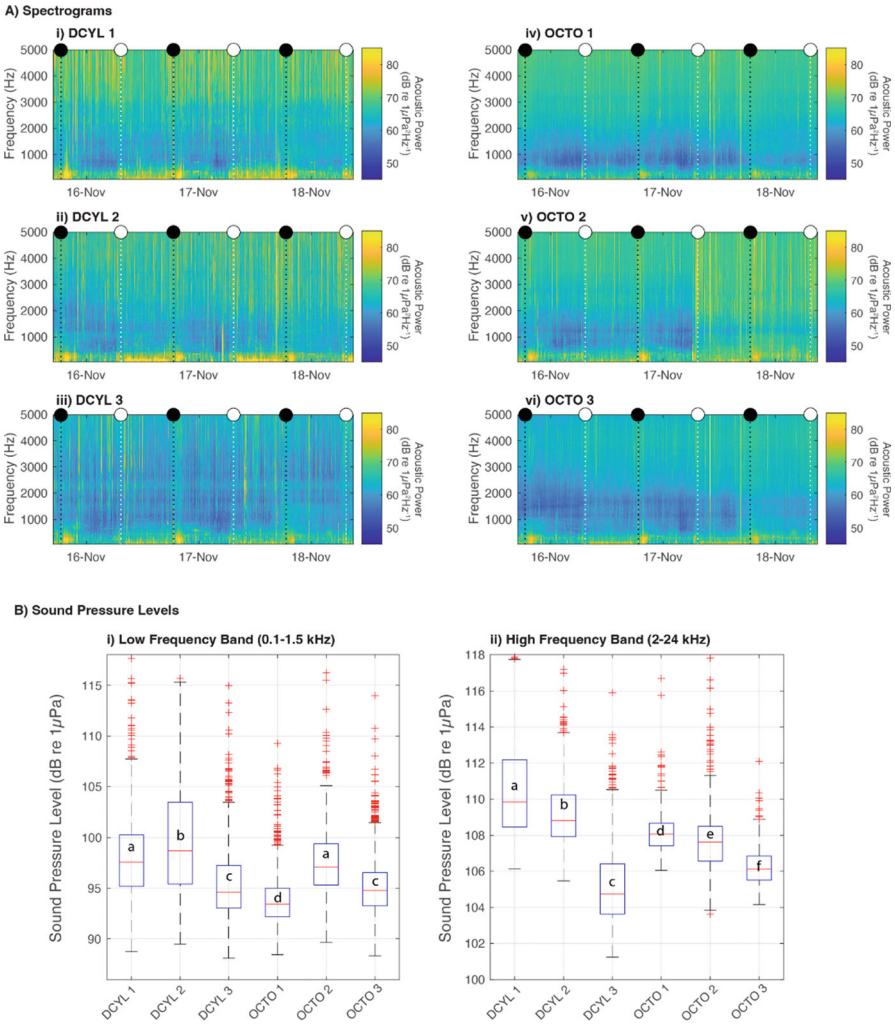



**Fig. 3** (**A**) Spectrograms showing frequency specific patterns for recordings made adjacent to the endangered *Dendrogyra cylindrus* (i–iii) and octocorals (iv–vi). Black and white circles denote sunset and sunrise, respectively. (**B**) Boxplots depicting medians and interquartile ranges for (i) Low frequency (0.1–1.5 kHz) and (ii) High frequency (2–24 kHz) bandwidths. Letters signify statistically significant differences between sites, based on a non-parametric multiple comparison post-hoc test

2 showed stronger dusk increases in low frequency SPL and had more acoustic activity during daytime hours compared to DCYL 3 (Fig. 3a(i–iii)). The day-night low frequency difference is less evident at DCYL 3 and the OCTO sites. Subtle diel patterns were also observed in the snapping shrimp-driven high-frequency band, with modest increases in snapping at night observed across sites (Fig. 3).

## Vertical Profiles

There was a clear vertical gradient of acoustic energy at sampled reefs (Figs. 4 and 5). At JR3, where 2-h continuous recordings were collected, sound pressure levels were greatest near the reef/benthos (10 m) and decreased when measured at 5 and 1 m from the sea surface (Fig. 4a, b). At 1 m depth, median SPL was ca. 6–7 dB lower, or about half the acoustic energy, than the bottom recorder. Variation was similarly lower at 1 m depth, compared to the deeper positions. However, these changes were not equally spread across frequencies. The benthic and mid-water positions showed greater, but variable, acoustic power in the low frequencies (Fig. 4).

For site JR3, at low frequencies (<400 Hz), acoustic power was ca. 5–8 dB higher closer to the reef, but frequency specific variability was relatively large (Fig. 4). At higher frequencies (>1000 Hz) dominated by snapping shrimp, acoustic power was ca. 3 dB lower near the benthos and similar at the mid-water and surface stations. There was also less variability between sites. Between ca. 500 and 1000 Hz, the surface generally showed higher sound levels, and variably was high, perhaps reflecting the effects of noise attenuating from the surface.

Conversely, Fig. 5 shows the results of vertical array recordings made at JR4 for a 12-h period. In this dataset, the mid-water recorder displayed the highest sound levels (Fig. 5b). Upon further inspection, it was determined that the ship thrusters contributed to this low frequency signal elevation, received most strongly at the 5 m depth hydrophone. The higher frequency bandwidth sound levels were relatively consistent with depth for this vertical array (Fig. 4b).

## Spatial Variation in Motion Across a Reef: The Drifter

The drifter spectrogram provides a unique ability to assess how a reef area soundscape varies spatially [(Lillis et al. 2018), Fig. 6]. Over the course of an hour, the

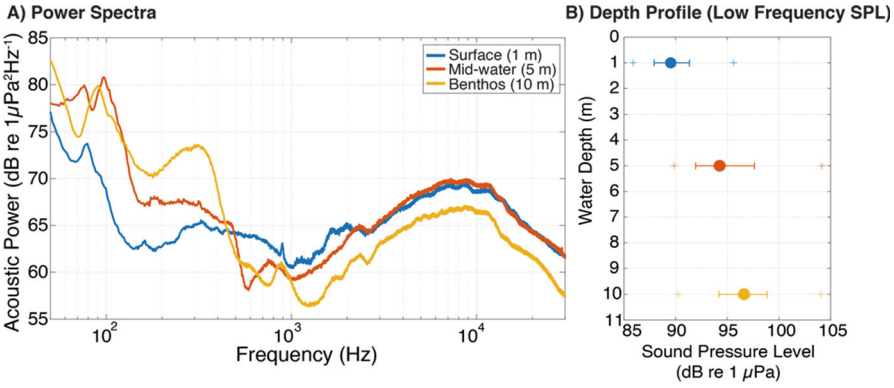



**Fig. 4** (**A**) Power spectra and (**B**) low frequency sound pressure level variation with depth measured at JR3 over 2 h of recording from the stationary vertical array

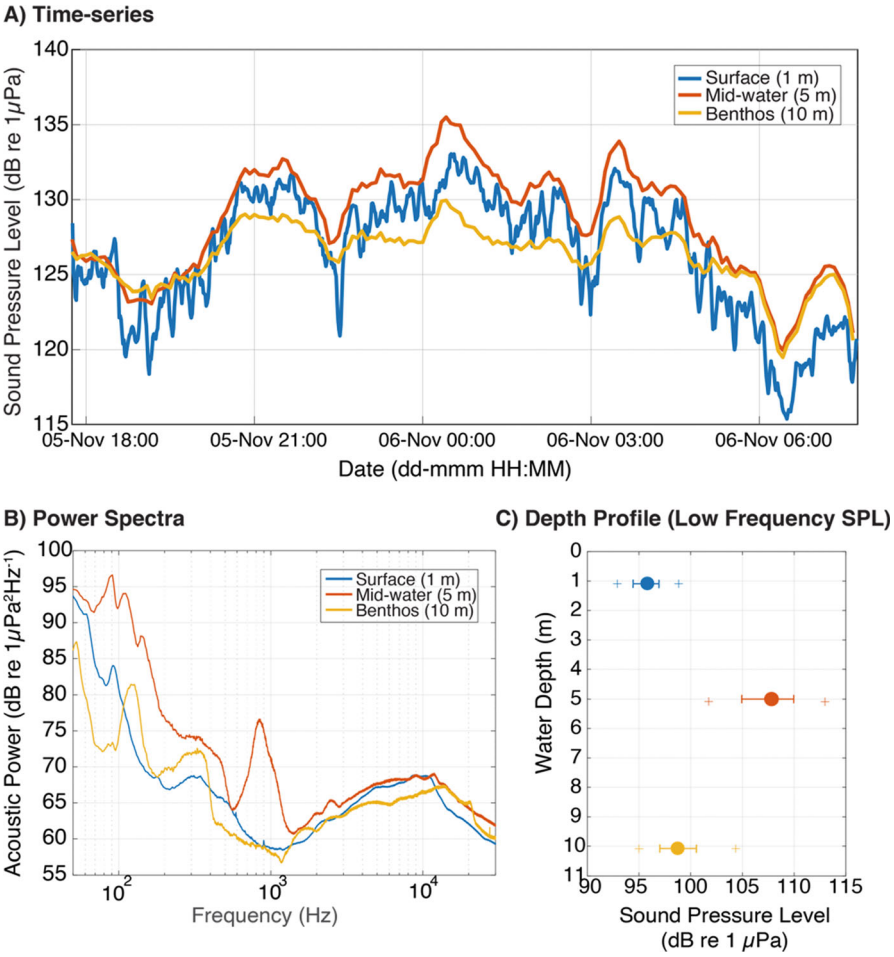



**Fig. 5** (**A**) Time-series, (**B**) power spectra, and (**C**) depth variation in low frequency sound pressure level measured overnight at JR4 from the stationary vertical array

drifter started on the coral reef but drifted off and away to areas of mixed benthic structures. High frequency energy was clearly elevated over the reef and this greatly influenced the full bandwidth analysis. Around 16:50 local time, the drifter approached an area of small patch reefs and subsequently the higher frequency snapping shrimp sound levels increased. Notably, just off the reef, around 16:22 and 16:28 schools of small jacks approached the drifter. At these times, low frequency SPLs peaked (Fig. 6a, b-ii), reflecting sounds produced by these schools. Otherwise, the low frequency fish sounds were generally consistent for the first 20–30 min, and then decreased substantially just off the reef (ca. 16:40), but slowly began to increase as the drifter approached the patch reef.

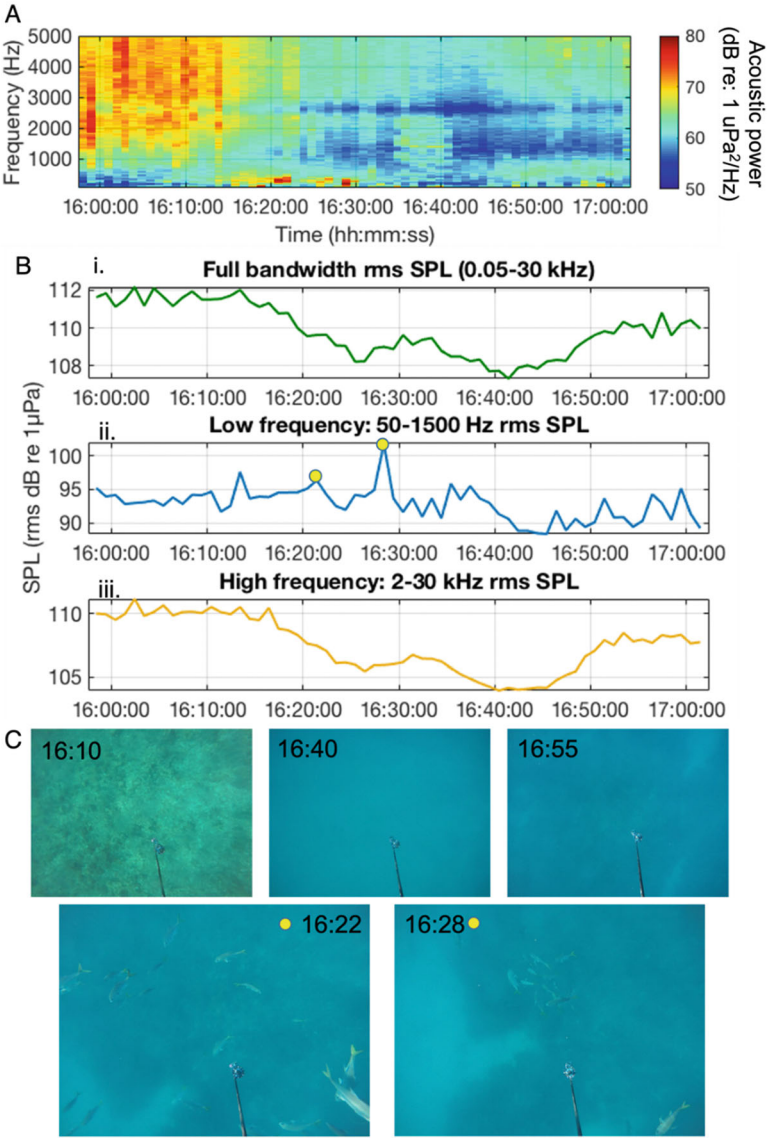



**Fig. 6** (**A**) Spectrogram of the drift; (**B**) i–iii corresponding spectra in the full, low, and high frequency bands, and (**C**) downward facing images at five timepoints during the drift. Note the increased high-frequency and overall sound levels over the initial reef habitat (16:10), reflective of an increase in snapping shrimp. There was increase in sound levels over reef areas, and particularly notable low frequency signals (highlighted by the yellow dots) when fish schools passed by

## Discussion

The goal of this research was to address spatial soundscape variation within a coral reef environment, particularly to establish if substantial variation exists that impacts passive acoustic monitoring, and to provide ecological context for the data collected. The results show evidence of substantial small-scale horizontal acoustic variation using complementary recording configurations: recorders deployed via benthic mooring in close proximity and via a low-noise drifter buoy. Multi-depth vertical arrays revealed that sound levels and frequencies also changed over small depth changes, with substantial particularly at lower frequencies (below ca. 1000 Hz). The suite of configurations was applied to holistically sample the reef environment, rather than focus on fewer locations for a longer period. This allowed us to make inferences regarding the reef soundscape over scales not typically monitored with PAM and emphasizes the importance of careful site selection when initiating an acoustic reef monitoring effort.

The soundscape of the dead pillar coral colony (DCYL 3) differed from the other two living colonies likely because a dead colony does not attract the same soniferous animal community. The structure of large *Dendrogyra* pillar coral is a unique community hub on the reef, providing habitat for fishes, which create nutrient hotspots and in turn increased coral cover additional fish local habitat (Shantz et al. 2015). Together, it is not surprising that these micro-habitats associated with the live *Dendrogyra* would foster a distinct soundscape with high levels of acoustic activity reflecting the community hub this structure provides. Reef soundscapes are expected to change concomitantly with deteriorating reef conditions caused by bleaching, diseases, and loss of corals (Lamont et al. 2022). However, it is important to note that in this chapter our measurements of adjacent soft coral areas also displayed unique and complex soundscape parameters, overlapping with the *Dendrogyra* SPLs and frequency patterns. It is clear that careful application of broad acoustic metrics and cautious interpretation of crude differences associated with reef variation is required. Nonetheless, the insights gained into loss of acoustic activity at a dead *Dendrogyra* colony has implications for broad, community-scale impacts of stony coral tissue loss disease (SCTLD), which has rapidly spread throughout the Caribbean causing nearly complete mortality of *Dendrogyra* corals (Neely et al. 2021). These results support the need for advanced acoustic analyses (e.g., machine learning) and sound libraries (Parsons et al. 2022) to enhance our ability to identify the drivers and understand the ecological relevance underpinning small-scale acoustic variation.

The paired camera-acoustics of the drifter provided a valuable method to visually observe the animals likely making certain sounds. This allowed us to video-capture what appears to be horse-eye jacks (*Caranx latus*), a predator fish, swimming around the recording system and suggests these could be the cause of concurrent higher low-frequency sound levels. These sounds appear to be biological in origin, not animals bumping hydrophones. Carangidae are one of the families with sound

production noted (Fine and Parmentier 2015) but its origin or function has not yet been well-described.

The vertical array provided a new assessment of the shallow coral reef environment demonstrating vertical differences in sound level and frequency at the scale of meters. These differences may be particularly important for understanding the depth distribution of acoustic cues for settling larvae. When near reefs, larvae can be at shallower depths (2–4 m) (Leis and Carson-Ewart 1999) but as they settle and move toward a reef (increasing depth), they may have more low-frequency sound cues available to them. Reef fishes select microhabitats such as species/morphotypes of corals and it is generally assumed that they need to be close to the reef to make such decisions (Sweatman 1983). The variation observed across a 10-m depth water column indicates that such vertical patterns are relevant to studies of the soundscape as an orientation cue and should be considered in measurement of salient larval cuescapes.

The recorded vertical variation of sound pressure level should be addressed further. These measurements were certainly in the acoustic near-field for the low frequencies, yet this is the reality for fish larvae as they swim over a reef. The variability was likely due, at least in part, to the unpredictability of sounds in the near-field (Kalmijn 1988). Indeed, closer to the reef and its sound sources, where soniferous fish and invertebrates are in higher abundance, variability was higher (Fig. 4c). The decrease of variability as shallower depths may be explained because the near-field variability effects would be limited to only the lower frequencies. Although there are few data of in situ swimming behavior of settling fish larvae, they appear to maintain a relatively stable (often relatively shallow) depth although this "stable" depth can vary by ca. $\pm$0.5 m or more (Irisson et al. 2010). This movement would allow them to potentially "sample" this variable acoustic environment to better assess the reef below. Such a mechanism would be quite valuable for fish settling at night.

To our knowledge, this is the first passive acoustic examination of coral reefs in Cuban waters. This environment was particularly compelling to document acoustically as the Jardines de la Reina reef system is relatively healthy, compared to many other Caribbean coral reefs. The region and some reefs within it have higher coral cover than other areas of the Caribbean and are also relatively abundant with large fishes and sharks (Ferguson et al. 2022). The sound dataset collected in Jardines de La Reina offers a starting point for further comparisons with other impacted and declining Caribbean reefs. It also offers an important baseline to this hub of biodiversity as it faces, as other Cuban reefs, ongoing impacts by chronic stressors and stochastic events (Caballero-Aragón et al. 2022).

Importantly, this study demonstrates that the soundscape of a single coral reef is not uniform in distribution. Rather, acoustic cuescapes vary horizontally across the reef and vertically with distance from the seafloor. Given the importance of sounds to reef fishes and invertebrates, these differences likely offer acoustic cues as animals seek to settle, find mates, or establish territories. The shelter that large corals provide to soniferous animals may support acoustic hotspots, given the increased density of fishes and invertebrates at these microhabitats and the potential interactions with the

confined space that the coral-shelter may provide. Understanding the variation in reef soundscape is also important as we seek to leverage these patterns for assessing reef communities. Thus, it is key to determine the diversity and structure of these sound-producer communities associated with reef microhabitats and the ecological function that they provide. Finally, understanding the spatiotemporal distribution of reef microhabitats should be a consideration and caveat to valuations from snapshot acoustic measurements. Indeed, this supports the need for longer-duration measurements as we seek to understand coral reef soundscapes and the communities they represent.

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# Soundscape Characterization in Chedabucto Bay on the Atlantic Coast of Canada, 2019–2021

30

Jessica E. Wingfield, Heather Breeze, Shihan Li, Jinshan Xu, and Emmaline C. Marotte

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**Abstract**

Fisheries and Oceans Canada's Coastal Acoustic Monitoring Project aims to better understand the underwater soundscape at coastal sites off Nova Scotia, Canada. Researchers began deploying passive acoustic monitors for this project in December 2018. In order to compare the contribution of various sources of sound to the soundscape at the different sites, they have taken a sound energy budget approach. Here they present the results from their longest-running site, Chedabucto Bay, near the approaches to one of the largest ports in Nova Scotia. The percent contributions of wind, vessels, and "other" sources to the total sound energy budget for the 1/3 octave bands centered at 100 Hz, 500 Hz, 1 kHz, 2 kHz, 5 kHz, and 10 kHz were calculated. Vessel noise was the dominant contributor to sound energy in the lowest frequency band in almost every month except between August and November 2020, during which irregular vessel movements are suspected to have impacted the accuracy of the sound energy budget estimations. This sound energy budget approach is useful for comparing different sites, as well as the same site over time, but should be complemented by other analyses that can present information on levels of sound energy.

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**Keywords**

Sound energy budget · Underwater soundscape · Marine acoustics · Noise · Nova Scotia · Passive acoustic monitoring

## Introduction

The Marine Planning and Conservation Program, Maritimes Region, Fisheries and Oceans Canada (DFO), initiated the Coastal Acoustic Monitoring (CAM) Project in December 2018 to better understand the types and levels of underwater sound at several coastal sites around Nova Scotia. The project focused on shipping noise to reflect the Government of Canada's priority to better understand the impacts of shipping on marine ecosystems (Transport Canada 2020). The project also aimed to acoustically detect baleen whales; more information on that aspect of the project can be found in Wingfield et al. (2022). The longest-running site of the CAM project is located on the southern shore of Chedabucto Bay (Fig. 1). The bay is located between mainland Nova Scotia and Cape Breton Island and is the pathway to one of the largest ports in Nova Scotia, the Strait Superport. Because of this, the bay is often frequented by commercial vessels. There are also several active fisheries in the bay, including mackerel (*Scomber scombrus*), shrimp (*Pandalus borealis*), lobster (*Homarus americanus*), and other shellfish (Serdynska and Coffen-Smout 2017; Rozalska and Coffen-Smout 2020). Despite this, researchers and local fishers were together able to select a deployment location where fishing does not regularly occur, avoiding the potential for disturbing the acoustic monitoring system deployed on the seafloor while still allowing for the effective recording of noise from nearby vessels. Several species of marine mammals have been sighted in the bay, including fin (*Balaenoptera physalus*) and sei (*Balaenoptera borealis*) whales (Gomez et al. 2020; Jacques Whitford Environmental Ltd. 2004). A large part of the bay is included within the Canso Ledges Ecologically and Biologically Significant Area due to the rich diversity of invertebrate, fish, and marine mammal species that occur there (Hastings et al. 2014).

A soundscape is a recording of all sounds that occur within an area, including from geological, biological, or anthropogenic sources (Havlik et al. 2022; Southworth 1967). A soundscape can therefore be analyzed to determine the type and temporal frequency of anthropogenic activities and weather events taking place in an area and even to assess species richness and general health of an ecosystem (Pijanowski et al. 2011). One way to summarize a marine soundscape is by producing a sound energy budget. In this approach, sound sources are ranked in order of greatest contribution to the overall soundscape (Miller et al. 2008). The researchers chose to characterize the soundscape around the Chedabucto Bay site using a sound energy budget approach similar to that used by Miller et al. (2008) and Nystuen et al. (2010). This allowed for the comparison of the contributions of vessels, wind, and all remaining sources (grouped as "other") to the overall soundscape across different months and between years, and the researchers felt that this method would be an informative way to summarize a previously unstudied soundscape.

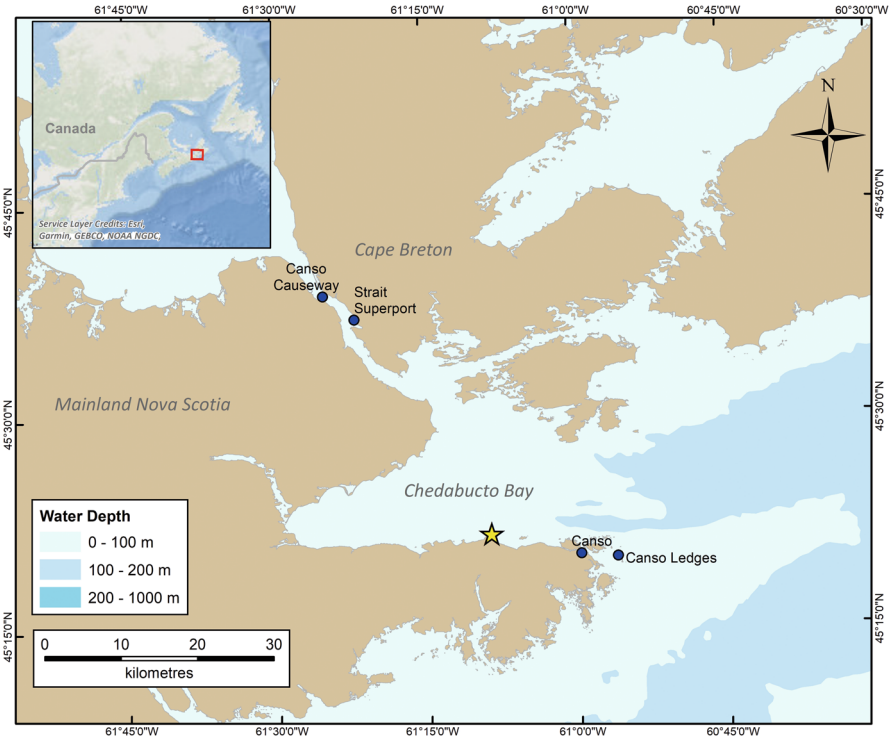



**Fig. 1** The deployment site (*yellow star*) and other points of interest (*blue circles*) within and surrounding Chedabucto Bay, Nova Scotia, Canada. Inset map credits: Esri, Garmin, GEBCO, NOAA NGDC, National Geographic, HERE, Geonames.org, and other contributors (2023). Main map credit: Natural Resources Canada (2023)

Previous acoustic monitoring efforts and ongoing projects carried out by DFO Science branch in Maritimes Region have focused on offshore areas using large and complex recording systems that require large vessels for deployment and recovery. Prior to the initiation of the CAM Project, the closest acoustic monitoring site to Chedabucto Bay was located approximately 93 km east of the mouth of the bay (referred to as "Stn 2," see Delarue et al. 2018, 2022). In their investigation of soundscapes off eastern Canada, Delarue et al. (2018) found that vessels and seismic surveys were the main anthropogenic contributors to ambient sound. At Stn 2 specifically, Delarue et al. (2018) found that vessel noise contributed the most energy to the overall soundscape in the spring, summer, and early fall. They did not detect any seismic survey noise at this site (Delarue et al. 2018). Much of the previous research on underwater noise in coastal Nova Scotia was carried out for national defense purposes, that is, to improve methods of detecting submarines and submarine communications (Chattopadhyay and Pecknold 2016; Osler and Pegg 2018) and to better understand the acoustic impacts of naval activities (e.g., Tollefsen and Pecknold 2012). Piggott (1964)

carried out the earliest acoustic study at a coastal Nova Scotia site, demonstrating a strong correlation between mean sound spectrum levels at multiple frequencies and wind speed. He deliberately excluded periods of time during which there were high levels of shipping noise, as the overall objective was to understand the influence of wind on ambient noise. Chattopadhyay and Pecknold (2016) examined underwater ambient noise in the Bedford Basin, Nova Scotia, for a short period of time in 2014 to better understand the environment for defense-related underwater communications.

While data collection continues, here the researchers discuss their sound energy budget approach and preliminary results for data collected at the Chedabucto Bay study site. To their knowledge, this is the first project to undertake acoustic monitoring within the bay. Major industrial projects have been proposed for this area, including additional marine terminals and a spaceport facility (Black Point 2022; Edwards 2022). These projects are likely to impact the soundscape of the bay in the coming years if they move forward. The data from this project could therefore be considered baseline data, and will serve as a valuable reference point should the aforementioned projects move forward.

## Methods

A SoundTrap (ST300 STD or ST500 STD, Ocean Instruments, New Zealand) mounted in a small, modified lobster trap was deployed for periods of approximately 2 months at a time from December 12, 2018, to April 20, 2021 in Chedabucto Bay, Canada (Fig. 1). A newly programmed SoundTrap was deployed immediately after the previous mooring was retrieved. This allowed for the very near continuous collection of data, even on recovery days, but meant that different instruments were used for consecutive deployments. The SoundTraps were set to record continuously at a sampling rate of 24 kHz with the PreAmp gain set to "high." The 1/3 octave bands centered at 100 Hz and 500 Hz were most sensitive to vessel noise. Researchers found the dispersion between these two bands to be greatest when a vessel was closest to the mooring. The dispersion could therefore be used as an identifying feature of vessel noise. The contribution of vessel noise to the soundscape was determined by calculating the energy added to the overall sound pressure level (SPL) by vessels (Fig. 2). Wind speeds were extracted from Environment Canada's High Resolution Deterministic Prediction System dataset. The minimum wind speed needed to produce white caps, indicators of breaking waves, is ~3.7 m/s (Callaghan et al. 2008). Therefore, when wind speeds were greater than 3.7 m/s, it was assumed that increases in SPL after vessel noise was removed were caused by wind (Fig. 3). The percent energy contributions of vessels, wind, and all remaining sources (grouped as "other") to the total energy budget in 1/3 octave bands centered at 100 Hz, 500 Hz, 1 kHz, 2 kHz, 5 kHz, and 10 kHz were calculated. For a more detailed description of the methodology, see Wingfield et al. (2022).

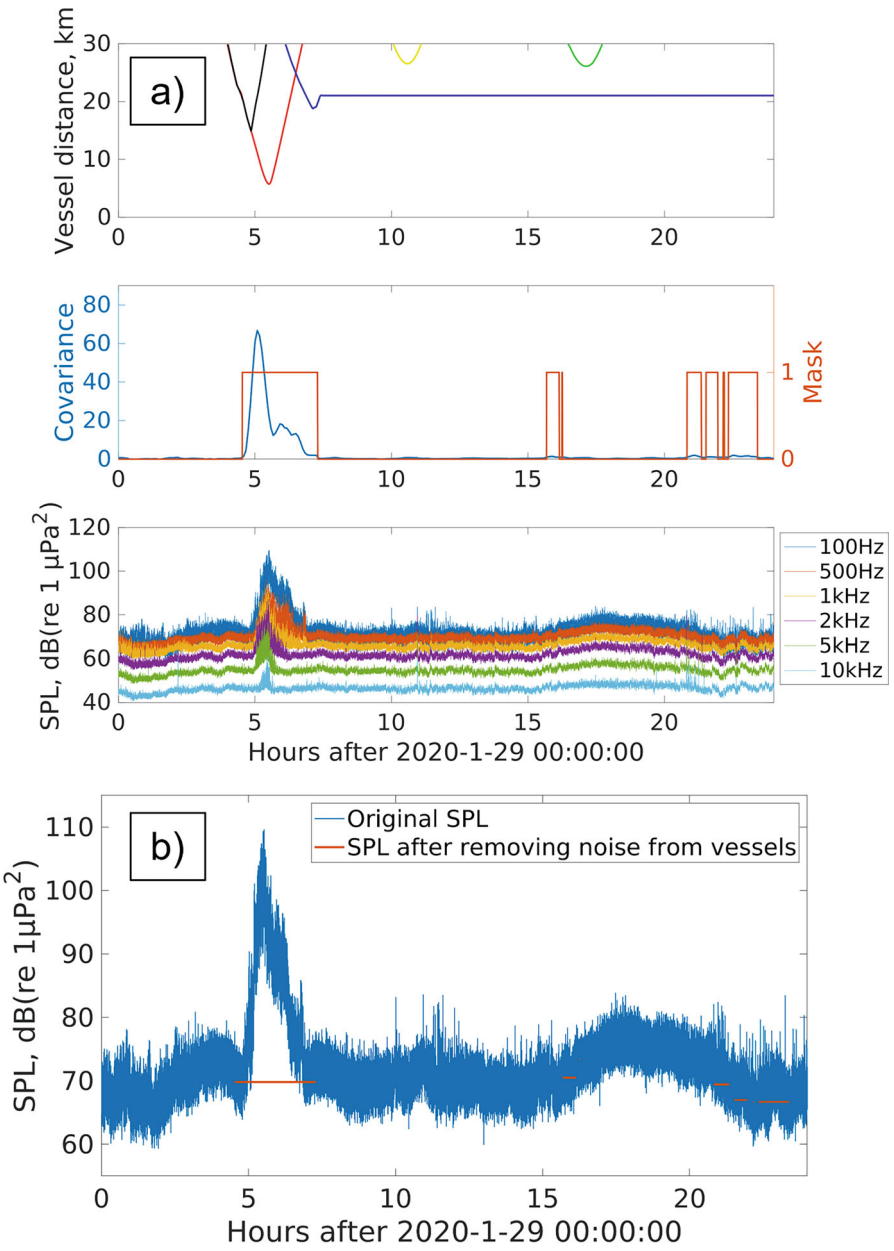



**Fig. 2** (**a**) The distance of vessels from the mooring location (top), the covariance between SPLs in the 100 and 500 Hz bands and resulting mask values (middle), and the SPL (bottom) over time (UTC) on January 29, 2020. (**b**) The original SPL and SPL after removing suspected vessel noise in the 1/3 octave band centered at 100 Hz on January 29, 2020 (UTC)

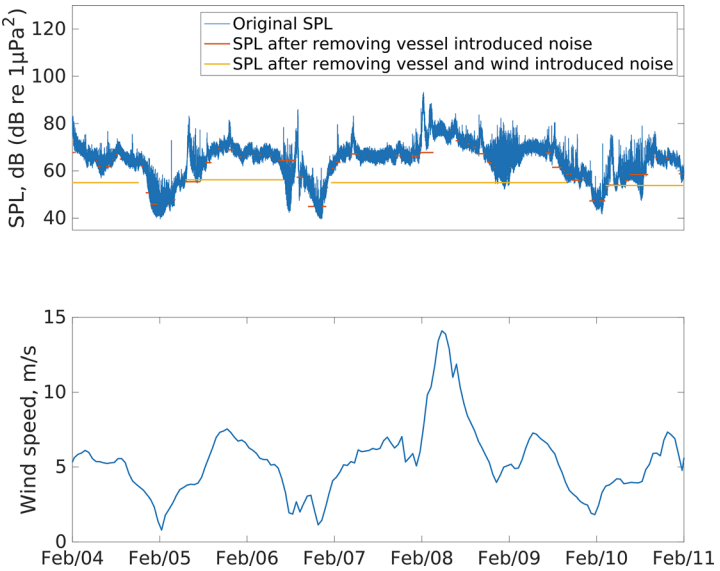



**Fig. 3** The original SPL, the SPL after removing vessel noise, and the SPL after removing vessel and wind noise (top) in the 1/3 octave band centered at 500 Hz and the wind speed (m/s) (bottom) from February 4 to 11, 2020 (UTC)

## Results

The researchers aimed to determine the contributions of vessels, wind, and all other sources (precipitation, biological noise, etc.) to the total sound energy budget for each month of the study period. Some of the preliminary results are discussed here. Overall, wind and vessels dominated much of the soundscape for the six frequencies examined. During the periods examined, vessel noise was the largest contributor to the sound energy budget in the lowest frequency band in almost all months, and in most frequency bands during summer and fall months, except between August and November 2020. During these months, there were several multi-day periods of elevated noise levels in the lower frequency bands caused by the prolonged presence of one vessel, the crane ship *Thialf*, and increased vessel traffic from its support vessels. The sound energy budgets for these months therefore misrepresent the contributions of each source. Vessel noise was likely a dominant contributor to the soundscape during this period, as it was in summer 2019. Wind was often the largest contributor to the sound energy budget in all but the 100 Hz frequency band during winter months (December to March). Fig. 4 shows the sound energy budgets for January 2019 and July 2020. These figures demonstrate the dominance of wind in the winter months and vessel noise in the summer months. For all monthly sound energy budgets and expanded results, see Wingfield et al. (2022).

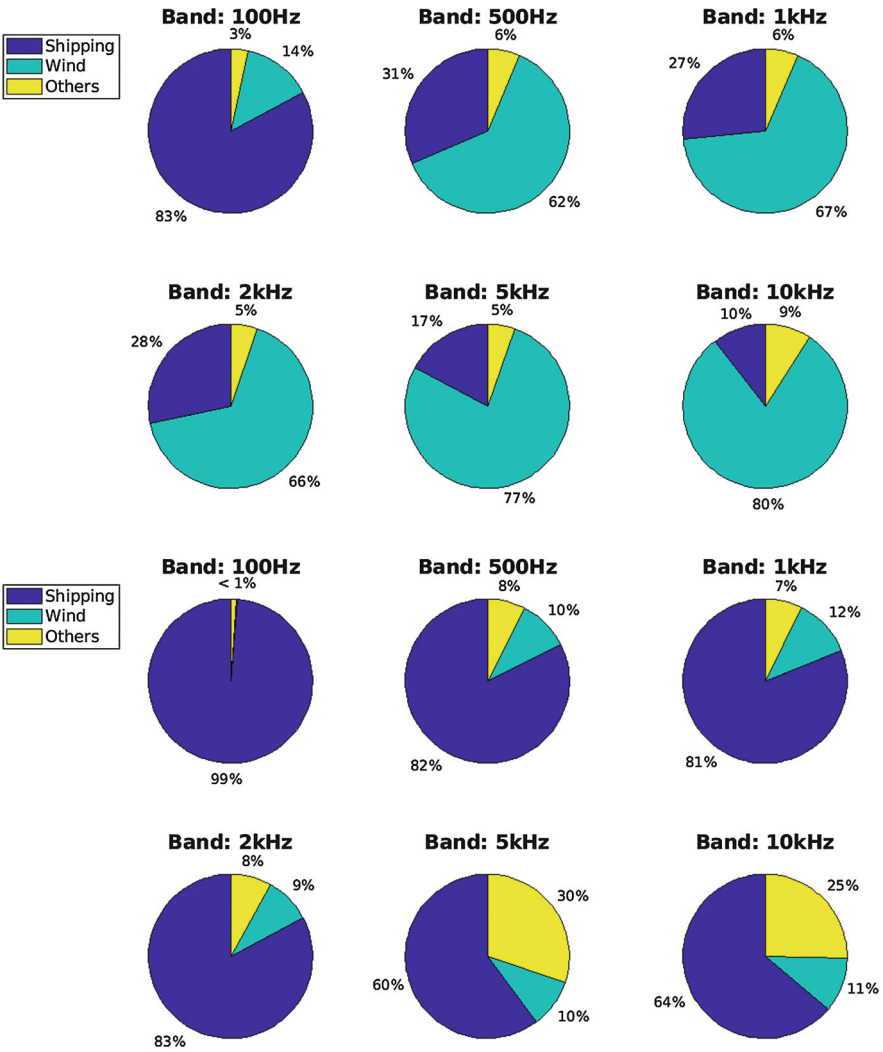



**Fig. 4** The estimated sound energy budgets for January 2019 (*top*) and July 2020 (*bottom*)

## Discussion

The researchers have demonstrated that it is possible to use data from relatively low-cost underwater acoustic recorders to characterize the soundscape of a coastal area regularly frequented by vessels. Vessel noise dominating the lower frequency bands was expected, as the Chedabucto Bay study site is within the approaches to one of Nova Scotia's busiest ports. While understanding absolute sound levels is

important in understanding potential impacts on marine life, characterizing the soundscape using a sound energy budget approach allows for the comparison of the relative contribution of different sound sources at different sites (see, e.g., Breeze et al. 2021) and could inform management measures.

If the new industrial developments proposed for the bay go forward, they are likely to cause an increase in the amount of vessel traffic in the area and therefore an increase in noise levels in the lower frequency bands. This may have implications for use of this area by baleen whales, as these whales communicate mainly in these same lower frequency bands. To investigate the potential impacts of noise on baleen whales, researchers continue to characterize the soundscape and are also analyzing daily and hourly baleen whale call occurrence (Wingfield et al. 2022). Baleen whale call occurrence will be compared to noise levels in future publications.

When this project was initiated, noise levels in the lower frequency bands were of most interest due to the focus on impacts of shipping. However, vessels also contribute to noise levels at higher frequencies, and the coastal construction related to the planned industrial developments (e.g., pile driving) are expected to produce noise at higher frequency levels. Monitoring at this site continues, and efforts have expanded to include an additional site on the north side of the bay. The researchers also plan to increase the sampling rate to record underwater sounds at higher frequencies. From this, a more complete picture of the soundscape of Chedabucto Bay will be developed.

**Acknowledgments** The success of this project is due in large part to the guidance and support received from local fishing associations. Thank you to the Guysborough County Inshore Fishermen's Association for assisting with site selection and for retrieving and deploying the moorings. Thank you also to Jim Theriault for his guidance and advice throughout the project.

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# Soundscapes from a Saltwater Marsh Creek Captured by a Hydrophone Array

31

Mark W. Sprague, Phillip Deville, and Joseph J. Luczkovich

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**Abstract**

A seven-hydrophone array was deployed in a shallow saltwater creek at the University of South Carolina Baruch Marine Field Laboratory in the North Inlet-Winyah Bay National Estuarine Research Reserve near Georgetown, South Carolina, USA, and sounds were recorded for 24 h from June 01 through June 02, 2017, to identify and localize soniferous animals in the water and to serve as a proof of concept for future studies. The array was calibrated by producing sounds at known locations and using triangulation to determine the hydrophone positions. Similarly, the calibrated hydrophone positions were used to determine the positions of soniferous animals from the signal delays between the different hydrophones using triangulation. The locations of three fish, including one that was moving as it vocalized, are shown.

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## Introduction

Soundscape recordings allow listeners to determine the presence and activities of soniferous animals in an area (Pijanowski et al. 2011), and these recordings have been used to learn about both terrestrial (Fischer et al. 1997) and aquatic environments (Parks et al. 2014; Luczkovich and Sprague 2022). If we want to learn about animals that we cannot see, we can listen to them. But most soundscapes recorded with a single hydrophone provide little information about the locations and movements of the sound-producing animals in the recording because hydrophones are omnidirectional at low frequencies. Therefore, a seven-hydrophone array was deployed in a very shallow saltwater creek to learn about the locations of sound producers and their movements.

The study site was Clambank Creek (33°20′2.57”N, 79°11′34.58”W), a saltwater creek at the University of South Carolina Baruch Marine Field Laboratory in the North Inlet-Winyah Bay National Estuarine Research Reserve near Georgetown, South Carolina, USA (see Fig. 1). The goal of the study was to determine which soniferous species were present in the creek and how the activity and locations of these sound producers changed throughout the day and to serve as a proof of concept to be used in future studies.

In order to determine the position of a sound source from an hydrophone array, the positions of the hydrophones, the delay between the arrivals of the same signal at

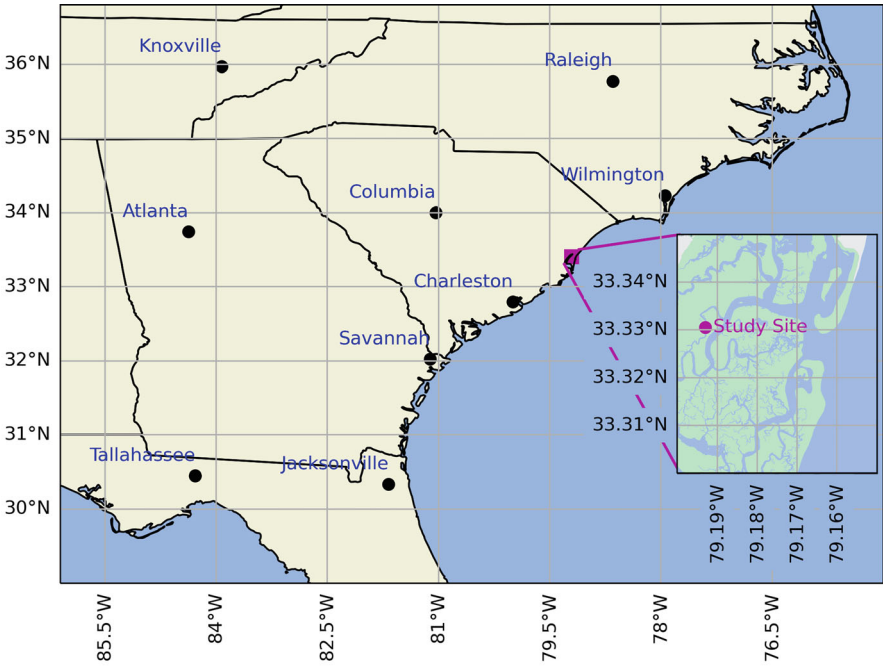



**Fig. 1** The study site shown on a map of the southeastern United States

the different hydrophones, and the sound speed are used to determine the source location using triangulation. This chapter describes how each of these parameters were determined to locate and track sound sources in this shallow water environment.

## Hydrophones and Recorder

The array consisted of seven HTI-96-Min hydrophones (High Tech, Inc., Long Beach, MS, USA). A 100 m, nine-conductor cable (conductors for voltage supply, ground, and hydrophone 1–7 signals) connected an eight-track simultaneous-sampling digital recorder (Zoom F8, Tokyo) and voltage supply on the shore to a junction box. Each hydrophone was connected to the junction box with a 30 m, three-conductor cable (with conductors for voltage, ground, and signal). The recorder made simultaneous 24-bit recordings from each hydrophone sampled at 44100 Hz and stored them on a SD card in the Broadcast Wave file format.

## Reference Positions

Four reference positions (see Fig. 2) were used to accurately locate the position of each hydrophone. At each reference position, a large metal target was positioned so it was visible throughout the site. The location of each reference position was measured using a handheld GPS receiver (Garmin GPSMAP 78sc, Olathe, KS, USA) with Wide Area Augmentation System-enabled Global Positioning System (WAAS-GPS).

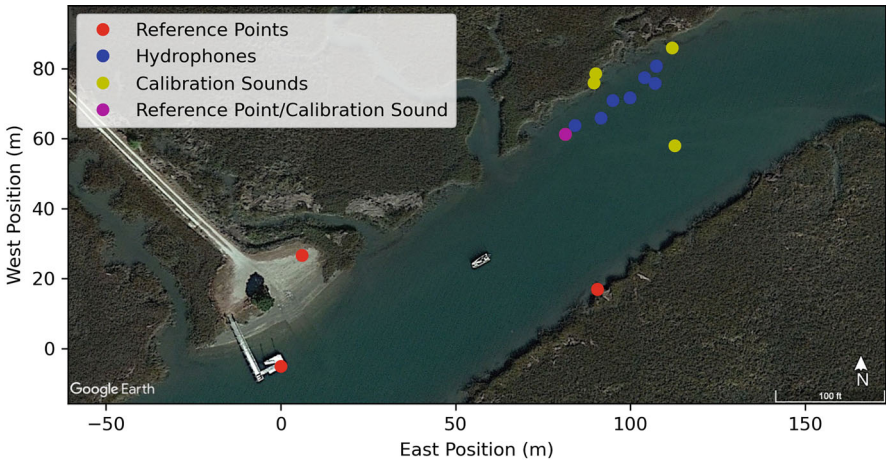



**Fig. 2** Positions of the reference markers, hydrophones, and calibration sounds at Clambank Creek. The Clambank water quality monitoring station is located on the pier on the bottom left of the image. (Satellite image obtained from Google Earth)

## Array Deployment

The array was deployed on June 01, 2017, in water depths of 1.00–1.70 m by wading into the water and mounting the hydrophones on rods connected to cinder block anchors placed on the creek bed. The water depth and time of day were recorded at each hydrophone site. Hydrophone positions were measured with the GPS receiver for validation of the hydrophone locations to be determined by the calibration procedure (section "Array Calibration"). Figure 2 shows the hydrophone positions at the study site. The array configuration was suboptimal with hydrophone locations influenced by bottom conditions and the ability to secure the hydrophone in place while wading.

## Array Calibration

To calibrate the array, impulsive sounds were produced underwater at five positions, determined by triangulation of laser rangefinder (Laser Tech TruePulse 360 °R, Centennial, CO, USA) measurements of the distances to the reference position and also using WAAS-GPS for validation. One of the calibration sounds was produced at the location of reference point 3 (see the reference point/calibration sound point in Fig. 2). The delays of the calibration sounds on the different hydrophone recordings were used to calibrate the array by solving sound propagation equations for the distances from the calibration sounds to each hydrophone and using triangulation to determine the hydrophone positions.

The small pier in Fig. 2 at the second reference point is the location of a National Oceanic and Atmospheric Administration (NOAA) tide and water quality measuring station (station 8662299 Clambank Creek Dock), allowing the determination of the water depth and water quality parameters. The local speed of sound in the water, which was determined with the equation developed by the United Nations Educational, Scientific and Cultural Organization Division of Marine Sciences (the UNESCO equation; Fofonoff and Millard 1983; Wong and Zhu 1995) using water quality measurements taken every 900 s (15 min). Sound speed values for times between the 900-s water quality samples were determined using a third-order cubic spline interpolation.

## Sound Recordings and Source Localization

Sounds were recorded for 24 h and identified using knowledge of the fishes in the estuary (Simpson et al. 2015) and by comparing them to recordings of captive fish. The locations of recorded fish were determined by measuring the signal time delays between each hydrophone recording and solving acoustic propagation equations for the source position using the interpolated local speed of sound in the water.

The soundscape recordings contained rich acoustic activity, including sounds produced by Atlantic croaker *Micropogonias undulatus*, bighead searobin *Prionotus*

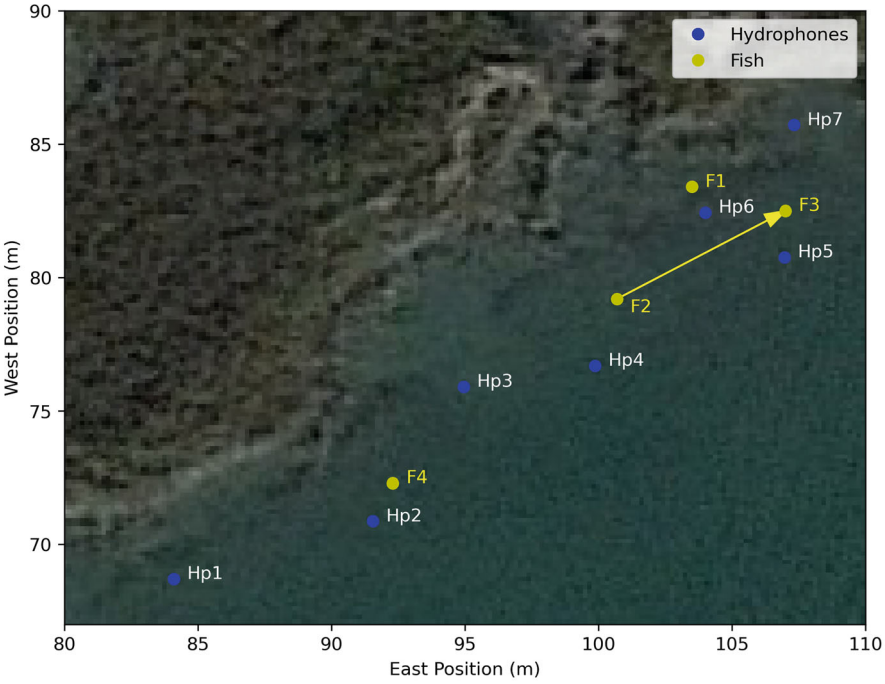



**Fig. 3** Hydrophone positions with fish detections at Clambank Creek. **F1**, a sea robin detected at 2017–06-01 T12:54 (UTC-4); **F2**, a spotted seatrout *Cynoscion nebulosus* detected at 2017–06-01 T18:29:21 (UTC-4); **F3**, the same spotted seatrout as F2 detected at 2017–06-01 T18:30:21; **F4**, a striped blenny *Chasmodes bosquianus* detected at 2017–06-01 T21:06:22. The arrow shows the motion of the spotted seatrout between F2 and F3. (Satellite image obtained from Google Earth)

*tribulus*, oyster toadfish *Opsanus tau*, silver perch *Bairdiella chrysoura*, spotted seatrout *Cynoscion nebulosus*, and striped blenny *Chasmodes bosquianus* as well as snapping shrimp (family Alpheidae). Using the time delays between the same signals in the array channels, the locations of many fish in the recordings were determined. The localized fish sounds shown in Fig. 3 include a bighead searobin at point F1, a spotted seatrout that was "followed" as it vocalized continually for over 60 s as it moved from point F2 to F3, and a striped blenny at point F4.

## Discussion and Conclusion

This chapter demonstrates that sound-producing fish can be located and tracked using a hydrophone array deployed in a suboptimal configuration in very shallow water. The measurements of the calibration sound positions with a laser rangefinder had sufficient accuracy to calibrate hydrophone positions acoustically with results that were consistent with independent WAAS-GPS measurements. The hydrophone positions and measured time delays of sounds in the recording were

used to locate and track vocalizing fish. In the future this technique will be used to locate groups of fishes producing sound and to determine how these fishes move throughout the day.

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# Studying the Soundscape of Shallow and Heavy Used Marine Areas: Belgian Part of the North Sea

32

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**Abstract**

The impact of anthropogenic sound on marine fauna is a growing concern, particularly in shallow, coastal, and heavily exploited marine areas such as the Belgian Part of the North Sea (BPNS). Understanding the ecosystem and its limits in these areas is necessary to protect these areas and ensure their sustainable use. To quantify this impact, characterizing and analyzing the soundscape is crucial. However, analyzing soundscapes in shallow and heavily exploited marine areas poses several challenges and particularities. Bio-fouling, flow-noise, unknown sound sources, and masking compromise propagation. This chapter provides an overview of the soundscape in the BPNS and the inherent challenges to measure and analyze it. Some of the challenges are exemplified using data collected in the framework of the LifeWatch Broadband Acoustic Network.

**Keywords**

Flow-noise · Bio-fouling · Masking · Shallow waters acoustics · Unknown sound sources

## Introduction

The growing human pressure which is applied to the ocean has raised concern. Consequently, among other pressure agents, it is critical to evaluate the effect of anthropogenic sound on marine fauna. These effects include physical and behavioral effects but also masking of sounds, both for sounds produced for communication and sounds used as cues to obtain information crucial for survival (Duarte et al. 2021). However, to quantify the pressure that anthropogenic sound exerts in the marine fauna and estimate its effects, it is first necessary to understand the sound sources in the soundscape and determine the contributions of human-made sound.

To this effect, soundscape analysis has been a growing field in recent decades. Soundscapes offer rich descriptions of composite acoustic environments, and they provide a more holistic view of the ecosystem than single event analysis. Describing and characterizing a soundscape can comprise one or multiple methods from listening to recordings, via visual inspection of spectrograms, to automated detection of signals of interest, and computation of several acoustic metrics (Schoeman et al. 2022). In this process, understanding natural variability of a soundscape is important for using soundscapes as an ecological and management tool to determine the potential effects of human activity (McKenna et al. 2021).

Often, soundscape ecology is based on the identification of sound sources and their subsequent classification into biophony, geophony, or anthropophony. As a consequence, when the sound sources are not known, or the anthropogenic sound is too loud, some of the traditional ways of analyzing soundscapes do not provide enough information for ecological assessment. For this reason it is necessary to develop new techniques to analyze soundscapes focused on shallow water highly

used marine areas. There has been some focus on developing more integrated approaches to characterizing soundscapes, for example, adding contextual information to understand possible sound sources (McKenna et al. 2021; Parcerisas et al. 2023), but there is still a knowledge gap.

The Belgian Part of the North Sea (BPNS) is located in the southern part of the North Sea. It only reaches 45 nautical miles offshore and it is very shallow. It has an average depth of 20 m and a maximum of 45 m. The BPNS is characterized by a unique subtidal sandbank system, which is very dynamic and has strong tidal currents (Kerckhof and Houziaux 2023). The sandbank system is subdivided into shallow sandbanks, sand mason worm banks, and gravel beds. Shipwrecks, which are artificial hard substrates in these sandbank systems, attract various species (Belgium Health, Food Chain Safety and Environment 2016). The BPNS is also one of the most exploited marine areas in the world. In an area that covers only 0.5% of the entire North Sea, activities such as shipping, blue energy, fisheries, and dredging converge. Every year, there are more than 300,000 shipping movements, ranging from commercial ships to pleasure boats. In addition, Belgium is the country with the highest percentage of offshore renewable space in the world (Belgian Federal Public Service (FPS) Health, Food 2020).

Highly exploited marine areas, such as the BPNS, have important economic value but are threatened by overexploitation of resources. Therefore, it is necessary to understand the ecosystem and its limits to protect it so that use can continue in a sustainable way.

In this chapter, we provide an overview of the particularities of shallow and heavy-explored soundscapes, and we present a list of some of the challenges which can arise when analyzing them. Furthermore, we exemplify them using data collected in the framework of the LifeWatch Broadband Acoustic Network (Parcerisas et al. 2021).

## Sound Sources

### Biophony

Biophony comprises all sounds produced by non-human organisms. Sound is an important source of information for many marine animals. For this reason, all vertebrates and many invertebrates are sensitive to sound (Duarte et al. 2021).

The BPNS can be characterized by a positive east-west gradient of species richness and densities in coastal waters (Cattrijsse and Vincx 1999). Furthermore, a clear coastal-offshore pattern is visible in the plankton communities (Aubert et al. 2022). In the BPNS, there are four main benthic communities linked to different seabed habitats. From these habitats, the sandbanks cover more than 80% of the North Sea, and they consist of gullies, flanks, and a sandbank top. Each part is home to one of the benthic communities. The highest biodiversity is found on the flanks and in the gullies. Further out at sea, there are also gravel beds, which consist of coarse sand and gravel bigger than 2 mm. They have an important function as a

breeding ground and nursery for some species, but the current species composition of the gravel beds has been impoverished by frequent fishing. There used to be banks of the native flat oysters, which create their own micro-habitat just like the sand mason worm. However, these were fished empty in just a few years from the end of the nineteenth century. Additionally, shipwrecks are artificial hard structures that are hotspots for biodiversity. During a study of ten shipwrecks in the BPNS, more than 200 animal species were observed (Zintzen and Massin 2010).

There are 140 different species of fish listed in the BPNS, and the largest group is the demersal species. The many sandbanks provide shelter and breeding ground for many species. All studied fishes have been shown to be sensitive to sound (Popper et al. 2019), although not all species use sound for communication. Nevertheless, sound production is vital for territorial defense, reproduction, and feeding for many species of fish. In a recent review about fish sound production, Looby et al. (2022a) identified 989 actively soniferous fish species, evenly distributed among all the geographical areas of the world. However, this represents a substantial underestimate of global actively soniferous fish diversity as over 96% of fish species lack published examination. The active sounds they produce are usually in the lower frequencies (<1 kHz), and are often described as "grunts," "growls," "thumps," "knocks", among others (Amorim 2006). In addition to active sounds, it is probable that all fishes produce passive sounds, such as feeding sounds, but it still remains unknown which ones do it often and have a recognizable signature. Due to the importance of sound for many species of fish, masking of their signals can have a large impact on their fitness (Popper and Hawkins 2019). Despite the importance of fish sounds, the field of fish bioacoustics has historically been less developed than for other taxa higher in the food chain (Looby et al. 2022a). Moreover, the studies on fish sound production are biased toward fish species from areas where visual ground-truthing is possible.

Marine invertebrates can produce and use sounds for a variety of behaviors. They can generate the sound unintentionally or deliberately for communication. The capacity to produce sounds is known in only three groups of marine invertebrates: bivalves, echinoderms, and crustaceans (Solé et al. 2023). From the crustacean group, only barnacles (Cirripeda) and decapods (Decapoda) have been observed to produce sound. However, many species have been described to be sensitive to sound. They are the group of animals living in the BPNS less studied in terms of sound production and reception, and there exist very few automated detectors focused on sound produced by marine invertebrates.

The most common marine mammal in the BPNS is the harbor porpoise (*Phocoena phocoena*), which is using echolocation to forage, navigate, and communicate. These echolocation clicks are very typical and concentrated around 132 kHz, making them a perfect target to study using passive acoustics (Gillespie et al. 2005; Todd et al. 2009; Verfuß et al. 2005). Harbor porpoises are present throughout the year, but are found in greater abundance between January and April. Other vocalizing cetaceans are the bottlenose dolphin (*Tursiops truncatus*) and the white-beaked dolphin *Lagenorbynchus albirostris*, which are also residents in Belgian waters. Furthermore, less vocal marine mammals are the grey and common seal (*Halichoerus grypus* and *Phoca vitulina*, respectively).

In Table 1 a non-exhaustive list of the species of the BPNS that have been reported to produce sound is provided, together with the available information about the sound they produce.

## Anthropophony

The most important source of anthropogenic sound in the BPNS is shipping. Farcas et al. (2020) reported that the shipping sound exceeded the modeled ambient noise by more than 50% of the time in most of the Belgian Exclusive Economic Zone (EEZ) and by more than 10% in the resting area. They also reported that the annual median ship noise excess in the BPNS is more than 20 dB from 63 Hz to 4 kHz for the shallower waters of the North Sea. For the deeper areas, the sound in these bands attenuated clearly in frequencies above 250 Hz. This means that there are very few places free from shipping noise in the BPNS.

Although knowing the total sound level generated by shipping is crucial for policymakers, shipping sounds can be very different from each other. Shipping noise has most of its energy at low frequencies (50–150 Hz), but sound can be up to 10 kHz (Zhang et al. 2020). Furthermore, several characteristics of the ship (type, size, speed, maintenance state, operational settings, etc.) influence its source levels and frequency distribution (MacGillivray and de Jong 2021). Small recreational vessels have a different sound signature than large cargo vessels, and anchored ships also contribute differently from moving ships. Furthermore, in Belgium there are also dredging and trawling activities, which are also audible in the soundscape and have a clear and recognizable signature. All these differences should be reflected in soundscape analysis for ecological purposes.

Most studies concerning shipping sound use data from the Automatic Identification System (AIS) combined with models of ship sound production (MacGillivray and de Jong 2021). In the North Sea, several propagation models have been developed, in the context of the JOMOPANS project, and AIS-based sound level models have been developed for the Dutch part of the North Sea (Sertlek et al. 2016). AIS data are commonly used to predict ship intensities on large spatial and temporal scales. They have been proven to be a valid source of information on global sound levels produced by shipping noise, which is useful for policy and management decisions (Farcas et al. 2020). Nonetheless, small vessels without AIS transponders and vessels switching their transponders off should not be underestimated in terms of sound pollution. This is particularly important in shallow waters, where the low frequencies are not propagated as well and therefore shipping contribution could be concentrated in higher frequencies generated by smaller recreational boats (Hermannsen et al. 2019).

In addition to the unintentionally generated noise, some shipping activity also produces some intentional sounds using sonar. Sonars use frequencies higher than those of sound unintentionally generated by ships, typically between 2 and 400 kHz, depending on the application.

**Table 1** Summary of all the known soniferous species of the BPNS

| Group | Scientific name | Common name | Sound production | | | |
|---|---|---|---|---|---|---|
| | | | Type | Bandwidth | Frequency peak | Comment |
| **Fish** (Looby et al. 2022b) | Anguilla anguilla | European eel | Click | 10 kHz | ** | ** |
| | | | Knock | ** | ~200 Hz | ** |
| | | | Thump | | | |
| | Brosme brosme | Tusk | Drumroll | ** | ~185 Hz | ** |
| | | | Thump | 100 to 800 Hz | ~100 Hz | ** |
| | Chelidonichthys cuculus | Red gurnard | Growl | 1500 Hz | ** | ** |
| | Chelidonichthys lucerna | Tub gurnard | Growl | 1500 Hz | ** | ** |
| | Clupea harengus | Atlantic herring | Grunt | ** | ~60 Hz | ** |
| | | | Knock | | | |
| | | | Thump | | | |
| | Conger conger | European conger | Passive | ** | ** | Feeding noises |
| | Eutrigla gurnardus | Grey gurnard | Growl | 100–2000 Hz | ~100 Hz | ** |
| | | | Grunt | 100–1500 Hz | | |
| | | | Knock | 100–1000 Hz | | |
| | Gadus morhua | Atlantic cod | Grunt | 500 Hz | ~70 Hz | ** |
| | Hippocampus guttulatus | Long-snouted seahorse | Click | 1500 Hz | ** | ** |
| | Hippocampus hippocampus | Short-snouted seahorse | Click | 1500 Hz | ** | ** |
| | Melanogrammus aeglefinus | Haddock | Hum | 600 Hz | ** | Multiple haddock will produce a rumble from 30 to 400 Hz |
| | | | Knock | | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| | Merlangius merlangus | Whiting | Passive | ** | ** | Feeding noises |
| | Merluccius merluccius | European hake | Knock | ** | ~50 Hz | Weak |
| | Molva molva | Ling | ** | ** | ** | Only morphological analysis, sound not described |
| | Myoxocephalus scorpius | Shorthorn sculpin | Groan | ** | ~76 Hz | Dull |
| | | | Growl | | | Sustained |
| | Pegusa lascaris | Sand sole | Passive | ** | ** | Feeding and other noises |
| | Petromyzon marinus | Sea lamprey | Active | ** | ** | Sound not described |
| | Pollachius pollachius | Pollack | Grunt | 700 Hz | ~100 Hz | ** |
| | Pollachius virens | Saithe | Thump | 400 Hz | ~10–60 Hz | ** |
| | Pomatoschistus | Gobies | Drum | ** | ~150–190 Hz | ** |
| | | | Thump | ** | ~83 Hz | ** |
| | Salmo salar | Atlantic salmon | Gurgle | 300–2000 Hz | ** | ** |
| | | | Moan | 500–1000 Hz | ** | ** |
| | Scomber scombrus | Atlantic mackerel | Grunt | ** | ** | Harsh |
| | Scophthalmus maximus | Turbot | Thump | ** | ~60 Hz | ** |
| | Sebastes mentella | Beaked redfish | ** | ** | ** | Only morphological analysis, sound not described |
| | Sebastes norvegicus | Golden redfish | ** | ** | ** | Only morphological analysis, sound not described |
| | Trachurus trachurus | Atlantic horse mackerel | Grunt | 300–5000 Hz | ** | ** |

(continued)

**Table 1** (continued)

| Group | Scientific name | Common name | Sound production | | | |
|---|---|---|---|---|---|---|
| | | | Type | Bandwidth | Frequency peak | Comment |
| | Trisopterus minutus | Poor cod | Passive | ** | ** | Feeding noises |
| | Zeus faber | John dory | Bark | 200–600 Hz | 312 ± 10 Hz | ** |
| **Invertebrates** (Solé et al. 2023) | Cirripeda | Barnacle | Peak pulses | ** | ** | 1–3 ms |
| | Homarus gammarus | European lobster | Buz | ** | ** | ** |
| | | | Rattle | | | |
| | Pecten maximus | Great scallop | Cough | 20–27 kHz | ** | Can be clusters or isolated |
| **Mammals** (Ainslie et al. 2009; Richardson et al. 1995) | Halichoerus grypus | Gray seal | Calls | 0.1–5 kHz | 0.1–3 kHz | 6 different call types (breeding period) |
| | | | Click | 0–30 kHz | ** | ** |
| | | | Hiss | 0–40 kHz | ** | ** |
| | | | Knock | ≤16 kHz | ≤10 kHz | ** |
| | Lagenorbynchus albirostris | White-beaked dolphin | Click | ≤325 kHz | ** | ** |
| | | | Squeals | ** | 8–12 kHz | ** |
| | Phoca vitulina | Harbor seal | Growl | 100–400 Hz | 100–250 Hz | Bubbly |
| | | | Click | | 12–40 kHz | ** |
| | | | Creak | 0.7–4 kHz | 0.7–2 kHz | ** |
| | | | Grunt | 0.1–4 kHz | ** | ** |
| | | | Groan | | | |
| | | | Social | 0.5–3.5 kHz | ** | Pulse duration 0.019–0.4 s |
| | | | Roar | 0.4–4 kHz | 0.4–0.8 kHz | ** |

| | | | | | | |
|---|---|---|---|---|---|---|
| | Phocoena phocoena | Harbour porpoise | Click | 2 kHz | ** | Not echolocation |
| | | | Click | 110–150 kHz | ~132 kHz | ** |
| | Tursiops truncatus | Bottlenose dolphin | Click | 110–140 kHz | ** | Echolocation |
| | | | Bark | ** | ** | ** |
| | | | Mew | | | |
| | | | Grate | | | |
| | | | Rasp | | | |
| | | | Yelp | | | |
| | | | Narrowband | 2 kHz | 0.3–0.9 kHz | ** |
| | | | Whistle | 0.8–24 kHz | 3.5–14.5 kHz | ** |

In addition to all the shipping lanes, Belgium is also the country with the highest percentage of renewable offshore space in the world (Belgian Federal Public Service (FPS) Health, Food 2020). Wind turbines do not produce loud sounds during operations, though the sound production of large offshore wind farms should not be underestimated because they elevate the ambient noise due to cumulative effect (Tougaard et al. 2020). During the construction phase, pile driving can reach very loud levels (Madsen et al. 2006). These impulsive sounds appeared to be correlated with a large-scale avoidance of the construction zone by porpoises, which led the Belgian Federal Government to implement a maximum threshold of 185 dB re 1 μPa (sound pressure level, zero to peak) at 750 m from the source for pile driving events. Therefore, since 2017, offshore wind farm constructors have developed sound mitigation measures that are applied during their construction phase to reduce the sound produced (Degraer et al. 2021; Norro 2018).

Other sources of anthropogenic sound present in the BPNS are sparkers (frequency range 200 Hz to 2 kHz), sub-bottom profilers (frequency range 2 to 20 kHz), and underwater explosions, which are characterized by very loud impulsive sound sources which are only active for a short period of time.

## Geophony

Some of the geophony sound sources in the BPNS are related to current: strong currents around hard structures at closer distances or very loud sources further away (several kilometers) such as coastal surf and tidal river currents. In Rogers et al. (2021), they relate these distant sound sources with the tide. Together with particle motion measurements, they propose that these low-frequency sound sources of 10–20 Hz and 20–40 Hz are possibly river mouths from some of the main Belgian, Dutch, and English rivers (Scheldt Estuary, Rhine Delta, North Sea Canal, and Thames River).

Current is also the main agent for sediment transport, where seabed material is brought into suspension by currents (Cotter 2008). The collision of particles generates high-frequency sound, which is largely dependent on the grain size and speed of the particles during impact (Thorne 2014). The theoretical peak frequency of the generated sound can be approximated according to the formula in Eq. 1 (Thorne 2014):

$$f_{peak} = 0.15\left\{\frac{E}{\rho_s(1-\sigma^2)}\right\}^{0.4}\frac{U^{0.2}}{a} \tag{1}$$

Where

for glass spheres: $E = 7.10^{10}$ $\mathrm{Nm}^{-2}$ is the Young's modulus, $\sigma = 0.2$ is the Poisson's ratio $\rho_s = 2500$ $\mathrm{kgm}^{-3}$ is the density.

And $U$ is the collision speed and $a$ is the radius.

Therefore, considering a collision speed between 0.2 and 0.6 $ms^{-1}$ (Vindenes et al. 2018) and a radius between 50 μm and 1 mm (Lee et al. 2012), the frequency of sound produced by sediment transport in the BPNS should be between 12 and 300 kHz.

Besides, current generates flow-noise, which is not actual sound which propagates but just pressure turbulence generated by the shape of the hydrophone and/or the anchoring system. In any case, even if it is not a sound wave, this turbulence is picked up by the hydrophone and has to be removed before analyzing the data. Flow-noise and the other sounds generated by the current dominate the lower frequencies of the BPNS soundscapes, below 100 Hz. An example of this effect can be seen in Fig. 2a.

Another source of nonbiological natural underwater sound is wind, which appears between 100 Hz and 25 kHz, and the sound level generated depends on its speed (Wenz 1962). The higher frequency components come from oscillating and bursting bubbles and splashes created by breaking wave crests (Cotter 2008). Bubbles generated by waves in shallow waters also contribute to the decrease in the propagation of other sound sources (Ainslie 2011).

Last, rain can also contribute significantly to underwater sound between 1 and 50 kHz. The sound is generated from the splashes of raindrops falling in the water. The generated sound depends on the drops size distribution, the wind speed, and the rainfall rate. In the lower frequencies, lightning is a less common event, but it is very loud compared to other sources of natural noise such as wind or rain (Ainslie et al. 2009).

## Common Soundscape Analysis Techniques and Their Applicability in Shallow Anthropogenic Marine Areas

As identified in the report of the International Quiet Ocean Experiment (IQOE) on marine bioacoustics standards (IQOE – Marine Bioacoustical Standardization Working Group 2021), there are currently no standards to measure, characterize, and analyze underwater soundscapes.

Regarding data acquisition, there are no international standards or procedures on how to measure underwater sound for ecological analysis. Different projects have published their best practices (Robinson et al. 2014; Fischer et al. 2021; European Commission. Joint Research Centre. Institute for Environment and Sustainability 2014), but different duty cycles, sampling frequencies, anchoring systems, measuring depth, and so on make it hard to compare results between them. There are also different approaches regarding calibration and data cleaning (Robinson et al. 2021; Ward et al. 2021). For frequency domain calculations, a frequency-dependent calibration can be applied, but this is difficult for calculations in the time domain. Regarding data cleaning, it is common to have artifacts when recording such as flow-noise, electronic noise from the instrument, or sounds produced by the anchoring system. Some approaches propose the removal of these artifacts, and in some studies, these are not reported.

Regarding data processing, there are no standards for analyzing and reporting these soundscapes, which makes them difficult to compare among each other. Some studies focus on temporal patterns, while others report broadband frequency spectrum per site. Nonetheless, some practices have become widely used to analyze

underwater soundscapes, as summarized in Schoeman et al. (2022). Some of these analyses focus on a general overview. They can be used to describe the soundscape of a particular habitat or to compare multiple habitats between them. Data are generally explored using spectrograms for shorter periods (minutes or hours) and the long-term spectral average (LTSA) for longer periods (days, months, or years) (Schoeman et al. 2022). Spectral Probability Density (SPD) plot is a common way to plot a soundscape summary, it gives an idea of how often certain frequencies appear at a certain sound pressure level (Merchant et al. 2013). However very informing, these features are multidimensional and therefore not straightforward to compare between sites and moments. To ease this problem, several acoustic indices have been developed. These indices summarize the acoustic information from a particular soundscape into a single value. They have been used to characterize the acoustic features of the soundscape and to test their relation to the structure of the ecological community, habitat quality, and ecological functioning in ecosystems (Sueur et al. 2014; Buxton et al. 2018). They have been successfully implemented in terrestrial ecosystems, but it is not clear whether they are robust in marine environments (Bohnenstiehl et al. 2018).

Another common way of analyzing soundscapes is to compute the Sound Pressure Level (SPL) broadband or at specific frequency bands of interest. These frequency bands are sometimes selected according to the expected sound sources of interest or known sensitivity of certain species, but third-octave bands or decidecade bands are also commonly used. Recently, Miksis-Olds et al. (2021) proposed the use of hybrid millidecade bands as the best current solution to share trends in ocean noise. Per contra, characterizing marine soundscapes using sound levels of frequency bands is not appropriate to answer other research questions, because it does not contain all the information about the frequency and temporal distribution of the soundscape and its unique acoustic characteristics. For example, when studying biophony in areas dominated by anthropogenic sound, averaging in frequency bins can lead to the fading of some shorter and less common biological sounds.

Apart from general trends in the soundscape, other analyses can be performed to identify known sound sources and to quantify their distribution and percentage. Usually, these are certain species calls and/or ship sounds. The detection of sound sources is often performed manually, but this approach is only feasible for small amounts of data. This is why over the last decades the interest in automatic detectors and classifiers has increased. These automatic source detectors are based on rule-based algorithms, where the user defines certain thresholds for hand-picked acoustic features, to fully unsupervised machine learning algorithms. In the BPNS, this is the case with the harbor porpoise and the dolphins, which can be studied because their vocalizations are known. Moreover, masking occurs mainly at frequencies below the echolocation clicks of harbor porpoises and dolphins. Therefore, these can be monitored using Passive Acoustics Monitoring (PAM), which is already happening within the framework of the LifeWatch Cetacean Network (Debusschere et al. 2015) with the use of C-PODs. These devices can detect the stereotypical clicks produced by harbor porpoises and store the metadata of these detections. The information of the spatio-temporal distribution of these mammals can then be used in impact assessments. However, the source

code used in the C-PODs to detect the echolocation clicks is not available, and therefore it is a "black box" which is hard to evaluate. To overcome this, there are several other open source algorithms that can be used on raw acoustic data or on the output of a SoundTrap Click Detector (OceanInstruments, New Zealand), such as PAMGuard (Gillespie et al. 2009) and D-PorCCA (Cosentino et al. 2019). These algorithms have to be applied after the raw acoustic data are collected because they are not embedded in a sensor. This has the advantage of also having access to the raw data and therefore to more information and easier performance evaluation, but the disadvantage of a higher battery consumption and storage need.

For shallow, highly used areas such as the BPNS, some of these approaches are not descriptive enough. For example, the Joint Research Center of the European Commission set root mean squared (RMS) sound levels in 1/3 octave bands centered at 63 and 125 Hz as indicators of ship noise (Tasker et al. 2010). In a shallow area like the BPNS, these frequencies are below the theoretical cut-off frequency in places where the water depth is shallower than 70 and 35 m, respectively. Therefore, these indicators could lead to misleading information of shipping sound. Additionally, RMS levels are sensitive to outliers, so a few very loud events will affect the mean (Merchant et al. 2016). Furthermore, other shipping activities such as trawling and dredging have distinct sound signatures from underway ships, and should also be considered.

The underwater soundscape of the Belgian Part of the North Sea has been very little studied. In general, most of the soundscapes analysis in literature is focused on describing natural or pristine areas, or to compare managed ecosystems with non-managed ones. Therefore, the tools developed so far to describe and analyze underwater soundscapes may not apply to highly exploited areas.

In the following subsections, we list some of the most common challenges and particularities of recording and analyzing underwater sound in shallow highly anthropogenic areas. To exemplify these challenges, we analyzed data from the LifeWatch Broadband Acoustic Network (Parcerisas et al. 2021). These data were collected using a RESEA 320 recorder (RTSys, France) together with a Colmar GP1190M-LP hydrophone (Colmar, Italy, sensitivity: −180 dB/V re $1\mu Pa$, frequency range −3 dB: 10 Hz to 170 kHz). All acoustic data were processed using the Python package pypam (Parcerisas 2022).

## Propagation

As summarized in Ainslie and Dahl (2014), in shallow water, propagation is influenced by multiple reflections from the seabed and sea surface, so geometrical spreading laws need to be modified. The resulting effect depends on the kind of source and its directivity. Several other studies have focused on trying to understand the propagation patterns in shallow marine areas and simplify their computation (Sertlek 2016; Sertlek et al. 2016; Sertlek et al. 2013). Furthermore, the cut-off frequency is the minimum frequency at which sound will propagate. For shallow water, the cut-off frequency can be approximated to (Jensen et al. 2011) (Eq. 2):

$$f_{cutoff} = Cw/\left(D * \sqrt{1-(Cw/Cb)^2}\right) \quad (2)$$

Where

$f_{cutoff}$ is the cutoff frequency
$Cw$ is the velocity of the sound in water
$Cb$ the sound velocity in homogeneous bottom
$D$ depth in meters

In the BPNS the cut-off frequency is somewhere between 100 Hz and 400 Hz, depending on the water depth. Due to the change of water depth generated by the tides, the cut-off frequency changes with the tide, and so do the propagation patterns. Consequently, the local water depth and bottom material have a clear influence in the propagation of further away sound sources (Ainslie et al. 2009). For this reason, some sources at lower frequencies do not propagate, and others are distorted when propagating simultaneously in the water column and the seabed.

On the other hand, in the BPNS the water mass is considered to be mixed, and therefore there are no big changes in salinity and temperature. This makes the sound speed constant throughout the water column (Ivanov et al. 2020). Furthermore, in shallow waters, the sounds generated at the sea surface by rain, wind, or waves have an influence in the entire water column and can be received by the benthic habitats. This is because there are more reflections and a smaller propagation range to attenuate the sound.

## Flow Noise and Bio-Fouling

Another challenge of measuring underwater sound in the BPNS is flow-noise. The turbulence generated by the current around the hydrophone or the mooring system produces pressure changes that are not actual propagating sound. This pseudo-noise, however, is picked up by the hydrophone anyway and can mask other sound in the lower frequencies below 1000 Hz, limiting the capability to reliably measure ambient noise (Bassett et al. 2014; von Benda-Beckmann et al. 2016). This turbulence depends on the current speed. In the North Sea, the tidal current is estimated to range from a few $cms^{-1}$ to 67.7 $cms^{-1}$ (Vindenes et al. 2018). Furthermore, the flow-noise recorded by the hydrophone depends on the shape and size of the hydrophone and/or the anchoring system.

When deploying bottom long-term observatories in the North Sea, the instruments will be exposed to bio-fouling. The growth of living organisms on top of the sensors can lead to a change in sensitivity (Heupel et al. 2008). The anchoring system is an artificial hard substrate that is sometimes rapidly colonized by many organisms, depending on the location and season. The fauna residing on top of the mooring therefore can affect the soundscape. Furthermore, the presence of bio-fouling on top of the hydrophone can lead to changes in sensitivity, particularly to the sensitivity of the hydrophone to pick up flow-noise.

We exemplify these particularities with data analysis from deployments in the BPNS. For this example, three frequency bands were selected for analysis: broadband (up to 24,000 Hz), low frequency (10–1000 Hz), and higher frequency (1000–10,000 Hz). For each of the band, the signal was filtered using a band-pass Butterworth filter of order 4, and data were down-sampled to twice the higher frequency limit. Next, the RMS value of the calibrated data was computed for each file, of equal duration (10 min and 15 s). For each processed point, the bottom current was retrieved from the COHERENS model for the North Sea (MFC ODNature RBINS 2023) and assigned using the Python package bpnsdata (Parcerisas 2021).

This process was done for all the recording stations (see Fig. 1). In some deployments, it could be observed that the low-frequency band was dominated by a source that is correlated with the current speed, which we assume is flow-noise (e.g. 2a). However, this pattern did not repeat in other deployments (e.g. 2b). This could be due to a different position of the anchoring system with respect to the current and the sand banks, or it could be due to the fact that the sound levels in 2022 were higher than in 2021, and these loud sources could be masking the flow-noise. When checking the AIS-based route density from EMODnet (EMODnet Human Activities 2022a), the average between March and May in the vicinity of the Grafton station is 173.75 routes/km$^2$/month for 2021 and 196.83 routes/km$^2$/month for 2022, so the increase in sound levels could be due to increased shipping.

In the deployments where flow-noise was present, most of them seemed to show a decrease in correlation with the current after a certain time. The hypothesis was that

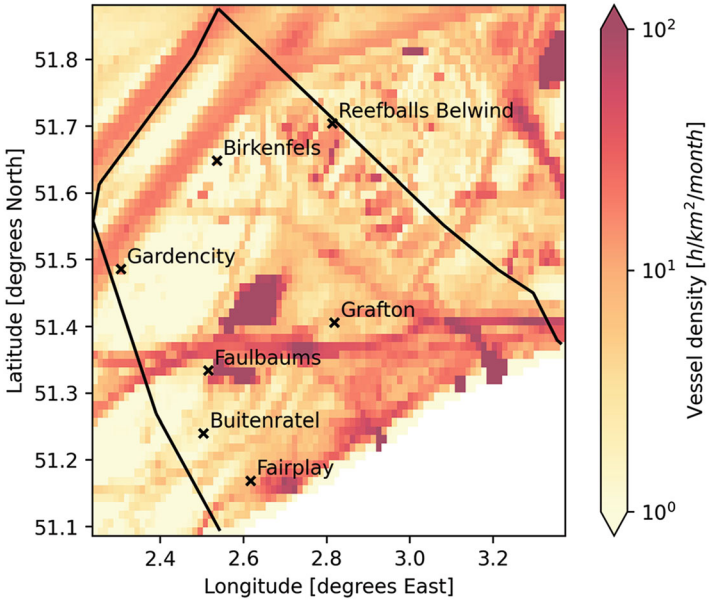



**Fig. 1** Vessel density from EMODnet (EMODnet Human Activities 2022b) in the BPNS, together with the station location of the LifeWatch Broadband Acoustic Network (Parcerisas et al. 2021)

the bio-fouling growing on top of the hydrophone would decrease the flow-noise picked up by the hydrophone or its sensitivity. A linear mixed effects regression as described in Eq. 3 was fitted to check whether the effect of the number of days deployed was significant. To this effect, only deployments with a total correlation higher than 0.4 between the sound pressure in the low-frequency band and the current were included. The deployments with a low correlation with flow-noise were assumed to be in a spot where currents were less strong because of the sandbanks or the protection of a nearby shipwreck. Because the effect of the bio-fouling was not expected to change proportionally to the days but rather at a

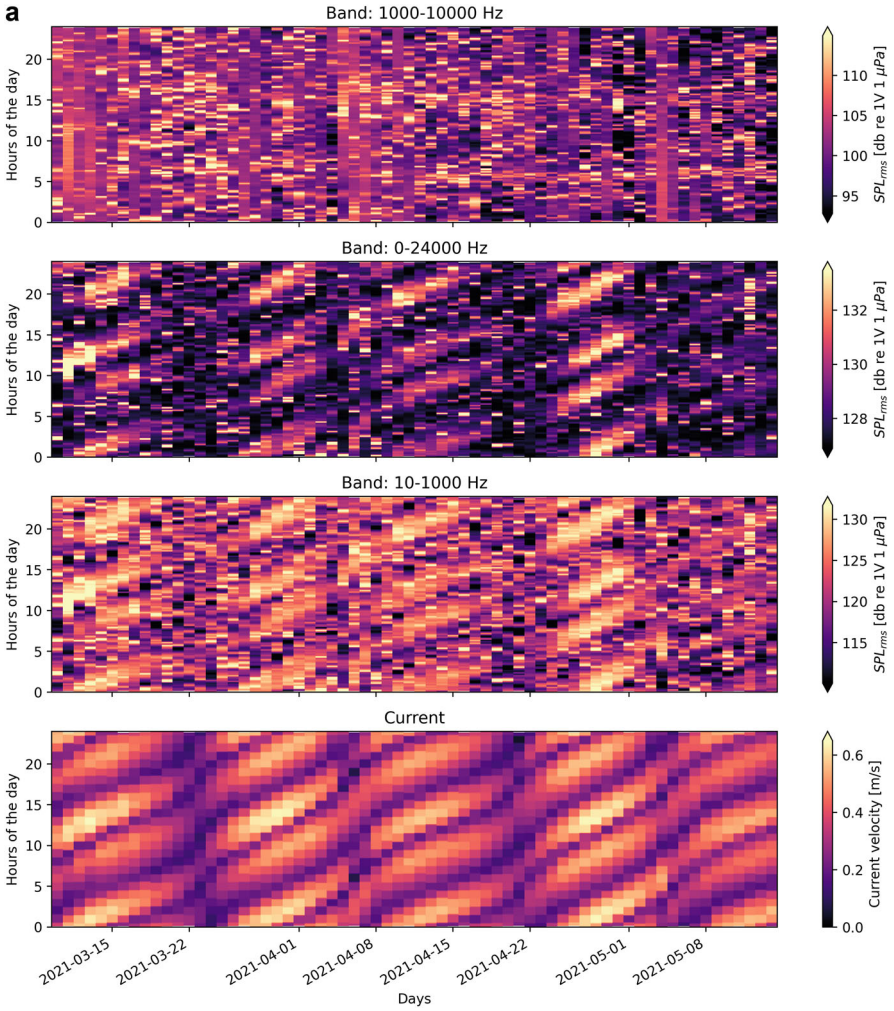



**Fig. 2** Root mean squared sound levels in station Grafton in different bands (three top), and total current bottom speed (bottom). Deployments performed in (**a**) 2021 (**b**) 2022

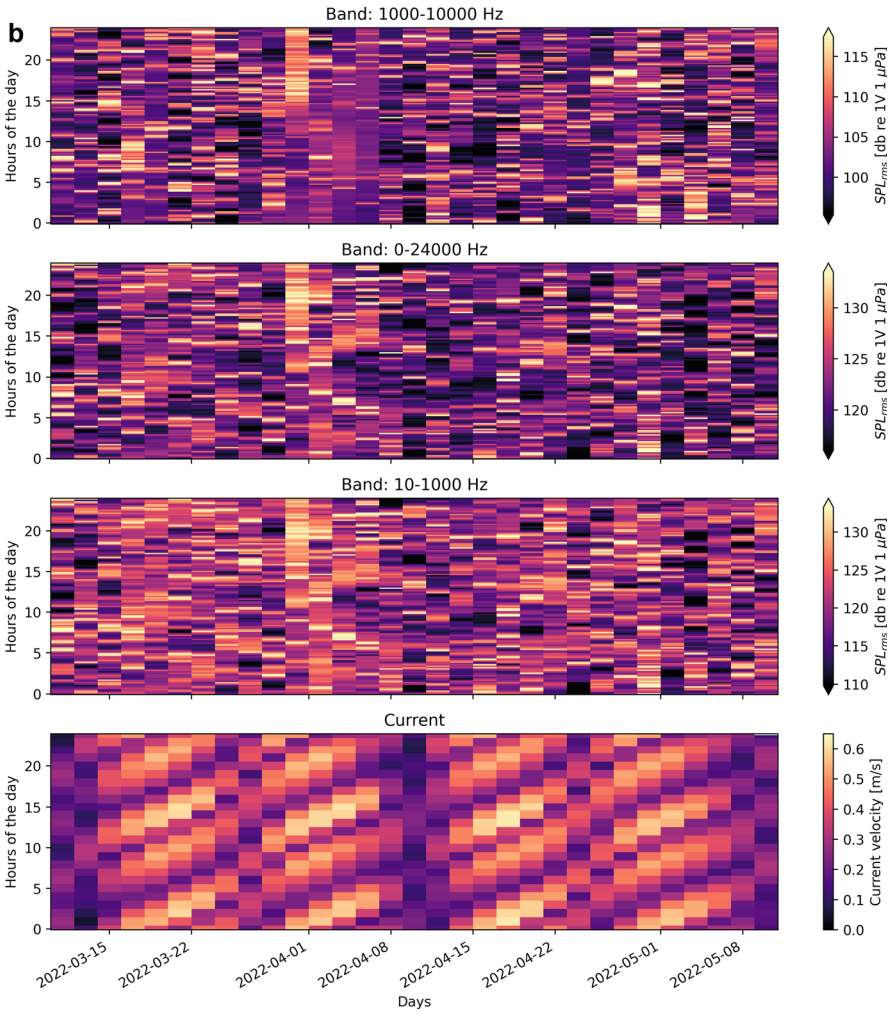



**Fig. 2** (continued)

decreasing rate, the variable expressing the days since deployment was converted to a logarithmic scale.

$$SPL_low_frequency \sim current + days_since_deployment + current : days_since_deployment + (1|deployment) \quad (3)$$

The results (see Table 2) showed that the days since deployment, the current and the interaction between the two were significant. The interaction and the days since deployment have a negative coefficient, which could be because the flow-noise is diminished due to bio-fouling (in time), and it has a bigger effect when the

**Table 2** Summary of the results of the fitted linear model. p-value of deployment: 0.17, not significant

| Mixed Linear Model Regression Results | | | | | | |
|---|---|---|---|---|---|---|
| Model: | Mixed LM | | Dependent variable: | | lowfreq_sound | |
| No. of observations: | 38056 | | Method: | | REML | |
| No. of groups: | 5 | | Scale: | | 25.9264 | |
| Min. group size: | 705 | | Log-likelihood: | | –115961.0987 | |
| Max. group size: | 17484 | | Converged: | | Yes | |
| Mean group size: | 7611.2 | | | | | |
| | Coef. | Std. Err. | z | P>\|z\| | [0.025 | 0.975] |
| Intercept | 111.782 | 2.066 | 54.117 | 0.000 | 107.733 | 115.830 |
| days_since_deployment | –0.265 | 0.079 | –3.355 | 0.001 | –0.420 | –0.110 |
| current | 27.598 | 0.783 | 35.250 | 0.000 | 26.064 | 29.133 |
| days_since_deployment: Current | –2.732 | 0.206 | –13.245 | 0.000 | –3.137 | –2.328 |

current is high than when it is low. However, this effect could also be due to sand burying increasing with time, change of depth because of the movement of the sandbanks, loss of sensitivity from the hydrophone for technical reasons, or other causes.

## Unknown Sound Sources and Low Visibility

A common technique to link sound to its producer is visual correlation, both in the lab, in the wild with cameras deployed together with hydrophones or in the wild with visual surveys. Thanks to these efforts, all the marine mammals species are nowadays known to produce sound, and many of their signatures can now be recognized without the need for a visual confirmation.

However, many other biological sound sources are still unknown, particularly those of fish and invertebrates. The examination of fish and invertebrates for sound production can be complex, difficult, irresolute, and inefficient. Laboratory experiments can be time and resource intensive and do not guarantee results (Looby et al. 2022a), and field work to correlate sounds with their producers in areas with high turbidity, such as the BPNS (Fettweis and Van den Eynde 2003), is often not possible due to low visibility.

Furthermore, in shallow waters the propagation patterns can be complex, so sound signatures can be distorted when received (Ainslie and Dahl 2014). This complicates the recognition of certain sounds. Moreover, due to the propagation patterns, it is also complicated to know which environmental event produces received sounds, and therefore the mathematical models for sound sources comprising the lower frequencies are not correlated with the received levels.

As an illustration for these unidentified sound sources, some examples were manually selected. In Fig. 3 the short spectrograms of these sounds can be found, illustrating sounds that could be from a biological source, but have not been identified at the moment.

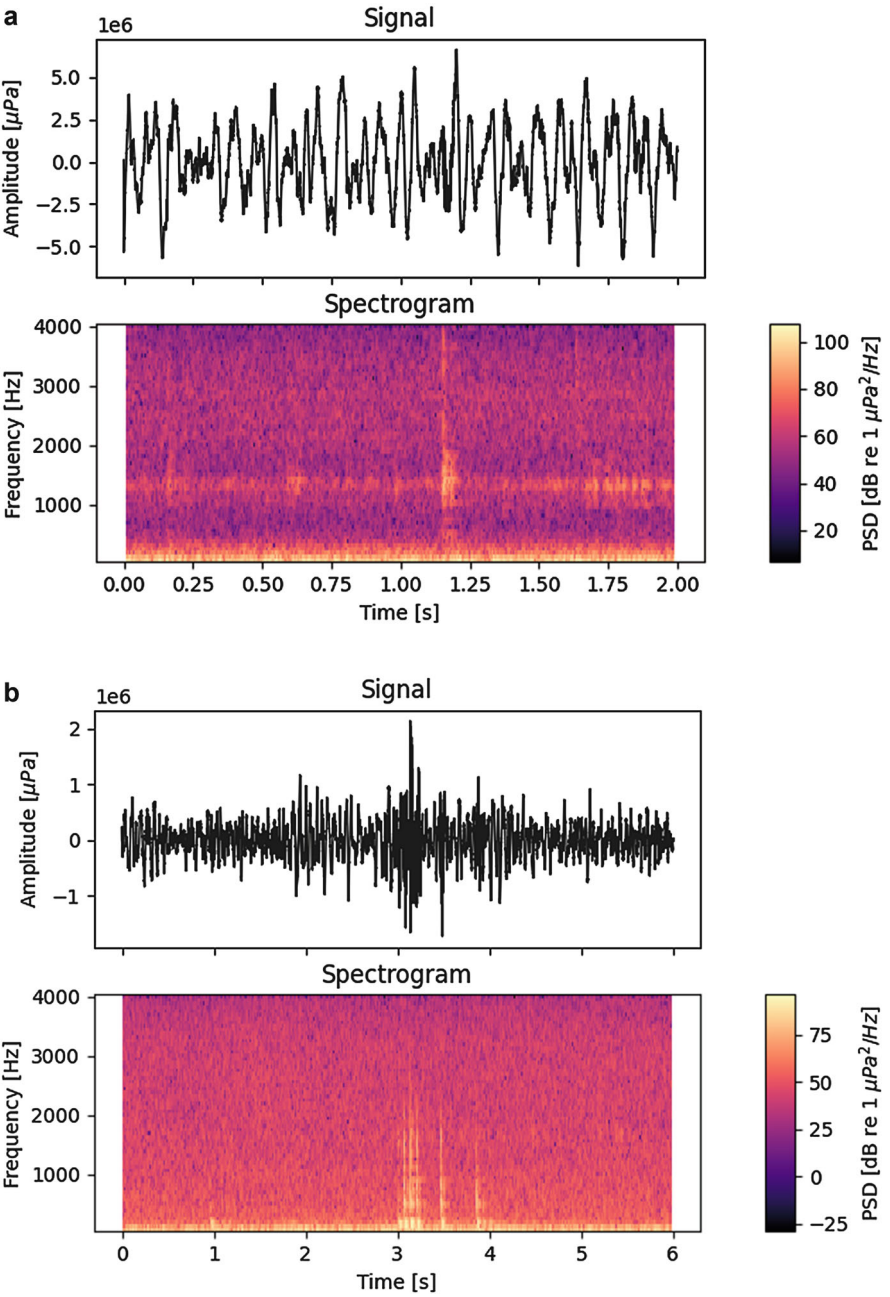



**Fig. 3** Several examples of unknown sound sources in the BPNS. For the short spectrograms computation, data were filtered using a band-pass Butterworth filter of order 4 between 10 and 8000 Hz, and data were down-sampled to 16,000 Hz. For the spectrogram computation, the length of the Fast Fourier Transform (FFT) was chosen to be 128, with a 70% overlap. The descriptions of the detected unknown sounds are (**a**) stridulation, possible source: crustacean, (**b**) single "drum," possible source: fish, (**c**) consecutive "pocs," possible source: unknown, (**d**) continuous "drumroll," possible source: fish

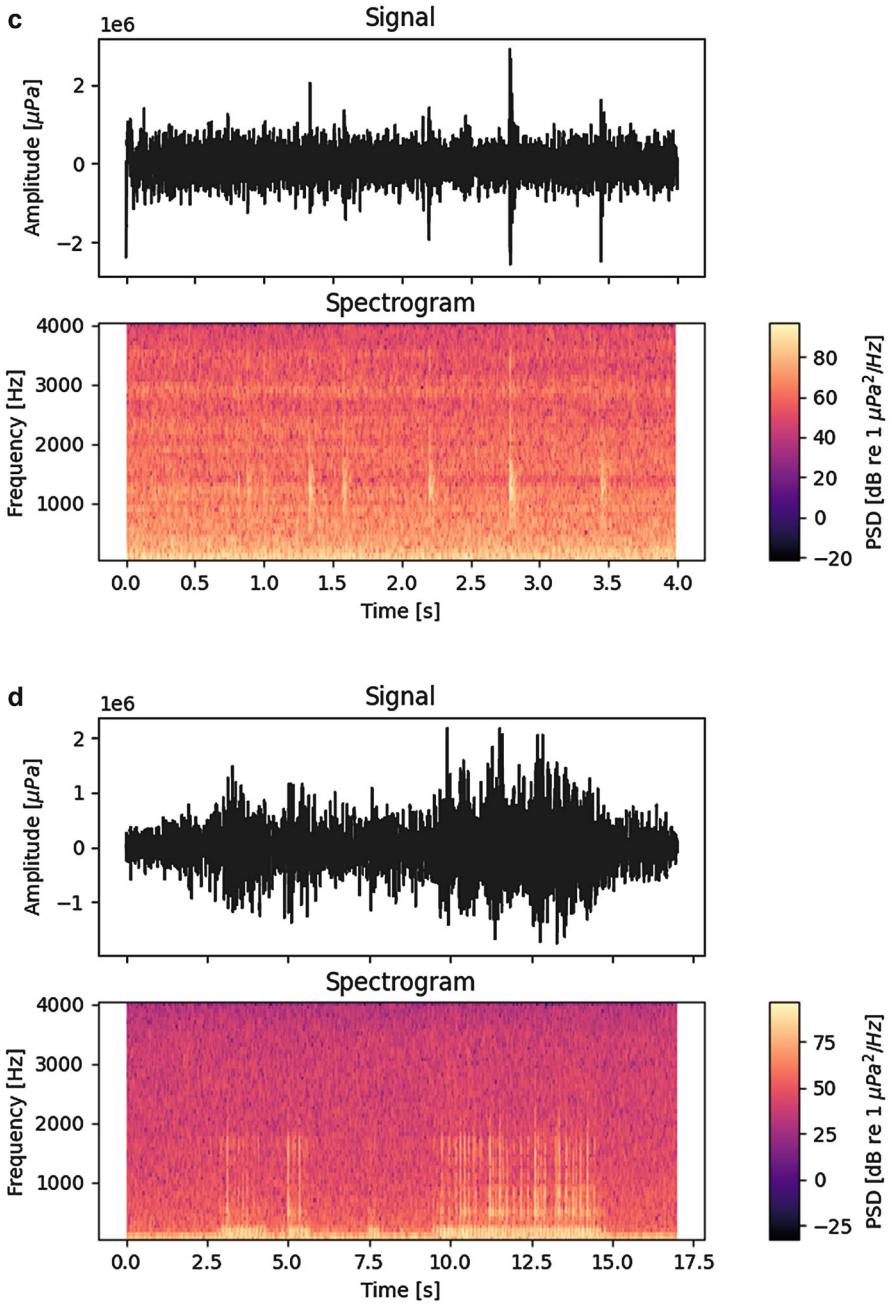



**Fig. 3** (continued)

### Masking

When interested in monitoring biodiversity using PAM, it is necessary to exclude almost constant anthropogenic noise to analyze biophony. This can be done by analyzing only quiet periods or with automatic denoising techniques. Shipping sounds have most of their energy from 50 Hz to 10 kHz (Zhang et al. 2020), which overlaps with most of the biological sounds produced by fishes, invertebrates, seals, and dolphins. This can lead to potential masking effect, where a louder sound source disrupts the reception of a weaker sound.

The sound produced by harbor porpoises is found at higher frequencies and is therefore not prone to be masked by human-made sounds. However, some sonar activities are in that frequency range, which are also detected by some click-detector algorithms such as the KERNO algorithm from the C-PODs (Chelonia Ltd., Penzance, UK) and could be a source of masking for these species.

Besides the impact that masking can have on marine fauna communication, it also complicates the soundscape analysis. Anthropogenic noise can decrease the ability of humans to detect certain sounds in short-term spectrograms. Furthermore, biological sounds can disappear in the general overview plots like LTSA or SPD because of their minor contribution to the soundscape compared to anthropogenic and geophonic sounds.

## Tackling Shallow Water Challenges

The challenges discussed above complicate marine soundscape analysis in shallow heavily used areas. Nevertheless, these can be tackled using pre- and post-processing techniques.

Concerning the technical limitations specific to the BPNS with respect to flow-noise, there are several ways of reducing its impact. One way of denoising flow-noise is to record simultaneously with an array of hydrophones and during the data processing discard the parts of the signal that are not correlated between the multiple channels. However, this approach can be costly due to the increased number of hydrophones needed and the increase in memory use. Furthermore, it also risks removing parts of the signal of interest (Beerens et al. 1999; von Benda-Beckmann et al. 2016). Some studies have been conducted to investigate mitigation techniques during recording (Wei et al. 2021), but there are still no accessible robust technologies that can be deployed in long-term monitoring stations that completely suppress flow noise. Finally, flow-noise is limited if the recording system is drifting (Merchant et al. 2015), because it moves with the water column. Therefore, drifting can be chosen as an acoustic recording strategy, which allows for a higher spatial resolution but a lower temporal one, but it is not applicable for long-term monitoring stations.

Regarding the deployment of equipment, in shallow waters it can be done without leaving anything behind on the sea floor (Goossens et al. 2020). Moreover, using this

type of moorings allows for practical and efficient procedures of deployment and recovery. The rigid design of the frame results in higher stability of the sensors compared to the common anchored buoy lines.

As a ground-truthing method to discover fish sounds and relate them to the species producing them, other methods besides direct visual observations can be used. For example, the European Tracking Network (ETN) aims to track aquatic animals across Europe using Telemetry, which uses acoustic tags to investigate the ecology and movement behavior of aquatic species in relation to their environment. This technology can thus be used to know when a certain fish species was present in the vicinity of the acoustic station. Therefore, the LifeWatch permanent acoustic receiver network in the BPNS (Reubens et al. 2017) combined with the LifeWatch Broadband Acoustic Network (Parcerisas et al. 2021) makes it possible to reduce the amount of acoustic recordings to be manually annotated to find fish vocalizations and could be used as a semi-ground-truthing mechanism.

Additionally, the recent development of the FishSounds database (Looby et al. 2022b) provides a framework to share knowledge about fish sounds, and the GLUBS initiative (Parsons et al. 2022) aims to create a collective and global open-access platform that provides a reference library of known and unknown biological sound sources. However, more research is still needed to verify and identify fish and invertebrate species that produce sound, document the behavioral contexts of their sound production, record and describe the acoustic characteristics of the sounds, and create a repository of sound production information and recordings (Looby et al. 2022b; Solé et al. 2023; Parsons et al. 2022).

Sounds which remain unidentified might be – or might not be – of importance for the marine fauna. Finding, reporting, and understanding the patterns of these unidentified sounds are of significant benefit to PAM and the assessment of underwater soundscapes. Furthermore, it can lead to an increase of the knowledge about the local acoustic community, as a proxy for biodiversity and ecosystem health. Documenting them before they are identified provides a baseline for their presence and extra information for later source identification.

Once more data is gathered on the sound production of fish and invertebrates, access to these open sound source libraries can lead to the further development of detectors and classifiers of fish (e.g. Urazghildiiev and Van Parijs 2016; Ibrahim et al. 2018; Vieira et al. 2015) and invertebrate sounds. Furthermore, with the advancement of machine learning tools, other approaches for marine sound identification can be developed which do not need annotated data or need little labeling effort. These approaches include unsupervised models, which have already been developed and applied to marine ecosystems, such as the one described in Ozanich et al. (2021). Other state-of-the-art machine learning algorithms with great potential to be applied to marine ecosystems use a semi-supervised approach including humans in the loop. These approaches are classified into active learning, in which the system remains in control; interactive machine learning, in which there is a closer interaction between users and learning systems; and machine teaching; where human domain experts have control over the learning process (Mosqueira-Rey et al. 2023).

When the sound sources of interest are partially masked, one possible way to extract the signal of interest is to apply source separation algorithms. Even though source separation has received comparatively less attention than automatic detection and classification of sounds, in this context, it is useful to develop source separation algorithms. This has already been applied to soundscape descriptions (Lin and Tsao 2020; Lin et al. 2021) and has also been applied to solve the cocktail party problem (Bermant 2021). Applying these tools to soundscape description in a standardized way could be an added value to the field.

Finally, soundscape analysis can be done through a more holistic approach, which does not depend on the recognition of sound signatures. The most common ways of general soundscape analysis are not always applicable to shallow highly anthropogenic areas. In recent years, new approaches have appeared, such as the categorization of soundscapes in an unsupervised way (Parcerisas et al. 2023), where the different soundscapes are grouped into groups of acoustically similar types, which are then correlated to environmental parameters using explainable artificial intelligence.

Sometimes noise can mask the biophony, but that does not mean that the biophony is not there. Therefore, there is a need to develop more PAM analysis techniques robust to anthropogenic noise, which are sensitive to acoustic signals of not so vocal communities like sandy benthic habitats.

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# Open-Air Acoustic Laboratory in a World Biosphere Reserve: The Case of Lake Saint-Pierre, Canada

33

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**Abstract**

The St. Lawrence River is a large river system in North America, connecting the Great Lakes to the Atlantic Ocean (~1500 km) and is a major commercial waterway in the world. The fluvial portion has a highly diverse mosaic of habitats that support ~100 species of freshwater fish. Fed by numerous tributaries of varying quality, the St. Lawrence includes three large, shallow, slow-flowing fluvial lakes that alternate with narrow, fast-flowing stretches and numerous islands sometimes grouped in archipelagos, all of which are exposed to various anthropogenic pressures. Lake Saint-Pierre (LSP), located downstream, has been recognized as a World Biosphere Reserve (UNESCO, 2000) for its rich biodiversity and extensive wetlands. This UN status does not, however, protect it from the intense pressures associated with agricultural practices, water discharge regulation and, of special interest to us here, acoustic pressures, some of which are of particular concern. First, between 1952 and 1999, the Canadian Department of National Defense fired more than 500,000 projectiles at nearly the entire southern shore of the SLP. It is estimated that 7880 pieces of unexploded ordnance –UXOs – (3%) could remain with an explosive charge of 0.5–7.0 kg. For public safety reasons, a clearance, expected to last 9 years, has begun. Some UXOs will be removed and detonated at a land-based site, while those deemed too hazardous to be moved will be detonated directly in the lake, with the expectation that effective mitigation measures will be taken. These operations involve a vast 16,000 hectare area of shallow, highly productive habitat used for spawning and rearing by approximately 40 species of fish, some of which are threatened. Second, the navigation channel, which is ~300 m wide and ~ 12 m deep, is used year-round by commercial vessels, whose noise contribution to the soundscape has never been characterized in these fresh waters. In addition, during the ice-free season, the LSP and its archipelago are frequented by numerous recreational vessels, whose acoustic effects have not yet been measured. Also, the exposure to winds and waves of this large shallow lake is likely to generate significant ambient noise fluctuations, which would deserve to be quantified. Finally, there is great interest in characterizing biophony in this unique environment and testing some of the effects of underwater sound (including particle motion) and ground vibration on fish and invertebrates, especially during bombshell explosions. It is in this context that a joint academic-department acoustic research program has been initiated, with the aim of preparing a first portrait of the soundscape in this World Biosphere Reserve and possibly proposing recommendations for mitigation measures regarding the most important anthropogenic noise sources.

**Keywords**

Sound · Impact · Fish · Freshwater · UXO · UNESCO Biosphere Reserve · Conservation

## Introduction

Since the 1950s, anthropogenic underwater noise has doubled every decade in the oceans and near shipping lanes (Frisk 2012; McDonald et al. 2006). Underwater soundscapes – defined here as the ensemble of acoustic signals characterizing a particular underwater environment – are therefore changing rapidly. The soundscape of many aquatic ecosystems is currently dominated by anthrophony events (sounds of anthropic origin, e.g., vessel noise, pile driving, or sonar), compared to the preindustrial era when soundscapes were dominated by biophony (sounds of biotic origin, e.g., sciaenid fish or snapping shrimps) and geophony (sounds of abiotic origin, e.g., rain, thunder, ice, etc.) (Duarte et al. 2021). Understanding the extent to which this change is affecting the behavior and life cycle events of aquatic fauna is crucial to maintain or restore the integrity of marine and freshwater ecosystems. It is therefore essential to quantify and mitigate the impacts of rapid transformations of underwater acoustic landscapes caused by human activities on fauna.

The acoustic dimension is an important feature of aquatic habitats. Sound travels faster and farther in aquatic environments than in the air and the aquatic fauna's capacity to hear and communicate through acoustic cues can be advantageous, especially in environments with limited visibility (e.g., under ice and turbid water). Fish have evolved a great variety of inner ear and accessory hearing structures that allow them to perceive sound, as well as a diversity of sound-generating mechanisms (e.g., drumming muscles or stridulatory bony structures). It is estimated that approximately two-thirds of fish species produce sound (Rice et al. 2022) and active sound production has been reported in a wide variety of behavioral contexts including reproduction (courtship and spawning), food source, or territory defense and predator avoidance (Ladich 2015).

Pervasive noise pollution has the potential to affect both sonic fish and species that hear sound and use it to navigate, feed, or detect potential threats. Impacts of shipping noise on fish include behavioral changes such as avoidance (Weilgart 2018), altered antipredation behaviors (Simpson et al. 2016), decreased parental care of newborns (Nedelec et al. 2022), as well as physiological impacts such as increased stress levels (Mills et al. 2020). Effects from underwater explosions are even more severe and can be expected to range from temporary or permanent anatomical damages to death (e.g., Dahl et al. 2020; Smith et al. 2022).

Although noise pollution is globally raising important concerns, its effects are little documented in freshwater environments and the underlying mechanisms are poorly understood. In Quebec (Canada) in particular, little scientific knowledge and data exists on the effects of shipping noise or underwater explosions on freshwater fish species and their habitat. This includes the St. Lawrence River fish communities. The St. Lawrence River is a large river system in North America, connecting the Great Lakes to the Atlantic Ocean (~1500 km) and is a major commercial waterway in the world. The fluvial portion has a highly diverse mosaic of habitats that support ~100 species of freshwater fish. Fed by numerous tributaries of varying quality, the St. Lawrence includes three large, shallow, slow-flowing fluvial lakes that alternate

with narrow, fast-flowing stretches and numerous islands sometimes grouped in archipelagos, all of which are exposed to various anthropogenic pressures.

Lake Saint-Pierre (LSP), located downstream, has been recognized as a World Biosphere Reserve (UNESCO 2000) for its rich biodiversity and extensive wetlands (Quebec; Fig. 1). The fish community of LSP, composed of approximately 90 species identified by the Ministère de l'Environnement, de la lutte contre les changements climatiques, de la Faune et des Parcs (MELCCFP, Québec, Canada), including about 20 species of precarious status, is potentially exposed to underwater noise from shipping. There are also native mussel beds at both ends of the LSP that have a precarious status. Although there is scarce information available on invertebrates hearing, recent studies point out that they are sensitive to particle motion and therefore can detect and react to sound in the water column and related physical vibrations near or within the substrate (see Solé et al. 2023 for a review). Indeed, there is evidence for a range of behavioral responses, and even higher mortality rates, in mussels exposed to air gun sound and substrate vibration (e.g., Roberts et al. 2015;

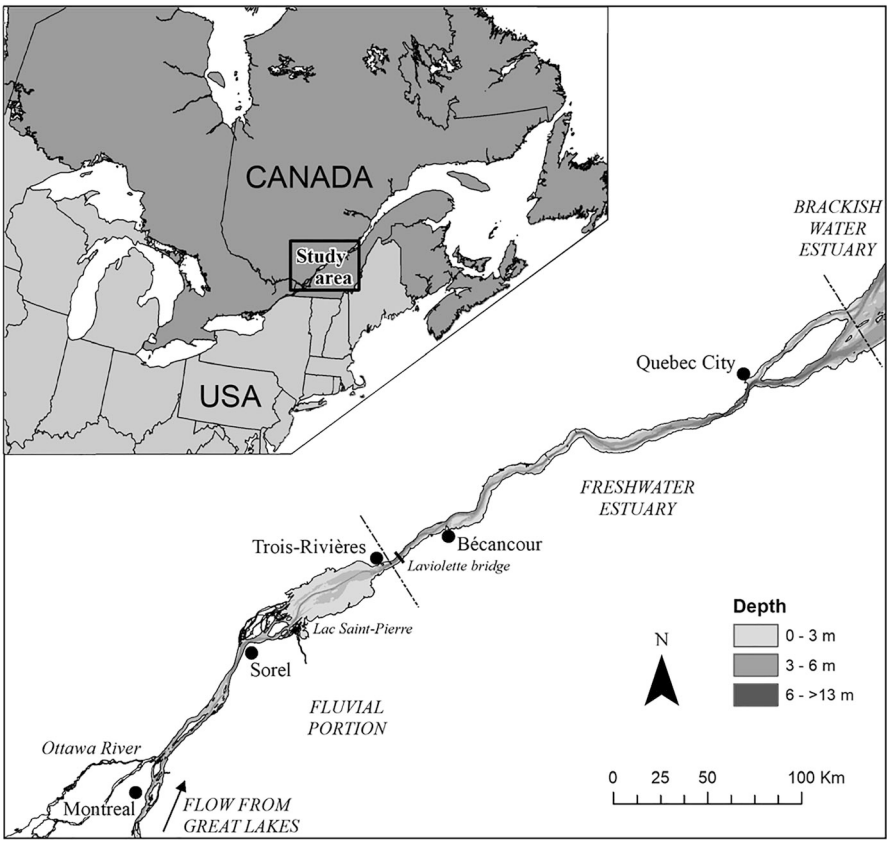



**Fig. 1** Great Lakes–St. Lawrence River basin and detail on the course of the St. Lawrence River

Day et al. 2017) and thus could potentially be threatened by the high power of underwater explosions.

The physical properties of LSP are unique, with a particular configuration that differs from an open ocean environment and makes it a true open-air laboratory for acoustic research. It is a freshwater fluvial lake, consisting of a vast and shallow floodplain (average depth is 3 m), crossed by a wide (~300 m) and deep (at least 11.3 m) navigation channel, dug in the clay and rock over 250 km between Montreal and Quebec City. In addition, there are horizontal gradients of conductivity, temperature, and turbidity reflecting the different water masses flowing in LSP from various tributaries.

The contrasting LSP bathymetric and physicochemical configuration hosts a rich aquatic fauna, which is exposed to noise sources, whose propagation and effects need to be studied. First, the seaway that crosses this fluvial lake supports approximately 4800 transits of merchant ships/year (Bernier, 2022). Secondly, this area is also very popular with hundreds of recreational fishermen and boaters attracted by the scenic beauty of the landscapes, whose frequent trips alter the underwater soundscape. Finally, the Canadian Department of National Defense tested numerous munitions in the highly productive aquatic habitats on the south shore of the LSP between 1952 and 1999, many of which remained unexploded. For public safety reasons, a cleanup needs to be undertaken, involving the removal of shells but also the in situ explosion of unexploded ordnances (UXOs), posing a threat to the physical integrity of animal and plant species, and their habitats.

These multiple acoustic pressures are the subject of recent research efforts led by academia (e.g., Univ. du Québec en Outaouais, Univ. du Québec à Trois-Rivières and Univ. Laval) and the MELCCFP (e.g., Barbeau 2023). The initial work highlights a pressing need to (i) establish acoustic baselines of the natural soundscape characteristics and of fish species bioacoustics (hearing capacity and sound production) and (ii) to assess the impact of vessel traffic and underwater explosions on resident and migratory species for which the St Lawrence River is a critical habitat.

The poster presented at the Aquatic Noise Congress (Berlin, July 2022) was intended to introduce the scientific community to a research program that had just started in 2022 in the fresh waters of the St. Lawrence River. This paper provides more details on the physics of this unique environment, which is described as an open-air laboratory where the following research objectives are pursued:

(a) Prepare an initial acoustic portrait of LSP to assess the exposure and vulnerability of fish communities to noise from commercial and recreational boating in this UNESCO Biosphere Reserve.
(b) Characterize the acoustic capacity of LSP's fish species.
(c) Make accurate measurements when detonating UXOs implying the deployment of high-sampling rate (>1 MHz) and high-pressure-resistant (~250 dB) hydrophones in close collaboration with the Canadian National Defense.
(d) Develop an acoustic propagation model to simulate the acoustic landscape in wildlife habitats of interest and different noise sources. To do so, several layers of information were collected, including topography, substrate, density of

submerged aquatic vegetation, and gradients created by water masses (conductivity, temperature, and turbidity).

(e) Evaluate the impacts of anthropogenic sounds on target species based on their level of exposure, auditory sensitivity, and behavioral changes.

(f) Recommend potential mitigation measures to reduce the acoustic footprint of the most important sources of anthropogenic noise that are identified as critical to sensitive aquatic wildlife.

## Geographical Context and Heterogeneity of the St. Lawrence River

The Great Lakes-St. Lawrence River watershed is one of the largest freshwater ecosystems in the world (Fig. 1, top panel). Located in the eastern region of North America, it contains about 84% of all North American freshwater (18% of the world's freshwater) and is home to about 30 million people. It is a natural migration route of approximately 1500 km that connects the Great Lakes to the Atlantic Ocean. It is also one of the most important commercial waterways in the world, contributing significantly to the economy of the entire St. Lawrence-Great Lakes region.

With a naturally contrasting topography and a complex hydrological network, the St. Lawrence River presents a mosaic of highly diverse habitats that are home to some 100 species of freshwater fish (Fig. 1). It is generally divided into three main sections: (i) a fluvial section extending from the outlet of the Great Lakes to Trois-Rivières (~350 km), in which Lake Saint-Pierre is located; (ii) followed by an estuarine section composed of a long freshwater estuary (~160 km) progressively influenced by the tide, which reaches a tidal range of 6 m near Quebec City, and then a brackish water section (~150 km), where salinity increases progressively from 0.2–25 ppt; and (iii) further out, a long marine section that joins the Gulf of St. Lawrence and the Atlantic Ocean (salinity >25 ppt; see Fig. 1).

The fluvial section presents a variety of aquatic habitats with three slow-flowing fluvial lakes (lakes Saint-François, Saint-Louis, and Saint-Pierre; Fig. 1), separated by narrow fast-flowing corridors and dotted with numerous islands and a few archipelagos. The heterogeneity of the habitats is accentuated by the contribution of numerous tributaries. These produce highly contrasting water masses that can be identified along the river, depending on their origin and their changing physico-chemical properties (temperature, conductivity, and turbidity). Water masses flow side by side with little lateral mixing for hundreds of kilometers (e.g., the clear green water of Lake Ontario, the brown from Ottawa River, and more turbid water of agricultural watersheds; Fig. 2), creating strong horizontal gradients that largely influence aquatic habitats (e.g., abundance of submerged vegetation and fish diversity) and presumably also sound propagation (suspended particles, temperature, and conductivity).

The entire course of the St. Lawrence River is of great interest for acoustic studies, but the focus here is on LSP, the largest and most productive of the fluvial lakes of the St. Lawrence River. LSP has been recognized as a World Biosphere

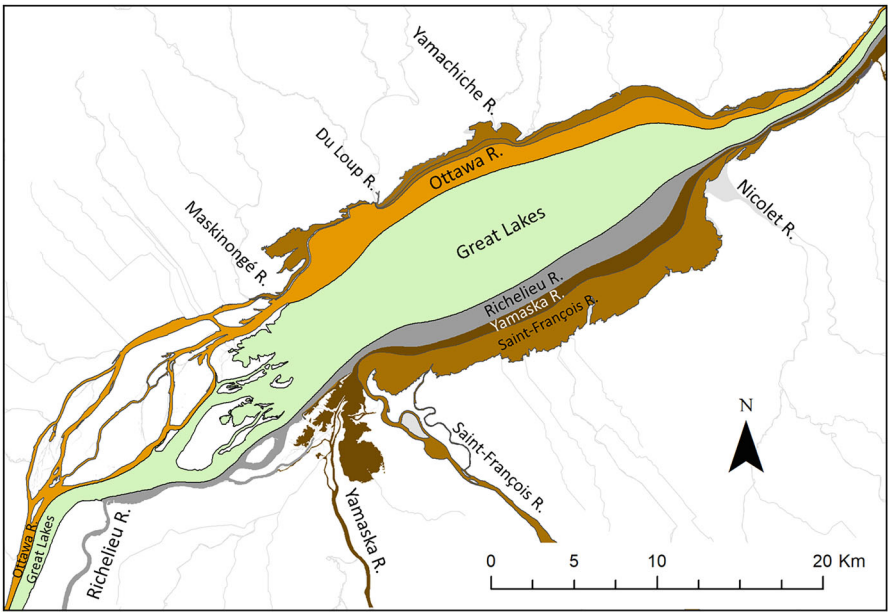



**Fig. 2** The study area is in Lake Saint-Pierre, the largest fluvial lake in the St. Lawrence River, where water quality is strongly influenced by the flow of several tributaries. These tributaries generate contrasting water masses, flowing side by side for tens of kilometers and creating strong horizontal gradients that largely influence aquatic habitats (e.g., abundance of submerged vegetation and fish diversity) and presumably also sound propagation (i.e., suspended particles, temperature, and conductivity). The current water mass simulation corresponds to an average hydrological event during the summer. (Adapted from Martin et al. 2016)

Reserve (UNESCO 2000) for its rich biodiversity and extensive wetlands. It is home to about 100 species of fish, nearly 300 species of resident and migratory birds and about 30 rare plants.

## Main Anthropogenic Pressures

Although the St. Lawrence River is vast and emits an impression of great strength, this does not make it sheltered to pressures of various kinds from human activities in its watershed. On a smaller scale, the UN status of LSP does not protect it from local anthropic pressures.

### Agriculture

Nearly 5000 ha of intensive annual crops are planted directly in the floodplain, destroying a significant area of critical habitats for many species, and diminishing

the production and diversity potential of the lake (Foubert et al. 2020). On a larger scale, land use in the LSP watershed shows intensive agriculture, resulting in high nutrient and contaminant loads and high turbidity.

## Hydrological Alterations

Dams and flow regulation, which have been progressively implemented in the Ottawa River and St. Lawrence River, for more than a century, have reduced the height and duration of floods, altering the natural hydrological regime to which aquatic life had been adapted for centuries (Foubert et al. 2020).

## Fishery Decline

LSP was home to a thriving yellow perch fishery that had to be closed in 2012 and extended until 2027, due to low recruitment and advanced degradation of aquatic habitats (Magnan et al. 2022).

## Anthropogenic Noise

There are acoustic pressures that have never been documented:

- **Commercial ships.** Commercial shipping is concentrated in the shipping channel. Approximately 4800 merchant vessels use the seaway through the LSP throughout the year. In the Montreal-Sorel section of the St. Lawrence River, a voluntary measure to reduce the speed of commercial ships was implemented in the fall of 2000 by the shipping industry to reduce erosion problems. Over a length of nearly 25 km in the Sorel-Varennes sector, this measure encouraged ship pilots not to exceed a speed of 10 knots upstream and 14 knots downstream. This measure is also a good study avenue to reduce ship underwater noise.
- **Harbors.** In the freshwaters of the St. Lawrence River, there are five main harbors (from upstream to downstream): Montreal, Sorel, Trois-Rivières, Bécancour, and Quebec City (Fig. 1). These harbors are central to Canada's internal and external trade. They have different functions and jurisdictions that will not be discussed here. The harbors of Sorel and Trois-Rivières are located at both ends of LSP and represent sources of anthropogenic noise to be assessed.
- **Bridge.** The Laviolette Bridge, located downstream from LSP, is the only bridge connecting the two shores between Montreal and Quebec City and linking the city of Trois-Rivières to Bécancour. It is an arch bridge with the longest steel structure in Quebec (1383 m and maximum deck height of 54 m), supported by large concrete foundations, many of which are in contact with the waters of the St. Lawrence River. With a traffic flow of approximately 42,000 vehicles per day, it represents a source of noise that deserves to be estimated.

- **Recreational boats.** During the summer months, LSP is also heavily visited by hundreds of fishermen and boaters attracted by the landscape beauty. Boaters are heavily concentrated along the slope of the navigation channel throughout LSP, as well as in the many smaller channels throughout the Sorel archipelago. The most intensively frequented locations will be targeted in this study.
- **UXOs removal.** Explosions of UXOs are expected in the next 9 years for public safety reasons. Indeed, between 1952 and 1999, the Canadian Department of National Defense fired over 500,000 rounds into the CYR 606 area, covering almost the entire southern shore of the LSP (16,000 ha; BAPE 2020). It is estimated that nearly 300,000 unexploded projectiles could end up on the bottom of the lake, of which 7880 (~3%) would contain explosive material (0.5–7.0 kg; maximum caliber 155 mm) and must be removed to reduce the risk to public safety. Some will be removed and detonated at a land-based site, those deemed too dangerous to be safely moved out of water will be detonated directly into the lake, hopefully with effective mitigation measures. This large CYR 606 area consists of shallow, highly productive habitats used by ~40 species of fish for spawning and juvenile growth, some of which are threatened. Native mussel beds have also been observed at both ends of LSP.

## Habitat Characteristics in Relation to Acoustics

The St. Lawrence River and more specifically LSP are real open-air laboratories because they present a strong heterogeneity in their aquatic habitats, which make them unique environments for acoustic studies. In addition to spatial heterogeneity, the physical characteristics of LSP also show temporal dynamics with (1) significant seasonal changes in river flow, space occupied by water masses from tributaries, water temperature, and submerged vegetation cover showing cycles of growth and senescence, and (2) also longer-term changes related to climate. All these changes influence the propagation of sound underwater and the exposure of aquatic fauna and their habitats.

### Bathymetry

The LSP is a widening of the St. Lawrence River, 30 km long by 12 km wide, with an average depth of about 3 m (Fig. 1). It consists of a large floodplain that extends toward the center of the lake with a gentle side slope. The center of the lake is occupied by a deep navigation channel 300 m wide and guaranteeing a minimum depth of 11.3 m for commercial ships, which was gradually dug out from the middle of the nineteenth century. The upstream portion of the lake is an ancient delta that formed an archipelago containing 103 islands and numerous small channels rich in biodiversity and used by boaters. The ends narrow rapidly, are deeper, and show fast currents $>1$ $ms^{-1}$.

## Hydrology

The flow of the St. Lawrence River comes mainly from the Great Lakes watershed, but it is largely influenced in the spring by the inflow from tributaries, including the Ottawa River, which amplifies the variability of the hydrological regime. In LSP, the average annual variation in water level is 2 m (Morin and Bouchard 2000), with a spring flood and a low water level during the summer period. In addition to seasonal fluctuations, climatic cycles with a period of 20–35 years also influence the hydrological regime of the St. Lawrence system (precipitation and evaporation), modifying the availability of water for natural and anthropogenic uses and modifying the acoustic configuration of LSP (depth and surface). The digging of the navigation channel was accompanied by the construction of important works along the St. Lawrence River which aimed at raising the water level to facilitate navigation during low water periods and at regulating the flow of the river to avoid the flooding of large urban centers in spring.

## Water Masses and Horizontal Stratification

The many tributaries are producing highly contrasting water masses that can be identified along the river, depending on their origin and their changing physicochemical properties (temperature, conductivity, and turbidity; Fig. 2). Water masses flow side by side with little lateral mixing for hundreds of kilometers (e.g., the clear green water of Lake Ontario, the brown from Ottawa River, and more turbid water from agricultural watersheds), creating strong horizontal gradients that largely influence aquatic habitats (e.g., abundance of submerged vegetation and fish diversity) and presumably also sound propagation (suspended particles, temperature, and conductivity).

## Substrate

Fine substrates such as clay, silt, and sand are dominant at LSP. They are of agricultural origin and transported primarily by tributaries because the LSP central water mass, which originates from the Great Lakes, is very clear. Coarser sediments such as gravels and pebbles are found near the fast-flowing channel, but also compact clay, which dates back to the Champlain Sea 10,000 years ago, and which was dredged during the digging of the channel. Other portions of the navigation channel between Montreal and Quebec City were dug directly in the rock (Fig. 3).

## Submerged Aquatic Vegetation

Until the mid-2000s, LSP bottom was covered by very dense and extensive beds of submerged aquatic vegetation occupying much of the water column. With increasing nutrient, contaminant, and turbidity loads, this submerged aquatic vegetation has greatly declined, either disappearing or being replaced by benthic algae (Giacomazzo

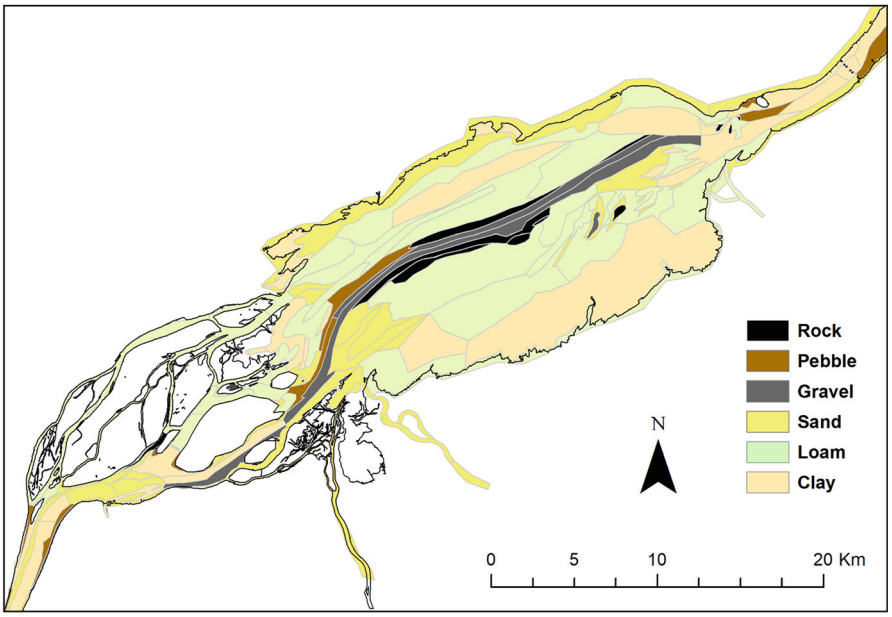



**Fig. 3** Substrate classification based on data from the literature and field campaigns, revealing that Lac Saint-Pierre is dominated by fine substrates such as clay, silt, and sand. The navigation channel is occupied by coarse substrates or has been excavated in clay. (Unpublished data from J. Morin, Environment and Climate Change Canada)

et al. 2023). It is important to consider the remaining vegetation cover as a factor influencing acoustic propagation (Fig. 4) and a potential acoustic refuge for some fish species (Wilson et al. 2013).

## Water Properties and Sound Propagation

Propagation loss in the LSP is defined by (i) the geometric dilution as the sound travels away from a source and (ii) the absorption of the sound waveguide by the surrounding medium. The geometric dilution is range dependent as it depends on the varying geophysical properties of the medium as the sound travels between two points (Collins 1995; Porter 2011). The geometric dilution is linked to the bathymetric profile, the height of the water column, the properties of the substrate, and gradients in speed of sound in the water. While the bathymetry remains the same under natural conditions (i.e., without human intervention), the height of the water column is determined by the hydrological regime (Morin and Bouchard 2000). The nature of the substrate can be expressed in terms of its geoacoustic properties: density and speed of sound in the substrate, shear, and compressional absorption coefficients (Jensen et al. 2011). The computation of the speed of sound in the water is a function of the water's conductivity ($\mho$), temperature ($t$), and depth ($z$) (Leroy et al. 2008). Table 1 presents preliminary results of field expeditions to the SLP

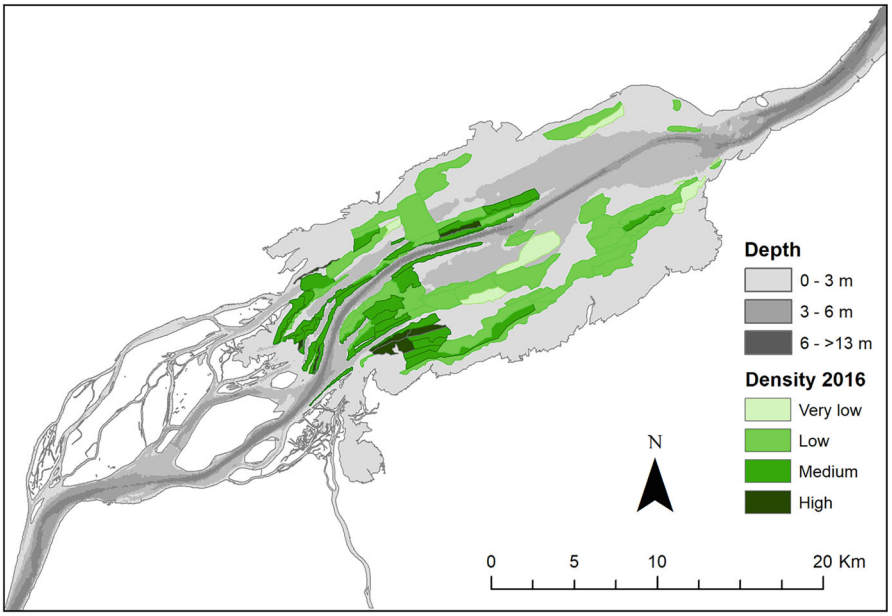



**Fig. 4** Density of submerged aquatic vegetation in 2016. (Adapted from J. Morin, Environment and Climate Change Canada, unpublished data)

**Table 1** Average chemical-physical properties of water ($\pm 1\sigma$)

| Season | $\mho$ | $t$ | $z$ | $S(\mho)$ | $c_w(t,S(\mho),z)$ |
|---|---|---|---|---|---|
| | (mS cm$^{-1}$) | (°C) | (m) | (ppt) | (m s$^{-1}$) |
| Spring | 0.203 ± 0.058 | 16.8 ± 0.9 | 1.705 ± 1.681 | 0.083 ± 0.025 | 1472.2 ± 2.9 |
| Summer | 0.247 ± 0.046 | 23.2 ± 1.2 | 1.226 ± 1.270 | 0.102 ± 0.020 | 1491.8 ± 3.4 |
| Fall | 0.241 ± 0.049 | 3.4 ± 1.1 | 1 | 0.099 ± 0.022 | 1418.9 ± 5.2 |

conducted to characterize the chemicophysical properties of the water from season to season. Spatial coverage extends from the north shore to the south shore.

Absorption by the water medium is caused (i) by boric acids and magnesium sulfate diluted in the water and whose concentrations are monotonically dependent on the water salinity S(℧) (François and Garrison 1982a, b) and (ii) by nondiluted suspended material in the water column due to thermo-viscous and diffusive interactions with the waveguide (Richards 1998). Suspended particles are characterized in concentration ($C_V$) using nephelometric turbidity measurements ($\zeta$) as shown in Table 2.

Absorption by the diluted content is highly dependent on the signal frequency and becomes significant only above 10 kHz. Absorption caused by nondiluted particles depends on their nature (more specifically, their volumetric density). In situ sedimentology and granulometry measurements are expected to take place in 2023 to identify and characterize the nondiluted suspended content in LSP.

**Table 2** Average turbidities of suspended sediments (±1σ)

| Season | $\zeta$ | $C_V(\zeta)$ |
|---|---|---|
| | (NTU) | (kg m$^{-3}$) |
| Spring | 4.835 ± 2.588 | 0.0016 ± 0.0009 |
| Summer | 8.440 ± 3.203 | 0.0028 ± 0.0011 |
| Fall | 5.126 ± 1.332 | 0.0017 ± 0.0004 |

## Fish Species, Habitat Type, and Noise Exposure

Of the 118 freshwater fish species living in Quebec, 79 have been reported in the LSP area by the MELCCFP monitoring campaigns using seines, gillnets, and trawls (1995–2019), of which 22 are sport species, 22 are commercial, and 7 have a precarious status (threatened or vulnerable). Trawl sampling has demonstrated that the navigation channel is not a desert and revealed several significantly different fish assemblages, whose composition is influenced by exposure to ship activities and habitat type (Mingelbier et al. 2019). Lake sturgeon is a dominant species in deep habitats, including the navigation channel, where shorthead redhorse, silver redhorse, channel catfish, and other species with known enhanced hearing sensitivity also occur. Among the deep environments, the pools are the richest in diversity and biomass, compared to the channel and its slope. Habitats exposed to ship activities had the lowest indices of diversity and abundance compared to unexposed environments. Shallow-nearshore habitats supported the most diverse fish communities, but not the greatest biomass, found in natural deep pools. Fish communities differed between the shipping channel and the shallow nearshore, suggesting that fish species are differentially exposed to various sources of noise in LSP, which is an avenue for further research. Depending on the species, it will be important to test whether fish with precarious status are exposed to noise and to explore causal relationships between low-diversity fish communities and noise intensity. Accurate description of noise exposure should help identify areas of higher intensity and acoustic refuges for fish, characterize noise in deeper pools, and potentially link noise to aquatic habitat connectivity, for example, for migrants such as lake sturgeon that primarily use the shipping channel to migrate through the river corridor. In addition, there is a unique opportunity to study the relevance and effects of noise propagated through the substrate, and its interface with the water column, on fish but also on invertebrates (e.g., indigenous mussels). Indeed, it is suggested in the literature that the possibility for noise propagating through the substrate to resurface further away in the water column at unexpected locations (e.g., Dahl et al. 2020). The explosion of UXOs will provide an excellent opportunity to explore such complex behavior and its ecological implications.

## Agenda for Future Research and Conservation

LSP is a UNESCO World Biosphere Reserve that is exposed to multiple anthropogenic pressures, some of which are acoustic pressures of ecological concern that raise several questions. The freshwaters of LSP present a diverse mosaic of habitats,

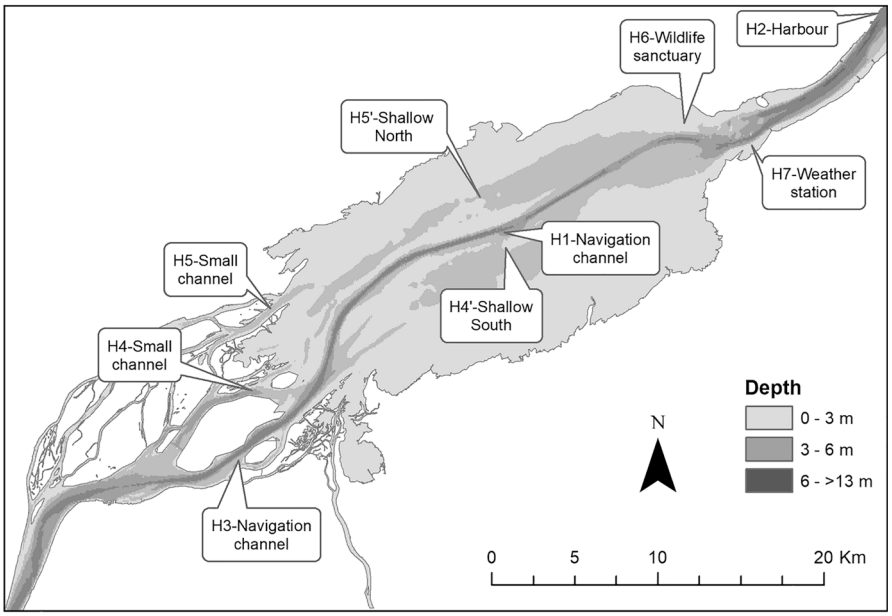



**Fig. 5** Acoustic recording stations during the 2022 season at Lake St. Pierre, showing the variety of environments and noise sources sampled: a harbor (H2), the deep navigation channel (H1 and H3), secondary channels frequented by recreational boats (H4 and H5), a wildlife refuge (H6), shallow environments (H4' and H5'), and near a weather station (H7)

where the soundscape has never been characterized and which, with its unique physical properties, represents a real open-air laboratory. The ongoing academic-departmental collaboration is an excellent opportunity to improve knowledge in the following research avenues (Fig. 5).

## Acoustic Portrait

Preparing an acoustic portrait in the freshwater of the LSP involves quantifying the main sources of anthropogenic noise, which have never been characterized in this area, from different navigation activities (merchant ships, recreational boats, and harbors) and paying particular attention to UXOs explosions. In this large shallow lake exposed to wind, waves, and ice, it would also be relevant to quantify geophony. Regarding biophony, an ongoing literature review on fish acoustic capacity (hearing sensitivity and sound-producing mechanisms) and an evaluation of available information on their generated sound patterns will allow us to assess species-specific vulnerability to anthrophony for the 118 freshwater fishes in Quebec. The plan is to (i) characterize the acoustic activities of a few key species (lake sturgeon and Atlantic sturgeon), by sampling their acoustic signatures in different natural situations (e.g., spawning grounds) and in captivity (e.g., fishermen's nets, aquariums, or

rearing) and (ii) develop a method using artificial intelligence applied to hydrophone recordings to automate the detection of acoustic activities of sturgeon in the St. Lawrence River. Finally, changes in the behavior of several focal fish species will be characterized in relation to changes in the soundscape by using fish positioning data from acoustic telemetry.

## Modeling

In a physical context with so much spatial heterogeneity and temporal changes, it is essential to develop an acoustic propagation model adapted to the situation of LSP to simulate the propagation of the main noise sources across different environmental conditions. This model will consider the physicochemistry, the submerged vegetation, and the substrate, and will be validated with field measurements. It will be useful to compare the exposure of deep and shallow habitats to very different sources of anthropogenic noise. On the one hand, the deep habitats of LSP are exposed to noise from commercial shipping and it would be relevant to verify if this noise propagates into the shallow habitats. Conversely, shallow habitats are exposed to UXO explosions, and it is questionable whether this intense noise propagates into the deeper environments further away from the explosions and whether the more distant mussel beds are exposed to such noise with sound propagation through the substrate.

## Conservation Measures

Based on these results, it will be possible to explore several avenues for conservation measures, including the following.

- **Navigation versus slowdown.** We intend to verify if the voluntary slowing down measure, initially motivated among commercial ships to limit bank erosion, could be applied to reduce the noise of these ships in the most ecologically sensitive sectors. Additional measures could be studied for recreational boats, particularly in the LSP archipelago, which is very busy in summer.
- **Explosions**. UXO explosions are an opportunity to improve knowledge in shallow freshwater and to present a critical analysis of the methods used by the Department of National Defense to estimate the lethal radius for fish species. This will also allow the coproduce mitigation plans and measures to abate the effects on aquatic fauna (e.g., bubble curtains). It will at the same time provide an opportunity to experiment with some of the effects of particle movement on fish and invertebrates, establish (long-needed) conservation thresholds regarding this parameter, and provide potential mitigation recommendations and guidance in this context.
- **Risk analysis.** Acoustic risk mapping for aquatic wildlife in Lake Saint-Pierre will be produced by combining the acoustic sensitivity of aquatic wildlife with

their exposure in various habitats. This will provide an opportunity to georeference sound levels, acoustically polluted environments, and to identify potential acoustic refuges. Recommendations for mitigation measures may be proposed regarding the most important anthropogenic noise sources.

The successful completion of this research and conservation agenda will rely not only on strong interdisciplinary scientific collaborations but also on the active participation of all stakeholders from both the public and private sectors.

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# Soundscape Assessment of a Deepwater (12 kHz) Multibeam Mapping Survey

34

Hilary Kates Varghese, Xavier Lurton, Michael E. Smith, Jennifer L. Miksis-Olds, and Larry Mayer

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**Abstract**

Soundscape studies can provide insight into how, and the extent to which, an acoustic environment has changed due to anthropogenic noise-generating

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activity. Here a soundscape study was conducted to understand the changing sound levels on the Southern California Antisubmarine Warfare Range during a deepwater ocean mapping survey using a 12 kHz Kongsberg EM 122 multibeam echo sounder. The exercise confirmed the expected temporal and spatial dependence of the mapping survey activity contribution to the soundscape. The change in sound levels was closely tied to a narrow angular sector around the mobile survey vessel, which meant the temporal impact at any given location was also confined to a brief time period that the vessel was in the corresponding area. The EM 122 signal was detectable above baseline sound levels in the acoustic records only when the survey vessel was within 17 km of the hydrophone receiver and was clearly and consistently associated with changes in the 12.5 kHz decidecade spectral density levels. The results provide important context for simultaneous works assessing beaked whale foraging behavior during the mapping survey, where no adverse change in foraging behavior was found.

**Keywords**

Multibeam echo sounder · Marine soundscape · Deepwater ocean mapping survey · Acoustic impact · Southern California Antisubmarine Warfare Range

## Introduction

In the context of anthropogenic noise and marine life, soundscape studies can provide insight into how, and the extent to which, an acoustic environment has changed due to anthropogenic noise-generating activity. When possible, baseline assessments are made prior to the activity of interest to provide a foundation upon which to assess the extent of change beyond expected natural variation. Soundscape studies are imperative for providing context to our understanding of the interplay between anthropogenic noise-generating activity, natural variability within the ecosystem, and marine life behavior.

The goal of the work in this chapter was to understand how the acoustic activity of a 12 kHz deepwater mapping survey impacted the overall marine soundscape of the Southern California Antisubmarine Warfare Range (SOAR). The motivation for this was two-fold. While modeling efforts (Lurton and DeRuiter 2011; Lurton 2016) show that the Kongsberg EM 122 multibeam echo sounder – the primary system used in the mapping survey discussed herein – should be below the currently accepted injury thresholds for marine mammals, a thorough assessment of its impact on the marine acoustic environment during typical operations had not been undertaken prior to the work in this chapter. Thus, the first motivation was to provide empirical observations of the spatial and temporal frequency-dependent impact on the marine soundscape of the acoustic sources (vessel sound and active acoustic sources) associated with a typical deepwater multibeam mapping survey. The second motivation for understanding the impact of the mapping survey on the acoustic environment was to provide context for parallel marine mammal behavior impact studies conducted during the same mapping survey (Kates Varghese et al. 2020, 2021).

Multibeam echo sounders (MBES) were developed to provide detailed maps of the seafloor. The original MBES utilized the natural radiation pattern of a long thin transmitting array to generate a narrow, angularly constrained swath of sound that propagates to the seafloor – wide across-ship, but narrow along-ship. Present-day systems have undergone significant technological advancement in electronic and beamforming capabilities, resulting in multi-swath and multi-sector capabilities which significantly increase the insonification capacities but also greatly complicate the transmission pattern. The active component of a MBES ping cycle is comprised of a train of short independent pulses with non-overlapping, narrow bandwidths. Each pulse in the ping corresponds to a specific sector in a given swath. State-of-the-art multibeam systems can be dynamically operated under various permutations of several operating modes: (1) single/dual swath, (2) auto-ping mode/manually selected mode, (3) depth-specified modes, (4) with/without motion compensation, (5) various angular swath widths, (6) gated continuous-wave (CW)/frequency-modulated (FM) pulses, among other options which may further complicate the radiation pattern (note: features 2 and 5 (The setting of the swath width on EM122 (when changing modes) may correspond to various angular coverage (narrowing in very-extra-deep modes) hence a change in the radiation characteristics.) will not impact the sound radiation characteristics) (Kates Varghese et al. 2019). Most of these parameters are dictated by the survey requirements and the operating environment. Dual swath coverage enables the user to obtain a higher density of pings in a fixed area, which can be important for the quality of data required for a survey and/or for the survey speed. Similarly, the use of various swath widths and motion compensation capabilities allow the user to adapt to rough seas and various survey speeds while ensuring quality data collection. Essentially the options available with state-of-the-art multibeam allow the user to adapt to dynamic conditions in a variety of ways. Therefore, the exact signal transmitted may vary from one survey to the next (Kates Varghese et al. 2019) and the ensonified area will be a function of the water depth, achievable swath width, firing mode, and applied motion compensation parameters.

The Kongsberg EM 122 deepwater MBES operates using either single or dual swath mode with 4–8 sectors per swath. The EM 122 can be manually or automatically set to different depth-dependent operational modes implying, as depth increases, higher source levels, longer pulse lengths, more sectors in a narrower angular coverage, and frequency-modulated pulses at deeper depths. Pulse lengths may vary from 3 ms (CW) in the shallowest depths (i.e., <450 m water) to 100 ms (FM) in very deepwater (i.e., up to 11,000 m). A typical deepwater survey (i.e., deep mode is defined as 1000–2600 m depth) operates in dual swath mode with eight sectors, each with pulse durations of 8–15 ms and inter-ping intervals of about 5–10 s.

## Methods

A deepwater ocean mapping survey was conducted over an approximately three-day time period at the SOAR. In order to characterize the marine soundscape modifications caused by the survey, several sound level metrics were computed and compared across time and space. The SOAR consists of a hydrophone array which spans

an approximately 1800 $km^2$ area and contains a subset of 89 bottom-mounted hydrophones. Acoustic data from all 89 hydrophones were collected from January 4, 2017, 20:13 until January 7, 2017, 06:55 UTC, which spanned a time period before, during, and after a deepwater MBES mapping survey of the SOAR. A subset of the acoustic data was used for characterizing the marine soundscape before and during the survey.

The bottom-mounted (~5 m above the seafloor) omnidirectional hydrophones are at depths from 850 to 1750 meters, have a roughly flat sensitivity response between 50 Hz and 48 kHz, and are sampled at a rate of 96 kHz (Smith 2019). Acoustic data from nine hydrophones were used in the work in this chapter. The raw acoustic data were extracted at a duty cycle of 5 min on/10 min off to reduce time and memory during processing. The acoustic data were then converted to various broadband and band-specific sound level metrics, including:

- Peak sound pressure levels (*SPLpk*), identifying the maximum pressure amplitude in a given time window and measured in units of dB re 1 μPa
- Sound exposure levels (*SEL*), measuring all of the energy within a specific (100 ms) time window, reported in units of dB re 1 $\mu Pa^2$ s
- Decidecade band levels (*BL*) with center frequencies between 50 Hz and 40 kHz, representing the mean square sound pressure spectral density level in each decidecade, or one-tenth of a decade band (ISO 18405), reported in units of dB re 1 $\mu Pa^2$/Hz

The *BL*s were chosen to directly align with the Atlantic Deepwater Ecosystem Observatory Network (ADEON) project (Ainslie et al. 2017). But, as no formal standard exists yet for computing and reporting broadband quantities, a decision was made to maximize the spectral observations available with this data set. Consequently, broadband was defined here as the 30 bands inclusive of the 50 Hz through 40 kHz center frequency decidecade bands (i.e., $n = -13$ to 16, Ainslie et al. 2017). A 100-ms time window was used to produce the sound level metric time series, selected because of its relevance to the duration of the MBES signals (typical magnitude of 10 ms). Sound levels weighted with the NMFS mid-frequency cetacean weighting function (2018) were also produced, but are not discussed further here. The interested reader is referred to Kates Varghese 2021 for details on that portion of the work.

Sound level metric time series were generated and partitioned into four analysis periods based on the anthropogenic activities that occurred during the overall time period examined. These analysis periods were defined as:

- No Activity (NA) – a time immediately preceding the mapping survey when no activity related to the mapping survey was occurring on the SOAR array;
- Vessel Only (VO) – a period when only the survey vessel was known to be on the array and all active acoustic systems related to mapping were off;
- Vessel and MBES (VM) – a period which included the presence of the survey vessel and the active EM 122 MBES on the SOAR array;

- Mixed Acoustics (MA) – a period that included the survey vessel on the array with transmitting Kongsberg EM 122, Kongsberg EM 712 MBES (40 kHz), Simrad EK 80 wide-band echo sounder (18 kHz), and a Knudsen sub-bottom profiler (3.5 kHz) at various times during this period.

Note: Due to the sporadic and intermittent use of the various active acoustic systems during the MA period, the ability to characterize the impact of the other active acoustic systems (excluding the EM 122) on the marine acoustic environment was limited and therefore is not further discussed here. The interested reader is referred to Kates Varghese (2021) for the results from this analysis period.

Aside from the known active acoustic sources used during the mapping survey, there were most certainly other acoustic sounds incidentally recorded on the array hydrophones. These may have included sounds inherent with the safe operation of the research vessel, radiated noise from other vessels and anthropogenic activity near the array, and/or biological, mechanical, or geophysical activity. There was no way of excluding other potential sound sources from the soundscape assessment of the survey activity. The presence of other sounds – anthropogenic or natural – is a reasonable expectation for the SOAR soundscape. It is worth noting that the survey took place during the first week of January when activity at the SOAR was very limited, if at all.

A purposeful selection of the *BL*s was chosen for more detailed analysis, representing specific acoustic sources expected to contribute to the SOAR soundscape. These were the *BL*s with center frequencies of 50 Hz, 500 Hz, 3.2 kHz, 12.5 kHz, and 40 kHz. The 12.5 kHz band (11.2–14.1 kHz) was selected because it encompasses the majority of the frequency content of the EM 122 MBES signal (11–13.25 kHz). This band may also be occupied by biological sounds, such as those from vocalizing marine mammals. The 40 kHz band (35.5–44.7 kHz) was chosen as it contains the peak frequency of beaked whale foraging clicks (Baumann-Pickering et al. 2013). (Note: It also contains the signal of the EM 712 echo sounder used during the MA period of the mapping survey (40 kHz).) The 50 Hz (44.7–56.2 Hz), 500 Hz (447–562 Hz), and 3.2 kHz (2.8–3.5 kHz) bands were chosen as they each represent a distinct regime of the low-frequency signals that often dominate a marine soundscape. The 50 Hz band represents vessel engine sound, the sound from propellers, as well as the low-frequency harmonics of these sources (Hildebrand 2009). The 500 Hz band includes other ship-radiated sound, as well as other natural and ambient sources such as marine mammals that communicate at low frequencies (Wenz 1962). The 3.2 kHz band was selected because it can contain higher frequency radiated vessel noise. (Note: it was also selected because it would contain significant energy from the sub-bottom profiler, nominal frequency of 3.5 kHz, used during the MA period. Reader is again referred to Kates Varghese 2021 for details pertinent to the MA period.)

A subset of nine hydrophones – numbers 14, 16, 19, 22, 45, 57, 63, 70, and 85 – were selected to provide a representative spatial coverage of the array, while avoiding peripheral (or edge) hydrophones (Fig. 1). There was roughly a 5–15 km horizontal distance between any two adjacent hydrophones examined.

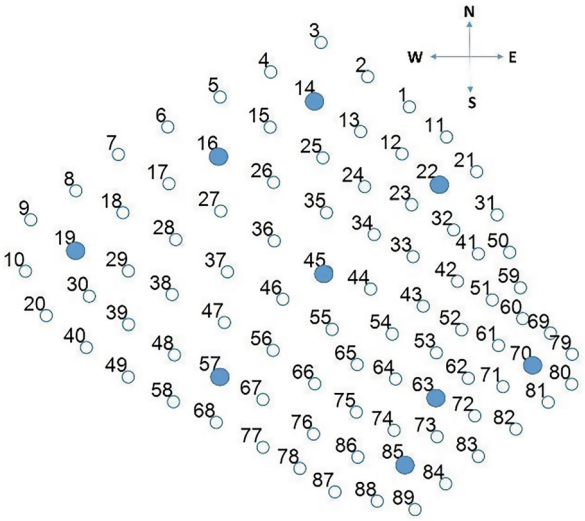



**Fig. 1** SOAR hydrophone array layout and the nine hydrophones examined in the work in this chapter (blue filled circles)

In order to understand how the MBES mapping survey impacted the local soundscape, the sound level metric time series, along with several spatio-temporal-spectral plots and animations, and sound files were reviewed to identify, describe, and characterize acoustic events in the SOAR during each of the analysis periods. In addition, the oblique range of the survey vessel to each hydrophone, and the expected transmission loss in the select frequency bands examined at those same ranges (based coarsely on spherical spreading loss and frequency-specific attenuation) were plotted along with the sound level metric time series, all at the same time resolution to aid in the interpretation of the source of the acoustic events with respect to the mapping activity. This was done for each of the nine selected hydrophones.

In addition to generating the sound level time series, sound level percentiles, i.e., 1, 5, 10, 25, 50, 75, 90, 95, and 99th, were computed for the following sound level metrics: *SPLpk*, *SEL*, 50 Hz, 12.5 kHz, 40 kHz *BL* for each analysis period and used to inform and focus the review of the acoustic activities in the data set. To facilitate the review, an acoustic event was defined *a priori* as a period in the *SPLpk* time series when the 95th percentile, for a given analysis period, was exceeded. The *SPLpk* time series had to return to or below baseline conditions – defined here as the 50th percentile of the *SPLpk* metric for a given analysis period – before a separate acoustic event was defined in the time series. Acoustic events were described by the maximum *SPLpk* value during an acoustic event. Changes in the other sound level metrics are discussed with respect to the 50th percentile level for the same metric and analysis period. The 50th percentile level for a given metric in a given analysis period is considered as, and referred to throughout this report, as the baseline sound level. For those *BL* metrics where the percentiles were not calculated, maximum changes in the sound levels during an acoustic event are discussed relative to a local baseline sound level, defined here as the approximate static sound level value for a given metric before and/or after an acoustic event.

Because of its location in the center of the SOAR and its proximity to repetitive passes of the survey vessel during the mapping survey, the review of the hydrophone 45 (see Fig. 1 for location) acoustic events is presented here as a representative example to aid in the discussion of the changing sound levels on the SOAR array. The intensive approach used for hydrophone 45 was modified based on the findings of this initial analysis for the other eight hydrophones. Details of that process and the results from the other eight hydrophones are freely available to the interested reader in Chap. 5 and associated appendices of Kates Varghese (2021).

## Results and Discussion

The No Activity (NA) period was based on the period of time prior to the survey vessel's presence on the SOAR array for which acoustic data could be obtained. This was an approximately 3 h and 40 min period from 1/4/17 20:13 to 23:51 UTC (note: all times are presented in UTC).

The Vessel Only (VO) period, which began 1/4/17 23:51 and ran until 1/5/17 08:16 for a total of 8 h 25 min, consisted of two vessel survey tracks across the hydrophone array from the south center edge to the north center edge of the array and back to the south. The ship's speed was approximately 10 knots throughout this period. All active acoustic sources were off. The track lines of the VO period are shown in blue in Fig. 2.

The VM period consisted of the primary mapping survey utilizing the hull-mounted Kongsberg EM 122, a MBES with a 12-kHz center frequency and source level of 239–242 dB re 1 μPa m. The VM period began at the end of the VO period on 1/5/17 at 08:16, and ran until 1/6/17 13:48, for a total of 29 h 32 min. The survey was conducted as a typical mapping survey for the water depths in the SOAR area, using a "mow-the-lawn" pattern (Fig. 2, red lines) with the EM 122 in deep, dual swath mode and CW pulses only. The survey was conducted in a calm sea state with

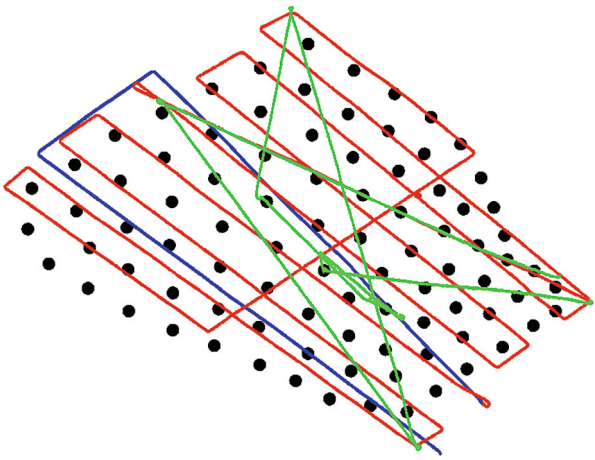


**Fig. 2** Track lines of the vessel during the analysis periods containing anthropogenic activity: blue lines correspond to the VO period, red to the VM period, and green to the MA period. (Discussed further in Kates Varghese 2021)

0.3–0.6 m (1–2 feet) swells. The vessel's survey speed was 10 knots except during turns when the speed was dropped to 5 knots.

Examples of the spectral probability density (SPD) animations (spectral-temporal-spatial animation_hydro45.pptx), and array-wide spatial animations iterating through the one-minute broadband and select decidecade band levels (array_wide_sound_levels.pptx, Smith et al. 2021), were used to inform the acoustic data set review are provided in the supplementary materials. The spectrograms for each of the events can be found in Chap. 5 of Kates Varghese (2021) and associated appendices.

Acoustic events in the hydrophone 45 dataset were triggered if the 95th percentile *SPLpk* threshold of each analysis period, listed in Table 1, were exceeded. Where possible, the maximum changes in the sound level metrics associated with an acoustic event were discussed with respect to the baseline sound level, i.e., the 50th percentile level, if calculated for a given metric (Table 1), or relative to the local baseline – the approximate static sound level for a given metric before and/or after an acoustic event (Table 2).

The detailed review of the acoustic time series data from the nine hydrophones on the SOAR provided useful insight about how the various anthropogenic activities related to the mapping survey contributed to the sound levels of the local soundscape. It also provided awareness about other sound sources present in the acoustic data that could not be linked to activity on the SOAR. The review of hydrophone 45, in particular, provided a representative example of the expected and unexpected changes in sound levels observed across the array. For example, during the NA and VO periods, there were epochs of low-frequency sound present in the hydrophone 45 data set, which was also identified in the larger review across all 9 hydrophones (see spl_peak_1min_viridis.mp4 animation in Supplementary). There were also instances where vessels, other than the survey vessel, passed nearby the array boundaries. These events were routinely detected on the northern and western edge hydrophones which was consistent with standard vessel traffic routes around the SOAR. There was also salient biological activity interspersed throughout the acoustic data on all of the hydrophones, which was particularly prominent in the changing 12.5 kHz and 40 kHz band levels. And finally, there were several unknown

**Table 1** Percentile values (dB re 1 μPa) for each analysis period used for annotating the sound level time series on hydrophone 45. *SPLpk* values given are in units of dB re 1 μPa, *BL* values given are in dB re 1 $\mu Pa^2$/Hz

| | Percentile value | | |
|---|---|---|---|
| Percentile | NA | VO | VM |
| *SPLpk* 50th (baseline) | 101 | 99 | 103 |
| *SPLpk* 95th (threshold for triggering an acoustic event) | 111 | 113 | 112 |
| 50 Hz *BL* 50th (baseline) | 68 | 60 | 55 |
| 12.5 kHz *BL* 50th (baseline) | 19 | 21 | 31 |
| 40 kHz *BL* 50th (baseline) | 13 | 14 | 16 |

**Table 2** Summary table of the acoustic events identified on hydrophone 45 during the study period, by analysis period. Table includes associated acoustic event number which corresponds with acoustic events annotated in Fig. 3, a description of the acoustic event and most prominent spectral changes, the distance of the survey vessel to hydrophone 45, and the most likely sound source, if identifiable. The asterisk in value 138* demarcates acoustic events during which a portion of the received signal was clipped due to limitations in the dynamic range of the hydrophone receiver (i.e., only sounds of 138 dB re 1 μPa or less were able to be fully captured)

| Analysis period | Acoustic event # | Acoustic event *SPLpk* maxima (dB re 1 μPa) | Context/spectral description | Oblique distance (km) of survey vessel to hydro 45 at maxima in acoustic event | Likely source |
|---|---|---|---|---|---|
| NA | 1 | 116 | Array-wide phenomenon; spectral energy greatest at very low frequencies (10's of Hz) | >30 | Another passing vessel |
| | 2 | 113 | Isolated to the western edge of the array; spectral energy greatest at very low frequencies (10's of Hz) | >30 | Another passing vessel |
| VO | 3 | 116 | array-wide phenomenon; spectral energy greatest at 5 kHz and below, tonal bands at 400 Hz, 1.2 kHz and 5 kHz | >20 | Uncertain |
| | 4 | 125 | Clear increase in all *BLs* starting when vessel was ~15 km away, clear decrease in *BLs* until vessel was again 15 km away; 50 Hz band increased by 26 dB; 40 kHz band increased by 8 dB | <2 | Survey vessel radiated sound |
| VM | 5 | 129 | Correlated with peak in 40 kHz band | 15 | Beaked whale foraging clicks |
| | 6 | 138* | corresponds with survey vessel transiting from the south to the north by hydrophone 45; sound levels were 25–35 dB | 5.3 | Survey vessel and MBES |

(continued)

**Table 2** (continued)

| Analysis period | Acoustic event # | Acoustic event *SPLpk* maxima (dB re 1 μPa) | Context/spectral description | Oblique distance (km) of survey vessel to hydro 45 at maxima in acoustic event | Likely source |
|---|---|---|---|---|---|
| | | | over baseline for the 12.5 kHz, 40 kHz, *SEL*, and *SPLpk* metrics; peak in 12.5 kHz band 33 dB over baseline | | |
| VM | 7 | 114 | Correlated with peak in 40 kHz band | >20 | Biological clicks |
| | 8 | 120 | Correlated with peak in 12.5 kHz band; 12.5 kHz band 15 dB over baseline | 10 | Survey vessel and MBES |
| | 9 | 114 | Very loud low frequency (50 Hz) noise, occurring every 20 s for about 1 s | >30 | Uncertain |
| | 10 | 115 | Elevated levels between 50–100 Hz; peak in 12.5 kHz band 7 dB above baseline | 14 | Vessel and MBES |
| | 11 | 133 | Low-frequency signal; short regularly spaced pulses in 40 kHz band | >25 | Another vessel? biological activity? |
| | 12 | 135 | Survey vessel passing from west to east by the hydrophone; peak in 12.5 kHz band 33 dB above baseline | 3.5 | Vessel and MBES |
| | 13 | 121 | Peak in 12.5 kHz band 12 dB above baseline | 12.5 | Vessel and MBES |
| | 14 | 123 | Peak in 12.5 kHz band 17 dB above baseline | 9.3 | Vessel and MBES |
| | 15 | 138* | Peak in 12.5 kHz band 33 dB above baseline; also peaks in the 50 Hz and 40 kHz band (i.e., local increases of 12–45 dB), with a visible increase in the other bands (i.e., local increases of 3–12 dB) | 4.7 | Vessel and MBES |

(continued)

**Table 2** (continued)

| Analysis period | Acoustic event # | Acoustic event *SPLpk* maxima (dB re 1 μPa) | Context/spectral description | Oblique distance (km) of survey vessel to hydro 45 at maxima in acoustic event | Likely source |
|---|---|---|---|---|---|
| VM | 16 | 138* | Peak in 12.5 kHz band 42 dB above baseline; also peaks in the 50 Hz and 40 kHz band (i.e., local increases of 12–45 dB), with a visible increase in the other bands (i.e., local increases of 3–12 dB) | 1.6 | Vessel and MBES |
| | 17 | 138* | Peak in 12.5 kHz band 35 dB above baseline; also peaks in the 50 Hz and 40 kHz band (i.e., local increases of 12–45 dB), with a visible increase in the other bands (i.e., local increases of 3–12 dB) | 5.7 | Vessel and MBES biological clicks? |
| | 18 | 132 | Marine mammal clicks centered around 40 kHz, as well as a longer, louder, and more irregular signal around 50 Hz; correlated with peaks in the 50 Hz and 40 kHz bands | 9 | Biological clicks |
| | 19 | 131 | Marine mammal clicks centered around 40 kHz, as well as a longer, louder, and more irregular signal around 50 Hz; correlated with peaks in the 50 Hz and 40 kHz bands | 24 | Biological clicks |

sources and signals identified, some of which were as loud as some of the closest passes of the survey vessel to the hydrophones.

The NA period, during which the survey vessel was not on the array, was meant to provide information about the background SOAR soundscape and variability without anthropogenic acoustic activity. There were several events on many of the hydrophones that indicated closely passing vessels, i.e., anthropogenic activity not

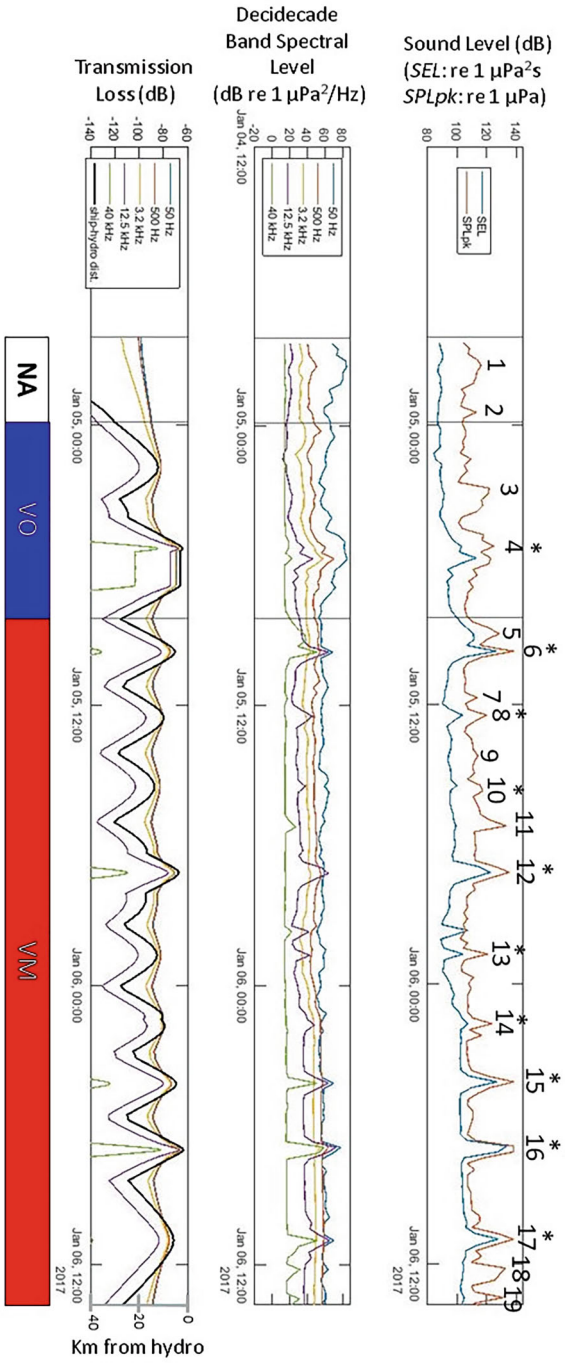



**Fig. 3** Sound level time series for hydrophone 45, data smoothed over 5-min window for ease in viewing. Time along the x-axis begins with the start of the NA period and runs to the end of the VM period. Top plot: Broadband sound level time series: *SEL* (blue), *SPLpk* (orange). Numbers correspond with the acoustic events identified. Asterisks indicate that the acoustic event was most likely attributed to the survey activity. Vertical lines on all graphs delineate the time periods of the three analysis periods, NA, VO, VM, also shown at the bottom of the figure, respectively. Middle plot: Select *BL* time series: 50 Hz (blue), 500 Hz (orange), 3.2 kHz (yellow), 12.5 kHz (purple), and 40 kHz (green). Bottom plot: The vessel's oblique distance from hydrophone 45 in kilometers (black line, right axis), and time series of modeled frequency-specific transmission loss from a hypothetical source if it were the same distance from hydrophone 45 as the vessel at that respective time (left axis). Color of frequency-specific transmission loss corresponds to the colors of frequencies described for the middle plot

related to the ocean mapping survey, during this period. It was also expected that this period would be stable and quiet in comparison to the other periods. However, there was a high amplitude, low frequency (visible in 50 Hz band) signal detected across the entire array over much of the NA period. Because the assumptions about this period were not met, a comparison of the NA period with the other analysis periods with these attributes in mind may be misleading. The assumption that the NA period was not influenced by substantial anthropogenic activity appeared to hold better for some hydrophones (i.e., 22, 45, 63, 57, and 70; closer to San Clemente Island and farther from the typical vessel traffic corridors) in comparison to the others. Ultimately, the characterization of the NA periods on each hydrophone and comparison to the periods with survey activity provided understanding of what the added input of a mapping survey would be on top of the typical ambient SOAR soundscape variability.

The close passes of the survey vessel alone had a very characteristic sound level signature indicative of ship-radiated sound, i.e., elevated levels across a broad range of frequencies. A spectral analysis of the specific signature of *R/V Sally Ride* in comparison to other vessels present in the acoustic data was not made. When the survey vessel alone was within 3 km of a hydrophone, the peaks in the 50 Hz, 500 Hz, 3.2 kHz, and 12.5 kHz bands were all about 15-26 dB over their respective local baselines, and this was similar for *SPLpk*. (Note: Event 4 on hydrophone 45 occurred when the survey vessel was within 2 km and was characterized by maximum increases in *SPLpk* and the 50 Hz band of 26 dB. There were local maximum increases of 8–23 dB in the other select decidecade bands.) Beyond 10 km, it was difficult to distinguish just the survey vessel in the sound level time series, and there were few instances to reliably characterize the sound levels at more intermediate distances.

A comparison between the VO and VM periods provides insight as to how the sound levels were influenced by vessel only related sound versus vessel sound plus multibeam signals. The two activities had different and distinguishable sound level signatures. For example, events 4 and 16 on hydrophone 45 represented the closest approach of the survey vessel during the VO and VM periods, respectively. In the VO period, the elevated sound levels in the five select decidecade *BL*s scaled similarly (i.e., the increase in levels parallel each other), whereas in the VM period, the close approach of the vessel and the actively transmitting EM 122 manifested as strong peaks in the high frequency bands (i.e., 12.5 kHz and 40 kHz) which surpassed the peaks in the lower frequency bands (i.e., 3.2 kHz and 500 Hz) (see Fig. 3, events 4 and 16). The 12.5 kHz and the 50 Hz bands were similarly loud. The peak in the 12.5 kHz band matches expectation since the EM 122 signal is a known sound source in this band. The energy in the 50 Hz band was most certainly driven by vessel-radiated noise.

The peak in the 40 kHz band was likely a measurement artifact related to the times when the EM 122 signal was clipped at the receiver and occurs only when the survey vessel comes within close range of the hydrophone. Because this is a likely measurement artifact, it is difficult to distinguish real signal content from aliased signal content due to clipping. Harmonics related to the EM 122 transmissions are possible due to harmonic distortion caused by the transducer, but are typically observed about 30 dB less than the fundamental frequency (as shown by dedicated

source transmission recordings (Smith 2019)). So although some energy from harmonics may be present, in this case, they cannot be separated from the clipping-related artifact, and are most certainly only a small contribution to the elevated 40 kHz band observed here. Moreover, source-generated harmonics at 40 kHz should attenuate much faster with range than the fundamental 12.5 kHz signal, and thus would only be detected at very short distances from the source. On the other hand, receiver-generated harmonic tones caused by clipping have intensity levels with magnitudes comparable to the signal's fundamental frequency, making them very likely to dominate over the source-generated harmonics.

With regard to the changes in the sound levels during the VM period, clipped peaks in *SPLpk* (i.e., at 138 dB re 1 μPa) were observed when the EM 122 was in use 6 km away or less from the hydrophones. These events had corresponding peaks in the 12.5 kHz band of 50–75 dB re 1 $\mu Pa^2$/Hz that were 19–42 dB over baseline, while the 40 kHz band was elevated by a similar magnitude (Note: hydrophone 45 was characterized by maximum increases of 25–35 dB in the 12.5 kHz and *SPLpk* metrics). The lower select decidecade *BL*s (50 Hz, 500 Hz, 3.2 kHz) were generally 10–15 dB over local baselines when the vessel was at close distances to the hydrophone (i.e., <6 km). When at distances of 6–10 km, peaks in *SPLpk* ranged between 107 and 127 dB re 1 μPa and the 12.5 kHz peaks were 30–54 dB re 1 $\mu Pa^2$/Hz, or 6–23 dB over baseline (Note: hydrophone 45 was characterized by increases of 15–20 dB over baseline in the 12.5 kHz band and *SPLpk* metrics). At this range, the other decidecade *BL*s were only elevated by about 3–6 dB, and there was no longer a similar magnitude peak in the 40 kHz band which is consistent with the observed absence of clipping (though some events may have elevated levels in the 40 kHz band because there were biological clicks present at the same time). Distinguishable peaks in the 12.5 kHz band time series that were clearly associated with the vessel and EM 122 activity were identified up to distances of 13.5–17 km (although EM 122 pulses were faintly visible in a few spectrograms out to 20 km). Within this range, *SPLpk* varied from 107 to 134 dB re 1 μPa with peaks in the 12.5 kHz band of 35–44 dB re 1 $\mu Pa^2$/Hz, or 2–14 dB over baseline (Note: for hydrophone 45, maximum increases in *SPLpk* and the 12.5 kHz *BL* were 7–12 dB over baseline). And fluctuations in the other select decidecade bands were less than 3 dB. However, at this range, the maxima in *SPLpk* could no longer be exclusively attributable to survey activity.

Undoubtedly, orientation of the vessel to the hydrophone also plays a major role in received sound levels since the main transmission beam of the MBES is strongly directional (1° aperture), downward-propagating, fan-like and oriented perpendicularly to the ship's axis. This likely accounts for many inconsistencies to the generalized patterns identified here based on a range-only explanation. Not only is the transmission directional, but there are multiple lobes that make up the transmission pattern. The main lobe of the transmission is by far the loudest and is generally directed vertically toward the seafloor, whereas side lobes are transmitted at oblique angles to vertical and are typically on the order of 20–30 dB lower than the main lobe. Thus, some of the variability in the detection of the EM 122 signal in the sound level time series was related to (1) the orientation of the survey vessel as it approached and/or departed the

hydrophone, and (2) whether the hydrophone was exposed to the main transmission beam, or to a side lobe. Nonetheless, no EM 122 signal could be discerned beyond a range of 17 km.

Although other signals and periods of elevated levels from non-survey activity could be detected on the hydrophones, the survey activity was by far the most common and salient source of acoustic events identified during the study period. The survey vessel alone was detected on seven of the nine hydrophones examined (not on hydrophones 14 or 70), and discernible for 1 (hydrophones 63 and 85) to 3.5 h (hydrophone 45) of the VO period. The survey activity during the VM period was intermittently discernable in the sound level time series for 7 h and 22 min hours on average, ranging from as few as 5 h (hydrophone 19) to at most 11 h and 45 min (hydrophone 45).

It is worth noting that the impact of the same mapping survey discussed herein was also examined with respect to the spatio-temporal foraging behavior of Cuvier's beaked whales (*Ziphius cavirostris*) at the SOAR in Kates Varghese et al. (2020, 2021). The results of those studies clearly showed that the mapping survey did not adversely affect Cuvier's beaked whale foraging. Animals stayed on the SOAR and continued to forage during the mapping survey in 2017, as well as during a second mapping survey with the EM 122 carried out in 2019. The characterization of the sound levels during the 2017 mapping survey as presented herein provides important context to those whale behavior observation works. Despite high sound levels transmitted by the MBES, no adverse changes were observed in a critical life behavior of a species that has been previously shown to be adversely affected in a very significant way by other sonar sources with significantly wider transmission lobes and longer signals (McCarthy et al. 2011; Manzano-Roth et al. 2016). The findings by Kates Varghese et al. (2020, 2021) are corroborated by the results presented here: the ensonification by the MBES mapping activity is confined to a limited space (defined by a narrow angular sector for the main beam and widespread sidelobes at lower levels) around the mobile survey vessel and thus can only have a transient effect on any single location; this is still reinforced by the low values of the signal durations and ping rates.

## Conclusion

This time series review proved to be an essential step for understanding the complex acoustic data obtained from recording a deepwater MBES mapping survey on an array of bottom-mounted hydrophones. Despite the limitations raised by the signal clipping upon reception, the exercise confirmed the expected temporal and spatial dependence of the survey activity contribution to the soundscape. The change in sound levels was closely tied to the specific radiation pattern of the MBES around the mobile survey vessel (short narrow-band signals inside a fan-shaped main sector of high intensity and widespread sidelobes at lower levels), which meant the temporal impact at any given location was also confined to a usually brief time period that the vessel was in the corresponding area.

- The EM 122 signal was detectable above baseline sound levels in the acoustic records only when the survey vessel was within 17 km of a hydrophone receiver (visible faintly in spectrograms up to 20 km from a hydrophone receiver).
- Clipping of received signals occurred at times when the survey vessel with transmitting EM 122 was less than 6 km from a hydrophone, preventing their true sound levels from being measurable at these ranges.
- Of the sound level metrics examined, the EM 122 signal most clearly and consistently manifested in the elevated 12.5 kHz decidecade band levels, and was often – but not always – associated with the maxima in the peak sound pressure level time series.
- Neither the EM122 nor the Sally Ride's other sonar systems or own radiated noise were the only detected sources during the survey. There was consistently biological activity, other vessels passing on or near the array, as well as other unidentified sound sources.
- The sound level characterization of the MBES mapping survey provides important descriptive and explanatory context for simultaneous observation works of marine mammal behavior during the same MBES survey.

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# Noise Pollution on the Singing Cycle of Humpback Whales and the Soundscape of the Gulf of Tribugá, Colombia

35

Maria Paula Rey-Baquero, Kerri D. Seger, and Andrea Luna-Acosta

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**Abstract**

Noise pollution in marine ecosystems has increased during the last decades. In the Pacific Ocean, like in other ocean basins, the most continuous anthropogenic source that contributes to the ambient acoustic environment is maritime transport. Noise produced by ships reduces the communication space of marine species and has various immediate and chronic effects. The study area for this project was the

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Gulf of Tribugá – a breeding ground for one of the humpback whale distinct population segments. The goal of the project was to quantify the contribution of humpback whale sounds and other acoustic sources to the soundscape, as well as to document the baseline soundscape. Acoustic samples were recorded at Morro Mico and Nuquí in 2018 and 2019. Both sites were dominated by biophony. In 2019, humpback whale song activity (minutes with song activity) peaked in the mornings at the Morro Mico site and in the evenings at the Nuquí site. This study has revealed that the Gulf of Tribugá demonstrates relatively lower presence of acoustic pollution generated by boats in comparison to other documented oceanic regions. Nonetheless, it is advisable to avoid increasing boat traffic during peak times of the humpback whale song cycle and tailored to the specific site.

**Keywords**

Biophony · Noise · Small boats · Soundscape · Humpback whale

## Introduction

Marine pollution produced by anthropogenic activities is closely related to the increase of human populations in coastal areas and domestic, agricultural, and industrial activities (Garay et al. 2001). There are different types of pollution, such as chemical pollutants, microplastics (Chitrakar et al. 2019), microbiological toxins (Zappalà et al. 2012), and noise (Shannon et al. 2016). Specifically in the last five decades, noise has increased significantly in the Pacific Ocean and one of the primary contributors to this problem is maritime transport (Hildebrand 2009; McDonald et al. 2006).

Noise produced by ships is a key contributor to reduced communication space of many marine species, from marine mammals to fish, likely causing chronic effects on individuals and populations (Putland et al. 2018). Ships have been considered the reason for a roughly 12 dB increase in low-frequency background noise levels in the Pacific Ocean (Hildebrand 2009), contributing to the masking of natural sounds (Kaplan and Mooney 2015). Noise produced by small boats, for instance from whale-watching vessels that stay generally close to shore to view easily accessible cetaceans, varies according to the flow of tourists since this parameter influences the number and size of boats used on ecotourism trips (Sprogis et al. 2020). In general, more and larger boats will create more noise pollution, particularly near ports (Chen et al. 2017).

A soundscape includes three fundamental elements: (1) anthropophony, which includes sounds associated with human activity; (2) biophony, which includes sounds produced by animals, like vocalizations (e.g., the song of whales) and nonvocal acoustic signals (e.g., feeding sounds of mollusks) (Kitting 1979); and (3) geophony, which are the sounds generated by physical events such as waves, earthquakes, rain, or storms (Krause 1987; Montgomery and Radford 2017; Pijanowski et al. 2011; Wang et al. 2019).

Some sources of noise, such as those from tourism boats, can have predictable activity cycles from predetermined schedules, while weather events (a natural source of masking) can be more sporadic (Fournet et al. 2018). Inconsistent source cycles

cause inconsistent sound level fluctuations in various frequency bands of the soundscape. In some cases, noise level changes could be driven by biological sound sources and in others by anthropogenic sources (Haver et al. 2019; Seger et al. 2015).

Many marine organisms, including invertebrates, fish, and mammals, contribute to the soundscape, with natural sounds such as whale vocalizations and fish stridulations (Pijanowski et al. 2011). This and many more examples showcase how a healthy soundscape is vital for many animals, allowing them to detect predators/prey, as well as navigate or protect young, thus facilitating critical interactions and dynamics of the ecosystem (Merchant et al. 2015; Remage-Healey et al. 2006). Marine mammals such as whales and dolphins use acoustic signals to facilitate some vital functions like communication, navigation, foraging, breeding, and predator avoidance (Haver et al. 2019; Richardson et al. 1995). Noise can affect them at individual and population levels. For example, physiological stress can inhibit developmental, reproductive, and/or immune functions (McDonald et al. 2008; Rolland et al. 2012).

The humpback whale (*Megaptera novaeangliae*) is a cosmopolitan species, found in all oceans (Clapham and Mead 1999). Humpback whales migrate long distances between tropical breeding areas and polar feeding areas (Clapham 1996, 2000). Individuals from Stock G (International Whaling Commission 1998) arrive in coastal waters of the Colombian Pacific to mate, give birth, and suckle their young (Flórez-González 1991).

Humpback whales produce considerable acoustic activity including songs emitted only by males, mostly during the breeding season, and social sounds, produced by individuals of both sexes and all age classes (Payne and McVay 1971; Rossi-Santos 2015; Stimpert et al. 2012). The song of the humpback whale has been studied, leading to various hypotheses regarding its functionality (Herman 2017). It has been proposed that the song could be used to mediate interactions between males, establish dominance (Darling and Bérubé 2001), attract females (Frankel et al. 1995; Tyack 1981; Winn and Winn 1978), maintain male spacing (Frankel et al. 1995), promote ovulation in females (Baker and Herman 1984), and assist in locating other whales (Frazer and Mercado 2000).

Some behavioral changes have been reported in humpback whales and other cetacean species in response to noise from whale-watching boats (Parsons 2012). For example, Scheidat et al. (2004) reported an increase in swimming speed of humpback whales in Isla de la Plata, Machalilla National Park, Ecuador, in response to tourism vessels. In Bahía Málaga in the Colombian Pacific, an increase in the speed of displacement was reported in correlation with the presence of boats, which was interpreted as avoidance behavior and could incur an additional energetic cost (Avila et al. 2015). Reactions to underwater noise can vary by source (Erbe 2012).

The Colombian North Pacific (specifically Chocó and the Gulf of Tribugá) is one of the most biodiverse areas in the world with high ecological value and complex climatic, geological, and biological histories (Velandia and Díaz 2016). However, the construction of a port in the area has been under consideration for some time. If the project is carried out it would create changes in the soundscape since underwater noise particularly increases near ports and along shipping routes due to a greater abundance of large boat traffic (Chen et al. 2017).

The purpose of the research reported here was to quantify the baseline soundscape and the diel and seasonal patterns of the humpback whale song in the Gulf of Tribugá under the current level of whale-watching and small fishing boats. The goal is to ultimately evaluate any changes in years to come no matter what infrastructure changes occur. This information could also serve to update management plans and environmental impact assessments by including acoustics, which was a topic not considered in the previous permit applications for the port.

## Methods

### Study Site

This study was conducted in the Colombian North Pacific and the sampling sites were: (1) Morro Mico (adjacent to the southern border of Utría National Natural Park), a tiny island reef a few hundred meters from the coast in the north of the Gulf of Tribugá, and (2) Nuquí, a village in the department of Chocó with a large river outpouring and the location of the airport that connects the area with the interior of the country (Fig. 1). The sites were chosen because of the presumed differences in their maritime traffic schedules as advised by experienced researchers in the area. Nuquí is a more central and commercial area while Morro Mico is closer to the Utría National Natural Park and is less traversed. Because of its relatively pristine condition largely to do its remoteness from large shipping lanes and its containing little

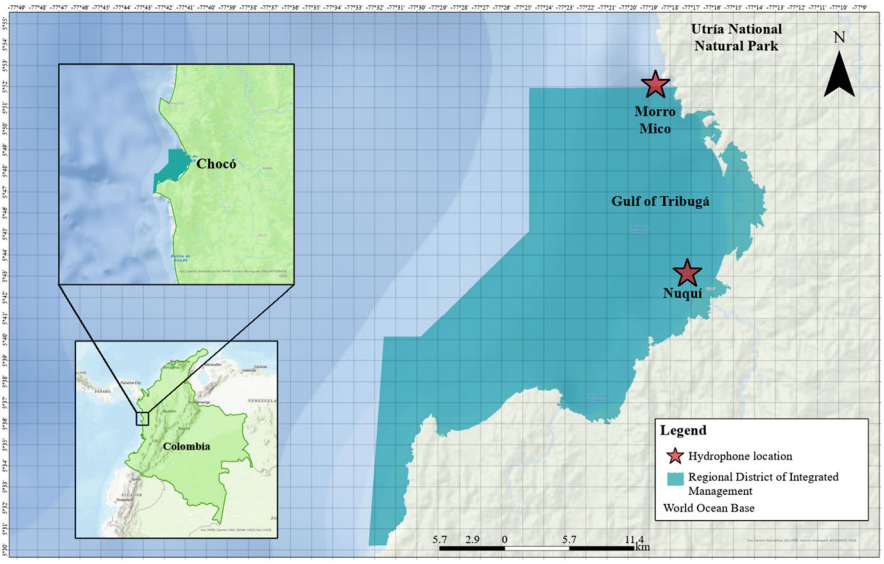



**Fig. 1** Study area in Colombia located in the department of Chocó. The stars indicate the locations (sites) of the recorders in Morro Mico and Nuquí

infrastructure, the entire Gulf of Tribugá was designated a Hope Spot by Mission Blue in 2019. Hope Spots are special places scientifically identified as critical to the health of the ocean. Additionally, Tribugá was named a Biosphere Reserve by UNESCO in 2023, signifying it as a "learning place for sustainable development."

Additionally, weather differences could potentially have been captured between these two sites, specifically differences in rainfall. The Gulf of Tribugá is characterized by being one of the rainiest regions in the world with average annual rainfall around 5000–7000 mm (Velandia and Díaz 2016). There is a trend that rainfall increases South to North across the Gulf of Tribugá, so a more northern study site could experience more sound from rain than a more southern study site, even though weather patterns in the Gulf are patchy microclimates (Velandia and Díaz 2016).

The Gulf of Tribugá Cabo Corrientes Regional District of Integrated Management Area has 60,183 ha and is administered by Codechocó, the policymaking entity responsible for managing natural resources based on sustainable human development with community participation. Therefore, conservation actions are combined with sustainable resource uses like whale-watching tourism. The national park and local conservation-minded people are interested in best practices of whale-watching boats to ensure that they protect the artisanal fishing economy of the area.

## Acoustic Data Collection

Acoustic data were collected using an Ecological Acoustic Recorder (EAR, Oceanwide Science Institute) in Morro Mico and Nuquí in 2018 and 2019 (Table 1). The instrument was programmed to record for 10 minutes on and 20 minutes off, or a 33.3% duty cycle. The sampling rate was 15.625 kHz. The hydrophones on the EARs had a sensitivity of 193.5 dB re 1 V/Pa with a 47.5 dB preamp.

## Spectrogram Analysis

Acoustic recordings were analyzed using spectrograms (Rossing 2007) generated in Raven Pro (version 1.5; Cornell Laboratory of Ornithology, Ithaca, New York) with a temporal resolution of ten minutes and a Fast Fourier Transform (FFT) of

**Table 1** Acoustic data collected in 2018 and 2019 at both sites

| Site | Year | Recorder | GPS | Dates | Days w/Data |
|---|---|---|---|---|---|
| Morro Mico | 2018 | EAR | 05°52′10.1″N 77°18′40.7″W | 22/10 to 28/11 | 38 |
| Morro Mico | 2019 | EAR | 05°52′10.1″N 77°18′40.7″W | 31/07 to 9/10 | 71 |
| Nuquí | 2019 | EAR | 05°43′12.2″N 77°17′10.1″W | 31/07 to 7/09 | 39 |

512 along with an overall bandwidth analysis. The representation in the form of a spectrogram displays time on the horizontal axis, frequency on the vertical axis, and the relative power at each time and frequency as a color depiction using a color scale within the spectrogram. The ten-minute files were manually analyzed every 6 h at 12:00, 18:00, 00:00, and 06:00. Any presence of the following six sound types were tallied in each minute: humpback whales, dolphins, fish, shrimp, wind/rain, and small boats engines. This per-minute tally of sound classifications provided presence time series of six sound types. These time series were then examined to identify the presence of biological cycles, such as diel patterns. Furthermore, a comparison was made between the compositions of the soundscape at each site, and the calculation of "noise budgets" was performed to illustrate the percentage that each type of sound was relatively present at each site. Because noise budgets are not yet widely used in policy decisions (Merchant et al. 2017) and considering that reports concentrating on anthropogenic noise emphasize the creation of these budgets (National Research Council 2003), the goal was to establish a measure of usable relative activity of sound sources at each environment. Percentages of total time when a sound source is present yields itself to an easily accessible pie chart that could be reasonably be tracked over the years to provide policymakers with a reference for how human noise fits into the overall noise budget of each region as time changes the infrastructure of the area (Merchant et al. 2020). For monthly comparisons, the sounds were divided by the total samples within each group, resulting in the generation of a presence rate.

### Statistical Analysis

To identify whether there was a significant difference between the monthly occurrence of each sound type at each site, chi-square tests were performed. Because the expected frequencies in some cells were too small to use the chi-square distribution, a Monte Carlo simulation was performed (Verzani 2014). The null hypothesis ($H_0$) stated that there would be no relationship between the months and the proportion of each sound's presence, while the alternative hypothesis ($H_1$) stated that there would be a relationship between the months and the sound source proportions. Statistical analyses were performed using R software (version 3.5.2). The chosen significant p-value was 0.05.

## Results

For both years (2018 and 2019) and both sites (Morro Mico and Nuquí), sounds from humpback whales, fish, shrimp, wind/rain, and small ecotourism/artisanal fishing and shrimping boats were detected. Time series of these sounds are shown for descriptive results to provide the reader with a "feel" for each year's and site's soundscape. The qualitative results are important, because they represent relevant observations that will allow for comparison of soundscape aspects into the future.

**Table 2** Sampling schedule for each year at both sites

| Site/Year | Month | Sample |
|---|---|---|
| Morro Mico 2018 | October | 76 |
| | November | 222 |
| Morro Mico 2019 | July | 4 |
| | August | 248 |
| | September | 240 |
| | October | 68 |
| Nuquí 2019 | July | 4 |
| | August | 248 |
| | September | 54 |

The recording months were different in each year and location due to climatic and logistical conditions. In 2018, 1681 recordings were collected over 38 sampling days at Morro Mico; in 2019, 3675 recordings were collected over 71 sampling days. In Nuquí the recorder failed in 2018, but in 2019, 1804 recordings were collected over 39 days. The sampling schedule for each year at sites Morro Mico (MM) and Nuquí (NQ) is presented in Table 2.

## Spectrogram Analysis

For both sites, the sources of anthropophony came primarily from small boats as well as from shrimping boats and the associated mechanical sounds of their net chains (Fig. 2). This was evident because these boats occupied a large part of the frequency band and had the typical u-shape indicative of the Lloyd's mirror effect observed in passing vessels. Geophony sounds came from wind or rain (Fig. 3). The sounds of storms (Fig. 3a) and raindrops (Fig. 3b) occupied a large part of the frequency spectrum, and the amount of rain was reflected in the intensity of the spectrograms. Finally, biophony came from the vocalizations of humpback whales (Fig. 4) and sounds of fish and snapping shrimp (Figs. 5 and 6). Neither dolphin clicks nor whistles were detected in any of the years or at either sites.

Between both sites differences were observed in the relative amount of contribution from snapping shrimp to the soundscape. Their presence in Morro Mico (Fig. 6a) for the 2 years was higher than at Nuquí (Fig. 6b), where they were barely evident in most of the acoustic samples. The cavitation sounds produced by the shrimp at Morro Mico for the 2 years were ever-present in the above 4.5 kHz band (Fig. 6a).

## Analysis of Soundscape

The dominant sound source in the 2018 (Fig. 7a) and 2019 (Fig. 7b) soundscapes at Morro Mico was snapping shrimp, followed in descending order by sounds produced by fish, humpback whales, wind/rain, and boats. The snapping shrimp sounds

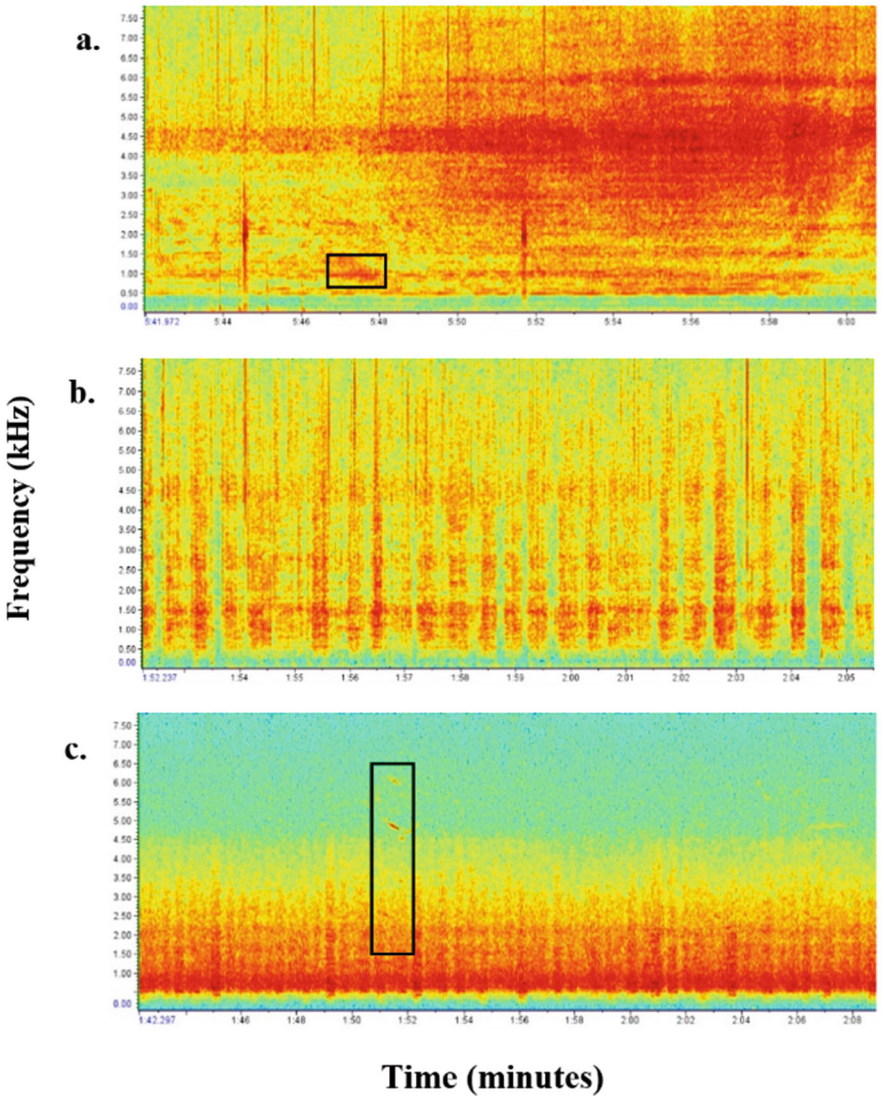



**Fig. 2** (**a**) Spectrograms of small ecotourist/artisanal fishing boats at Morro Mico on 08/09/2019 12:00 hours and (**c**) Nuquí on 08/09/2019 at 06:00 hours. (**b**) Shrimping boats at Morro Mico on 08/28/2019 at 00:30 hours. Black squares denote social calls of humpback whales

were nearly constant. On the other hand, for Nuquí in 2019 (Fig. 7c), the dominant sounds were humpback whales, followed by sounds produced by fish, boats, wind/rain, and shrimp. While the relative proportions of the different categories of sounds did not vary between years in Morro Mico, they did vary between the locations of

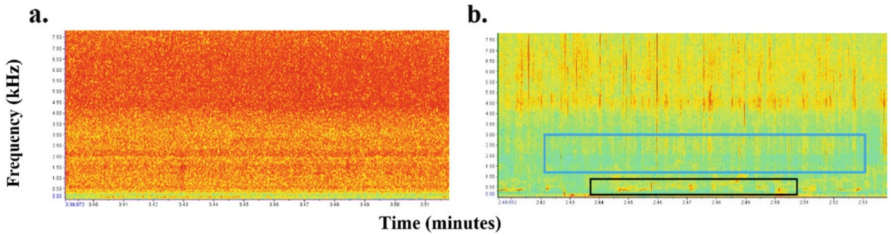



**Fig. 3** (**a**) Spectrograms of a storm at Morro Mico on 08/06/2019 at midnight and (**b**) rain drops at Morro Mico on 08/19/2019 at 18:30. The blue box denotes the drops, while the black rectangle denotes the song of at least one humpback whale (from 0–1 kHz). Sound energy above 4 kHz is primarily from snapping shrimp at Morro Mico

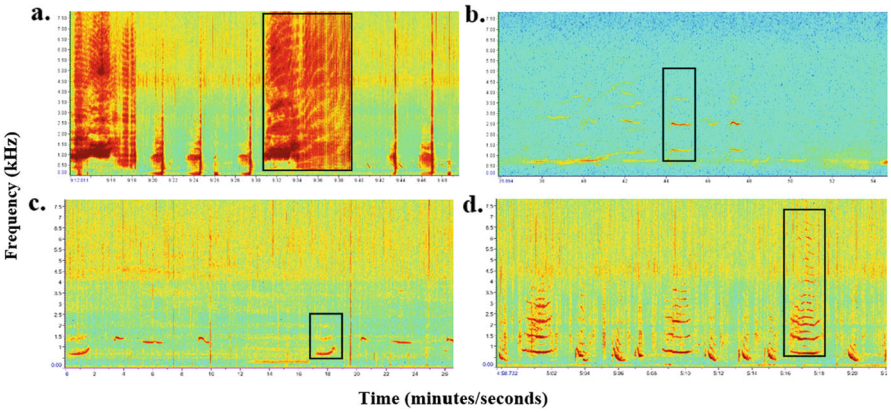



**Fig. 4** Spectrograms of some humpback whale song units (framed in black boxes)

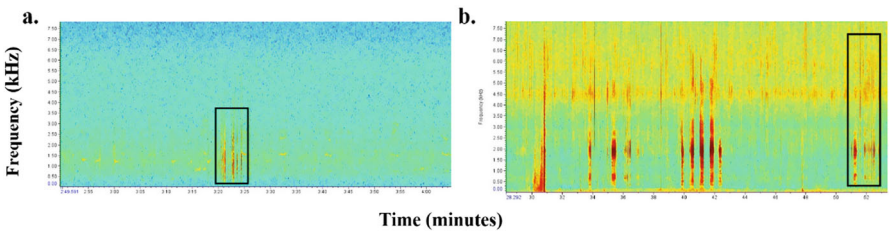



**Fig. 5** (**a**) Spectrograms of sounds produced by fish at Nuquí on 08/01/2019 at midnight and (**b**) at Morro Mico on 08/24/2019 at 12:00 hours, framed in black boxes

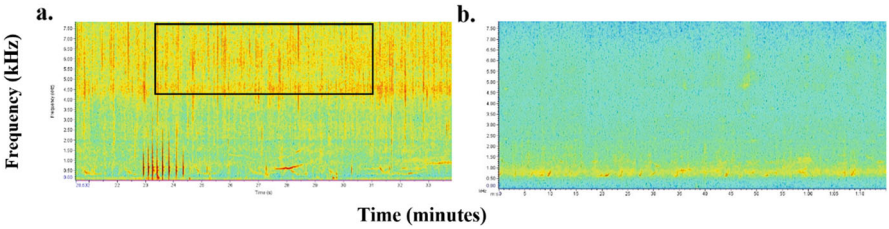



**Fig. 6** Spectrograms of shrimp sounds (block box) at (**a**) Morro Mico on 08/09/2019 at 06:00 hours and (**b**) and their absence at Nuquí on 08/09/2019 at 06:00 hours

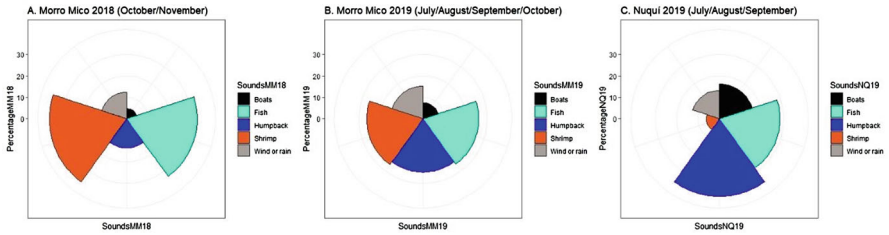



**Fig. 7** (**a**) Pie charts (or proportions) of sounds found in Morro Mico for the year 2018, in the combined months of October and November; (**b**) Morro Mico 2019 in the months of July, August, September, and October; and (**c**) Nuquí 2019 in the months of July, August, and September

**Table 3** Sound source presence percentages (rates) for Morro Mico and Nuquí in 2018 and 2019

| Site | Year | Humpback (%) | Fish (%) | Shrimp (%) | Wind/Rain (%) | Boats (%) |
|---|---|---|---|---|---|---|
| Morro Mico | 2018 | 14 | 33 | 35 | 12 | 5 |
| Morro Mico | 2019 | 25 | 26 | 26 | 15 | 8 |
| Nuquí | 2019 | 36 | 28 | 6 | 13 | 16 |

Morro Mico and Nuquí. The proportional values for each sound source across site and year are shown in Table 3.

Presence rates, which correct for sampling bias toward some months (Fig. 8), show that humpback whales were less prominent later in the year, so overall noise budgets (Fig. 7) of a season do not necessarily represent noise budgets per month.

In 2019 at Morro Mico (Fig. 8b), October compared to November would suggest a large difference in sound source types in the soundscape, but in 2018 all months July–October were relatively similar in their noise budgets. This illustrates that a potential soundscape shift at Morro Mico begins in November, which does coincide with humpback whale southward migration commencing.

In 2019 at Nuquí, July to September (Fig. 8c) had relatively stable noise budgets. However, a peak in snapping shrimp activity in August occurred and could be studied in the future when year-round data can be collected.

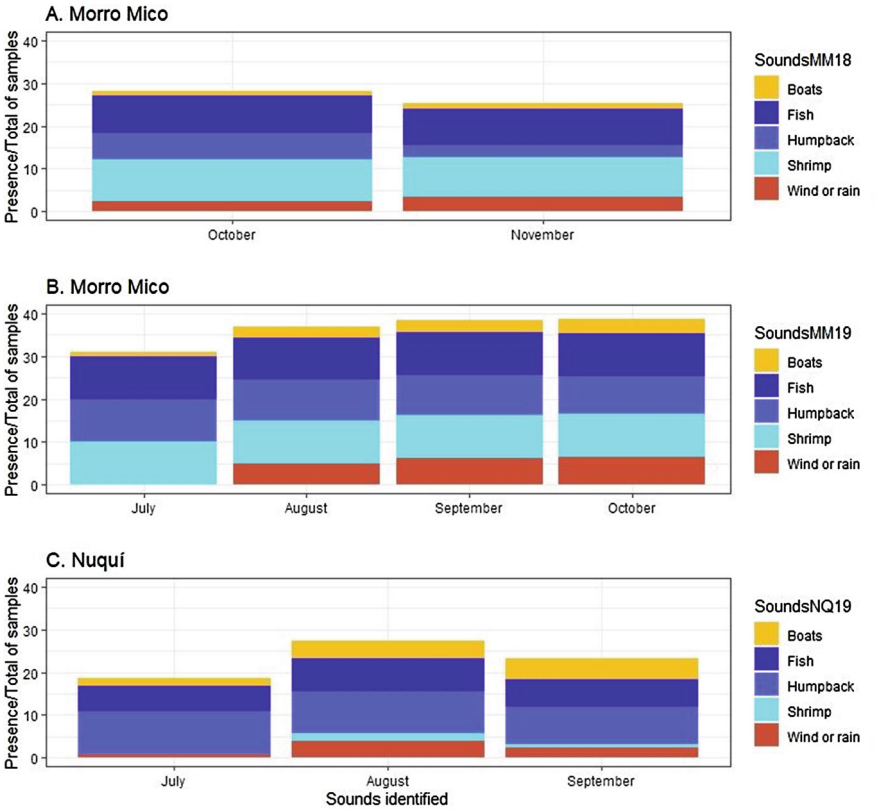



**Fig. 8** Presence rates for each sound source in total samples per month for Morro Mico in (**a**) 2018 and (**b**) 2019, and (**c**) Nuquí for 2019

In brief, for Morro Mico in 2018, presence rates were similar between months except for humpback whale sounds, which occurred less often in November. For Morro Mico in 2019, rates were also similar between months except for July when boats had lower presence rates than the other months. For Nuquí in 2019, the same lower presence rate of boats in July was detected, likely because of fewer recording days. Wind/Rain presence rates were different between months insomuch that they peaked in October or November at Morro Mico but in August at Nuquí. This could be because Nuquí was not sampled in October or November and/or it shows the existence of microclimates in the Gulf of Tribugá. Anecdotally, a common joke is that each village in the Gulf "has its own cloud," and the collected data show that could be true. Presence rates for each sound source in total samples per month are shown in Table 4.

A chi-square test ($X^2_x = 0.994$, $p < 0.05$) revealed that there was no statistically significant relationship between the sound rates of each group and the months at each

**Table 4** Presence rates for each sound source in total samples per month for both sites

| Site/year | Month | Shrimp | Fish | Humpback whales | Wind/ Rain | Boats |
|---|---|---|---|---|---|---|
| Morro Mico 2018 | October | 9.8 | 8.8 | 6 | 2.4 | 1.1 |
| | November | 9.3 | 8.6 | 2.7 | 3.5 | 1.3 |
| Morro Mico 2019 | July | 10 | 10 | 10 | 0 | 1 |
| | August | 9.9 | 9.7 | 9.5 | 5.1 | 2.7 |
| | September | 9.9 | 9.9 | 9.3 | 6.3 | 2.9 |
| | October | 10 | 10 | 8.8 | 6.6 | 3.3 |
| Nuquí 2019 | July | 0 | 6 | 10 | 0.8 | 1.8 |
| | August | 1.9 | 7.8 | 9.8 | 3.8 | 4.3 |
| | September | 0.6 | 6.4 | 8.9 | 2.5 | 4.7 |

site. In brief, the rates of sound sources that occurred did not vary significantly between months.

In 2018 at Morro Mico, the highest absolute amount of humpback whale vocal activity (primarily singing but occasionally social calls) was detected in 180 of the November files recorded at 18:00 hours (Fig. 9a). In 2019, the highest number of files with detected humpback whale sounds (613) occurred at 06:00 hours in August (Fig. 9b). Despite humpback whales having a lower presence rate in November than in October 2018 (Fig. 9a), their sounds filled more files in November than in October. Morro Mico in 2018 had a slight diel cycle for humpback whale songs with a sunset peak. Morro Mico in 2019, however, did not exhibit this same evening peak of activity, but had a slight sunrise peak instead. In 2019 at Nuquí (Fig. 9c), the highest number of files with humpback sounds (620) were detected at 18:00 hours in August. This is the same month as the peak at Morro Mico, but the opposite time of day. In summary, the diel cycle of singing of humpbacks (previously documented in Hawaii [Au et al. 2000] and Mexico [Seger 2016] as more active near midnight and less during the day) was more evident in Morro Mico in 2018 than at either site in 2019, but at dusk instead of midnight.

The number of boat sounds detected at Morro Mico in 2018 (Fig. 10a) filled 376 files and were most commonly detected at midnight. At Nuquí (Fig. 10c), there was a total of 1317 files with boat sounds and they were most common during the day. It is important to consider that, for the year 2019, in both Morro Mico and Nuquí, the recordings from the month of July are only from one day because deployment occurred on 31 July and recordings began at 12:00 hours, preventing any data from existing at midnight and 6 am for July.

## Discussion

This study provides qualitative and quantitative descriptions of soundscapes at two sites located in the Gulf of Tribugá. Both sites have relatively few contributions from noise generated by tourism and fishing boats. Currently, there are no published

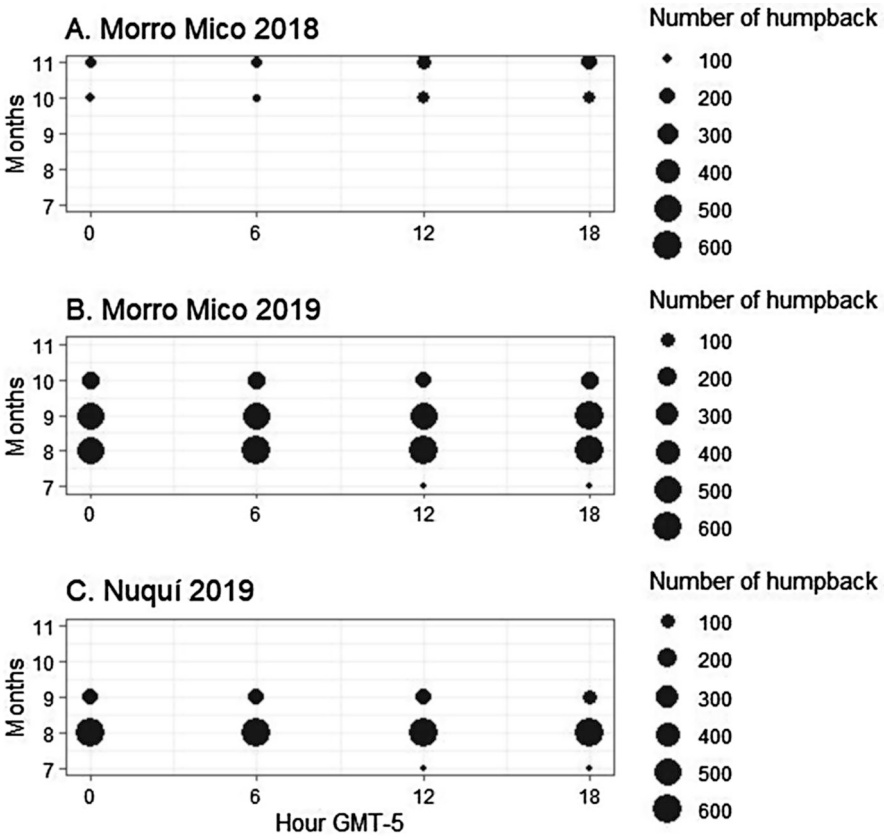



**Fig. 9** Number of files with humpback whale sounds (usually songs) detected during four time periods (00:00–00:10 and 00:30–00:40, 6:00–06:10, 12:00–12:10, and 18:00–18:10). The number of files of the intervals were added, plotted, and represented in the top hours for each of the months sampled by site. From top to bottom: Morro Mico 2018 (October and November), Morro Mico 2019 (July, August, September, and October), and Nuquí 2019 (July, August, and September)

studies about the Gulf of Tribugá's soundscape and the singing cycle of humpback whales that inhabit the Gulf during the boreal summer months. However, behavioral responses of humpback whales have been studied in the presence of whale-watching boats in Bahía Málaga in the Colombian Pacific (Avila et al. 2015), as well as in other South American countries such as Ecuador and Brazil (Scheidat et al. 2004; Sousa-Lima and Clark 2008, 2009). These previous studies are relevant to the current one since the whales that visit the Gulf of Tribugá may cross on the same feeding grounds near Antarctica.

A pattern was evident in Morro Mico's soundscape for both years sampled. Shrimp, fish, and humpback whale sounds were nearly equally dominant until November (2018) when the contribution of humpback whale song reduced, likely due to their migration to Antarctica commencing (Clapham and Mead 1999). The

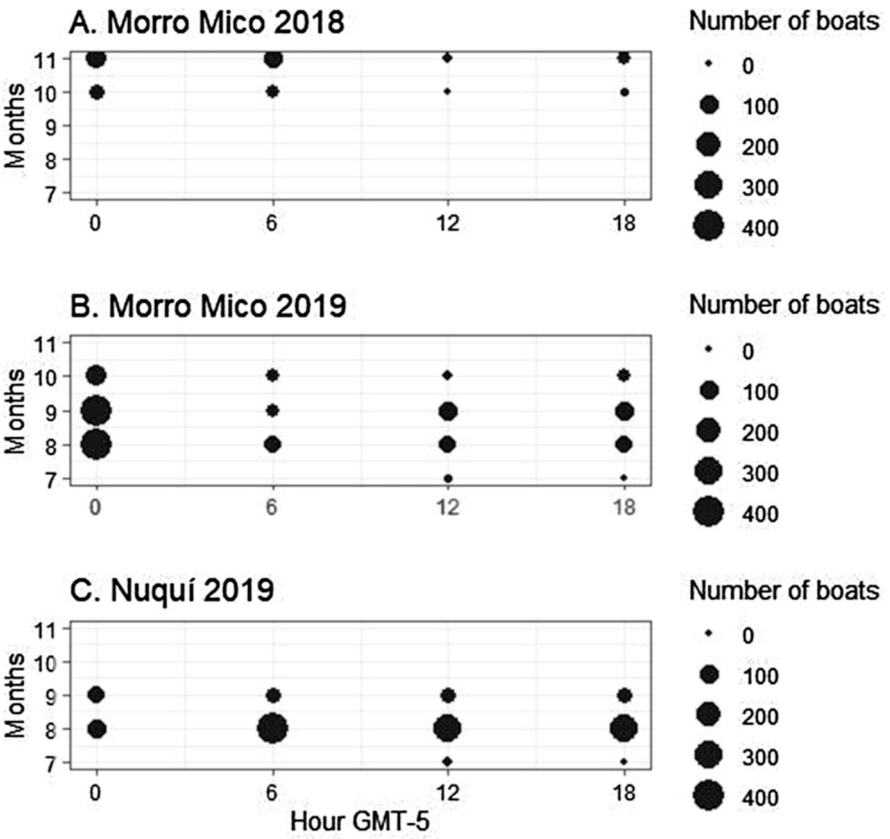



**Fig. 10** Number of files with boat sounds detected during four time periods (00:00–00:10, 06:00–06:10, 12:00–12:10, and 18:00–18:00) for each of the months sampled at (top) Morro Mico 2018 (October and November), (middle) Morro Mico 2019 (July, August, September, and October), and (bottom) Nuquí 2019 (July, August, and September)

peak humpback whale acoustic presence was in July, and then their vocal activity was relatively constant in the Gulf of Tribugá from August until sometime in October. Most previous photo ID research in the area that visually tracks humpback whale demographics has ended by October (Acevedo et al. 2017), so it was assumed that the whales also migrated en masse in October. But comparing October 2018 and 2019 to November 2018 at Morro Mico, it now appears that the mass migration south commences sometime in November.

The soundscape of Morro Mico in 2018 contained consistent snapping shrimp acoustic activity. The presence of shrimp can be a bioindicator for habitat quality and a way to monitor the conditions of the soundscape and the ecosystem. The sounds in a reef could reflect significant ecological changes (Watanabe et al. 2002). For example, the snapping activity of shrimp and the relationship with changes in temperature and dissolved oxygen concentrations have been studied in other

ecosystems (Watanabe et al. 2002). Because this project is the first acoustic study of the Gulf of Tribugá, it provides a baseline amount of activity, but not comparison to assert the relative health of Morro Mico and Nuquí ecosystems. However, if a port is built or if other infrastructure is built in the future, or as climate change continues, this study will serve the important purpose of being that which future studies can be compared against for relative health assessments of the ecosystem(s). Tracking physical variables of the ecosystem, too, like pH or temperature or the sound speed profile, would aid in tracking soundscape shifts into the future.

Finally, sounds of dolphins were not detected at either location, and possible explanations for this are that (1) the location of the hydrophone was not near where dolphins tended to be as they have been documented further south and away from the coast (Noreña 2020), or (2) the sampling rate of 15,625 Hz was too low to capture their whistles and clicks.

Nuquí in 2019 was distinguishable from Morro Mico by having a more variable soundscape. Although, in the absence of anthropogenic disturbances, it was quieter and more monotonous than Morro Mico. This is likely because in Nuquí the seabeds are mostly sedimentary but near Morro Mico they are rocky (Velandia and Díaz 2016), so there are no small coral communities for snapping shrimp and fish to live near Nuquí and contribute to its soundscape. Furthermore, sound attenuation should be greater around Nuquí because sediments with lower porosity and smaller particles (like mud and clay) create greater attenuation (absorb sound waves more) than sediments with higher porosity and larger particles (like rocks and gravel) (Buckingham 1997).

The cycles of humpback whale singing were different between Nuqui and Morro Mico, which was a key interesting discovery in this project. There are a number of reasons why this could be the case. In 2019, samples with whales' songs peaked during the mornings in Morro Mico but in the evenings in Nuquí. This may be an indication that the whales are shifting locations up and down the coast to sing in different places during the night and day, where further north in the Gulf is a better singing location in the morning, but further south is a better singing location in the evening. This is not impossible as whales are mobile animals and can easily cover the distance between Morro Mico and Nuquí in a few hours. If song is a territorial display, though (Seger et al. 2016), and the singers are not moving along the coast to sing in various locations, but choosing to sing at the same location day after day, males within recording range of Morro Mico may prefer to sing more in the mornings than males within recording range of Nuquí. Finally, there could be a binary onshore/offshore trend like documented in Munger et al. (2012) where all singers near Morro Mico are doing so closer inshore in the mornings but singing closer inshore to Nuquí in the evenings. Since there is an uptick in vessel traffic at Morro Mico in the middle of the night and at Nuquí in the middle of the day, singers in each area could be shifting to sing near shore during the relatively lower boat traffic hours at each location. Whether different diel cycles exist at each site (16.9 km apart), or the singing whales shift between the two locations on a daily schedule cannot be determined from single recorders, but would be an interesting future project if arrays or vector sensors can be deployed.

Another possible explanation in these different peak times of singing activity at each recording location is that singer distribution and activity is driven by the locations of other humpback whales and not from an individual singer's decision-making process. Nuquí could be a location where the mothers and calves go at night once the daily peak in boat traffic has slowed down, and they return to Morro Mico in the mornings to avoid the increase in vessel traffic coming out from the largest village of the region. If the males are using song to attract these females, their singing activity may follow the daily routes that mothers take with their calves. However, this would be counter to the theory of "aural camouflage" described by Seger et al. (2015) whereby grey whale mothers with calves seek out areas with high snapping shrimp activity to use as a masking device to be able to communicate at close ranges with their calves but not be detected by males in the immediate vicinity. If mothers and calves preferred areas with lots of snapping shrimp sounds, and males sang in areas close to adult females, song would be concentrated only in Morro Mico and not in Nuquí. A third explanation could be based on the observation that males stop singing when they join with other males (Darling et al. 2006; Tyack 1981) and start singing when they disaffiliate into singletons (Kobayashi et al. 2021). If Nuquí is at the center of the Gulf, then there may be more male whales joining and stopping to sing each morning, causing a dip in singing activity after sunrise but an increase in the evening near Morro Mico when they spread out again to sing. These possible explanations all make assumptions about singing trends that cannot be corroborated with visual surveys because previous work in the Gulf of Tribugá has not tracked movement patterns overnight, has not investigated whether there are temporal patterns of resightings of animals between the two monitoring locations of this research, and has not expanded past the Gulf's borders to determine whether the humpback whale song may have a broader trend that cannot be captured with just two recording stations. Further research, including a count of songs within each time frame, would be required to distinguish between the presence/absence of singing activity in the files.

Overall, the sounds that substantially dominated the Nuquí soundscape's noise budget were those of humpback whales, fish, and boats. Humpback whale songs were detected more often at Nuquí than at Morro Mico and represented a larger percentage of the noise budget. The lack of snapping shrimp at Nuquí is likely what leaves room in the noise budget for humpback whales to occupy a larger percentage of it. A possible explanation for the greater detection of songs in Nuquí is that, in the absence of boats, the acoustic environment was quieter, so a whale song would be easier to detect if there wasn't snapping shrimp to mask its reception like at Morro Mico. Even though song from whales and noise from boats would both be attenuated by the muddy sediments, the higher attenuation of sound at Nuquí compared to Morro Mico might also reduce the amount of noise from far away boats and whales reaching the EAR, making it easier to detect whales within detection range when there were no boats within detection range.

The coincidence that a larger portion of the Nuquí noise budget being humpback whale song implies that the area with the highest humpback whale vocal activity detected in the Gulf of Tribugá is also the area with the highest boat activity. Thus,

high human activity appears to be overlapping with a critical singing area for the whales. The effect of this might be seen in the different diel cycles of humpback whale song at each site. This analysis found that the diel singing cycle of humpback whales was more evident at Morro Mico in 2018 than at Nuquí or Morro Mico in 2019 but was backward since the peak did not occur at midnight like in other locations around the Pacific Ocean–Hawaii and Mexico (Au et al. 2000; Seger 2016). Instead, Morro Mico in 2018 had more singing activity at 12:00 and 18:00 hours than at 00:00 and 06:00 hours. Some authors suggest that vision in daylight plays a key role in the formation of competitive groups, but at night males can focus effort on vocal activity as the main mating strategy (Au et al. 2000; Helweg and Herman 1994). If true, high boat activity during peak singing hours could cause the humpback whales to either relocate to an area with less boat traffic, or to shift their diel singing cycle to peak when there is less disturbance. The shift to more singing in the daylight hours at Morro Mico in 2018 exemplifies this latter possibility as it does coincide with a potential avoidance of the higher nighttime boat traffic at Morro Mico. However, this trend did not hold true in 2019 as Morro Mico still had the most boat traffic at 00:00 hours, but there was no clear diel singing cycle from the whales. The closest thing to a peak in singing activity that occurred at Morro Mico in 2019 was that a large number of files with song present in them occurred at 06:00 hours, but only in August. The lack of a midnight-peaking singing cycle is interesting given that the midnight peak in boat activity is obvious.

Nuquí in 2019 similarly showed no clear diel singing cycle even though boat activity there did drop overnight. The most acoustic files containing singing activity was centered at 18:00 hours in August, again suggesting that even some boat noise at night might be too much for humpback whales to maintain a midnight-peaking singing cycle at Nuquí. According to Seger (2016), it was observed that their site of study with the most boat activity did not contain a diel singing cycle, either, so humpback whales may have a threshold of masking from boats that causes them to adjust their singing cycle accordingly or to vacate an area with too much disturbance. A final explanation could be that humpback whales from the Southern Hemisphere, or at least from Stock G, have a naturally different diel singing cycle than those from the Northern Hemisphere that breed in Hawaii and México.

The noise generated by boats was higher at Nuquí than in Morro Mico both in terms of percent of the noise budget and in absolute counts of files with boat noise present. In Caribbean marine parks, it was found that, the lowest percentages of time that vessel noise existed in their soundscapes were between 20% and 30% but did sometimes reach up to 99% (Heenehan et al. 2019). The most that boat noise was detected at Nuquí in 2019 (the highest for this study) was 16% of the time, which is still considerably lower than that documented in some marine parks in the Caribbean area. Therefore, the Gulf of Tribugá can be characterized as an area with relatively low levels of noise pollution compared to the Colombian Caribbean coast, which in some areas has operations that produce intense underwater noise through seismic testing, platform construction, and the constant drilling and removal of the land (Vásquez 2019). This emphasizes the claim that Tribugá is still a "relatively pristine" environment as compared to the current status of vessel traffic in other areas of the

world today. To the best of current knowledge, no scientific investigation has compared soundscapes between Colombia's Pacific and Caribbean coast waters.

The Gulf of Tribugá is part of the administrative area of Chocó, which is one of the places in the world with highest rainfall and associated winds (Velandia and Díaz 2016). This is evident in the monthly precipitation reports for the year 2019 by the Institute of Hydrology, Meteorology, and Environmental Studies (IDEAM- acronym in Spanish) (Instituto de Hidrología, Meteorología y Estudíos Ambientales (IDEAM) 2019). Because of the rain, the communication space for marine animals may be masked by natural hydrological and geophysical factors. The results of this research showed there were more acoustic files with rain sounds during 2019 than 2018. According to the IDEAM monthly report, for the months of August, September, and October of 2019, the monthly amount of precipitation was above that observed for the same months in 2018 (Instituto de Hidrología, Meteorología y Estudios Ambientales (IDEAM) 2019). The sound of rain occupied the entire frequency spectrum when present, sometimes intensely enough to mask sounds even as intense as humpback whale song. Also, it is known that wind is an important source of contribution to soundscapes (Wenz 1962) as it makes the surface of the ocean choppier, producing more bubbles, and causing their cavitation sounds to radiate downward with cosine directionality. Waves also have a sharp pressure pulse when they impact the surface, followed by the resonance of the gas bubbles generated by the impact (Hildebrand 2009) contributing to the overall geophony.

Within the data there were many instances of biological sounds and boat sounds that occurred at the same times, illustrating temporal, spatial, and spectral superpositions and causing masking effects in the communication not only of marine mammals (Clark et al. 2009; Cholewiak et al. 2018; Erbe et al. 2016, 2019), but also of fish (Smott et al. 2018). This could generate combined masking effects by the rain on the ocean surface and the sound of the boats, potentially masking biologically relevant signals by two sound sources at once (Montgomery and Radford 2017). Since boat sound decreased at night, the combined masking effect could be more intensely present for longer periods of time in the daylight hours, with the effects of wind and/or rain only persisting through the nighttime hours. This would mean the reduction of the communication space of singing humpback whales would vary throughout a 24-h period.

Within the datasets of this paper, fish sounds were consistently present at both sites and years. Even though neither the fish nor the whales notably stopped calling/singing completely at any time in the datasets, possible impacts such as increase in stress hormones from receiving high noise levels could affect vital functions and change communication functionality, as has been found in right whales (Rolland et al. 2012). Future work investigating stress levels of whales during days and times with lower or higher sound levels close to the Morro Mico or Nuquí sites would help answer this question. Also, more in-depth studies about specific acoustic reactions of fish and whales when boats are nearby compared to when they are not would be useful to understand how much and in what ways animals are responding to fluctuations in background noise levels in their soundscapes.

To that end, Rey-Baquero et al. (2021) used a subset of these data to determine whether humpback songs changed after a boat pass compared to the song structure and rhythm before the boat passed. They found that song structure changed roughly 50% of the time during a boat pass, so was not statistically significant in terms of a reaction to boat noise, but the song units and space between song units had a statistically significant difference after a boat passed as compared to before. Also, Brown et al. (▶ Chap. 76, "Frequencies of Humpback Whale Song Units (*Megaptera novaeangliae*): Adjustments to Small Boat Noise") tested whether humpback whales in the Gulf of Tribugá changed the frequency content of their songs during and after boat noise as compared to before it was present. They found that out of 38 comparisons, song frequencies significantly changed nine times. Similar studies on frequency content would be important to provide a more holistic understanding of humpback whale singing reactions to boat noise.

More boats or larger vessels in the Gulf of Tribugá, especially if an international port were to be built, would increase background noise levels in the soundscape. In fact, Huertas-Amaya et al. (▶ Chap. 77, "Acoustic Models to Consider Noise Pollution from a Possible Port on Humpback Whales in the Gulf of Tribugá, Colombia") quantified how much more area of the waters off Colombia could be ensonified by new shipping lanes. Noise produced by large vessels has been found to impact acoustic-based interactions for whales, fish, and other marine animals, especially in protected areas or areas of special biological interest such as sanctuaries and marine parks (Hatch et al. 2008; Stanley et al. 2017). Also, disturbances and impacts from smaller vessels, such as fishing and whale-watching boats, have been reported because they contribute to masking (Rey-Baquero et al. 2021). Thus, with both large and small vessels, the effect of heightened background noise would be cumulative (Cholewiak et al. 2018). Vessels have been found to have the ability to reduce the communication space for fish and whales, even though there is natural variability in background sound (Putland et al. 2018; Rey-Baquero et al. 2021).

The impacts of tourism whale-watching boats have been studied in different marine mammals and places around the world (Parsons 2012). Some of the changes studied in humpbacks are acoustical (Erbe et al. 2019) while others are behavioral (Avila et al. 2015; Sousa-Lima and Clark 2009; Schuler et al. 2019). In feeding areas, it has been found that the noise of the boats can affect foraging, causing impacts at the individual and population levels (Blair et al. 2016). Although the effect of boats on the humpbacks in Tribugá is unknown, the impacts could be similar to other places, affecting ecological, reproduction, and breeding processes. In addition, whale-watching of marine mammals has gained great economic importance, especially in the interior of the Gulf of Tribugá (Velandia and Díaz 2016). If the industry grows, the masking effect could increase due to the number of boats and frequency of tours, so responsible whale-watching practices should include acoustic considerations in their guidance. This is considered a relatively easy lift in the area since boat operators already receive training that they largely abide by, so adding acoustical considerations would also likely receive high compliance.

## Conclusions and Perspectives

Both the study years and sites were dominated by biophony, arising from sound sources such as the songs and social calls of humpback whales, and sounds of shrimp and fish. Overall, this study has shown that the Gulf of Tribugá contains relatively low presence of boat noise/acoustic pollution as compared to other documented areas of the oceans. Monitoring is recommended to document any increase of human noise disturbances, especially in sites such as Nuquí which present greater potential for more contributions from sounds of boats as the population of Chocó grows. Interestingly, the humpback singing cycle peaks did not follow the midnight peaks of other Pacific breeding grounds, which could be attributed to: (a) the coincident peak boat activity that would make it harder to detect song even when peaking; (b) a behavioral shift toward the time of day when there were fewer boats, thus illustrating a possible adaptation to the human noise diel cycle; or (c) a possibility that Stock G whales from the Southern Hemisphere may naturally sing on a different diel cycle than their Northern Hemisphere counterparts.

Communicating the results of this study could help in regulating noise levels and keeping them to a minimum as infrastructure grows along the Pacific Coast of Colombia. A key recommendation would be to avoid an increase in boat traffic at peak times of the humpback song cycle – a time that would depend on the site. For future studies, it is recommended to evaluate the effects of anthropophony on the soundscape, using other metrics such as percentiles in decidecade or one-third octave bands (Wang et al. 2019). Including percentiles or analyzing more hours could better represent the soundscape components and more easily identify the intensity contributions of each group and help determine where and when the humpback whales are singing most of the Gulf of Tribugá. Finally, continuing with passive acoustic monitoring allows management authorities to prioritize protection of conservation areas before possible disturbances due to human activities, especially when any proposals for the construction of a mega maritime port or other large infrastructure project exists.

Future studies should extend the acoustic sampling season to times when humpback whales are in their feeding grounds (November–May). Visual observations from local fishermen and tourist boats have recently documented pilot whales and killer whales in the Gulf of Tribugá, so it would be interesting to determine if sounds from these animals become more prevalent, or are at least more detectable, when humpback whales are not around. Sampling rates of the recorders would need to increase to capture the delphinid contribution to the soundscape, though. Future work would benefit by having more than one recorder at a single site so that locations of boats and all sound sources can be determined.

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# Assessment of the St. Lawrence Estuary Soundscape

36

Jeanne Mérindol, Pierre Cauchy, Guillaume St-Onge, and Cédric Gervaise

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**Abstract**

The St. Lawrence Estuary is renowned for the abundance and diversity of marine mammals that come to feed in summer or live all year round in its waters. It is also a major seaway linking the Great Lakes to the Atlantic Ocean. Due to the high maritime traffic in this seaway, negative impacts on marine mammals are an issue to be studied. In the current context of sustainable development and protection of marine species at risk in Canada, such as the beluga whale (*Delphineptarus leucas*) and blue whale (*Balaenoptera musculus*), it is essential to determine the noise levels to which these species are exposed. In this chapter, the underwater soundscape recorded from August to October 2021 by the Marine Acoustic Research Station (MARS) – consisting of four vertical arrays of three hydrophones deployed in the middle of the Laurentian Channel – is analyzed. In addition to recording the ship's signatures, they continuously acquire ambient noise created by all traffic and cetacean emissions to provide a complete picture of the acoustic context of the St. Lawrence Estuary. An analysis of the relative contribution of wind and traffic noises to ambient noise at three different 1/3-octave bands has been achieved. At low frequencies (50 Hz), used by blue whales' long-range communication signals, traffic noise dominates the soundscape. At mid-frequencies (300 Hz), used by humpback whales' short-range signals, a mixed regime is observed. At high frequencies (6300 Hz), used by belugas' whistles, wind noise dominates the soundscape.

**Keywords**

Acoustics · Ambient noise · Canada · Cetaceans · Noise exposure · Quebec · Ships · Soundscape · St. Lawrence

## Introduction

### Whales and Shipping Lane in the St. Lawrence

The importance of the Estuary and Gulf of St. Lawrence for the abundance and diversity of marine mammals has long been recognized. The abundance of marine mammals in the Gulf was first reported by Jacques Cartier during his voyage to the New World in 1534. The diversity of marine mammals in the St. Lawrence Estuary is composed of two orders: cetaceans (Mysticetes and Odontocetes) and pinnipeds (Odobenidae and Phocidae). Up to 16 species of cetaceans and pinnipeds may visit the Estuary in any given season. The region boasts resident populations of beluga and harbor seals (*Phoca vitulina*), as well as migratory populations of minke whales (*Balaenoptera acutorostrata*), humpback whales (*Megaptera novaeangliae*), fin

whales (*Balaenoptera physalus*), and blue whales (*Balaenoptera musculus*), which use the Estuary as a summer feeding ground due to its high concentrations of forage species. These species frequent the Estuary and contribute to its underwater soundscape (Kingsley and Reeves 1998; Edds et al. 1987). The St. Lawrence system is also a major shipping lane linking the Atlantic Ocean to the Great Lakes which has effects on local marine life, including exposure to underwater noise (Chion et al. 2017; Gervaise et al. 2015; Gervaise et al. 2012) and collisions with marine mammals (Krause 2008).

## Effects of Noise on Whales

In the current context of sustainable development and the protection of species at risk in Canada, such as the North Atlantic right whale (*Eubalaena glacialis*) (Gervaise et al. 2021; Meyer-Gutbrod et al. 2018; Brillant et al. 2017; Elvin and Taggart 2008), blue whale, fin whales, and the beluga whale (*Delphineptarus leucas*) (Lesage 2021; Vergara et al. 2021; Fisheries and Oceans Canada 2020; Bailleul et al. 2012a, b; Department of Fisheries and Oceans 2012), there is a need for research that investigates the effects of noise on marine mammals.

## MARS Project

The MARS project (http://www.projet-mars.ca) is an applied research project that aims to understand and measure the underwater noise radiated by ships to propose relevant methods for its reduction, in synergy with Canadian shipowners. The MARS project scientific program is based on the development and deployment of world-class instrumentation in the St. Lawrence Estuary near Rimouski (Fig. 1). An underwater acoustic recording station for measuring the acoustic signature of ships operating on the St. Lawrence, hereafter called "the station" (Fig. 2), was deployed in the middle of the Laurentian Channel from August to November 2021. It is a cabled instrument that records continuously for a frequency between 5 Hz and 50 kHz. Data is sent via the surface buoy, which establishes a connection with Rimouski via the cell phone network. In addition to measuring the acoustic signature of ships, this data also enable observation of the local soundscape.

## Sound Levels Recorded by the MARS Station

The underwater soundscape is very rich with sounds generally classified into three categories: biophony, geophony, and anthropophony (Krause 2008). The objectives of this chapter are to quantify the sound levels in the St. Lawrence recorded by the MARS station during the 3 months of the operating period and to identify the main sources that generate them. To achieve these objectives, the ambient noise will be first analyzed, and the sound levels measured at the station will be quantified by frequency band. Then, the sources responsible for these sound levels (geophony or anthropophony) will be identified and the environmental parameters best describing

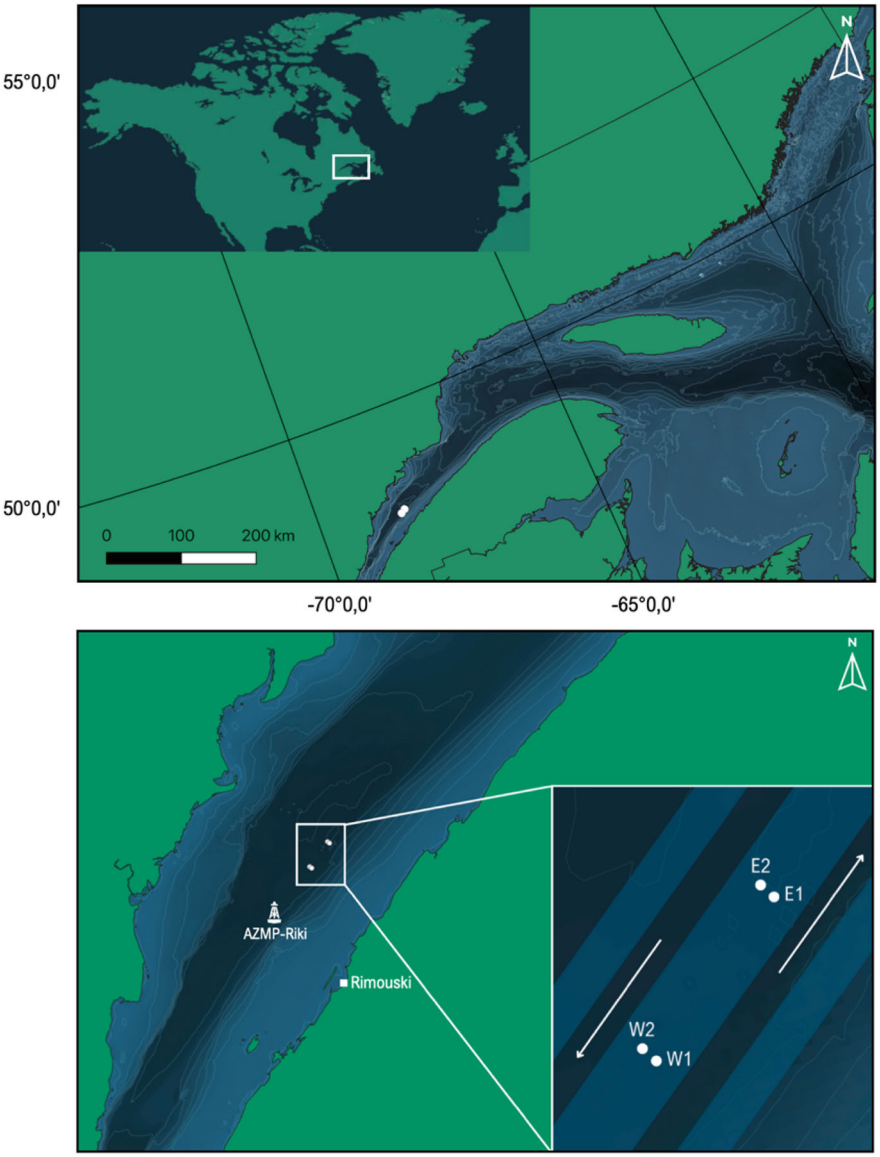



**Fig. 1** Implementation of the MARS acoustic station in the center of the traffic separation lane of the St. Lawrence seaway, offshore Rimouski. *Up:* the white points are the location of the hydrophones. *Below:* the white arrows illustrate trajectories of ships. AZMP-Riki is an oceanographic and meteorological buoy designed by Multi-Electronics (MTE) and operated by Fisheries and Oceans Canada for the Atlantic Zone Monitoring Program (AZMP), located 9 km away from the station (Pêches et Océans Canada – Fisheries and Oceans Canada 2022). The bathymetric datasets come from NASA.

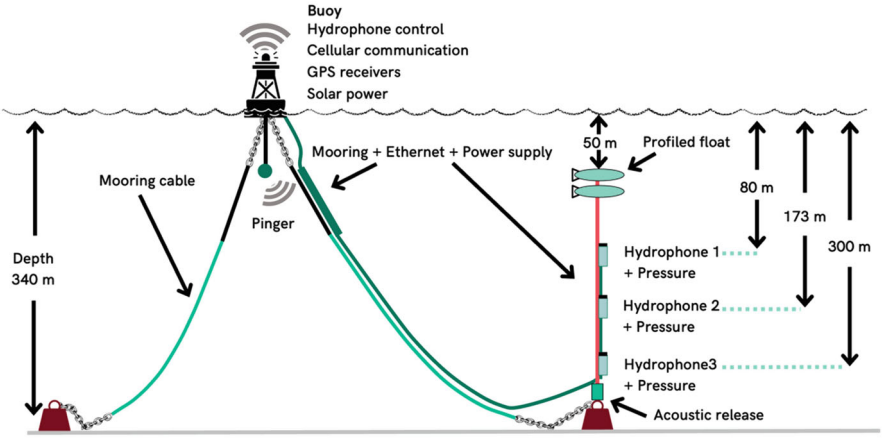



**Fig. 2** Geometry of a MARS station antenna

them will be defined: surface wind speed for geophony and distance to the nearest ship for anthropophony. The sound levels will be described in the three chosen 1/3-octave bands (50 Hz, 300 Hz, 6300 Hz), representative of the wide bandwidth covered by cetacean sounds.

## Materials and Methods

### AIS Data

Traffic data were collected using an Automatic Identification System (AIS) receiver, installed on shore at 20 km from the station, allowing detection of ships within 100 km. Time series of longitude, latitude, and speed over ground have been collected as well as other relevant information such as length, width, draft, type, and age of each vessel. A speed criterion was applied to remove vessels with a speed less than 2 knots to avoid bias caused by vessels stopped in nearby harbors. From the position of each vessel, the distance between the acoustic station and the nearest vessel was calculated with a time resolution of 5 min. To focus on traffic noise only, data collected when wind noise is not prevalent (wind speed $<5$ m.s$^{-1}$) was considered for analysis.

### Wind Speed Data

Wind speed was recorded by the AZMP-RIKI oceanographic and meteorological buoy (Fig. 1) designed by Multi-Electronics (MTE) and operated by Fisheries and Oceans Canada for the Atlantic Zone Monitoring Program (AZMP) located 9 km away from the station (Pêches et Océans Canada – Fisheries and Oceans Canada 2022). From August to October 2021, time series of the 60-s average surface wind

speed was extracted with a time resolution of 30 min. From the wind speed recorded by the PMZA-RIKI buoy, a 120-min moving average was applied. To focus on wind-generated noise only, data collected in the absence of nearby ship, when the nearest vessel was at least 30 km away from the station, was considered for analysis (Fig. 5b).

## Acoustic Recordings

The acoustic station has been designed to be ANSI/ASA S12/64-2009 compliant (ANSI/ASA 2009). It consists of four moorings (E1, E2, W1, W2) forming two gates spaced eight kilometers apart anchored between the two shipping lanes of the St. Lawrence, off Rimouski at approximately 48°N and 68°W (Fig. 1). It records continuously during its operating period. Each mooring consists of a surface buoy and a vertical array of three acoustic recorders (Fig. 2). The buoy provides energy to the recorders and communication with the shore. The acoustic recorders are calibrated and placed at 80 m, 173 m, and 300 m, respectively, along a vertical array. The line is held vertically by floats. Floats and cables are profiled to reduce current-induced vibrations. The moorings themselves generated underwater noise. This noise takes the form of periodic knocking and oscillations, and it is more intense during periods of high current. The proportion of noiseless measurements varied between 18% (antenna W1, bottom hydrophone) and 68% (antenna E1, bottom hydrophone). The E1 mooring had noise-free measurement proportions of 52%, 62%, and 68% for the three hydrophones from top to bottom. In this chapter only the data collected by the bottom (300 m) hydrophone of E1 mooring was analyzed. It is composed of an AURAL M3 recorder developed by MTE-OpDAQ comprising a Geospectrum hydrophone with a sensitivity of $-174.5 \pm 3$ dB re 1 V/μPa for a frequency between 5 Hz and 50 kHz, a gain of 18 dB, and an acquisition scale $\pm 2.5$ V, 24 bit. The raw acoustic data files from all four moorings were transmitted once per week through the cellular network. The complete acoustic dataset was also accessed after recovery of the system at the end of the operating period. It is composed of 20,308 5-min .wav files, sampled at 16 kHz, representing ~1692 h (~70 days) of recordings and 326.22 GB. For this chapter, noise levels in third octave bands (Third Octave Level (TOL), dB re 1 μPa) were extracted using the PAMGuide software (Merchant et al. 2015), using a 1-s Hann windows and 50% overlap. The minimum TOL value on a 5-minute sliding window was extracted to provide a smoothed time series of the background noise level. Central frequencies of the three 1/3-octave bands are, respectively, 50 Hz, 316 Hz, and 6309 Hz. Spectrograms were generated using Matlab (0.5 s Hann window, 50% overlap) to allow visualization of the dataset.

## Focus on Three Third Octave Bands

For this chapter, the analyses focused on three 1/3-octave bands. The low frequencies (50 Hz) are representative of the frequencies used by the large baleen whales: the blue whale and the fin whale. The medium frequencies (300 Hz) are representative of the frequencies used by the other baleen whales: the humpback whale, the

North Atlantic right whale, and the minke whale (*Balaenoptera acutorostrata*). And the high frequencies are representative of the frequencies used by the toothed whales: sperm whale (*Physeter macrocephalus),* northern bottlenose whale (*Hyperoodon ampullatus*), killer whale (*Orcinus orca*), Atlantic pilot whale (*Globicephala melas*), beluga whale, Atlantic white-sided dolphin (*Lagenorhynchus acutus*), white-beaked dolphin (*Lagenorhynchus albirostris*), and harbor porpoise (*Phocoena phocoena*), and seals: harbor seal (*Phoca vitulina*), harp seal (*Pagophilus groenlandicus*), and gray seal (*Halichoerus grypus*). The construction of scatter plots of sound pressure level (SPL) in dB re 1 µPa versus wind speed, distance from the nearest vessel and SPL versus wind speed for vessels located more than 30 km from the station were performed to describe the recorded noise levels for these three 1/3-octave bands in relation to these covariates.

## Results

### Sound Pressure Levels

The median and percentile distributions of the noise levels recorded by the station were calculated (Fig. 3). A ~10 dB re 1 µPa dispersion between the SPL values of the 25th and 75th percentiles of the third-octave band 300 Hz and 6300 Hz was observed, which is consistent with previous observations in the area (Simard et al. 2010). The typical spectral slope of wind noise is visible at all percentiles of the distribution in the high frequencies, from ~500 Hz (Fig. 3), and the median level was

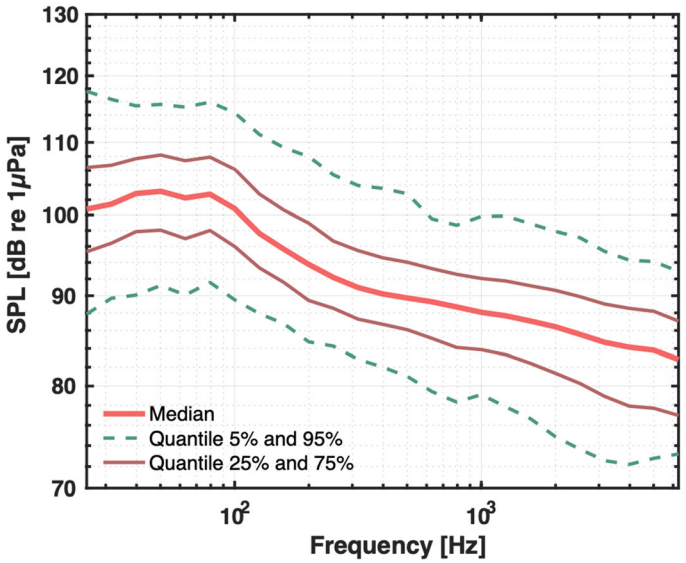



**Fig. 3** Percentiles of noise levels recordings by the station from August to October 2021 with the median received noise level in the 25 Hz–6 kHz band (black dotted line)

82.8 dB re 1 μPa in the third-octave band 6300 Hz. The typical signature of maritime traffic noise centered at 50–100 Hz (Wenz 1962) is visible on all percentiles of the distribution, and the median level, 103.1 dB re 1μPa, in the third-octave band 50 Hz corresponds to the values proposed for heavy traffic noise by the Wenz curves (Wenz 1962) (Fig. 3). A strong contribution of traffic noise to our observations is to be expected, the station being positioned between two shipping lanes, at 2 km from each, with a daily traffic in the Lower St. Lawrence Estuary during the 3-month period of about 22 ± 3 ships per day in both directions (Gervaise et al. 2012).

## Wind Noise

The correlation between the time series of noise level and wind speed − calculated with the *corr* Matlab function (MATLAB Version: 9.11.0.1809720 (R2021b) Update) − shows three frequency ranges where the relationship between TOL and wind speed appears to differ: 10–100 Hz, 100 Hz–1000 Hz, and the frequencies >1000 Hz (Fig. 4). Noise levels at 50 Hz, 300 Hz, and 6300 Hz were analyzed with regard to surface wind speed (Fig. 5a). At low frequencies, no tendency is noticed, and the dispersion of ~25 dB re 1 μPa is coherent with the dispersion of the ship source level observed by Simard et al. 2016, in the area. The tendency is partial at mid-frequencies and the pattern observed is more complex: at reduced wind speed (<5 $m.s^{-1}$), noise level is dominated by traffic noise (~25 dB re 1 μPa spread and no trend); at high wind speed (>5 $m.s^{-1}$), noise level is clearly influenced by wind speed (Fig. 5a). At high frequencies, wind-generated noise dominates the soundscape, therefore, noise level increases noticeably with wind speed as previously observed (Cauchy et al. 2018; Mathias et al. 2016; Vagle et al. 1990; Wenz 1962) and the observed tendency (Fig. 5a) suggests that at 50 Hz the wind noise is weak and is masked by anthropophony. While at 300 Hz, wind noise is masked by anthropophony for wind <5 $m.s^{-1}$ and wind-generated noise dominates the soundscape at 6300 Hz.

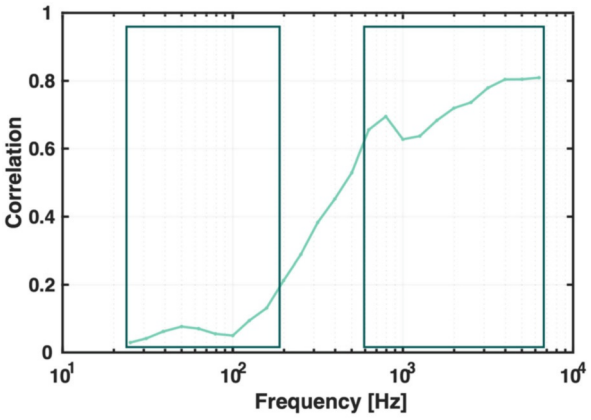



**Fig. 4** Correlation between the wind speed and observed noise levels recordings by the station from August to October 2021 based on 1/3rd octave band

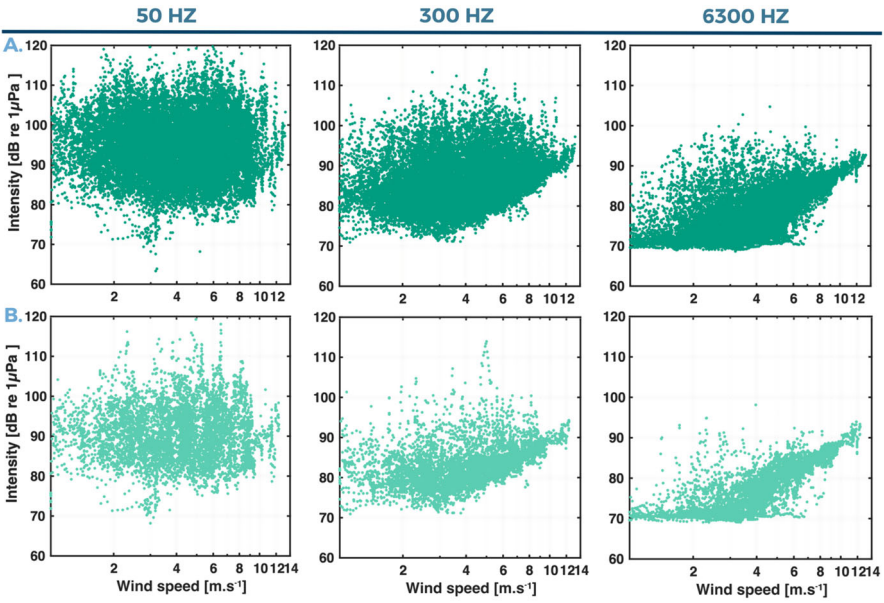



**Fig. 5** Distribution of noise levels at 1/3-octave bands 50 Hz, 300 Hz, and 6300 Hz as a function of wind speed recorded by AZMP-RIKI (**a**) complete dataset, (**b**) in the absence of ships in 30 km around the station (central frequencies of the three 1/3-octave bands are, respectively, 50 Hz, 316 Hz, and 6309 Hz)

The influence of the wind in the low frequencies is still not visible with the cleaning data, noise level likely being dominated by distant traffic, as stated by Wenz (1962). At medium frequencies, the relationship between wind speed and noise level becomes clearer: noise generated by local traffic is removed and the dependency of noise level with high wind speed (>5 $m.s^{-1}$) is highlighted. Finally, the dependency of high-frequency noise level with wind speed stays clear.

## Traffic Noise

Noise levels at 50 Hz, 300 Hz, and 6300 Hz were analyzed with regard to the distance of nearest ship to the station (Fig. 6). The noticeable spreading of the noise levels, of ~25 dB re 1 µPa, is coherent with the dispersion of ship source level observed by Simard et al. (2016) in the area. Despite a low tendency, distance to the nearest vessel seems to partially explain the SPLs considering the logarithmic decrease in noise levels with distance from the nearest vessel to the station in the three frequency bands. In fact, as the ship moves further away, noise levels decrease up to a certain limit, different for the three frequencies considered (Fig. 6). Therefore, other parameters must be considered such as the high variability of emitted noise (Simard et al. 2016) and the number of ships within a given radius (Magnier and Gervaise 2020).

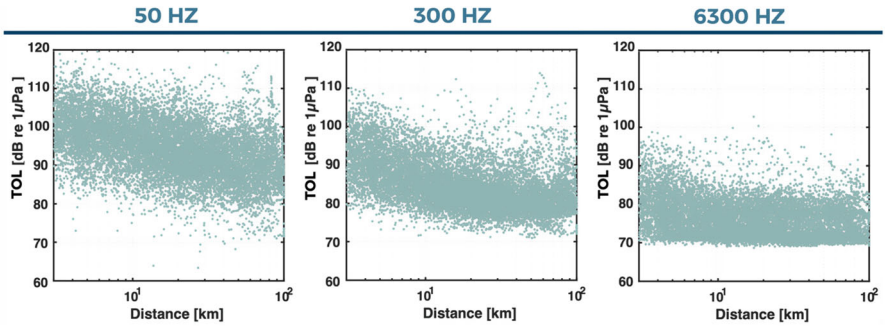



**Fig. 6** Distribution of noise levels at 1/3-octave bands 50 Hz, 300 Hz, and 6300 Hz as a function of distance of the nearest vessel from the station in wind speed $<5$ m.s$^{-1}$ conditions

## Discussion

### Distant Maritime Traffic Noise Dominant Above 200 Hz

Noise from distant traffic dominates the observed soundscape up to 200 Hz in all weather conditions. It is more complicated at mid- and high frequencies because the wind sometimes dominates. Further analysis will allow estimation of vessel's acoustic footprint, as the range over which vessel-generated noise exceeds the natural ambient noise (wind-generated), and the noise generated by ship category.

### Wind Noise Dominant Above 500 Hz

Co-located observations of wind speed and underwater noise level show a clear relationship above 500 Hz. Further analysis could consider wind direction to account for effects of a developed sea state, a fully developed sea being only possible at the station location under NE or SW winds, along the direction of the St. Lawrence Estuary. Tidal currents should also be considered to analyze their influence on sound levels. Access to similar recordings from the MARS hydrophones deployed at 80 and 170 m depths will provide valuable information regarding the depth dependency of wind-generated noise (Cauchy et al. 2018; Vagle et al. 1990).

### What Noise Levels Are St. Lawrence Whales Exposed to?

Up to 16 marine mammal's species use the St. Lawrence throughout the year (McComb-Turbitt et al. 2021; Druskat et al. 2019; Simard et al. 2019; Ramp et al. 2015; Teloni et al. 2008; Richard et al. 1990). The diversity of marine mammals in the St. Lawrence Estuary is composed of migratory populations of humpback whales, minke whales, blue whales, and fin whales, which use the estuary as a

summer feeding ground due to its high concentrations of forage species (Edds et al. 1987; Kingsley and Reeves 1998), and St. Lawrence resident populations of beluga whales and harbor seals (*Phoca vitulina*), which use the estuary as a summer feeding ground due to its high concentrations of forage species. In fact, this is the region with the highest krill aggregation documented to date in the Northwest Atlantic (Simard 2009). Six species have been identified in this chapter: blue whales (18–90 Hz), fin whales (15–100 Hz), humpback whales (100–710 Hz), minke whales (50–2000 Hz), sperm whales (1–30 kHz), and beluga whales (2–150 kHz) (Fig. 7).

The MARS station is important for two types of analysis. First, a long-term acoustic observatory such as this can provide means of monitoring animal populations, i.e., their seasonal occurrence patterns, migration timing, and relative abundance. And secondly, given the wide range of frequencies used by whales (15 Hz–150 kHz), it is ecologically important to know what levels of noise they are exposed to. Impacts of underwater noise on whales such as belugas include behavioral changes affecting foraging efficiency, such as avoidance and disturbance (Gomez et al. 2016), changes in vocalization behavior (Lesage et al. 1999), communication masking affecting the animal's ability to socialize and locate prey (Erbe et al. 2016), and hearing loss (Finneran et al. 2002; Schlundt et al. 2000).

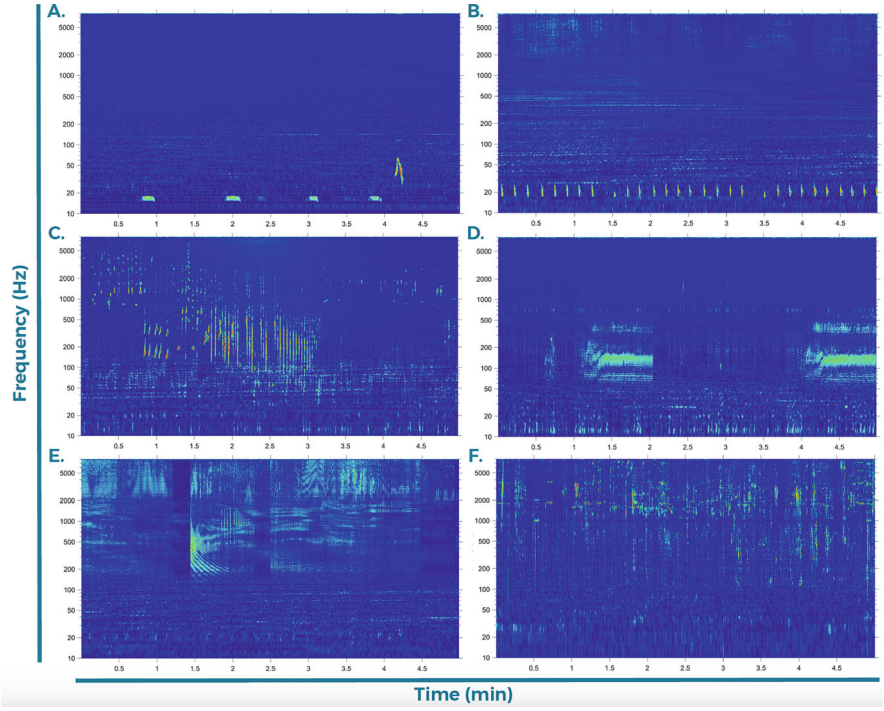



**Fig. 7** Spectrograms of cetacean sounds detected by the MARS station. (**a**) Blue whale, (**b**) Fin whale, (**c**) Humpback whale, (**d**) Minke whale, (**e**) Sperm whale, and (**f**) Beluga whale

Most of these studies have used behavioral attributes such as changes in site frequentation, dive patterns, swimming speed, movement orientation, herd cohesion, and dive synchronization to indicate possible disturbance or stress caused by marine traffic (Richardson et al. 2013). Numerous studies have examined the effects of high levels of underwater noise on the vocal behavior and hearing ability of marine mammals (Southall et al. 2008, 2019; National Marine Fisheries Service 2018; Lesage et al. 1999) and consensus has emerged; while less consensus has been established on behaviors and masking (Erbe et al. 2022). Given that marine mammals rely on the acoustic sensory channel for many of their activities, it is even more important to improving our knowledge of their tolerance thresholds and listening range, to target the frequencies at which anthropogenic noise needs to be attenuated to limit the effects on their functions that depend on underwater ambient noise levels.

## Conclusion

### Influence of Sound Sources in the St. Lawrence Soundscape

The influence of local traffic and wind speed on noise level at low (50 Hz), medium (300 Hz), and high (6300 Hz) frequencies have been assessed. The main results of this study are that distant maritime traffic noise is dominant above 200 Hz and wind noise is dominant above 500 Hz. In conditions of wind $<5$ m.s$^{-1}$ the traffic noise dominates and wind $>5$ m.s$^{-1}$ the wind noise dominates. For the high-frequency whistles and echolocation clicks of belugas, the wind noise dominates the soundscape.

### What Knowledge Can Provide the MARS Project?

The MARS project simultaneously deployed 12 acoustic loggers at four locations and three different depths in the Laurentian Channel. Three months of acoustic recordings from one hydrophone were analyzed to provide a qualitative description of the soundscape observed at the station. These data were compared to meteorological data (wind speed) and anthropogenic activities (shipping traffic) in the St. Lawrence. Analyses based on acoustic recordings from the MARS station provide a basis for studying the evolution and variations of the soundscape of the St. Lawrence. They also include preliminary studies for analyses of sound propagation, the acoustic footprint of ships, and the effect of anthropogenic sound on marine mammal communications and listening ranges.

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# Anthropophony Effects on Acoustic Metrics in a Marine Soundscape

37

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The original version of the chapter has been revised. The chapter was inadvertently published with incorrect name of the first author and this is corrected now. A correction to the chapter can be found at https://doi.org/10.1007/978-3-031-50256-9_194

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**Abstract**

Anthropogenic noise is known to influence acoustic environments and the habitats of marine species. Acoustic indices used to evaluate relative contributions of and trends in anthropogenic noise have had mixed success when applied to marine environments. This study assessed which acoustic metrics represent the level of anthropogenic activity within Abrolhos Bank, the most important humpback whale (*Megaptera novaeangliae*) breeding site in the western South Atlantic. Metrics included power spectral density (PSD), long-duration false-color (LDFC) spectrograms, ribbon plots, and nine acoustic indices. Metrics were calculated within a control area with low vessel traffic and a treatment area with high vessel traffic. Contributing sound sources included vessels, humpback whales, and fish species. The treatment area had higher PSD levels, indicative of high vessel noise conditions. Normalized difference soundscape index (NDSI), acoustic events (EVN), and acoustic complexity index (ACI) scored lower for higher levels of anthropogenic noise. ACI was influenced by high levels of humpback vocal activity and should be used with caution as a diversity proxy. LDFCs clearly confirmed the presence of singing humpbacks, fish choruses, and vessels, but only humpback songs were evident in ribbon plots. Findings in this chapter confirmed that PSD and spectral probability density plots are the current best way to characterize anthropophony within marine soundscapes.

**Keywords**

Passive acoustic monitoring · Long-term monitoring · Acoustic index · Humpback whale · *Megaptera novaeangliae*

## Introduction

A major challenge we face today is finding how to evaluate both biodiversity and human disturbance in the mosaic of natural to urban areas to determine conservation priorities in a world that is suffering rapid changes due to anthropogenic impacts. This task is especially hard for cryptic species and remote habitats, for whom classical ecological measurements are time-consuming and limited by human and financial resources (Sueur et al. 2008). In this perspective, acoustic recordings play an increasingly important role in monitoring trends within environments, both in terrestrial and marine ecosystems (Gasc et al. 2013; Pieretti et al. 2017; Lindseth and Lobel 2018; Alcocer et al. 2022).

For decades bioacoustic research focused on specific species and/or vocalizations. However, different passive acoustic monitoring (PAM) technologies have gained greater autonomy and are now capable of recording at higher sampling rates continuously for long periods of time (Sousa-Lima et al. 2013; Havlik et al. 2022). This scenario enables and forces us to look not just at a specific species but at the ensemble of sounds emanating from a whole ecosystem. These sounds may be derived from animals (zoophony, sensu Ferreira et al. 2018), from geophysical

events (geophony) or from human activities (anthropophony) (Krause 1987), and the sum of all these acoustic events is called "soundscape," defined as "the temporal and spatial distribution of sound through a landscape, reflecting important ecosystem processes and human activities" (Pijanowski et al. 2011).

The main challenge of using a soundscape scale is the choice of the adequate method to use, as analytical metrics continue to be developed and improved. Traditional measurements include broadband sound pressure level (SPL) and spectral analysis such as power spectral density (PSD) and fractional octave band levels (1/3 octave band) (Merchant et al. 2015). A more recent approach is the use of acoustic indices to summarize some aspect of the spectral-temporal distribution of acoustic energy in a recording (Towsey et al. 2014; Alcocer et al. 2022). The indices may be derived from waveforms, spectrograms or can be based on the relative contribution of the components of a soundscape (zoophony, geophony, and anthropophony), aiming to reflect different ecological aspects of a soundscape.

So far, results have been showing that weighted combinations of indices yield more ecologically useful information than single ones (Towsey et al. 2014). However, acoustic indices have shown inconsistent performance of correlation with diversity metrics (Alcocer et al. 2022), being influenced by transitory or permanent background noise (Sueur et al. 2014; Gasc et al. 2015; Bohnenstiehl et al. 2018). Another bias is the fact that the studies developed so far were mainly designed for terrestrial environments, despite the ecological importance of sound for several marine species (Tucker et al. 2014; Sueur et al. 2014; Fuller et al. 2015; Ferreira et al. 2018; Havlik et al. 2022). Furthermore, most of the studies performed in marine habitats were directed to coastal and shallow areas (having fish and crustaceans as major contributors to zoophony), concentrated in the northern hemisphere, with very little inspection of mammal contributors (Staaterman et al. 2013; Kaplan et al. 2015; Harris et al. 2016; Bertucci et al. 2016; Havlik et al. 2022; Alcocer et al. 2022). In addition, a visualization tool called long-duration false-color (LDFC) spectrogram was developed to facilitate the ecological interpretation of long-duration audio recordings (Towsey et al. 2014) and has nearly not yet been applied to marine ecosystems.

Hence, the goal of this study was to analyze how different acoustic indices react to two levels of anthropophony in a marine environment, guiding future uses of such measurements. For this purpose, PSD and LDFC analyses were included, and recordings made simultaneously in two areas were compared: one with high level of vessel traffic (treatment) and another outside the main navigation route (control). The low-frequency recordings (up to 1000 Hz) show male humpback whale (*Megaptera novaeangliae*) singing as the major contributor to zoophony, while vessel noise is the major contributor to anthropophony. This low-frequency spectrum is interesting because it is where major acoustic overlap between biotic and abiotic sources (geophysical and human-generated noise) occurs and hence may help elucidate to what extent each index is affected by abiotic noise rather than by biological sounds, a troubling factor already highlighted before for aquatic environments (Parks et al. 2014).

## Material and Methods

### Study Area

Study area was situated at the Abrolhos Marine National Park, a national protected area inside the northeast portion of the Abrolhos Bank (IBAMA/FUNATURA 1991). Abrolhos Bank is a sediment-capped volcanic bank in the east coast of Brazil (16°40′–19°30'S) with an approximate area of 30,000 km$^2$ and shallow mean depth of 30 m (Fainstein and Summerhayes 1982). It consists in an enlargement of the continental shelf with the bottom covered of coral reefs, mud, sand, and calcareous algae, and it comprises the most extensive and richest area of coral reefs in the Southwestern Atlantic (Leão and Kikuchi 2001). Abrolhos Bank is also the most important humpback whale breeding site in the western South Atlantic Ocean (Siciliano 1997), being a preferred concentration area of this population, estimated to be around 20,400 individuals (Bortolotto et al. 2017). Inside the national park, fishing is forbidden, and there are some restrictions for vessel traffic that mainly consists of small touristic boats (IBAMA/FUNATURA 1991; Hoyt 2001).

### Sound Data Acquisition

The sound data used in this study was acquired during the year 2005 (Sousa-Lima et al. 2018). Approximately one-third of the Abrolhos Marine National Park (314 km$^2$) was acoustically monitored using two arrays of four marine autonomous recording units ("MARUs" developed by the K. Lisa Yang Center for Conservation Bioacoustics of the Cornell Laboratory of Ornithology) fixed at the bottom floor at depths of around 20 m. The arrays were deployed northwest of the Abrolhos archipelago, one at an area of high level of vessel traffic (treatment, at 38°43′3.785"W, 18°0′47.237"S) and another in an area of little to no vessel traffic (control, at 38°43′43.777"W, 17°56′21.061"S) (Sousa-Lima et al. 2018), and one recorder from each array was selected for analysis (Fig. 1). Recording areas, though relatively close, were considered independent since low-frequency sounds are

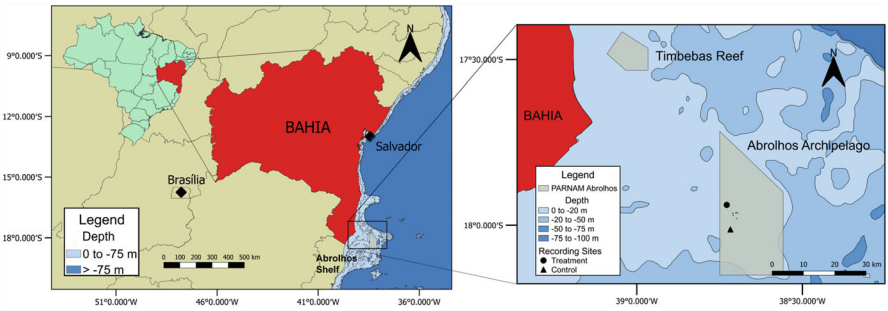



**Fig. 1** Map of the study area. Location of the recording sites, control (triangle, coordinates X 529881 Y 8008340), and treatment (circle, coordinates X 528717 Y 8016522) inside of PARNAM Abrolhos (Abrolhos Marine National Park), Abrolhos Bank, BA/Brazil

**Table 1** Properties of the recording equipment deployed for sound data acquisition (Sousa-Lima et al. 2018). Data was subsequently used for the analysis of soundscape measurements presented here

| | |
|---|---|
| **Recording advice** | Marine autonomous recording units (MARUs) |
| **Recording site** | Abrolhos archipelago, PARNAM Abrolhos, BA/Brazil |
| **Control position** | X 529881 Y 8008340 |
| **Treatment position** | X 528717 Y 8016522 |
| **Projection system** | UTM zone 24S |
| **Recording period** | 15 Jul 2005–01 Oct 2005 (79 days) |
| **Sampling scheme** | Continuously for 24 h |
| **Sample rate** | 2000 Hz |
| **Bit depth** | 12 bits |
| **Standard gain** | 23.5 dB |
| **Hydrophone sensitivity** | −165 dB re: 1Vrms/1μPa |

greatly attenuated in shallow depths (Forrest et al. 1993) and the same singing humpback whale individual could not be heard simultaneously in MARUs from the two different arrays. The arrays were programmed to record continuously during the breeding season of humpback whales, from 15 July to 01 October of 2005, at a sampling rate of 2000 Hz (Table 1).

## Sound Data Analysis

To produce calibrated measurements and calculate the power spectral density (PSD) of the recordings, the template code *PAMGuide* (Merchant et al. 2015) was used under MATLAB (MATLAB 2022). PSD is a standardized spectrum of sound levels across frequencies, estimating the strength of energy variation as a function of frequency. PSD is considered as one of the most comprehensive analyses of the sound-level distribution (Merchant et al. 2015; Lindseth and Lobel 2018). In *PAMGuide*, the following settings were used: Mh of −165 dB re: 1 Vrms/1 μPa, gain of 23.5 dB, vADC of 1.414 (estimated value), chunk size of 300 s, linear scale, Hamming window, and Welsh average of 1200s, resulting in a 10-min resolution analysis of the 79 days for each area.

For the calculation of acoustic indices and generation of LDFC spectrograms, the open-source software *QUT Ecoacoustics Analysis Programs* was used under Windows Command Prompt (Truskinger et al. 2014; Towsey et al. 2018); the indices used are listed in Table 2 (see Towsey 2017 for detailed information on indices calculation). LDFCs are based on three indices known to capture distinct characteristics of sound that are mapped to red-green-blue colors and displayed in spectrogram-like format. The calculation of the acoustic indices and production of LDFCs were performed for each 24 h using a 1-minute resolution (hence, each day was primarily split in 1440 1-minute segments), using the following settings: FFT length of 512, Hamming window, low-frequency bound of 100 Hz, mid-frequency bound of 900 Hz, high-frequency bound of 1000 Hz, and 0% overlap. LDFCs were based on ACI, temporal entropy (ENT), and EVN: ACI is more sensitive to complexly structured calls having multiple levels of amplitude modulation, ENT responds to acoustic events of

**Table 2** Definition and properties of the acoustic indices calculated for the investigation of anthropophony influence on acoustic index performance in a marine soundscape. See Towsey (2017) for detailed calculation

| Index | Name/formula (if available) | Definition/calculation |
|---|---|---|
| **ACI** | Acoustic complexity index (degree of complexity)<br>$ACI[f] = \sum i\lvert A_{if} - A_{i-1,f}\rvert / \sum iA_{if}$<br>$i$ index over frames<br>$f$ index over freq. bins | The average absolute fractional change in spectral amplitude between two adjacent intensities, presented as an average of all frequencies, widely used as a measure of diversity. See Pieretti et al. (2011) for index idealization, though Towsey (2017) modifications were used on the index calculation performed (only the mid-band is used) |
| **BGN** | Background noise | An estimate of the background noise in each 1-minute recording. It is set equal to the mode of the energy distribution in the waveform envelope, according to the method of Lamel et al. (1981) |
| **SNR** | Signal-to-noise ratio | The decibel difference between the maximum envelope amplitude and the background noise in any minute segment |
| **ACT** | Acoustic activity | The fraction of noise-reduced decibel envelope which exceeds 3 dB above the level of background noise |
| **ENT** | Temporal entropy (acoustic energy concentration in time)<br>$H[t] = -(\sum ipmf_i \, x \, log_2(pmf_i))/log_2 N$<br>$ENT = 1 - H[t]$<br>$pmf$ probability mass function<br>$i$ index over frames<br>$N$ number of frames | The squared amplitude values in each frequency bin are normalized to unit area and treated as a probability mass function; classically it is a measure of energy "dispersal" through time, and here it is converted into a measure of energy concentration (ENT = 1 – H[t]). Derived from the noise-reduced amplitude envelope |
| **EVN** | Acoustic events (Count of Acoustic events per minute)<br>$\theta = 3$ dB | The number of acoustic events (decibel value in a bin that crosses a 3 dB threshold) per minute in each noise-reduced frequency bin. Derived from the decibel envelope |
| **EAS** | Entropy of the averaged spectrum<br>EAS = 1 – H[a] | Calculates the average of the mid-band spectra then calculates its entropy as described for temporal entropy. A measure of the "concentration" of mean energy through the spectrum, expected to be like the Gini index. Derived from the noise-reduced frequency spectrum |
| **NDSI** | Normalized difference soundscape index (index of health quality)<br>NDSI = (zoophony – anthropophony)/(zoophony + anthropophony) | Estimates the level of anthropogenic disturbance on the soundscape by computing the ratio of anthropophony to zoophony in a sound file. Ranges between −1 (prevalence of anthropophony) and 1 (prevalence of zoophony). See Kasten et al. (2012) for index idealization and calculation |

(continued)

**Table 2** (continued)

| Index | Name/formula (if available) | Definition/calculation |
|---|---|---|
| **SPD** | Spectral peak density (Density of peaks in Spectrum) | A measure of the average number of cells in the mid-frequency band of a 1-minute spectrogram that is identified as being local maxima (when its decibel value exceeds a 6 dB threshold, being greater than the values in the frequency bins on either sides). Derived from the decibel spectrogram. Considered as an "ecological index" that attempts to be an acoustic measure of species richness (Towsey 2017) |

concentrated energy such as pulses or croaks, and EVN responds to high pulse rates (Towsey et al. 2018). False-color spectrograms were built with 2345 frames (frame duration = 25.58 ms) and 256 frequency bins (frequency resolution of ~3.91 Hz).

In addition, ribbon plots of the 79 days were made for control and treatment areas. Ribbon plots are a vertically stacked series of 24-hour LDFCs whose vertical resolution is reduced (1440 × 32-pixel ribbons), in a way that is possible to attach and visualize several months of audio recordings (Phillips et al. 2017). Hence, it is possible to reveal progressive patterns of changes occurring over time at the cost of losing detailed information contained in LDFCs.

## Statistical Analysis

For a description of the areas, a general inspection of daily spectrograms was performed, and the proportion of minutes containing the presence of vessels and humpback whale sounds was estimated based on visual and aural inspection of audio files in 2-minute window spectrograms with no overlap and 1024 FFT.

To understand how the acoustic indices respond to different levels of anthropophony and zoophony, a customized dataset consisting of recordings with different proportions of anthropophony (vessel noise) and zoophony (humpback whale vocalization) was generated. Recordings from the complete, real dataset were used to create ten 1-minute samples of each of these combinations: V100 (100% of the recording with vessel noise), V75W25 (75% of the recording with vessel noise and 25% with humpback whale vocalizations), V5W5 (50% of the recording with vessel noise and 50% with humpback whale vocalizations), V25W75 (25% of the recording with vessel noise and 75% with humpback whale), and W100 (100% of the recording with humpback whale). All the acoustic indices (Table 2) for the customized dataset were calculated, and the results were compared with those from the real dataset.

After exploratory data analysis (Zuur et al. 2010) indicating heterogeneity and temporal autocorrelation of data, a Wilcoxon signed-rank test was used to test for pairwise differences between the areas. For such, a random minute was selected every 3 h (to avoid autocorrelation of the same singing whale, following the procedure in Sousa-Lima and Clark 2008), and 30 random days of the entire recording period were

sampled for pairwise comparison, using a significance level of 0.05. Statistical analysis was performed under R statistical program (R Development Core Team 2022) using the packages dplyr (Wickham et al. 2022), cowplot (Wilke 2020), and ggplot2 (Wickham 2016). A synthesis of the performance of each acoustic index for each area was also generated, as recommended by Lindseth and Lobel (2018).

## Results

A total of 1896 h were analyzed for each sampled area. In the control area, 0.31% of the minutes had the presence of boat noise, while 98.79% of minutes had the presence of humpback whale sounds; in the treatment area, 8.28% of the minutes had the presence of boat noise, while 97.22% had the presence of humpback whale sounds. Hence, both areas were highly dominated by humpback whale vocalizations in the period recorded, whereas the main difference between them was the anthropophony level due to vessel traffic.

In addition, other acoustic events were found through the visual inspection of daily standard spectrograms. Those were continuous events that lasted hours (from 2 to 6 h), found in both areas during the nighttime, usually presenting 3 or 4 harmonics between 50 and 300 Hz. The events were identified as fish choruses, although further confirmation by specialists is recommended (Mcpherson et al. 2016; Pieretti et al. 2017).

## Power Spectral Density (PSD)

Power spectral density analysis resulted in a different pattern for both areas, highlighted by the spectral probability density (SPD) (Fig. 2). Below 400 Hz, the

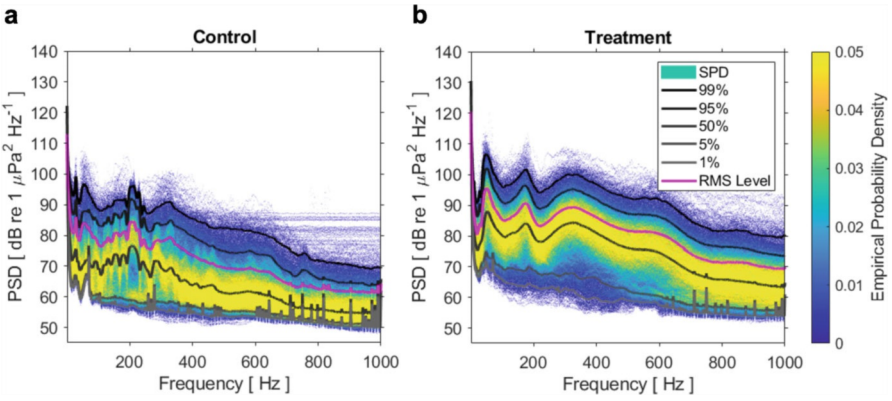



**Fig. 2** Power spectral density (PSD) analysis for control and treatment areas. PSD was calculated over the range of 0–1000 Hz using a 10-min resolution average of the long-term soundscape data from Abrolhos Bank/BR. Two recording areas were used, (**a**) control (lower level of vessel traffic) and (**b**) treatment (higher level of vessel traffic). RMS (root-mean-square) level: pink line; percentiles, gray-colored lines; spectral probability density (SPD), color-graded (yellow are values more likely to occur while dark blue are values less likely to occur)

treatment area showed higher sound levels, with three peaks of ~90 dB re 1 $\mu Pa^2$ $Hz^{-1}$ at around 50 Hz, 180 Hz, and 350 Hz, while the control area showed an irregular pattern of several small peaks between 100 and 300 Hz and bimodality of the spectral probability distribution at 200 Hz (two different sound levels with similar probabilities, around 60 and 80 dB re 1 $\mu Pa^2$ $Hz^{-1}$). Above 400 Hz, SPD of the control area indicated sound levels likely to be around 60 dB re 1 $\mu Pa^2$ $Hz^{-1}$, while in treatment sound levels were higher at around 80 dB re 1 $\mu Pa^2$ $Hz^{-1}$ until falling to ~65 dB re 1 $\mu Pa^2$ $Hz^{-1}$ above 650 Hz.

## Index Performance on the Customized Dataset

In the customized dataset, ACI was the index that best separated the categories, increasing its values consistently with increase of zoophony proportion on the recording (Fig. 3). Other indices that presented such a tendency were EVN, ENT, SPD, ACT, and signal-to-noise ratio (SNR), while entropy of averaged spectrum

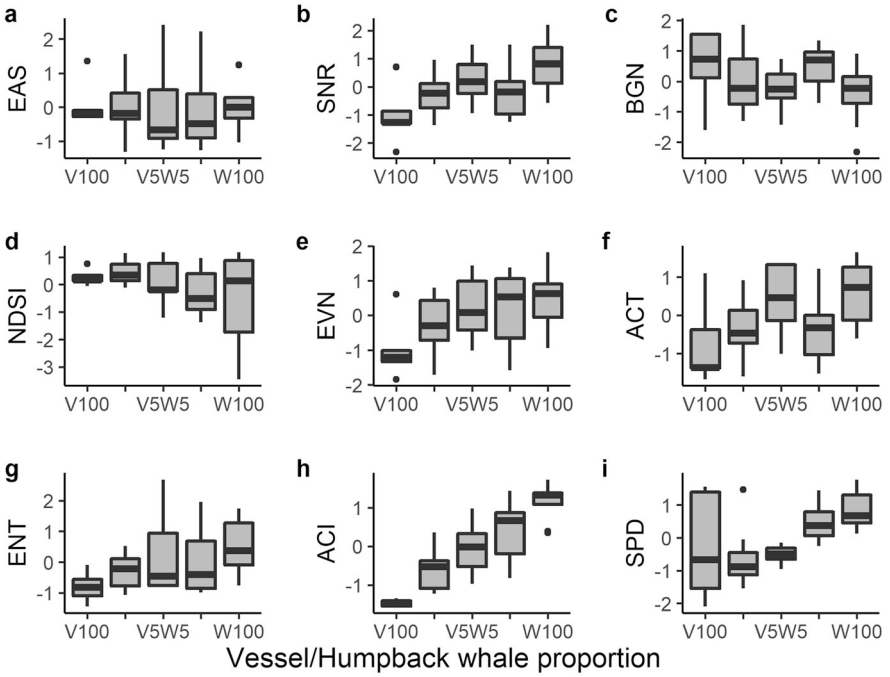



**Fig. 3** Acoustic index performance in different proportions of anthropophony and zoophony: V100 (100% vessel noise), V75W25 (75% vessel noise and 25% humpback whale vocalizations), V5W5 (50% vessel noise and 50% humpback whale vocalizations), V25W75 (25% vessel noise and 75% humpback whale), W100 (100% humpback whale). Indices results are normalized to mean 0. (**a**) Entropy of averaged spectrum (EAS), (**b**) signal-to-noise ratio (SNR), (**c**) background noise (BGN), (**d**) normalized difference soundscape index (NDSI), (**e**) acoustic events (EVN), (**f**) acoustic activity (ACT), (**g**) temporal entropy (ENT), (**h**) acoustic complexity index (ACI), (**i**) spectral peak density (SPD)

(EAS), NDSI, and background noise (BGN) showed a rather constant or inverted result between the levels (e.g., higher scores in V100 level).

## Index Performance on the Complete Dataset

EVN ($p = 0.017$), EAS ($p = <0.001$), and NDSI ($p = <0.001$) showed higher levels in the control area, despite having a lot of overlap between the results of both areas (Fig. 4). ACI had higher values in treatment ($p < 0.001$), as well as SPD ($p = <0.001$) and ACT ($p = <0.001$). SNR ($p = 0.58$) and ENT ($p = 0.66$) did not show any significant difference between the areas, although ENT had higher scores in control when compared to treatment (note the skewed distribution of ENT data). Table 3 is a summary of the performance of each acoustic metric.

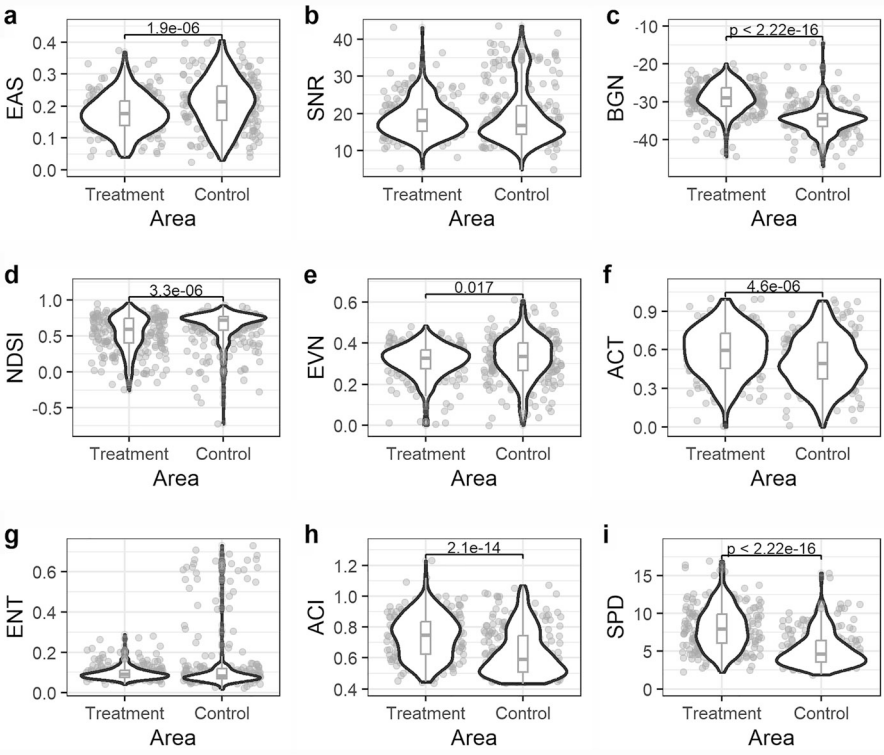



**Fig. 4** Wilcoxon pairwise comparison of index performance in control and treatment areas. Indices results are normalized to mean 0. (**a**) Entropy of averaged spectrum (EAS), (**b**) signal-to-noise ratio (SNR), (**c**) background noise (BGN), (**d**) normalized difference soundscape index (NDSI), (**e**) acoustic events (EVN), (**f**) acoustic activity (ACT), (**g**) temporal entropy (ENT), (**h**) acoustic complexity index (ACI), (**i**) spectral peak density (SPD)

**Table 3** Summary of acoustic metrics in control and treatment areas of Abrolhos Bank/BR during humpback whale breeding season (data by minute)

| Acoustic metric | | Control (N = 113,760) | Treatment (N = 113,760) |
|---|---|---|---|
| **EAS** | Mean ± SD | 0.22 ± 0.09 | 0.18 ± 0.06 |
| | Median (Q1, Q3) | 0.22 (0.16, 0.28) | 0.17 (0.14, 0.22) |
| | Min – Max | 0.00 – 0.83 | 0.00 – 0.58 |
| **ENT** | Mean ± SD | 0.15 ± 0.16 | 0.10 ± 0.04 |
| | Median (Q1, Q3) | 0.08 (0.07, 0.13) | 0.09 (0.08, 0.11) |
| | Min – Max | 0.00 – 0.78 | 0.01 – 0.55 |
| **SNR** | Mean ± SD | 19.51 ± 7.50 | 18.79 ± 4.55 |
| | Median (Q1, Q3) | 17.14 (14.03, 23.24) | 18.19 (15.62, 21.30) |
| | Min – Max | 0.76 – 52.15 | 2.90 – 49.22 |
| **BGN** | Mean ± SD | −34.10 ± 3.87 | −28.58 ± 4.07 |
| | Median (Q1, Q3) | −34.22(−35.95, −32.52) | −28.48(−31.05, −25.94) |
| | Min – Max | −52.15 – 7.53 | −49.22 – 7.97 |
| **NDSI** | Mean ± SD | 0.64 ± 0.23 | 0.55 ± 0.29 |
| | Median (Q1, Q3) | 0.72 (0.58, 0.77) | 0.63 (0.40, 0.77) |
| | Min – Max | −0.89 – 1.00 | −0.92 – 0.99 |
| **EVN** | Mean ± SD | 0.32 ± 0.10 | 0.31 ± 0.08 |
| | Median (Q1, Q3) | 0.32 (0.27, 0.38) | 0.32 (0.27, 0.37) |
| | Min – Max | 0.00 – 0.64 | 0.00 – 0.57 |
| **ACT** | Mean ± SD | 0.50 ± 0.20 | 0.58 ± 0.19 |
| | Median (Q1, Q3) | 0.49 (0.36, 0.64) | 0.59 (0.45, 0.72) |
| | Min – Max | 0.00 – 1.00 | 0.00 – 1.00 |
| **ACI** | Mean ± SD | 0.63 ± 0.18 | 0.74 ± 0.16 |
| | Median (Q1, Q3) | 0.57 (0.50, 0.72) | 0.73 (0.62, 0.85) |
| | Min – Max | 0.40 – 1.66 | 0.40 – 1.46 |
| **SPD** | Mean ± SD | 5.14 ± 2.47 | 7.84 ± 2.84 |
| | Median (Q1, Q3) | 4.39 (3.48, 6.02) | 7.70 (5.73, 9.82) |
| | Min – Max | 0.45 – 21.50 | 1.08 – 21.06 |

## LDFCs and Ribbon Plots

When visually comparing standard spectrograms with false-color spectrograms (Fig. 5), the pattern of humpback whale singing activity is easily identifiable in both plots (marked boxes), but even more so in LDFCs, where whale singing appears as a broadband pattern with predominantly yellow (interaction between ACI/red and ENT/green) and pink (interaction between ACI/red and EVN/blue). Vessel passages and fish choruses were detected by the blue color of the EVN index. Also, in LDFCs, electrical interferences generated by the recording system are visible as ENT/green marks, especially in the upper and lower frequencies.

Ribbon plots evidenced the lack of periodicity in the presence of humpback whale singing activity, as these events are found in both areas at any time of the day (Fig. 6). It is also possible to see that the treatment area has brighter and stronger pink colors than control, indicating stronger singing activity of humpback whales (Fig. 6).

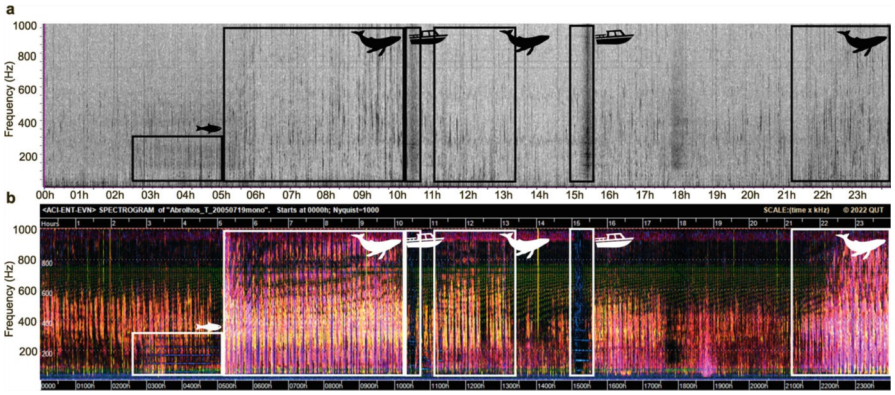



**Fig. 5** One day of recording from the treatment area is represented as a standard spectrogram (**a**) and long-duration false-color (LDFC) spectrogram (**b**). Boxes are drawn around sound events in each recording caused by fish choruses, vessel passages, or male humpback whale singing activity. For both representations, x-axis = time (24 h), y-axis = frequency (1000 Hz). For LDCFs, z-axis = ACI, ENT, and EVN index values assigned as red-green-blue code; for standard spectrograms, z-axis = acoustic energy in dB (gray color-coded), FFT = 512

On the other hand, ribbon plots did not show any of the temporal patterns expected to be visible in this long-duration recording period, such as a variation between night and day or any influence of moon and tidal cycles in the soundscape (shown in Sousa-Lima and Clark 2008). Also, the expected patterns of diurnal vessel noise and nocturnal fish choruses were not evident. These blue events were much more subtle than humpback singing events and were probably missed when the LDFCs lost frequency resolution to generate the ribbon plot.

## Discussion

The field of ecoacoustics has been exploring ways to assist the monitoring of terrestrial and aquatic environments. Acoustic indices that summarize soundscape characteristics have shown mixed performance on capturing ecological information. Results in this chapter support the need of using a combination of different acoustic metrics for marine soundscape characterization. Other studies have also used a combination of acoustic indices and standard acoustic measurements (Pieretti et al. 2017; Blondel and Hatta 2017), describing sound levels and its variations over different bandwidths and timescales (Merchant et al. 2013).

PSD and SPD patterns obtained for Abrolhos marine soundscapes corroborate that complementary metrics are useful in revealing the underlying noise-level distribution of different habitats. In this study, treatment and control areas presented different peaks of SPD in the frequency band of 100–600 Hz, with the treatment area also showing higher sound levels (Fig. 2). This result was expected once this is the

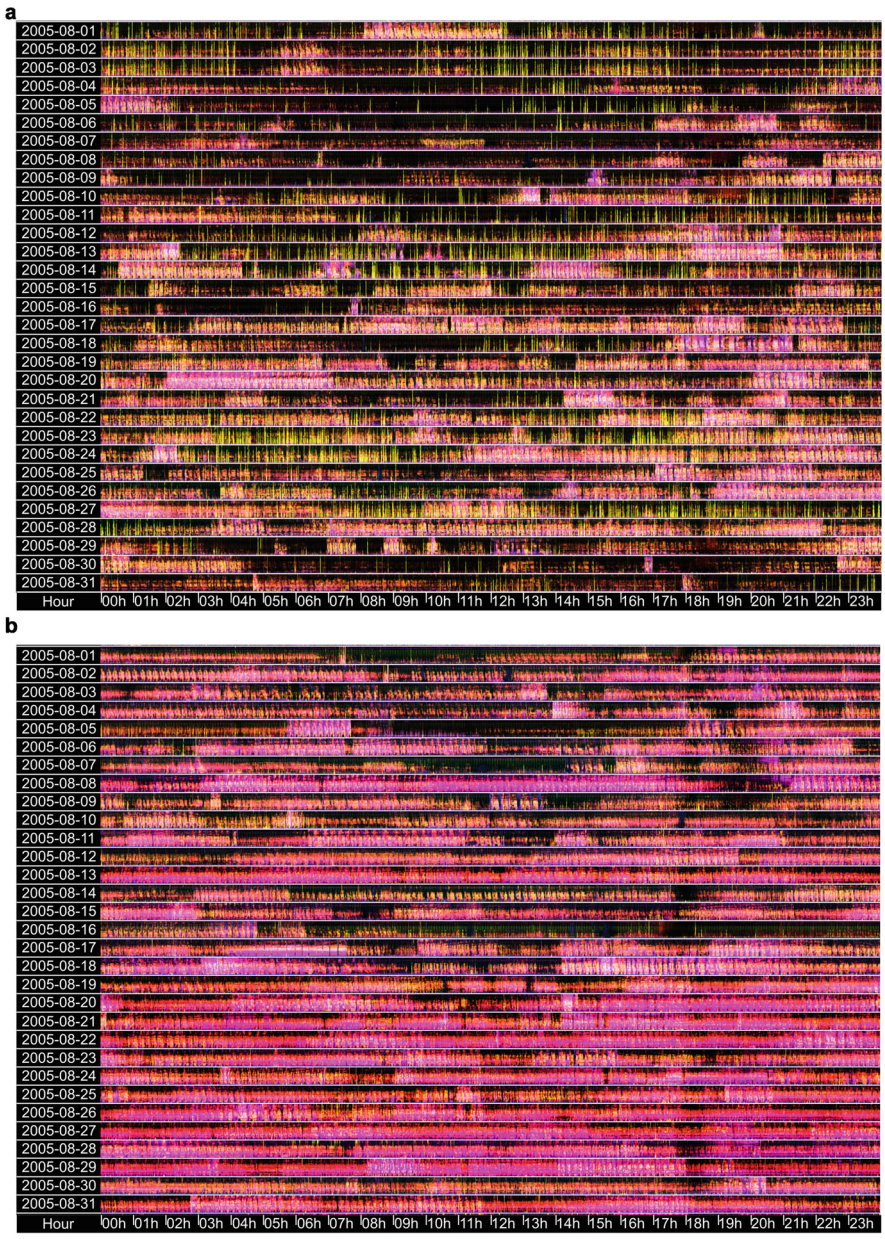



**Fig. 6** Ribbon plots of one month of continuous recording of Abrolhos soundscape for control (**a**) and treatment (**b**) areas. X-axis, time (hour); y-axis, frequency (1000 Hz as 32-pixel resolution; each string is a recording day); z-axis: ACI, ENT, and EVN indices with values assigned to red-green-blue code

frequency range where vessel noise is more intense (Urick 1983; Hildebrand 2009; Fernandes 2014; Mcpherson et al. 2016). In the control area, there was a bimodality in the SPD distribution at frequencies around 200 Hz (where engine noise is greater) that might be explained by the transient occurrence of vessel noise dragging the modal tendency ~20 dB greater than the primary mode, whereas in the treatment area vessels are more frequent and result in a unique mode with higher levels (Merchant et al. 2013). In addition, the control area had several small peaks between 100 and 300 Hz, which can be the reflection of a higher diversity of animal signals (zoophony).

Acoustic index performance was variable. In the complete, real dataset, NDSI, acoustic events (EVN), and entropy of the averaged spectrum (EAS) showed higher values in the control area (Fig. 4). The NDSI had mostly positive values in both areas, correctly demonstrating that both areas had a high level of biological sounds due to humpback whale vocal activity, yet the control area showed an overall higher score, probably due to less masking of humpback whale sounds. This result, even for such a narrow-frequency bandwidth, initially corroborates with the applicability of this index as an estimate of the level of anthropogenic disturbance also in marine soundscapes (Kasten et al. 2012; Machado et al. 2017; Alcocer et al. 2022). However, the same index did not perform as well in the customized dataset, showing relatively similar values between the different proportions of anthropophony/zoophony and only a higher variability in the 100% zoophony recordings. This inconsistency is probably due to the frequency bands selected for the present analysis, with the midband having a major overlap between zoophony and anthropophony (in contrast to the usual bands selected for terrestrial environments, anthropophony at 1–2 kHz and zoophony at 2–8 kHz), indicating an obvious caveat when selecting frequency bands according to the sampling frequency of each specific study (Kasten et al. 2012). EVN (a measure of the number of acoustic events) performed well for both complete and customized datasets, scoring higher for higher zoophony, although LDFCs showed that the index was also capturing the noise of vessel engines (see discussion below), yet with lower scores. As for the spectral entropy (EAS – a measure of energy concentration in the frequency spectrum), the control area showed higher and more variable values. However, in the customized dataset, EAS showed similar values in all levels. This result agrees with the reduced reliability already discussed for the classical H index (a measure of acoustic dispersion and not acoustic concentration) when measuring communities with a few sonic species (e.g., a single species producing sounds covering a broad frequency spectrum, which is the case in this study) (Sueur et al. 2008). Arguably, H has been one of the best performing indices in capturing ecologically relevant information (Alcocer et al. 2022), so it is possible that its applicability is useful in environments where animals produce broadband signals and the analyses' bandwidth is limited, as in this study.

The acoustic complexity index (ACI), background noise (BGN), activity index (ACT), and spectral peak density (SPD) showed higher values at the treatment area, indicating that anthropophony has biased these indices (Fig. 4). BGN is a simple measure of ambient background noise, showing expected higher levels for the area

with higher anthropogenic noise (treatment area). ACT is a simple measure of the number of acoustic events with a high sound level. These levels might be due to both biological and non-biotic sound sources, which can explain the higher score in treatment. However, ACI is the index most widely used for biodiversity appraisal in marine ecosystems (McWilliam and Hawkins 2013; Staaterman et al. 2014; Blondel and Hatta 2017; Lindseth and Lobel 2018; Bohnenstiehl et al. 2018; Alcocer et al. 2022). ACI assumes that biotic sounds are complex and amplitude modulated, whereas human-generated noise are usually constant (Pieretti et al. 2011). ACI has already demonstrated concerning biases such as being sensitive to geophony (Pieretti et al. 2011), to species identity (Gasc et al. 2015), to specific vocalization features (e.g., call rate and call type), and to the spectral resolution of analyses (Bohnenstiehl et al. 2018). Some authors have already suggested new indices derived from adjustments of ACI, such as the Acoustic Variability Index, which proved to be a good indicator of shipping noise (Mcpherson et al. 2016). For terrestrial habitats, background noise effects could be mitigated with threshold cutoff filters (Farina and Pieretti 2014); this simple approach is almost always impossible for marine ecosystems where there is a lot of overlap between biological and non-biological sound sources (Parks et al. 2014), especially at limited frequency ranges. At a first glance, the results in this chapter might suggest that ACI was incorrectly accounting for vessel noise as biological sounds, yielding unreliable higher values for the treatment area. Nonetheless, a clearer interpretation is achieved with the help of LDFCs to visualize features of interest with manual inspection and validation of the long-term recordings.

One of the main benefits of using LDFCs is that it enables a rapid identification of prominent soundscape features without having to listen to hours of recording (Phillips et al. 2017). Through this visualization, it was quite clear that humpback whale singing activity was the major feature dominating Abrolhos Bank soundscape during the recorded period, both in the control and treatment areas. However, vessel noise was not as evident as expected, nor were fish choruses, which were relatively easy to observe through standard spectrograms. Both events appeared as subtle blue events (captured by EVN) in LDFCs (Fig. 5), clarifying that ACI did not account for vessel noise, but rather, to a possible higher intensity of whale singing activity in the treatment area. The customized dataset also helped assign higher ACI scores to a high proportion of zoophony and lower ACI scores to a high proportion of anthropophony. According to Fernandes (2014), the treatment area within the Abrolhos Bank shows high concentration of vocally active whales (which is also the same area that overlaps with the highest vessel traffic). Hence, ACI might be biased by specific species and its call rate (humpback whales, in this case) and hence should be used with caution as a proxy for richness and diversity, as suggested by other studies (Gasc et al. 2015; Bohnenstiehl et al. 2018; Alcocer et al. 2022). The same might have influenced SPD, which was developed to be a measure of acoustic richness.

SNR and ENT showed no distinction between the study areas. However, in the customized dataset, there was a gradual increase with increasing zoophony proportion for both indices. This suggests that the result of the indices on the complete

dataset was possibly driven by the prevalence of humpback whale signals in both areas, despite higher anthropogenic noise and humpback whale vocal activity in the treatment area. Also, ENT presented a skewed distribution to higher levels in the control area, in accordance with the significant result seen for EAS (another similar index but calculated from the frequency spectrum while ENT is derived from the amplitude envelope). These indices have been used in weighted combination with others in a finer temporal resolution to create detectors of specific long-duration acoustic events, showing useful applications (Towsey et al. 2014).

The temporal patterns of the Abrolhos soundscapes studied in this chapter were visually assessed by the ribbon plots. Ribbon plots highlighted constant humpback whale singing activity, emphasizing the continuous occurrence of long-duration singing events throughout the 24 h. As reported by Sousa-Lima et al. (2018), humpback whale singing activity in Abrolhos ranges from 30 min to 20.5 h nonstop. This visualization tool might be only useful for spotting seasonality in longer datasets. It would be interesting to compare recordings made year-round, including humpback whale nonbreeding periods, when fish choruses should have a more important role in the Abrolhos soundscape.

## Conclusion

This chapter shows that a combination of traditional acoustic measurements with acoustic indices is recommended for the analysis of underwater soundscapes, in agreement with previous studies (Staaterman et al. 2013; Lindseth and Lobel 2018). PSD and SPD analysis proved to be useful in revealing the underlying noise-level distribution of different habitats and should be included in soundscape characterization. Among the acoustic indices, NDSI and ACI appeared to be less useful to account for anthropophony but are potentially useful to measure zoophony. However, ACI was impacted by humpback whale singing activity and should be used with caution as a measure of diversity within single-species acoustically dominated soundscapes. LDFCs highlighted humpback whale singing activity and demonstrated to be an interesting tool for the interpretation of index performance, and future applications should take into consideration different indices' combinations.

This study represents a step toward establishing robust protocols for long-term acoustic monitoring, fulfilling the passive acoustic monitoring promise of being an easy and accessible tool for conservation.

## Cross-References


- ► Noise Pollution on the Singing Cycle of Humpback Whales and the Soundscape of the Gulf of Tribugá, Colombia
- ► Soundscape Characterization in Chedabucto Bay on the Atlantic Coast of Canada, 2019–2021

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# Section IV

# Hearing and Auditory Assessment: General

# Value of a Kurtosis Metric in the Evaluation of Noise-Induced Hearing Loss

38

Wei Qiu

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**Abstract**

The fundamental problem with current noise exposure criteria (e.g., ISO 1999) is their reliance on an energy metric to quantify an exposure, which completely ignores the effects of temporal characteristics of noise exposure known to be important in affecting noise-induced hearing loss (NIHL). Noise usually has a rather complex temporal structure, which can be expressed as a combination of continuous steady-state noise and various transient impulsive noises. A large number of animal experiments and epidemiological studies have shown that complex noise presents a greater hearing loss risk than continuous noise. Therefore, a metric is needed to account for this increased risk when analyzing hearing loss caused by complex noise. Over the last 20 years, multi-national collaborative teams have addressed the issue of complex noise exposure. Both animal and human research studies have found that the kurtosis statistic, a metric that describes the

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probability for a noise signal to include high-level impulsive events, can account for the risks observed for groups with complex exposures. Kurtosis holds great promise in addressing the complexity associated with measuring and characterizing such exposures. Many issues focused on kurtosis can be applied not only to terrestrial mammals but also to marine mammals. A better understanding of the role of the kurtosis metric in NIHL may lead to its incorporation into a new generation of more predictive hearing risk assessments for noise exposure.

**Keywords**

Noise-induced hearing loss · Complex noise · Kurtosis of noise exposure · Kurtosis-adjusted noise exposure level

## Introduction

Noises of the same or similar energies and spectra can have very different effects on hearing because of their different temporal structures. Studies (e.g., Hamernik and Qiu 2001; Hamernik et al. 2003, 2007; Qiu et al. 2007, 2013) have shown that temporal variables need to be considered to adequately assess the effects of non-Gaussian complex noise on hearing. High levels of complex noise exposure are common in industrial environments. Complex noise exposure cause much more damage to hearing than continuous steady-state noise at the same equivalent sound level, and existing noise standards (e.g., ISO 1999) underestimate hearing loss due to complex noise. (Zhang et al. 2021a). Numerous animal experiments have shown that exposure to complex noises produces more sensory cells loss than exposure to equivalent level steady-state continuous noise (e.g., Dunn et al. 1991; Lei et al. 1994; Lataye and Campo 1996; Hamernik and Qiu 2001; Harding and Bohne 2004; Hamernik et al. 2003, 2007; Qiu et al. 2006, 2007). The results of animal experiments have been confirmed by epidemiological studies (e.g., Sulkowski et al. 1983; Taylor et al. 1984; Theiry and Meyer-Bisch 1988; Zhao et al. 2010; Xie et al. 2016; Zhang et al. 2022). These results also prompt us to re-examine the application of the equal-energy hypothesis (EEH), which forms the basis of current noise exposure standards (e.g., ISO 1999). The issues raised above, while directly relevant to humans and other terrestrial mammals, are also relevant to marine mammals and might even be considered for fishes (e.g., National Marine Fisheries Service 2016). While this kind of work has not been done for marine mammals, it might be highly relevant and should be examined.

Industrial workers are often exposed to complex noise environments. Complex noise consists of a background Gaussian noise with embedded high-level transients, as shown in Fig. 1. Transients can be impacts or noise bursts. These transients usually have different peak intensities, inter-transient intervals, and durations. These noise characteristics are known factors that affect the outcome of noise exposure.

Since current noise standards only use energy measures and do not include the temporal characteristics of noise, and since there are countless noise exposures with the same energy and spectrum but very different temporal structures, the inclusion of

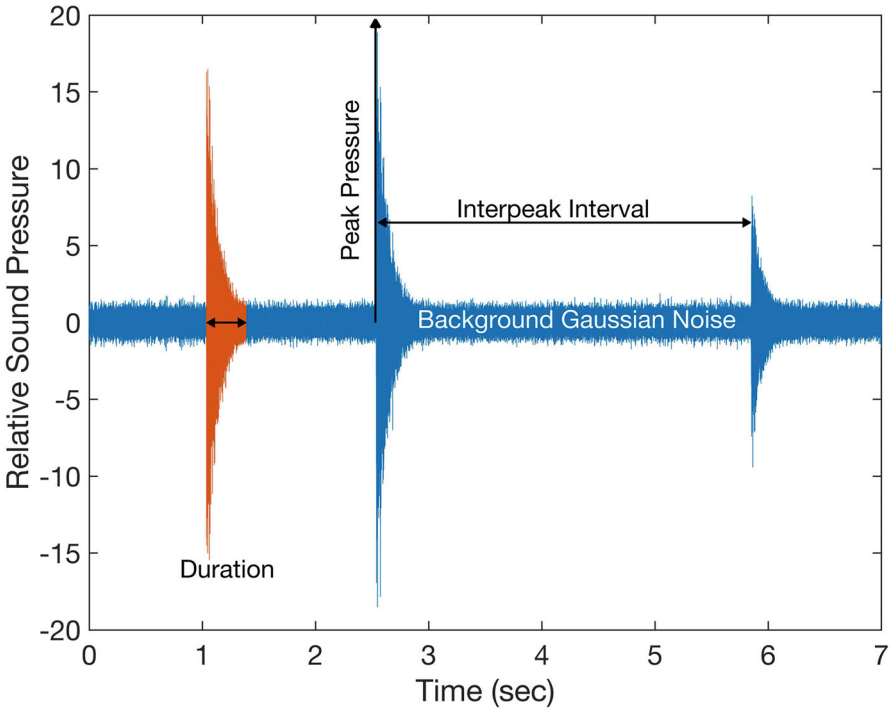



**Fig. 1** An example of complex non-Gaussian noise. Compound noise is composed of Gaussian background noise and various high-level transients superimposed. The transients could be impacts or noise bursts of varying peak intensities, inter-transient intervals, and durations. (Adapted from Qiu et al. 2020)

a metric related to the temporal variables of noise would be a useful adjunct to the energy metric. One such metric is kurtosis.

## The Definition of Kurtosis

Kurtosis is a statistical measure that describes the extreme value of the data relative to the two ends of the Gaussian distribution. In the case of noise, kurtosis describes the impulse property of noise, i.e., the degree of deviation from continuous, steady-state noise (Murphy and Qiu 2021). Theoretically, kurtosis is defined as the ratio of the fourth-order central moment to the squared second-order central moment of a distribution. For a sample of $n$ values, the kurtosis of noise ($\beta$) is calculated as:

$$\beta = \frac{1}{n}\sum_{i=1}^{n}(x_i - \bar{x})^4 \Big/ \left(\frac{1}{n}\sum_{i=1}^{n}(x_i - \bar{x})^2\right)^2$$

where $x_i$ is the $i$th value of noise amplitude and $\bar{x}$ is the sample mean.

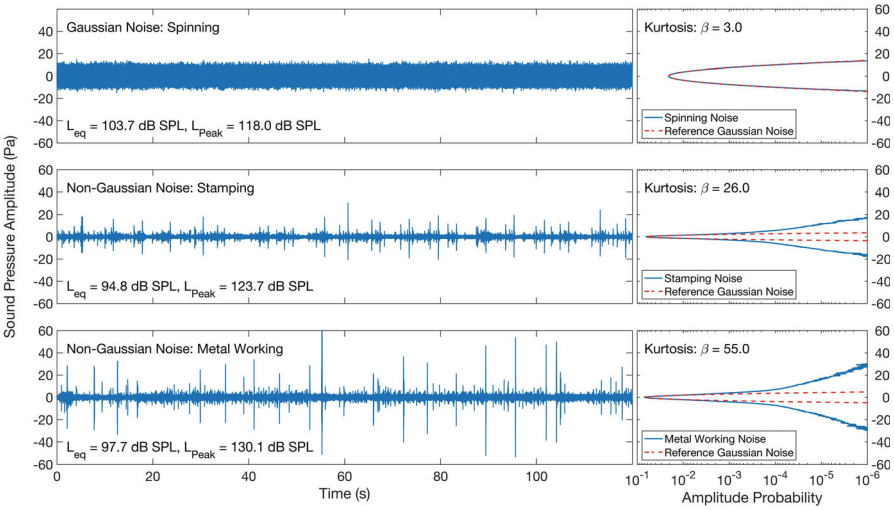



**Fig. 2** Comparison waveforms (left) and amplitude probabilities (right) from three industrial noises. Note: $L_{eq}$ is the equivalent sound pressure level of noise, and $L_{peak}$ is the peak sound pressure level of noise. (Adapted from Qiu et al. 2020)

All the variables that characterize a non-Gaussian noise, such as transient peaks, inter-transient intervals, transient durations, crest factor, etc., affect the kurtosis value. The kurtosis of a continuous steady-state (Gaussian) noise is 3. The kurtosis value of complex noise is greater than 3. Figure 2 shows three examples of industrial noise waveforms, their kurtosis values, and corresponding amplitude probability histograms.

## Evidence of Kurtosis Differentiating Hearing Loss Caused by Complex Noises with Different Temporal Structures

Qiu et al. (2013) designed animal (chinchilla) experiments to determine to what extent auditory system trauma, as quantified by sensory cell loss (percentage of outer hair cell loss, %OHC Loss) is affected by differences in the temporal structure of various complex noise exposures all having the same spectral energy. Four groups of animals ($n = 8$ for each group) were exposed to an overall SPL of 97 dB(A) noises (for 19 days) with different noise kurtosis values equal to 3, 25, 50, and 100, respectively. The 19 days of exposure simulated 3 weeks of typical occupational noise exposure, i.e., 8 h a day, 5 days a week. The results are shown in Fig. 3. The figure clearly shows that noise with the same sound pressure level but a different temporal structure can produce different degrees of hearing loss. From the chinchilla model, hearing loss is proportional to kurtosis in that threshold shift, and hearing loss increase as kurtosis increases for a fixed noise energy level. In other words, noise

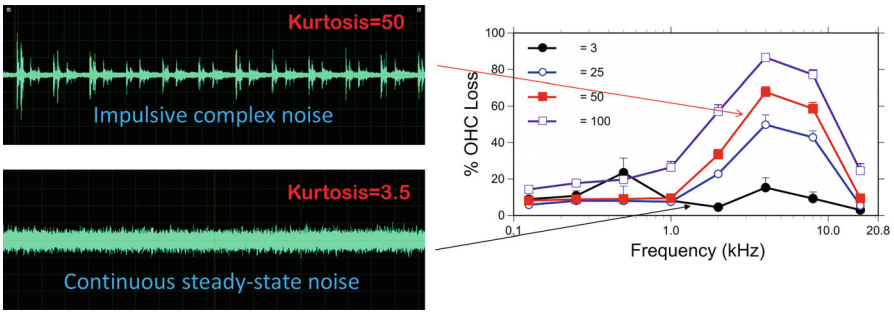



**Fig. 3** The group mean percent outer hair cell loss (% OHC loss) for groups exposed to the $\beta(t) =$ 3, 25, 50, or 100 complex noise at 97 dB SPL for 19 days. The right panel of the figure was adapted from Qiu et al. (2013)

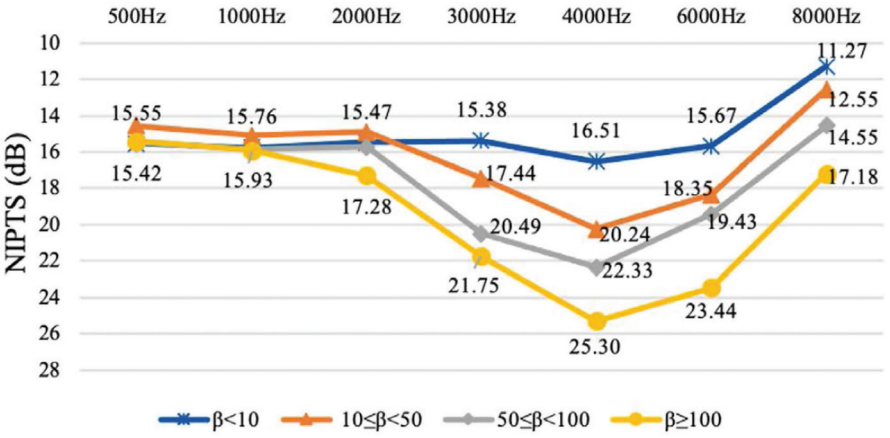



**Fig. 4** The relationship between noise-induced permanent threshold shift (NIPTS) and kurtosis category for the cumulative noise exposure in the range of 85 to 100 dBA·year. (Adapted from Zhang et al. 2021b)

kurtosis differentiates the degree of hearing loss caused by complex noise with the same energy but different temporal structures.

An epidemiological study was conducted in China by Zhang et al. (2021b) to investigate the relationship between occupational noise-induced hearing loss (NIHL) and the kurtosis of noise. The data of workers from six industrial settings were collected. Figure 4 shows the relationship between noise-induced permanent threshold shift (NIPTS) and kurtosis category for the cumulative noise exposure in the range of 85 to 100 dBA·year ($n = 1962$). The NIPTS at 3, 4, 6, and 8 kHz were sorted according to the level of kurtosis. Put another way; the NIPTS increased with the increase of kurtosis level, which is consistent with the results of animal experiments (as shown in Fig. 3). It follows that kurtosis, as an adjunct to the energy metric, is a potentially effective metric for assessing hearing loss caused by complex noise exposure.

## Kurtosis Metric Needed in the Evaluation of Hearing Hazards Associated with Industrial Noise Exposure

Zhang et al. (2021a) analyzed hearing measurements and shift noise exposure data for 2333 workers from 34 industries in China. These workers were exposed to various types of noise ranging from 85 to 100 dBA, and their exposure durations ranged from 1 to 35 years. Figure 5 shows the age, exposure duration, noise level exposed, and noise kurtosis distributions of workers.

The effects of noise level and kurtosis on the measured NIPTS for two exposure durations are shown in Fig. 6. For the D $\leq$ 10-year group, the measured NIPTS increased with the increase of kurtosis at similar noise levels. For the D $>$ 10-year group, the effect of kurtosis on measured NIPTS was no longer as pronounced as it was during the first 10 years of exposure. However, the measured NIPTS in the highest kurtosis group ($K_4$) were still significantly higher than those in the other three lower kurtosis groups. The results show that the kurtosis value plays a more important role in evaluating the NIHL of workers with exposure time less than or equal to 10 years compared to workers with exposure time greater than 10 years. However, when the kurtosis value is greater than a certain value ($\beta > 75$, namely, high impulsive noise), the damage to hearing caused by this type of noise is not only rapid but also continues with the extension of noise exposure duration.

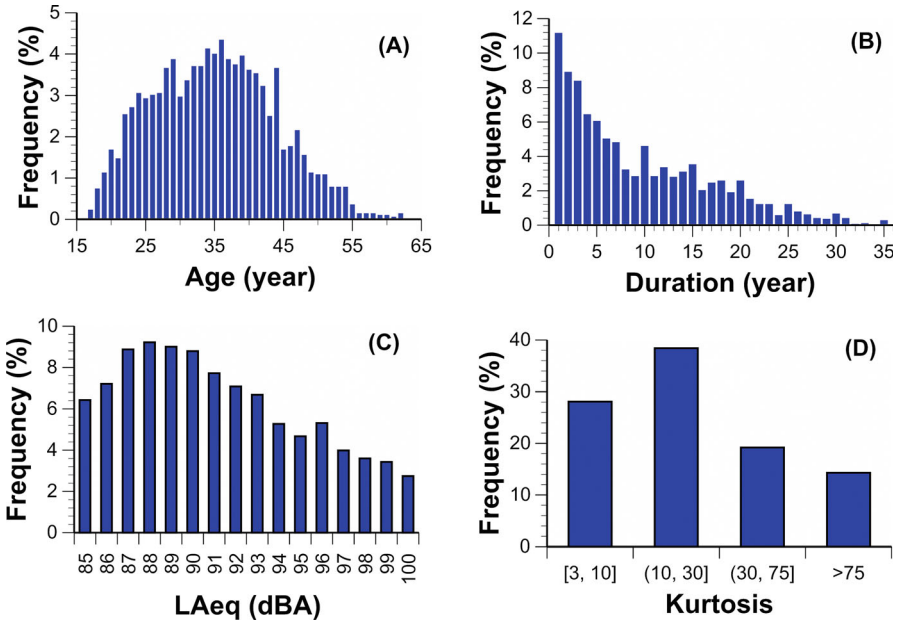



**Fig. 5** Distributions of (**a**): age; (**b**) exposure duration; (**c**) A-weighted equivalent sound pressure levels ($L_{Aeq,8h}$); and (**d**) kurtosis level in the 2333 noise-exposed workers. (Adapted from Zhang et al. 2021a)

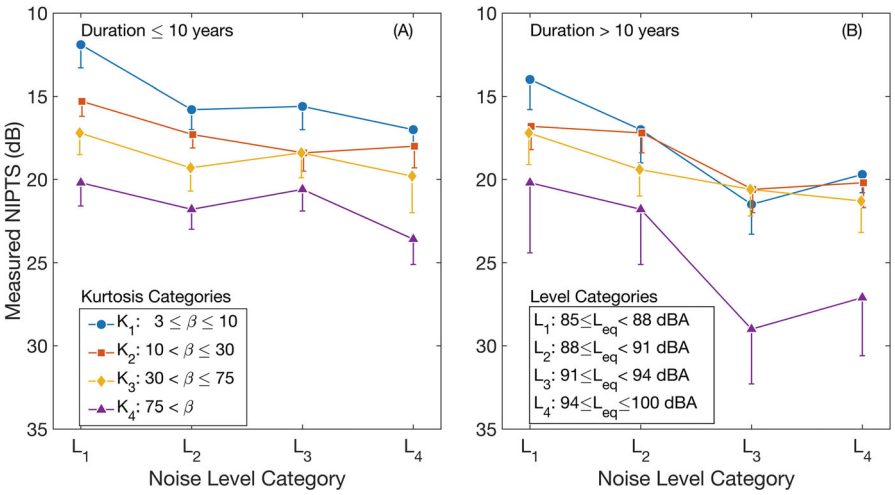



**Fig. 6** The estimated marginal means of the measured NIPTS at each kurtosis value across test frequencies for four noise level bins in two different exposure durations: (**a**) less than or equal to 10 years. (**b**) greater than 10 years. Error bars, standard error of the means. (Adapted from Qiu et al. 2020)

## How to Use the Kurtosis Metric in the Evaluation of NIHL in Industrial Settings

In order to practically use the kurtosis metric for hearing protection purposes in industrial settings, the following issues need to be addressed:

1. How to measure noise kurtosis in industrial settings?
2. What is the optimal window duration for kurtosis calculation?
3. How should kurtosis be normalized to converge over work shifts?
4. How to use the kurtosis metric in NIHL evaluation?

### How to Measure Noise Kurtosis in Industrial Settings

To obtain noise kurtosis, the noise waveforms must be recorded. The recording needs to have high fidelity: at 48 kHz sampling rate, 32-bit resolution in our studies, and a microphone capable of measuring the high-intensity sound levels typical of occupational noise exposures.

### What Is the Optimal Window Duration for Kurtosis Calculation

Tian et al. (2021) investigated the correlation between window duration for kurtosis computation and the accuracy of NIHL prediction using a Chinese industrial

database ($n = 2110$). Workers were divided into high- and low-kurtosis groups based on kurtosis values calculated over different window durations. Two algorithms were used to find the optimal window duration. Algorithm 1 applied the Mann-Whitney U test to compare the difference in group hearing threshold levels to obtain the optimal window duration to best distinguish these two groups. Algorithm 2 used a support vector machine to predict the NIHL under different window durations and found the optimal one by testing the performance of the model through the area under the curve. The results showed that 60 s was an optimal window duration.

## How Should Kurtosis Be Normalized to Converge Over Work Shifts?

A sampling rate of 48 kHz is used for noise recording. The length of the calculation window for the recorded noise signal kurtosis is 60 s, and the shift-long noise is calculated without overlap. For a sample of n values, kurtosis can be calculated as:

$$\beta_j = \frac{1}{n}\sum_{i=1}^{n}(x_i - \overline{x})^4 \Big/ \left(\frac{1}{n}\sum_{i=1}^{n}(x_i - \overline{x})^2\right)^2$$

where $x_i$ is the $i$th value of noise amplitude and $\overline{x}$ is the sample mean.

The mean of the measured kurtosis values, $\beta_j$, at every 60 s is calculated

$$\beta_{\mathrm{N}} = \frac{1}{N}\sum_{j=1}^{N}\beta_j$$

and used as the kurtosis of noise exposure ($\beta_{\mathrm{N}}$).

## How to Use the Kurtosis Metric in NIHL Evaluation?

An algorithm using kurtosis to adjust the measured equivalent A-weighted 8-h noise exposure level ($L_{\mathrm{Aeq,8h}}$) proposed by Goley et al. (2011) is applied. The formula is as follows:

$$L'_{Aeq,8h} = L_{Aeq,8h} + \lambda \log_{10}\frac{\beta_N}{\beta_G}$$

where $\beta_{\mathrm{N}}$ is the kurtosis of noise, $\beta_{\mathrm{G}}$ is the kurtosis of Gaussian noise, which is equal to 3, and $\lambda$ is the adjustment coefficient. The Goley's model uses noise kurtosis to directly correct the measured noise energy. The greater the kurtosis, the greater the amount of correction. Goley et al. (2011) found the adjustment coefficient $\lambda = 4.02$ when using the chinchilla data from Auditory Research Laboratory, SUNY Plattsburgh.

Using NIPTS at test frequencies 3, 4, and 6 kHz ($\mathrm{NIPTS}_{346}$) as the indicator of NIHL, Zhang et al. (2022) found the optimal kurtosis adjustment coefficient ($\lambda = 6.5$)

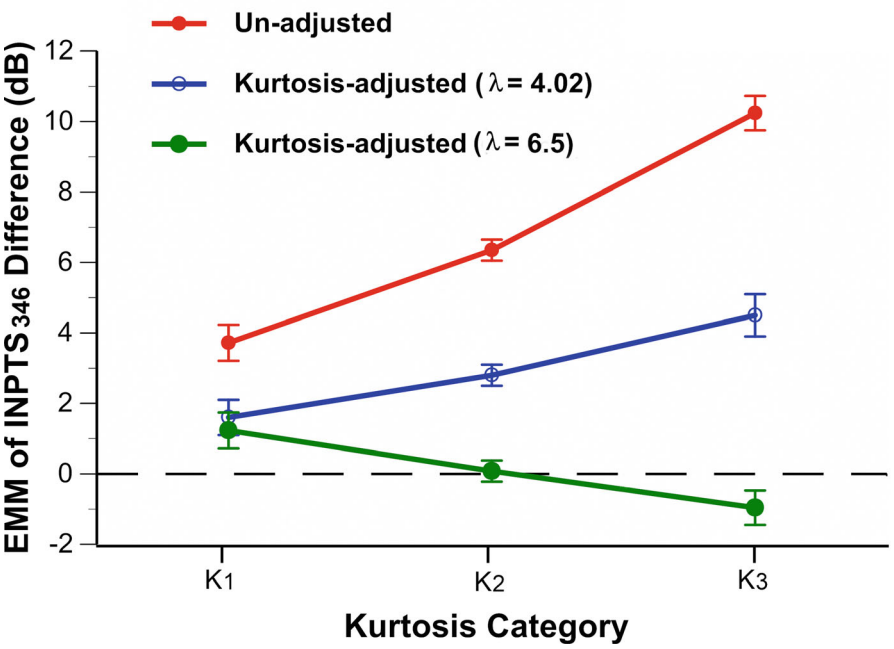



**Fig. 7** The estimated marginal mean of underestimated $NIPTS_{346}$ by ISO 1999:2013 model for three kurtosis levels under un-adjusted and kurtosis-adjusted noise levels. Error bars: standard error. The kurtosis value ranges are $K_1$:3 $\leq \beta_N \leq$ 10; $K_2$: 10 $< \beta_N \leq$ 50; $K_3$: $\beta_N >$ 50. (Adapted from Zhang et al. 2022)

for Goley's formula based on noise recordings and hearing measurements collected from workers ($n = 2601$) in industrial settings. Figure 7 displays the estimated marginal mean (EMM) of underestimated $NIPTS_{346}$ for each kurtosis level before and after kurtosis adjustment. It can be seen from Fig. 6 that the unadjusted kurtosis levels (red line) exhibit an increasing error in the prediction of $NIPTS_{346}$ as kurtosis increases. The EMM of underestimation can reach as high as 10 dB for the high kurtosis group. The kurtosis adjustment using $\lambda = 4.02$ (from the chinchilla model) provides a better estimate than the unadjusted noise levels but still exhibits an increasing error as kurtosis increases (blue line). The best-fit value, $\lambda = 6.5$ (from human data), exhibits an error of about $\pm 1$ dB (green line) as the kurtosis level changes. The kurtosis-adjusted $L_{Aeq,8h}$ with an adjustment coefficient of 6.5 can predict the high-frequency NIHL more accurately.

## Conclusion

Because non-Gaussian complex noise is common in the industry, the temporal characteristics of noise (i.e., kurtosis) must be taken into account in evaluating occupational NIHL. The temporal characteristic (i.e., kurtosis) can be used to adjust

noise level (via Goley's formula). The greater the value of kurtosis, the larger amount of the adjustment, which means the higher risk of hearing loss. The results of the epidemiological study demonstrated that these adjustments are feasible and necessary.

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# Information Use and Disturbance Across Sound, Light, and Smell in Animals

39

Annebelle C. M. Kok

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**Abstract**

Anthropogenic disturbance is affecting the sensory environment. Acoustic, visual, and chemosensory modalities are changing, impacting information use of animals across taxa. In order to effectively mitigate these impacts, we need to understand how disturbance in those sensory modalities are similar, and where they diverge. Animal information use follows the general principle of a cue that is produced by a sender and perceived by a receiver, which may alter its behavior in response to that cue. At the same time, acoustic, visual, and chemosensory modalities diverge in whether parts of this process involve active production and in whether the receiver is actively creating cues that it can perceive. Sensory pollution can take place through masking and changes in signal processing speed, which overarches sensory modalities. Getting better understanding of how acoustic, visual, and chemosensory pollution affect animal information use will help unravel the general mechanisms underlying the effects of sensory disturbance on information use in animals.

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Disturbance · Behavior · Sensory modality · Sound · Light · Chemosensory

## Introduction

The increased industrialization of human society is changing acoustic, visual, and chemosensory modalities in the environment (Frisk 2012; Naidu et al. 2021; Sanders et al. 2021). Since the Industrial Revolution, ambient noise levels in both terrestrial and aquatic environments have increased due to human-generated sounds (Miksis-Olds and Nichols 2016; Slabbekoorn 2019). At night, the sky and the ocean are illuminated through artificial light sources (Falchi et al. 2019). Chemical pollution has changed the composition of lakes and rivers (Håkan Olsén 2011).

These changes have an impact on information use in animals. For instance, increasing noise levels lead to masking of intraspecific communication (Erbe et al. 2016; Slabbekoorn et al. 2010), artificial light at night increases conspicuousness of prey to predators (Delhey and Peters 2017; Santos et al. 2010), and decreases in seawater pH levels affect the use of chemosensory cues by fish (Ferrari et al. 2011). On top of that, animals often have to cope with an accumulation of impacts from multiple sensory disturbances at the same time (Crain et al. 2008). This highlights a need for understanding how animals are disturbed by various forms of sensory pollution.

While the sensory modalities in which animals operate can vary, there are general mechanisms that underlie the way animals use these sensory cues (Sebeok 1977). A cue or signal is – inadvertently (cue) or intentionally (signal) – produced by a sender or abiotic factor and is received by a receiver (Laidre and Johnstone 2013). The receiver processes the cue and potentially adjusts its behavior. This process can only take place if the sender is able to produce the cue and the receiver is able to perceive the cue produced by the sender – which depends in part on the spectrum of sensitivity and the intensity threshold (Fay 1974). If one of those conditions is not met, the cue is not perceived. Humans, for example, cannot perceive the UV-color cues present on flowering plants, because UV light falls outside of their spectrum of perception (Silberglied 1979).

Similar to sensory information use, disturbance through sensory pollution is governed by mechanisms that overarch sensory modalities (Dominoni et al. 2020). The knowledge that various forms of sensory information use and sensory pollution operate through the same mechanisms gives us the opportunity to examine how they are the same, and more importantly, where they diverge.

## The Sender

The sender can either actively – by production – or passively – by reflection – use a sensory modality to send a signal. Acoustic and chemosensory signals are typically actively produced (Bossert and Wilson 1963; Busnel 1977). An animal produces a

call or releases a chemical compound (e.g., pheromones (Karlson and Lüscher 1959)) that transmits through the environment to hopefully reach a receiver. In addition to direct transmission, chemosensory signals can also be left to linger on other objects, giving the sender the possibility of sending the signal without actively being present to send it (Bossert and Wilson 1963). Terrestrial animals often use this kind of chemosensory signaling to mark territory boundaries (Campos et al. 2017). There are only few species that actively produce light to send a signal, most of them in the marine environment (Haddock et al. 2010). Light is usually used passively, by changing the way the light reflects off a body through coloration, through movement, or by changes in posture (Hailman 1977). For instance, cichlids (*Labeotropheus fuelleborni*) use body coloration to signal their potential to be a mate (Pauers and Mckinnon 2012).

In contrast to light, sound and chemicals are only rarely used passively to send signals. One example of passive acoustic signaling comes from plant-animal interactions. Bat-dependent plant species have evolved leaf shapes that can act as acoustic reflectors (Schöner et al. 2016). In the marine environment, no such intentional passive acoustic signals are known. Rather, fish and cephalopod prey may inadvertently reflect the sound of echolocating toothed whales, thereby betraying their position (Au 1997). Given the ubiquity of sound use in the marine environment, and the relative absence of light, one would expect more species to use acoustic reflection for sending signals. Are there biological trade-offs that negate the benefit of such a signal type? Or are we simply unaware of such examples?

## The Receiver

Signal reception is usually passive, although animals can actively enhance their probability of receiving a signal. They can do so in multiple ways, for instance, by moving away from potentially masking noise sources, by turning toward the sound source (Dent et al. 1997), by having directional sensing organs (Ebata et al. 1968; Von Békésy 1964), or by moving toward a location with better signal reception, i.e., closer to the source or to a point with higher propagation potential (Brumm and Slabbekoorn 2005). One notable exception to this rule is echolocation (Griffin 1953; Schevill and McBride 1956). An animal that is echolocating actively sends a signal in order to be able to act as a receiver of the echo of that signal (Au and Simmons 2007). In a similar fashion, light-producing animals may use their light source to see by. For instance, lanternfish (*Myctophidae*) are thought to use their bioluminescence to illuminate prey (de Busserolles and Marshall 2017; Herring 1985). Would a similar mechanism be possible in the chemosensory modality? And if so, is it used by animals?

## Disturbance

Both sender and receiver of sensory signals seem to follow general underlying rules, regardless of the sensory modality in which they operate. Senders usually actively produce a signal, while receivers passively receive it. However, upon closer

examination, there are differences between sensory modalities, especially for sending visual signals and receiving chemosensory signals. We should take these similarities and differences into account when we examine the mechanisms of sensory disturbance.

## Masking

One of the ways in which acoustic disturbance can affect animal information use is through masking. Masking is defined as a decrease in signal-to-noise ratio (SNR), such that the signal or cue is no longer discernible from the background noise (Luther and Gentry 2013). A specific requirement for masking in mammals is that the signal occurs in the same frequency band as the noise. If the signal is outside of the frequency range of the noise, it will be detectable regardless of the noise intensity. It is unclear if a similar mechanism applies to fish and invertebrates (Hawkins and Popper 2017; Popper and Hawkins 2019). Masking has been studied extensively for acoustic disturbance, both for terrestrial and aquatic animals (Brumm and Slabbekoorn 2005; Erbe et al. 2016; Popper and Hawkins 2019; Shannon et al. 2016; Slabbekoorn et al. 2018). Many vertebrates evolved coping mechanisms for masking in natural conditions, which work to varying degrees for anthropogenic disturbance (Brumm and Slabbekoorn 2005), but similar mechanisms seem to be absent in invertebrate taxa (Brumm and Zollinger 2011).

Similar to acoustic disturbance, visual disturbance can also act via masking. In aquatic environments, increasing turbidity leads to increased visual noise, which reduces the possibility to detect visual signals. Increased water turbidity mainly impacts visual predators, who are less likely to successfully catch their prey in more turbid waters (Ortega et al. 2020). Water turbidity may increase from material runoff or resuspension of the sediment through extreme rainfall and storms, which are predicted to increase due to climate change (Cózar et al. 2005; IPCC 2014; Schälchli 1992; van der Sluijs et al. 2011; Weyhenmeyer et al. 2016). In contrast to turbidity, artificial light at night (ALAN) will decrease masking of visual cues, which will make prey more conspicuous to visual predators. Nocturnally foraging waders, for example, changed their foraging effort to artificially light experimental sites and changed from mixed foraging strategies to visual foraging (Santos et al. 2010). Similarly, changes in the visual background can increase conspicuousness of prey that use the background as camouflage (Cuthill et al. 2019).

Chemosensory disturbance is often used in the field of ecotoxicology, where changes in behavior are used as biomarkers to assess the toxicity of a chemical compound (Carvan et al. 2008). Masking is one of the mechanisms through which this behavior changes, when an introduced chemical compound masks existing chemical cues from con- or heterospecifics. For instance, female sword-tail fish (*Xiphophorus birchmanni*) lost their preference for conspecific male olfactory cues in polluted stream water (Fisher et al. 2006). However, another possible explanation for the results is a physiological change in the olfactory receptors of the fish due to the pollutant (Fisher et al. 2006; van der Sluijs et al. 2011). In that case, it would not

be masking, but a temporary impairment of the receptive organ that created the effect. Alternatively, it has been proposed that chemical pollutants may change the structure or abundance of chemical cues, which could lead to a decrease in signal detection (Fisher et al. 2006; van der Sluijs et al. 2011). This would decrease the signal to noise ratio by decreasing the amount of signal available. The alternative, an increase in the noise, has not been proposed but could be possible, depending on the specificity of the olfactory receptors and the nature of the chemical disturbance.

## Changes in Signal Processing Speed

Acoustic disturbance has been shown to alter the response latency of multiple animal taxa to cues (Chan et al. 2010; Chan and Blumstein 2011; Simpson et al. 2015, 2016; Voellmy et al. 2014). More specifically, sound can distract animals (Chan and Blumstein 2011; Rosa and Koper 2018), potentially leading to increased predation (Simpson et al. 2016). However, this is not always the case. In some studies, acoustic disturbance was actually found to have an opposite effect: Instead of a longer response latency, the hoatzin bird (*Opisthocomus hoazin*), dwarf mongooses (*Helogale parvula*), prairie dogs (*Cynomys ludovicianus*), and three-spined sticklebacks (*Gasterosteus aculeatus*) all showed a shorter response latency under influence of noise (Karp and Root 2009; Kern and Radford 2016; Shannon et al. 2016; Voellmy et al. 2014). The precise mechanism that operates behind this dichotomy is not well understood. It could be related to the perceived risk of the disturbance, but this has not been tested explicitly (Frid and Dill 2002).

Distraction by visual disturbance has been suggested for nocturnal insects (Owens et al. 2020). Phototactic insects show – sometimes fatal – attraction to artificial light sources, rendering them more vulnerable to predation risk by visual predators that operate close to the light source. Nocturnal moths were equally easily caught with light traps while being exposed to playback of bat sounds as with silent light traps, suggesting absence of the typical acoustic avoidance flight to bat sound when exposed to artificial light (Hügel and Goerlitz 2020). However, conclusive evidence is lacking, as the study did not test whether avoidance flights would be reduced for moths exposed to light and sound as compared to sound alone. Furthermore, the attraction of insects to artificial light sources at night is not typical of the distraction mechanism as seen for acoustic disturbance, where the disturbance influences response latency to another, biologically relevant, cue. The question of changes in signal processing speed induced by visual disturbance therefore remains open.

Chemosensory pollution often leads to changes in signal processing speed of chemosensory cues. Air pollution impaired the learning of olfactory cues in honeybees (*Apis mellifera*) (Leonard et al. 2019). Background odors distracted moths (*Manduca sexta*) from the direction of scented flowers, increasing the time they took to localize the flower (Riffell et al. 2014). Chemosensory pollution might also affect responses to other sensory cues. However, care must be taken to ensure that changes in behavior due to chemosensory pollution are not due to physiological changes caused by the chemical compounds.

## Conclusion

Anthropogenic disturbance can have many different forms, but the ways in which sound, light, and chemosensory pollution impact information use of animals could be similar. Comparing animal information use from a mechanistic perspective helps unravel existing knowledge gaps. If sending and receiving a signal typically works via the same principle, why then do we sometimes see differences between sensory modalities in how signals are used? Is visual perception using bioluminescence as widely used as echolocation, and does it have the same functionalities? Is there a chemosensory equivalent, and if not, why not? Understanding how these mechanisms overlap and where they diverge will give further insight into animal information use, and more importantly, into the disturbance of that information use. This knowledge is necessary to increase the potential for effective mitigation of disturbance in a sensory world.

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# Section V

# Hearing and Auditory Assessment: Fishes

# Finite Element Modeling of Effects of Acoustic Stimulation on Fish Otoliths

40

Chong Wei and Robert D. McCauley

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**Abstract**

Early work has shown that exposure to high-intensity sounds can damage the sensory epithelia of fish ears and impair their hearing. However, experiments suggest that when the acoustic stimulus impinges on the ears from different directions, the damage level varies significantly. Therefore, it was hypothesized that the angle of the incident sound might play a part in the propensity to cause damage. To test the hypothesis and gain insight into the differential movement of the otolith relative to the epithelia, finite element models based on microCT data of the saccular otolith of a bight redfish (*Centroberyx gerrardi*) was built, which modeled the otolith motion to acoustic stimulations from the horizontal and vertical directions. Sound pressure level, relative displacement, acceleration, and shear stress at the surfaces of the otolith and macula (water-otolith boundary) were compared. The results suggest that the incident angle of the sound and the

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geometry of the sulcal groove play an important role in causing different damage levels to the fish ears.

**Keywords**

Otolith · Macula · Finite element modeling · MicroCT · Relative motion · Hearing · Displacement · Acceleration

## Introduction

The bony fishes have two ears with three pairs of otoliths: utricular (lapillus) otoliths, lagenar (asteriscus) otoliths, and saccular (sagittal) otoliths. In most fish species (except for the otophysans), the saccular otoliths are the largest ones. The three otolith organs are involved in hearing, which can be responsive to waterborne sound energy (Popper and Schilt 2008; Braun and Grande 2008; Schulz-Mirbach et al. 2013). The hair cells of the sensory epithelium are connected to the dense calcareous otolith through the otolithic membrane (a gelatinous and fibrous matrix) and lie in sulcal groove along the otoliths. When an acoustic stimulus is imposed on the otolith, due to the different densities between the otolith and macula, a differential motion is generated which causes deflection of the hair cells (Popper and Fay 1999). This deflection creates neural impulses in the innervating eighth cranial nerve that is proportional to the amplitude and direction of hair cell bending (Flock and Wersall 1962; Hudspeth et al. 2000).

Previous behavioral research has shown that continuous noise and pure tones can cause temporary hearing threshold shift (TTS) to fish (Popper and Clarke 1976; Popper et al. 2005). Exposure to high-intensity pure tones (e.g., over 180 dB re 1 μPa) for a period of time may damage the sensory hair cells of the fish ears and eventually impair fish hearing (Scholik and Yan 2001, 2002; Hastings et al. 1996; Smith and Monroe 2016). For example, in the study of McCauley et al. (2003), a group of pink snappers (*Chrysophrys auratus*) were held in cages and were exposed to multiple shots from air gun signals (a short, sharp, low-frequency sound made by venting high-pressure gas). They reported damage to parts of the ears of the exposed fish. The damaged sensory hair cells were not repaired or replaced up to 58 days after air gun exposure. McCauley et al. (2007) studied hearing damage in the tropical gold band snapper (*Pristipomoides multidens*) using fish caught in commercial traps set on the seabed. They also reported that significant damage to the ears of fish was observed when the fish was exposed to a seismic pass.

These experiments were followed up by partly contradictory results of effects on the hearing abilities by the air gun signals. In the first scenario which was the experiment conducted by McCauley et al. (2003), the exposed fish was located at 3 m depth in 10 m water depth, and then a single 20 cui (cubic inch, e.g., 1000 cui = 16.4 L) air gun at 5 m depth passed within 5 m off the fish. There were no constraints on the direction of arrival, horizontal and near vertical primarily from the surface bounce. It was reported that the fish ears were severely damaged, specifically on the dorsal and ventral sides of the macula (McCauley et al. 2003).

By contrast, in the second scenario, the exposed fishes were in small cages held at 7 m depth in 60 m of water. The seismic source, at 5 m depth, passed within 45–800 m of the cages (McCauley et al. 2008). In this case, the only vertical arrival would be the bounce off the seafloor; however, as the seabed was hard limestone, a poor reflector, the energy arriving at the fish from below would have been low compared to the direct horizontal arrival and distorted given the phase shifts from the seabed. Given the shallow nature of the source and receiver, the dominant arrivals were near horizontal with little to no energy arriving from above the fish. The damage level on the fish ears was much lower compared to the first experiment. The received air gun signal pressure metrics were similar between the two experiments. One possible explanation for the partly contradictory results could be the difference in fish species. There could be variations among the species in terms of the shape of the otolith (Dahl et al. 2020; Jenkins et al. 2022). However, the fish species in both scenarios have similar otolith shapes (the caudal groove is narrow and mostly horizontal, located at the midpoint of the otolith); therefore, we hypothesize that the angle of the sound impinging on the saccular otolith affects otolith motion patterns and causes different damage levels to the hair cells.

Here micro-computed tomography (microCT) scanning was performed on the saccular otoliths of a bight redfish (*Centroberyx gerrardi*) and constructed finite element (FE) models to simulate the saccular otoliths' response to the acoustic stimulations from the horizontal and vertical directions. This imaging-based finite element modeling (FEM) has been widely used to study the mechanisms involved in sound production, propagation, and reception in the heads of several animal species, including toothed whales (Aroyan 2001; Cranford et al. 2014; Wei et al. 2017, 2018a, b), baleen whales (Cranford & Krysl 2015; Tubelli et al. 2018), and marine fishes (Salas et al. 2019a, b; Krysl et al. 2012; Wei et al. 2022). It has become an important complementary tool for traditional experimental methods due to its advantage of studying the complex acoustic processes as sounds interact with animals' anatomical structures. Using the modeling techniques, this study computed the motion of saccular otoliths in the presence of horizontal and vertical stimulations, respectively. The sound pressure levels (SPLs), displacement, acceleration, and the shear stress on the surface(s) of the saccular macula and otolith were compared to test our hypothesis, especially with regard to damage to certain regions of the saccular macula observed in earlier studies and the previous partly contradictory results of effects on the hearing abilities of anthropogenic noise.

## Methods

### XCT and MicroCT Scans

Medical CT scanning on the full body of bight redfish specimens was firstly performed. The specimens were purchased from a fish market and were scanned in 3D by X-ray computed tomography (XCT) using a Siemens SOMATOM Definition

AS medical scanner, as shown in Fig. 1a. The scan was conducted using 0.6 mm spiral acquisition, and the slice increment was set as 0.1 mm.

However, after examining the data, it was found that the resolution of the CT data was not high enough to provide detailed geometry information of the saccular otolith (e.g., sulcal groove). Therefore, a dissection was conducted to cut off the otolith within the bony capsule for microCT scanning. The microCT scanning was performed at the Australian Resources Research Centre (ARRC, Kensington, Western Australia) using an Xradia VersaXRM-500 microCT scanner. The Xradia VersaXRM-500 provides high-resolution 3D X-ray imaging with submicron

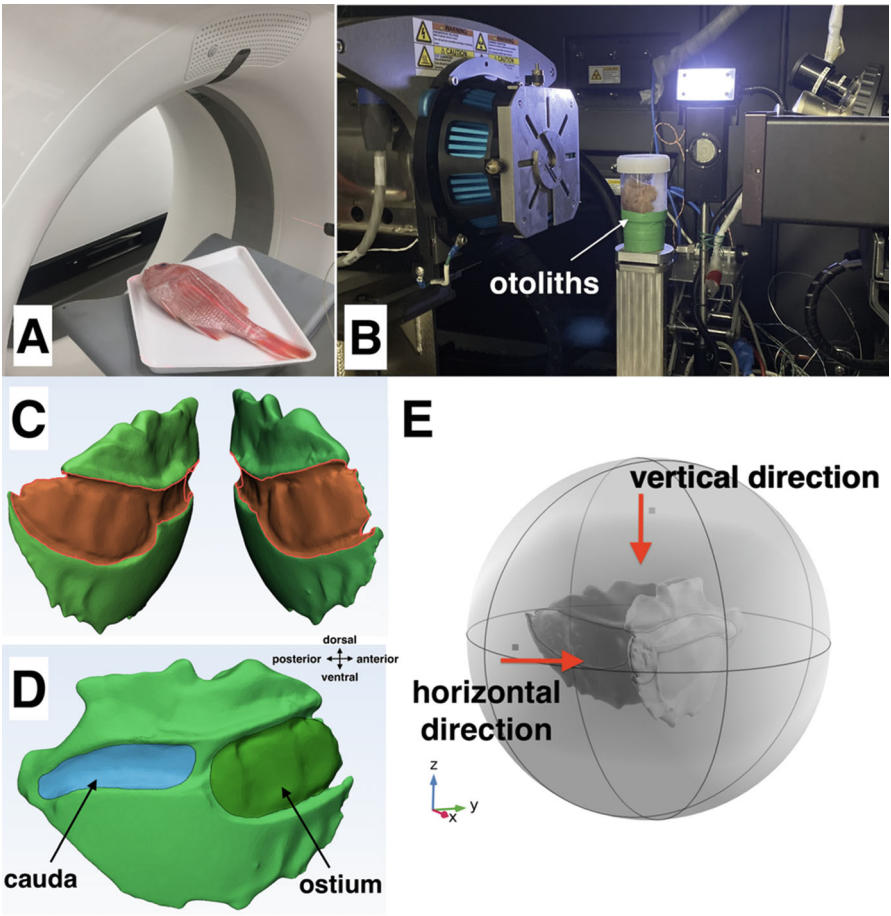



**Fig. 1** CT imaging scans and model reconstruction. (**a**) Medical CT scanning on a full body of fish. (**b**) MicroCT scanning on the fish otoliths within the bony capsule. (**c**) 3D reconstruction of a pair of saccular otoliths of a bight redfish. (**d**) 3D reconstruction of the left saccular otolith. The saccular otolith is parted into cauda and ostium: the light blue region represents the cauda and the dark green region represents the ostium. (**e**) The geometry of the 3D FE model

resolution. It facilitates nondestructive observation of the internal structure of objects, such as the saccular otoliths. The otoliths were firmly fixed in a plastic container during the scanning, as shown in Fig. 1b. In the microCT scanning, the isotropic resolution was set as voxel size 29 μm, and the power setting was set as 80 kV × 87 μA. Cone beam reconstruction was performed on the object, and the data was saved on a hard drive as a matrix of 1000 × 1024 pixels per image in TIFF format. The images were converted to DICOM format later and imported into Horos™ (Horos Project, Geneva, Switzerland) for data analysis and 3D geometrical model reconstruction. Figures 1c and d display the 3D reconstruction of the saccular otoliths. The regions in orange are the sulcal grooves. The otoliths were exported as a stereolithography (STL) file after optimization and modification (e.g., smoothing, removing overlapping and self-intersections, etc.).

## Finite Element Model Construction

STL files were imported into the COMSOL Multiphysics modeling software (Stockholm, Sweden) for finite element analysis (FEA) and corresponding data analysis. The microCT data only contains the capsule with otoliths in it, to ensure the saccular otoliths maintain the correct orientation as in an actual live fish; the orientation of the saccular otoliths was carefully adjusted based on the XCT data of the full fish. According to Schilt et al. (2011) and Krysl et al. (2012), most fish tissue has similar acoustic properties (e.g., density and elasticity) to the surrounding water; therefore a water sphere was set outside of the otoliths while the acoustic stimulation arrived, as shown in Fig. 1e. The diameter of the sphere was set as 30 mm, and the center of the sphere was located at the approximate center of the otoliths. A low-reflecting boundary condition (Bérenger 1994) was applied to the sphere, simulating the sound transmitting in free space with minimum boundary reflections. The sound source was located at a distance of 6 mm from the edges of the otoliths on the two main axes (it was assumed that the fish was ideally parallel to the water surface when the incident sounds arrived). As there were no boundary reflections from the water sphere surrounding the otolith, the experimental setup emulated long-range stimulation of the otolith system, while allowing a simplified and realistic model structure. A force was applied at the source to create vibration from the horizontal and vertical directions, creating two cases (horizontal case, H case; vertical case, V case) for later comparisons. The frequency was swept from 25 to 1600 Hz in steps of 25 Hz according to the fish hearing sensitivities and the signal characteristics of the air gun signals (Fay 1988; McCauley et al. 2003; Popper et al. 2005).

COMSOL's free mesher was used to map the entire model using the tetrahedral mesh. COMSOL has built-in detection for small features and narrow regions in a complex geometry. Based on the mesh refinement results, the size of the element was set between 0.12 and 1.6 mm, whereas the minimum wavelength of the impulse sound was 0.93 m in the model, ensuring the solution had sufficient numeric precision.

The finite element Pressure Acoustics-Frequency Domain module coupled with solid mechanics and an Acoustic-Structure Boundary was applied for the models. When acoustic waves propagate within the liquid medium, the longitudinal waves can be written as follows:

$$\frac{1}{\rho_0 c_s^2}\frac{\partial^2 p}{\partial t^2} + \nabla\cdot\left(-\frac{1}{\rho_0}\nabla p\right) = 0 \tag{1}$$

where $p$ is the sound pressure, $\rho_0$ is the density (kg/m$^3$), and $c_s$ is the speed of sound (m/s). The density $\rho_0$ is included in the equation because of its variations in different computational domains within the model. For the harmonic solution of the pressure $p(\boldsymbol{x}, t) = p(\boldsymbol{x})e^{i\omega t}$, with the angular frequency $\omega$ (rad/s), Eq. 1 can be simplified to the following:

$$\nabla\cdot\left(-\frac{1}{\rho_0}\nabla p\right) - \frac{\omega^2 p}{\rho_0 c_s^2} = 0 \tag{2}$$

While the acoustic waves interact with the solid medium, the multiphysics coupling provides and assigns the boundary conditions for the two-way acoustic-structural coupling between the liquid (e.g., water) and the solid (e.g., fish otoliths and bones). The automatic boundary condition set the boundary load $\boldsymbol{F}$ (force/unit area) on the otoliths to the following:

$$\boldsymbol{F} = -\boldsymbol{n}_s p \tag{3}$$

where $\boldsymbol{n}_s$ is the outward-pointing unit normal vector seen from inside solid domain, while on the liquid side the normal acceleration experienced by the liquid is set equal to the normal acceleration of the solid, meaning that:

$$-\boldsymbol{n}_a\cdot\left(-\frac{1}{\rho_0}\nabla p\right) = a_n \tag{4}$$

$$a_n = (\boldsymbol{n}_a\cdot\boldsymbol{u})\omega^2 \tag{5}$$

where $\boldsymbol{n}_a$ is the outward-pointing unit normal vector seen from inside the liquid, $a_n$ is normal acceleration of the solid surface in the liquid domain boundary, and $\boldsymbol{u}$ is the calculated harmonic displacement vector of the solid structure.

The following parameters were set for the models: (a) sound speed values of water outside the otoliths as 1483 m/s, whereas the density values were set as 998 kg/m$^3$. (b) The density and sound speed of the otoliths were set as 2970 kg/m$^3$ and 3000 m/s, respectively (Schulz-Mirbach et al. 2018, 2019; Salas et al. 2019b; Campana and Thorrold 2001). Young's modulus of the otolith was set as 66 GPa (Ren et al. 2013), and Poisson's ratio was set as 0.3, a value commonly used for bone (Tubelli et al. 2014).

## Results and Discussion

Some fish species can detect sound pressure, and all fish species are primarily detectors of particle motion, which can be expressed as displacement, velocity, or acceleration (Popper and Hawkins 2018). The otoliths act as elastic objects to vibrate when they are driven by the incident stimulus, creating displacement between themselves and the coupled hair cells on the macula lying in the sulcus. Figure 2a compares the distributions of SPL and displacement on the saccular otolith surface when the sound wave (frequency = 500 Hz) arrived in the H and V cases. For the SPL distribution, most energy was distributed in the anterior side of the otoliths in the H case, whereas most energy was distributed in the dorsal side of the otoliths in the V case.

Figure 2a also shows that the displacement distribution between the H and V cases is different. In the H case, higher displacement regions were visible around the anterior side of the otoliths and the direction of the displacement toward the midline of the fish. By contrast, in the V case, higher displacement regions were visible around the dorsal side of the otolith and the direction of the displacement toward the

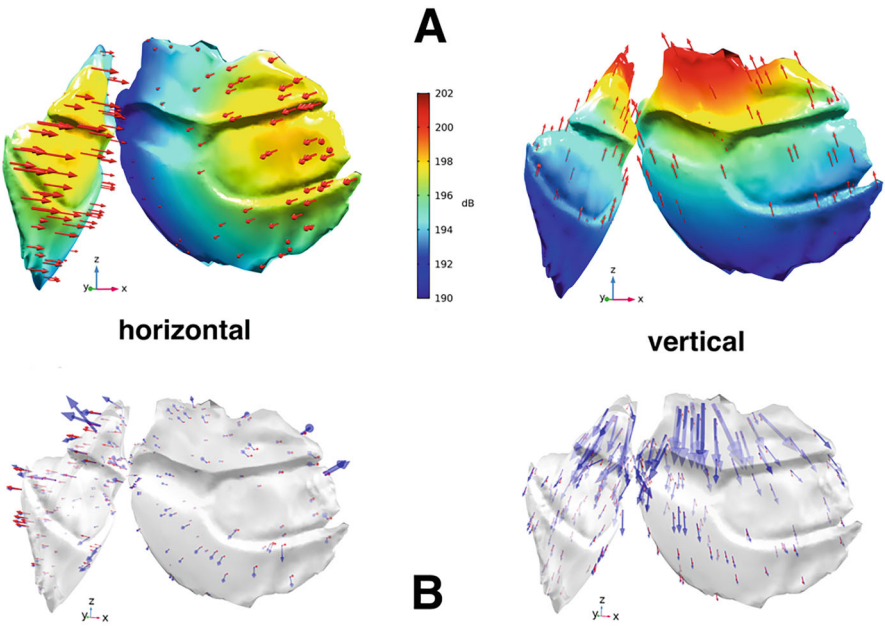



**Fig. 2** Distributions of SPL and displacement of the otolith, as well as the accelerations of the membrane and otolith in two cases (frequency = 500 Hz). The arrows indicate both the direction and magnitude. (**a**) The sound field and displacement field on the otoliths. (**b**) The acceleration distributions of the membrane and otolith. The blue arrows represent the accelerations of the membrane surfaces, and the red arrows represent the accelerations of the otolith surfaces

dorsal side of the otoliths. In both cases, the accelerations of the membrane (blue arrows) were higher than those of the otolith (red arrows). Moreover, the directions of the acceleration between the otolith and coupled membrane were also different in the H and V cases (Fig. 2b). The results demonstrated that the relative motion between the membrane and dense otolith differed between H and V cases when the sound waves arrived.

To further compare the differences of SPL, displacement, and acceleration on the sulcal groove in the two cases, the modeling data was derived to plot the 2D contours on the sulcal groove, as shown in Fig. 3.

Figure 3 displays roughly symmetrical contours. On both otoliths, as sound arrived from the horizontal direction, the regions of maximum SPL and displacement exhibited in the ostium regions. Beyond the maximum regions, the magnitudes decreased along the posterior direction. By contrast, as sound arrived from the vertical direction (Fig. 3a), the maximum regions of SPL and displacement were observed around the dorsal side of the grooves.

The contours of the acceleration on the otolith surface were similar to those of the displacement on the otolith surfaces in the two cases (Fig. 3c). Both cases had significantly different contours compared to the corresponding acceleration contours on the macula (water-otolith) surfaces, showing the differential movement between the saccular macula and dense otolith. For the acceleration distributions on the macula boundary (Fig. 3d), the regions of maximum acceleration were displayed at the edge of the rostrum in the horizontal stimulation. By contrast, the regions of maximum acceleration were distributed at the dorsal and ventral edge of the cauda when the sound wave arrived in the vertical direction.

To better visualize the relative motion between the otolith and saccular macula in two acoustic stimulations, two receiving points were placed on the water-otolith boundary. One point was located in the cauda region (PointC), and one point was located in the ostium regions (PointO) (Fig. 4). The results in Fig. 4 showed that accelerations on the water-otolith (macula) boundary (dashed lines) of corresponding points were all higher than those on the otolith (solid lines) in the two cases, suggesting that the dense otolith lagged behind relative to the motion of the macula due to its inertia (Dijkgraaf 1960; Popper and Lu 2000). When the sound arrived from the vertical direction, the differentials of the acceleration between the otolith and macula of both points were greater than those in the H case, suggesting stronger differential motions between the otolith and macula in both the cauda and ostium regions in the vertical sound arrival. The differential motion deflected the hair cells and, if amplitudes were high enough, may cause damage to the ears. A stronger differential motion might cause more severe damages. It could be one of the reasons causing the partly contradictory results from the previous experiments (McCauley et al. 2003, 2008), in which the severe damages were found in the experiment allowing vertical acoustic stimulation. The fish species used here and in the experiments of McCauley et al. (2003, 2008) had common otolith shapes with a long mostly horizontal caudal groove along the inside otolith side which was comparatively narrow and deep at the midpoint of the otolith. The different shape of otoliths and their caudal groove among other fish species may mean these results do not translate to all fish species.

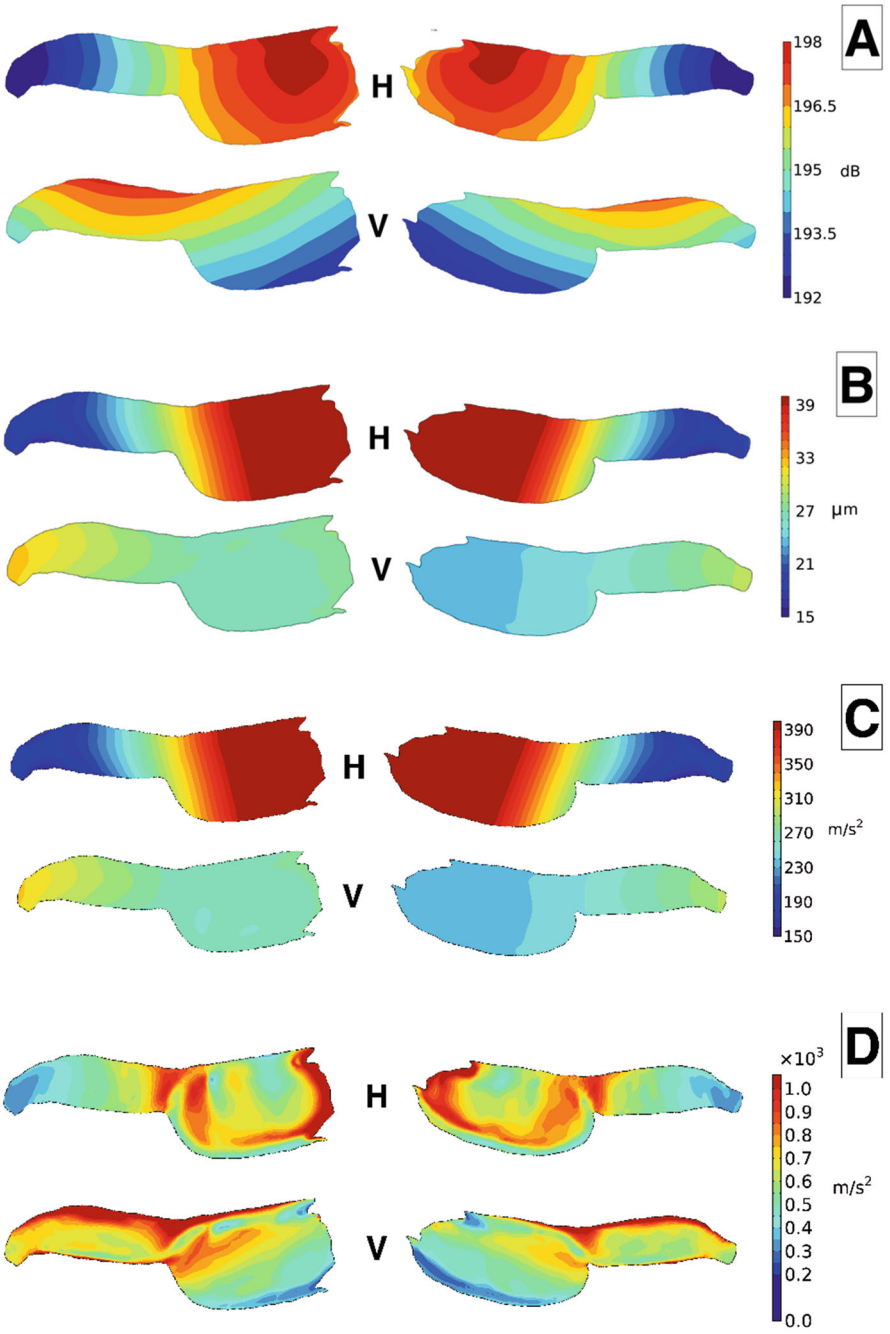



**Fig. 3** Contours of SPL, displacement, and acceleration on the sulcal grooves in the two cases. The color bars show the absolute values. (**a**) SPL contours on the otolith surfaces. (**b**) Displacement contours on the otolith surfaces. (**c**) Acceleration contours on the otolith surfaces. (**d**) Acceleration contours on the macula surfaces

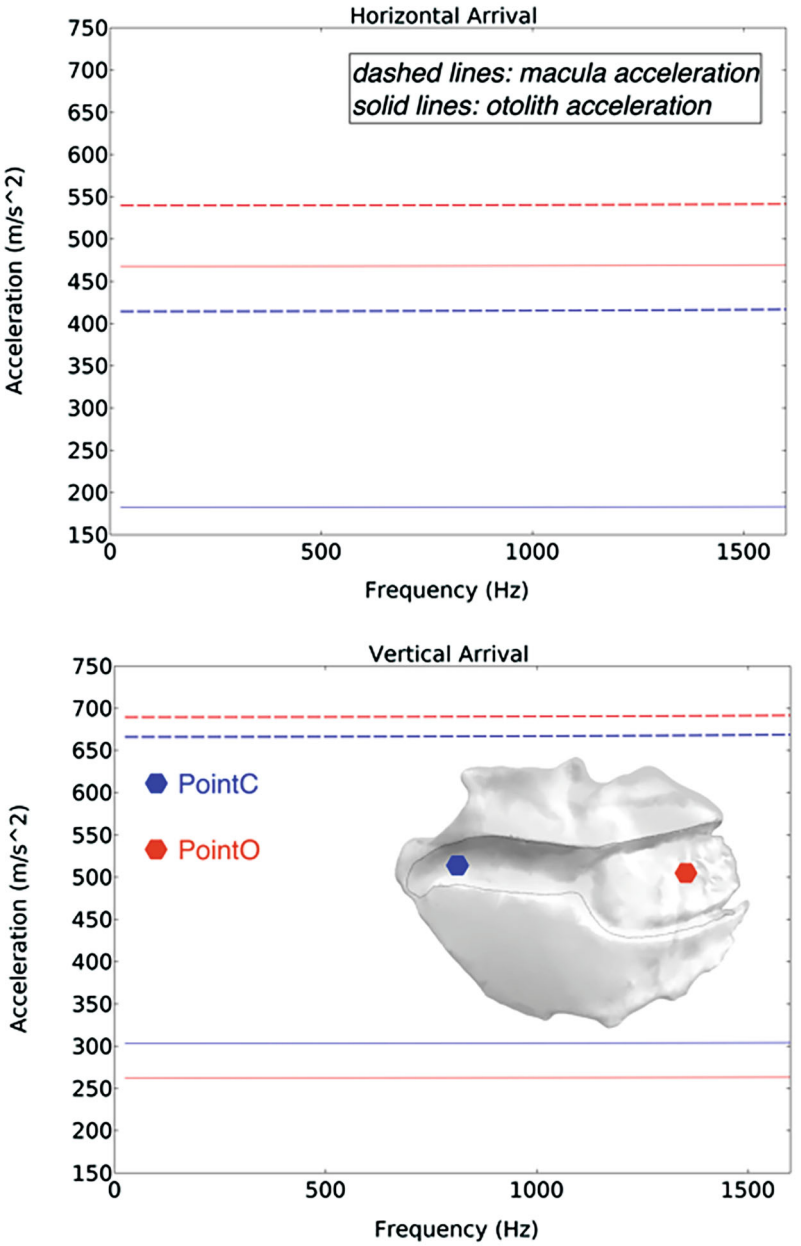



**Fig. 4** Comparison of accelerations of the otolith and macula (water-otolith boundary) from 25 to 1600 Hz in the two cases. The blue lines represent the data from the point in the cauda region, and the red lines represent the data from the point in the ostium region. The dashed lines represent the accelerations on the surface of the water-otolith, and the solid lines represent the accelerations on the surface of the otolith

The otolith is connected to the macula with the hair cells in it through the elastic membrane; the ciliary bundles could be either deflected or compressed as the sound wave impinges from different angles (Schulz-Mirbach et al. 2019). Here a force analysis was attempted to study the stress exerted on the surface of the otolith in the two cases. Figure 5a shows the shear stress distribution on the surfaces of both otoliths. As the sounds arrived from the vertical direction, significantly higher-stress regions were exhibited in the cauda regions of both otoliths. The results matched well with the spatial mapping of the hair cell damage on the exposed fish from the previous experiment (McCauley et al. 2003, Fig. 5b), in which most damage occurred in the cauda region closer to the dorsal/ventral edges when an exposed fish experienced air gun signals with an almost equal vertical to horizontal energy component.

According to the morphometric results obtained from the microCT data, the groove was relatively deep (~0.6 mm) and narrow (~2.4 mm) when looked at from the top and shallow and long (~16.6 mm) when looked at from the front. Because of this biological configuration of the otoliths, when an acoustic stimulation arrives from the horizontal direction, it will tend not to drive the membrane into the sulcal groove edges since there is more space for it to move. By contrast for stimulation in the vertical direction, there is not enough space for the membrane to go except to push against the inside of the sulcal groove. Therefore, the results from this study suggested that fishes with an otolith shape similar to the bight redfish might be able to endure sound energy arriving in the horizontal direction rather than the vertical direction. Again, the results are most likely species-specific; a different fish species with a distinct otolith shape may lead to different results (Smith et al. 2022).

The model in this study only simulated a pair of the saccular otoliths immersed in the water. However, the saccular otoliths of the bight redfish sit inside an otic capsule

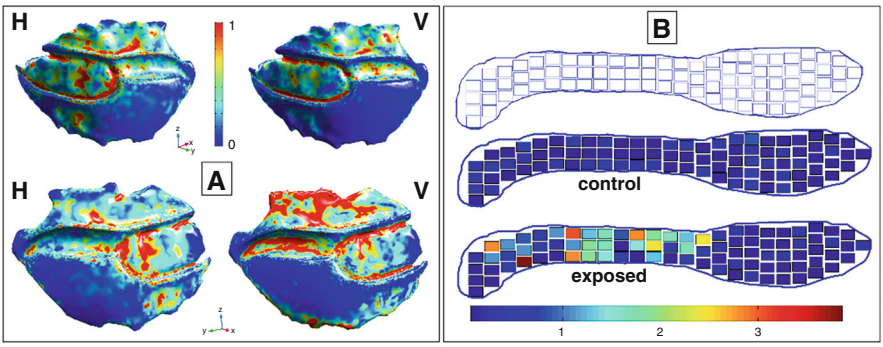



**Fig. 5** (**a**) Comparison of the relative shear stress on the otolith surfaces in the two cases, the color bar represents the magnitude of the normalized shear stress. (**b**) The spatial mapping of the hair cell damage on the pink snapper (*Pagrus auratus*) macula from air gun signals with an almost equal vertical energy component compared with horizontal. (Modified from McCauley et al. 2003, Fig. 2.62). The color code is a scale of severity as the mean number of ablated cells in that grid space (gross hair cell damage only). The middle image shows the results of the control fish, and the lower image represents the results of the exposed fish

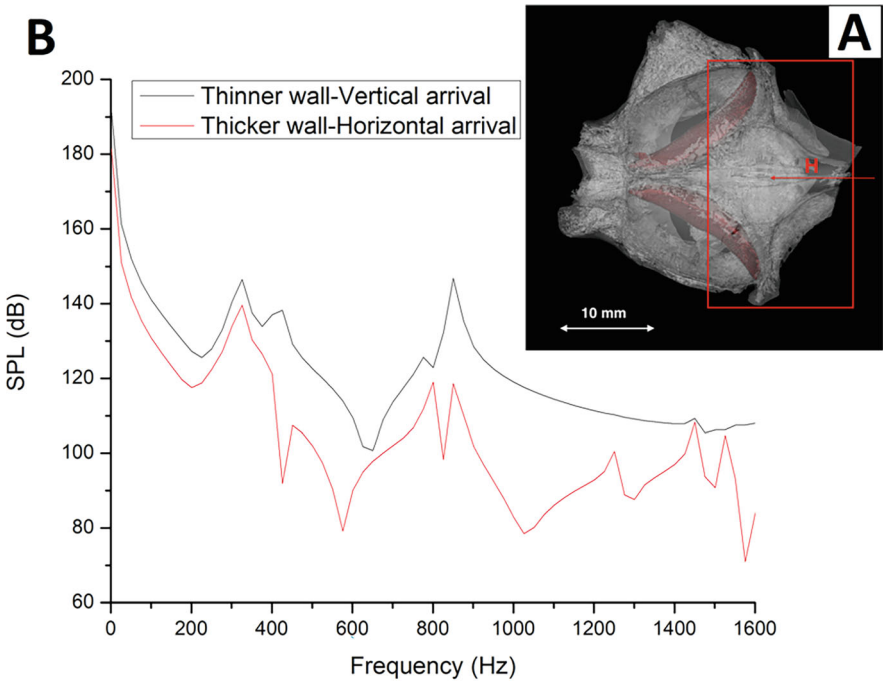



**Fig. 6** (**a**) 3D reconstruction of the capsule (white) containing the saccular otoliths (red). The transparency level represents the different thicknesses. (**b**) Comparison of the SPLs inside the capsule when sound penetrates through the thinner wall (vertical direction) vs. thicker wall (horizontal direction)

(Fig. 6a). The wall of the capsule in the head of the bight redfish is thicker in the horizontal direction compared to the vertical direction, as circled in Fig. 6a. The thickness of the wall could cause different attenuations when the sound penetrates through the capsule. A test was run by setting a receiving point in the center of the capsule and compared the SPLs of the receiving point when sounds arrived from the horizontal direction through a thicker wall and the vertical direction through a thinner wall (Fig. 6b). The results showed that the SPL was reduced in the entire frequency range from 25 to 1600 Hz if the sound penetrated through a thicker capsule wall in the bight redfish's head. This may further explain why less damage to the hair cells was found when the air gun signals were induced from the horizontal direction in the previous experiments.

As noted above, it is important to note that the morphology of fish otolith (e.g., size, geometry, the angle it is tilted in the capsule, etc.) varies among species. Therefore, it would be interesting to model other otolith shapes for this topic and consider what the shapes mean for hearing in different species. The morphology of the otolith may also change in the same species due to different ages and food sources. In addition, there are many other variables, such as the angle of arriving sound energy (caused by the location of the sound source, depth of water, bottom

type), the source of the stimulus (pulsed sound vs. pure tone), the hair cell orientation patterns, etc. It is important to consider these variables in any future studies to gain a deeper understanding of the effect of acoustic stimulus on fish otoliths.

## Conclusions

In this study, 3D finite element models based on the microCT data of a pair of saccular otoliths were constructed to investigate the motion of the otoliths in the presence of acoustic stimulations from the horizontal and vertical directions. The results show that a sound wave that arrives from different directions causes different magnitudes of the differential motion between the sensory macula and otolith in the fish ears. This motion difference plus the geometry of the otoliths plays an important role in terms of how the fish otoliths may respond to incoming sounds. The results from this work may potentially aid researchers in studying the hearing of some key fish species (e.g., commercially important species) and may feed into environmental policy with regard to the management of intense noise sources such as the noise from petroleum seismic survey air gun arrays, which are often in conflict with commercial fisheries.

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# Dopamine in the Auditory System of Vocal Toadfishes: Potential Adaptation for Noisy Aquatic Environments

41

Kelsey N. Hom and Paul M. Forlano

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**Abstract**

Anthropogenic noise negatively impacts the lives of aquatic animals including their ability to reproduce and survive. However, some animals are still able to thrive within these increasingly noisy environments and therefore must have some adaptations that allow them to attend to salient signals within noise. All major classes of vertebrates have cholinergic neurons in the hindbrain that function as auditory efferents, meaning that they project to the inner ear. Even less is known about dopamine neurons that project to the inner ear, but studies in rodents have proposed that one role of both types of efferents is to protect the ear from damage due to noise. Therefore, research in aquatic systems should focus on animals that use acoustic communication in reproductive contexts, like the oyster toadfish (*Opsanus tau*), which are regularly exposed to high levels of anthropogenic sounds. Examining the role of modulatory neurotransmitters, like dopamine, within the auditory system, will provide important insights to understanding neural adaptations for survival in environments exposed to increasing levels of anthropogenic noise.

**Keywords**

*Opsanus tau* · *Porichthys notatus* · Oyster toadfish · Plainfin midshipman · Efferents · Neurochemical protection · Acetylcholine · Acoustic communication · Teleost fish · Sound

## Introduction

Noise in the aquatic environment has only increased in the last 60 years (Ellison et al. 2012) and may come from various sources including impact pile driving, seismic air guns, vessels, and wind turbines (Slabbekoorn et al. 2010; Popper and Hawkins 2019). Anthropogenic noise is typically low in frequency and broadband, which is within the hearing ranges of many aquatic mammals and fishes (Slabbekoorn et al. 2010; Popper and Fay 2011) and can have an adverse effect on an animal's ability to communicate, feed, reproduce, and evade predators (Codarin et al. 2009; Simpson et al. 2016; Shannon et al. 2016). Intense noise can induce temporary and even permanent threshold shifts in both fishes and marine mammals (Amoser and Ladich 2003; Nachtigall et al. 2003; Finneran et al. 2005; Brietzler et al. 2020). Researchers have used several experimental methods to understand the negative effect of noise by measuring behavioral changes and hearing sensitivity. While these are necessary and valid measures for understanding the effects of noise on an animals' hearing sensitivity, these measures do not account for biochemical changes in the brain, which in turn modulate sensory, motor, and cognitive functions. More recently, research has focused on the role of neurochemical modulation of the inner ear and

central auditory areas that could be involved in filtering noise and attenuating acoustic damage.

All major classes of vertebrates have neurons in the hindbrain that contain the neurotransmitter, acetylcholine, and project to the inner ear (Köppl 2011). The proposed function of these efferents includes protection of the inner ear from overstimulation, increasing signal to noise ratio, and detection and attention to sounds (Tomchik and Lu 2006; Maison et al. 2013; Boero et al. 2018; Guinan 2018). Thus, one level where filtering noise occurs is in the inner ear via modulatory feedback from the brain (Tomchik and Lu 2006; Köppl 2011). In addition to cholinergic efferents, fishes and mammals are documented to have dopaminergic efferent neurons, which project to the inner ear and could therefore modulate acoustic signals at the level of the peripheral auditory system (Darrow et al. 2006; Forlano et al. 2014). In rodent models, dopamine neurons in the lateral olivocochlear (LOC) nucleus inhibit cochlear afferents. LOC neurons upregulate dopamine after sound conditioning, a low-level stimulus presented prior to a damaging stimulus, which is predicted to be involved with reducing damage at the level of the auditory nerve (Niu and Canlon 2002). Reducing dopamine (DA) receptors within the cochlea can increase vulnerability of hair cell damage (Maison et al. 2012) to acoustic trauma. Damaging noise triggers DA synthesis in the LOC, and it is predicted that DA protects the inner ear from overstimulation (Lendvai et al. 2011; Wu et al. 2020). Intense noise can induce irreparable damage to hair cells, which are not able to regenerate in mammals (Bermingham-McDonogh and Rubel 2003). Apart from recent studies in toadfishes (see below and Forlano and Sisneros 2016), virtually nothing is known about the role of dopamine in modulation of the inner ear under natural environmental conditions.

The inferior colliculus, or auditory midbrain in mammals, receives input from dopamine and other catecholaminergic (e.g., noradrenergic) neurons. In mammals, most of the dopamine terminals, or puncta, in the inferior colliculus come from the A11 population of dopaminergic cells located in the diencephalon (Takada et al. 1988; Nevue et al. 2016; Batton et al. 2019; see Wilson and Apawu 2022). A recent study using fast scan cyclic voltammetry and immunocytochemistry found a decrease in dopamine release and a decrease in the inhibitory D2 receptor distribution of the external cortex and central nucleus of the inferior colliculus after noise exposure (Wilson and Apawu 2022). The exact role of dopamine within the inferior colliculus is still unknown; however, disruptions and changes to dopamine within the central and peripheral nervous system can affect auditory processing (Matragrano et al. 2012; Hoyt et al. 2019).

Virtually all studies that have investigated the function of auditory neuromodulators during noise exposure have been conducted in terrestrial (rodent) models. Thus, there is a need to understand adaptations of the central auditory system of aquatic animals in the presence of environmental sounds, specifically in fishes that have an audiometric range within the frequency of anthropogenic noise. This is especially important for animals that require sound for communication and reproduction. Some fish species leave an area when noise is present (Slabbekoorn et al. 2010); however, animals that either cannot or will not leave their given

location, most likely have adapted to live within the noisy environment. Here it is proposed that central dopamine neurons and their influence over central and peripheral auditory processing may be a key adaptation in aquatic animals to survive in noisy environments.

## Toadfishes as a Model Species

*Opsanus tau*, the oyster toadfish, is a soniferous fish that relies on acoustic signals to communicate with one another and attract mates. These animals are known to reside along the US eastern seaboard, including populations that live along urban cities like New York City (Anderson et al. 2008) where commercial vessel traffic is particularly heavy (Fig. 1). Male toadfish establish nests and emit an advertisement call, a "boatwhistle," to attract female mates (Gray and Winn 1961; Fine 1978). Boatwhistles are detected via the lateral line, saccules, and other end organs of the inner ear (Radford and Mensinger 2014; Cardinal et al. 2018; Fay and Edds-Walton 1997; Maruska and Mensinger 2015). Males are responsible for egg care and maintain high nest fidelity (Maruska and Mensinger 2009; Mensinger 2014). This may make the toadfish more susceptible to the negative effects of noise since they may not evacuate the area when intense noise is present (Faulkner et al. 2018). It is

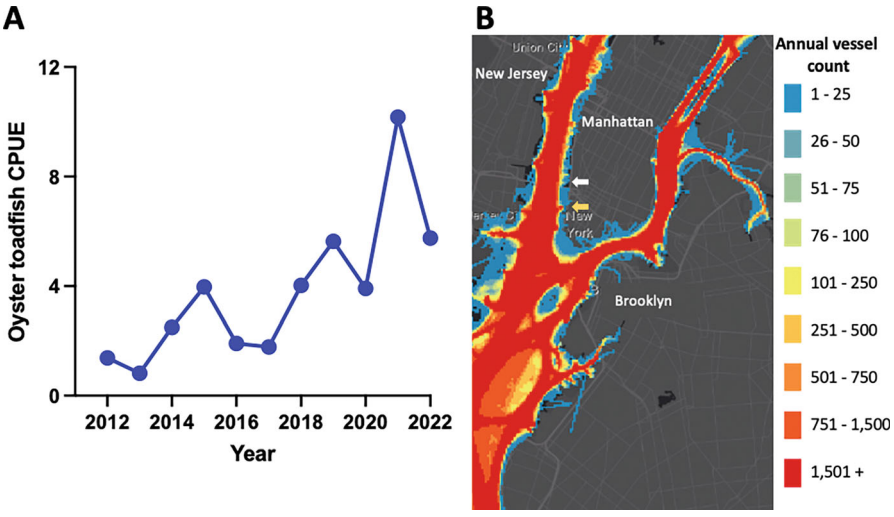



**Fig. 1** (**a**). Data from un-baited crab traps set at Hudson River Park (Pier 26/Pier 40, New York, NY; yellow and white arrow, respectively in B) and checked regularly to monitor population of fishes within the Hudson River system. Pier 26 and Pier 40 are where oyster toadfish are collected and counted by Hudson River Park's River Project, NY. The catch per unit effort (CPUE) was calculated for oyster toadfish populations (data courtesy of Hudson River Park, NY). (**b**). Automatic identification system (AIS) tracking information was collected from commercial vessels and large recreational boats. The map shows boat traffic within New York and New Jersey waters from 2021. (Image of map with boat traffic courtesy of MarineCadastre.gov and NOAA)

proposed that animals that stay in an area with increased noise may have increased stress levels and decreased reproduction (de Jong et al. 2020; Cartolano et al. 2020). However, there is evidence that oyster toadfish populations along some urban areas, like New York City, remain unchanged (see Fig. 1a), and perhaps diminished reproductive success does not occur in this species. For this reason, oyster toadfish may have adapted to these increasingly noisy environments (see Fig. 1b), in part, via modulation of their inner ears by auditory efferents in the brain.

Previous studies have used various toadfish species to examine the effects of noise on behavior and hearing sensitivity. Behavioral playback experiments conducted on the Lusitanian toadfish (*Halobatrachus didactylus*), within the same family as oyster toadfish (Batrachoididae), found that exposure to boat noise decreased the communication distance, interfered with male-to-male interactions, decreased calling rates of males (Alves et al. 2016; Alves et al. 2021; Amorim et al. 2022), and decreased reproductive success (Amorim et al. 2022). Researchers measured the effects of an outdoor musical festival on circulating cortisol in Gulf toadfish (*Opsanus beta*) and found significant increases in the stress hormone 7 h after the onset of music (Cartolano et al. 2020). Similar behavioral studies in oyster toadfish found that exposures to boat noise decreased calling behavior (Krahforst et al. 2016; Mackiewcz et al. 2021) and can potentially cause these animals to increase the amplitude of their calls, also referred to as the Lombard effect, in noisy environments (Luczkovich et al. 2016).

Auditory evoked potentials conducted on the Lusitanian toadfish found a decrease in auditory sensitivity during ship noise playbacks (Vasconcelos et al. 2007), most likely because the ship noise was within the hearing range of the toadfish. Additionally, ship noise can interfere with the detection of the boatwhistle due to masking (Alves et al. 2021). Oyster toadfish exposed to high intensity boat noise for at least 1 h experienced a temporary threshold shift (TTS) of a minimum of 8 dB SPL for at least 3 days measured by auditory evoked potentials (Rogers et al. 2020). Anthropogenic noises that cause significant threshold shifts could impact the females' ability to locate males, and therefore have a negative impact on reproductive success. Most of the literature examining the effects of noise on fishes, especially toadfishes, has focused on behavioral changes (calling patterns, amplitude) or TTS as measured by auditory evoked potentials. However, an understanding of responses at the circuit and neurochemical level to strong anthropogenic sounds may provide insight into neural adaptations that allow certain species to thrive despite increasing environmental noise.

## Recent Studies in Plainfin Midshipman Fish Support a Role for Dopamine in Peripheral Auditory Processing

The plainfin midshipman fish (*Porichthys notatus*), another toadfish in the family Batrachoididae, is found along the coast of the US Pacific Northwest and northern California and is a well-studied model system for understanding neural and hormonal mechanisms underlying vocal-acoustic behavior. Male midshipman fish

establish nests but produce an advertisement call, the "hum," lasting minutes to hours in comparison to the oyster toadfish (200–650 ms, Tavolga 1958; Mensinger 2014) to attract female mates (Brantley and Bass 1994). Females localize courting males by their hum, deposit their eggs in the nest, and return offshore, while males alone care for the eggs and developing fry (Forlano et al. 2015a). Female midshipman fish exhibit a seasonal increase in frequency sensitivity of the inner ear coincident with reproductive state (Rohmann and Bass 2011; Sisneros 2009; Sisneros and Bass 2003) that is driven by seasonal changes in hormones (Sisneros et al. 2004). In female midshipman, seasonal changes in saccular hair cell density (Coffin et al. 2012) and potassium channel expression (Rohmann et al. 2013) contribute to this plasticity. More recently, seasonal changes in dopamine innervation of the saccule (Forlano et al. 2015a; Perelmuter et al. 2019) is proposed as a major regulator of peripheral auditory sensitivity in the inner ear of this species. While seasonal changes in catecholamine fiber innervation was also demonstrated in auditory hindbrain and forebrain areas, its function is unclear (Forlano et al. 2015a).

## Projections in the Central and Peripheral Nervous System

### Cholinergic and Dopaminergic Efferent Projections to the Saccule

Cholinergic and dopaminergic neurons in the hindbrain and forebrain, respectively, send (efferent) projections to the inner ear. In teleost fishes, the octavolateralis efferent nucleus (OE) sends cholinergic projections to the saccule, the largest endorgan of hearing (SE, Fig. 2; Bass et al. 2000; Forlano et al. 2014), the other end organs of the inner ear, and lateral line (Köppl 2011; Tomchik and Lu 2006). Neurobiotin tract tracing studies demonstrate that most, if not all, OE neurons project to the saccule (Bass et al. 1994; Bass et al. 2000; Forlano et al. 2014) and these cholinergic projections to the fish saccule are inhibitory in nature (Furukawa 1981; Sugihara 2001).

Neural tract-tracing experiments have also demonstrated that dopaminergic neurons in the periventricular posterior tuberculum (TPp; diencephalon) project directly to the saccular epithelium and to cholinergic neurons in the OE (Forlano et al. 2014; Perelmuter and Forlano 2017). It is unknown whether the robust dopamine innervation onto OE somata and dendrites are inhibitory or excitatory in nature (Forlano et al. 2014); however, experiments in zebrafish (*Danio rerio*) have shown dopamine projections from TPp have an excitatory function on lateral line neuromasts (Toro et al. 2015). TPp is a conserved dopamine nucleus in the forebrain and is homologous to A11 cells in mammals (Tay et al. 2011), where these dopamine neurons project to the inferior colliculus and auditory brainstem (Nevue et al. 2016).

In midshipman fish, reproductive females show greater dopaminergic innervation on OE cholinergic somata and dendrites in comparison to winter females (as indicated by tyrosine hydroxylase immunoreactivity, a marker of dopamine synthesis (TH-ir; see Forlano et al. 2014; 2015a). Similarly, in the TPp, there is an

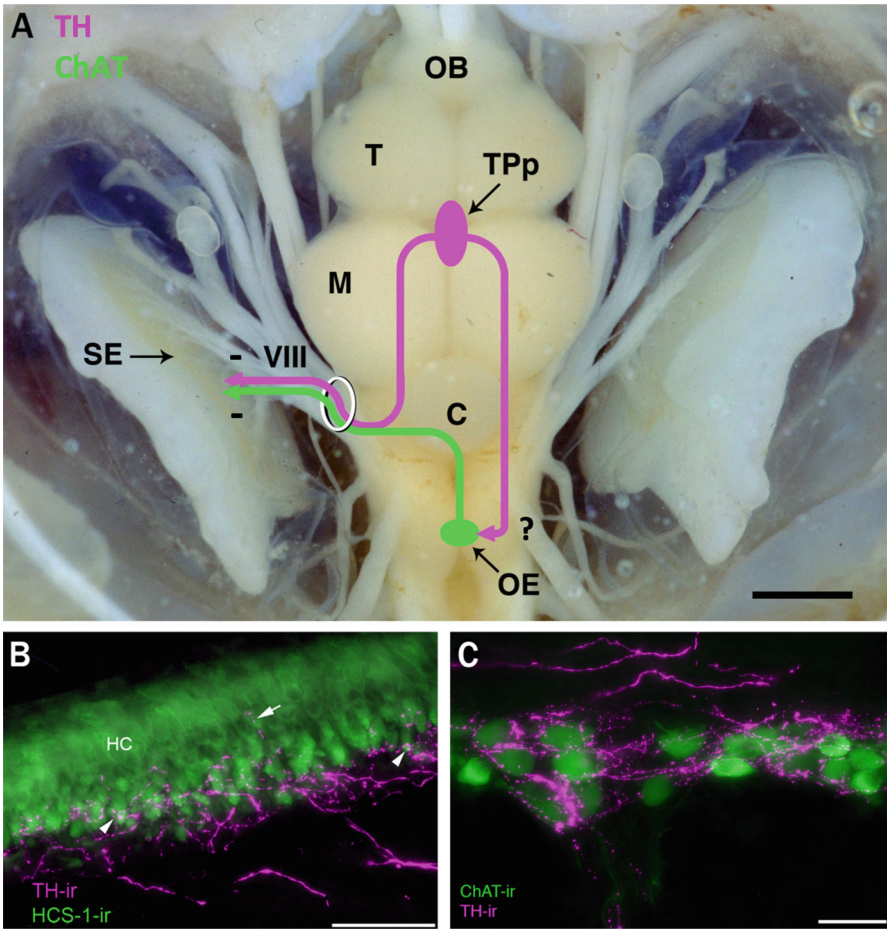



**Fig. 2** (**a**). dorsal view of the plainfin midshipman (*Porichthys notatus*) brain and inner ears. The saccule (SE) receives direct dopaminergic input (TH, magenta) from the periventricular posterior tuberculum (TPp) and receives cholinergic projections (ChAT, green) from the octavolateralis efferent nucleus (OE) via the eighth nerve (VIII) (Forlano et al. 2014; Perelmuter and Forlano 2017). Both of these inputs to the saccule are inhibitory(−) (Perelmuter et al. 2019; Furukawa 1981). The TPp also sends robust dopaminergic input to the OE but its action is unclear (indicated by "?"). (**b**). TH-ir fibers (magenta) enter the base of the SE and form punctate swellings (white arrowheads) around the base of hair cells (HC) (hair cell-specific marker, otoferlin-ir, green). (**c**). Beaded TH-ir fibers extensively wrap around OE neurons (ChAT-ir). Scale bar equals 1.5 mm in (a) and 50 μm in (b and c). Additional abbreviations in A: cerebellum (C), midbrain (M), telencephalon (T), olfactory bulbs (OB).

increase in TH-ir area of dopaminergic cells in summer females in comparison to winter females (Forlano et al. 2015a). In contrast, the saccular epithelium shows a decrease in TH-ir puncta (putative release sites of dopamine) in summer females in comparison to winter females (Forlano et al. 2015a). In the saccule, dopamine

produces a dose-dependent inhibitory effect on hair cell receptor potentials and thus decreases the sensitivity of the inner ear of midshipman fish (Perelmuter et al. 2019). Importantly, a seasonal decrease in dopamine innervation in the saccule coincides with an increase in frequency sensitivity and at the level of the hair cell in reproductive females (Perelmuter et al. 2019). Additionally, D2a receptors, known to be inhibitory in nature, are significantly decreased in the summer (Perelmuter et al. 2019). Therefore, a seasonal release of inhibitory input to the saccule will prime females for greater hearing sensitivity to locate mates in the summer (Perelmuter et al. 2019). However, more research is needed to elucidate the exact role of dopamine signaling and receptor expression in the inner ear during vocal communication in variable acoustic environments.

## Dopaminergic Cells in the TPp Are Responsive to Calls from Conspecifics

When TPp dopamine neurons are active during the summer reproductive season is an ongoing area of investigation. Importantly, sound playback studies coupled with using cFos immunohistochemistry (as a marker of neural activity) have shown that dopaminergic neurons in the TPp are responsive to conspecific vocalizations in type I (courting) males (Petersen et al. 2013), type II (sneaker) males (Ghahramani et al. 2018), and in females while attending the speaker emitting a synthesized hum (Forlano et al. 2017). Importantly, the TPp receives auditory input from the midbrain torus semicircularis via the periaquaductal gray (PAG, midbrain vocal-acoustic center) with which it shares bidirectional communication (Kittelberger and Bass 2013). In turn, the TPp projects to the nearby auditory thalamus (central posterior nucleus) and is widely connected to the basal forebrain, indicating the TPp is especially positioned to coordinate sensory-motor integration and behavior in response to acoustic stimuli (Petersen et al. 2013; Forlano et al. 2015b).

Taken together, changes in TPp dopaminergic innervation of the saccule and OE likely contribute to a seasonal increase in peripheral auditory hearing sensitivity; TPp may also function to modulate attention and motivation to salient acoustic social signals (Forlano et al. 2015a; Forlano et al. 2017; Perelmuter et al. 2019). In addition, cholinergic efferents (OE) are active in midshipman when they are calling, which is proposed to reduce stimulation of the saccule and lateral line by self-generated noise, thereby increasing sensitivity to external sounds during vocalization (Weeg et al. 2005). Cholinergic (OE) stimulation in another species, the *Dormitator latifrons,* supports its role increasing signal-to-noise ratio in auditory and lateral line systems (Tomchik and Lu 2006). Therefore, gaining a better understanding of the interplay between the dopaminergic and cholinergic efferent systems in contexts when relevant social signals need to be extracted from noisy soundscapes will be important to understanding their function in acoustic communication.

## Preliminary Studies Suggest Changes in Dopamine Synthesis Occur in the Auditory System of Toadfishes under Variable Acoustic Conditions

To better understand under what conditions there are changes in the synthesis and release of dopamine at their central and peripheral auditory targets, studies were conducted in midshipman and oyster toadfish measuring the phosphorylated version of TH (pTH), a proxy for dopamine synthesis and release (Xu et al. 1997), at the level of the saccule, OE, TS, and the source neurons (TPp) under various sound playback conditions.

Preliminary studies using female midshipman fish examined the effect of playback on catecholamine synthesis in the auditory system. Animals were exposed to 15 min of either the hum, brown noise control (within the frequency bandwidth and equivalent amplitude of the hum), or ambient control conditions. A significant decrease in enzyme activity measured by pTH-ir puncta intensity (terminals or putative dopamine release sites) was found in the hum conditions in comparison to noise and ambient controls in the OE and TPp (Hom et al. 2019). While no change in pTH-ir intensity was observed in the saccule, a decrease in pTH-ir puncta area was seen in hum condition compared to ambient control. It is hypothesized that a decrease in direct or indirect (via OE) action of inhibitory dopamine signaling to the saccule may increase peripheral auditory sensitivity when hearing conspecific vocalizations.

Another preliminary study by Hom et al. (unpublished observations) exposed oyster toadfish to either 30 min of ambient conditions or recorded boat noise used in a previous study that induced TTS after an hour-long exposure (Rogers et al. 2020). In the OE, there was a decrease in pTH-ir area ($\mu m^2$) in the noise condition in comparison to the control condition (Fig. 3). In contrast, TPp neurons showed an increase in pTH-ir area in the noise condition in comparison to controls. Similarly, in

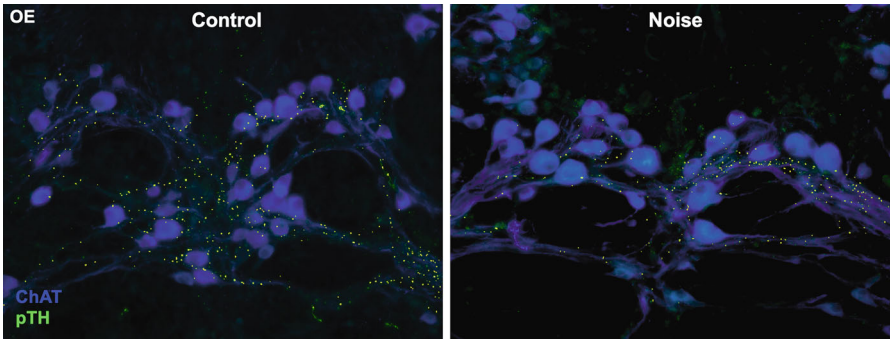



**Fig. 3** Micrographs of the OE depicts ChAT-ir somata (blue) and pTH-ir puncta (green) of both the control (ambient sound) and noise (continuous boat noise) conditions. There is significantly greater pTH-ir puncta area in the control versus the noise condition

the saccule, the noise condition showed a trend toward greater numbers of pTH-ir puncta. These results support a hypothesis that a downregulation in pTH puncta on the OE could increase inhibitory action of cholinergic projections to the saccule while an increase in pTH in the TPp in the presence of noise suggests an upregulation of dopamine synthesis in the neurons that may project to the saccule, similar to studies in rodents (Lendvai et al. 2011; Wu et al. 2020).

Overall, the release of dopaminergic inhibition of the saccule is proposed to enhance detection of salient signals in the reproductive season (Perelmuter et al. 2019). This work aligns with studies in rodents that show dopamine has an inhibitory effect on cochlear afferent neurons (d'Aldin et al. 1995, Lendvai et al. 2011, Ruel et al. 2001), and can result in preventing excitotoxicity in the cochlea (Ruel et al. 2001). Additionally, rodents exposed to intense noise exhibited an upregulation of TH-ir in the efferent auditory pathway and cochlea, confirming that dopamine is necessary for protecting against damaging noise (Wu et al. 2020; Niu and Canlon 2004). In the presence of noise, an increase of dopamine in the saccule could inhibit sensitivity to the inner ear, potentially protecting the saccule from damage. Furthermore, TPp dopamine neurons could exert additional control of saccular inhibition (and potentially other endorgans) via OE modulation, in the presence of intense noise. This type of multilayered regulation of the inner ear, could serve as an important adaptation for communication in complex underwater soundscapes.

## Ongoing and Future Studies

More research within the field of aquatic noise needs to be conducted on animals that have adapted to live in an increasingly noisy environment. While most of the research on the effects of anthropogenic noise on aquatic animals have focused on behavioral changes and potential hearing loss, it is important to also examine the neurochemical changes that might be occurring in these animals. Soniferous fish that use acoustic communication for reproduction, like the oyster toadfish, are a great model species to answer these types of research questions.

Future studies will examine neurochemical changes in both the central and peripheral nervous system of fishes, specifically measuring differences in catecholaminergic synthesis (Hom et al. unpublished observations) between fish in quiet versus noisy environments. Playback experiments testing animals in their natural environments or comparing populations from quiet and noisy soundscapes could illuminate how animals have adapted to their environment. This research would be pivotal in understanding how fish are evolving in louder soundscapes.

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# Zebrafish as a Model in Bioacoustics: Investigating Sensory Responses to Noisy Environments

42

Raquel O. Vasconcelos

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**Abstract**

The zebrafish, *Danio rerio*, has become an important model in hearing research to investigate the mechanisms underlying inner ear development, hair cell regeneration, and ototoxicity of environmental agents. However, limited information is known on its natural soundscape and acoustic conditions in captivity, as well as on the effects of noise exposure on the auditory system.

This chapter covers the present knowledge on (1) zebrafish natural soundscapes, (2) acoustic conditions in typical zebrafish housing systems, and (3) noise-induced hearing loss and inner ear damage in this model system. An overview is provided within a critical perspective considering the few existing studies and experimental limitations, while highlighting several open questions.

**Keywords**

Anthropogenic noise · Acoustic stress · Sensory adaptation · Fish

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## Introduction

Understanding how changes in the environment affect organismal responses and their adaptive potential is paramount in the current scenario of global change (Kelley et al. 2018; Fox et al. 2019). Fish species are increasingly being challenged by changes in their sensory environments due to anthropogenic activities and climate change (van der Sluijs et al. 2011; Slabbekoorn et al. 2010). Human activities are causing a near-ubiquitous and unprecedented increase in environmental noise, which has become an important component of global change (Swaddle et al. 2015).

Anthropogenic noise often overlaps with hearing sensitivity ranges of fish species (Slabbekoorn et al. 2010), thus masking acoustic cues (Vasconcelos et al. 2007) and causing a multitude of other detrimental effects such as auditory impairment, inner ear damage, physiological stress, and behavioral disruption (Nedelec et al. 2015; Kunc et al. 2016; de Jong et al. 2018). The impact of these changes on fish sensory systems, and the ultimate consequences for fitness and evolution, are largely unknown (Kelley et al. 2018).

The zebrafish, *Danio rerio* (Cyprinidae), has become an important model organism to investigate hearing and deafness (Bever and Fekete 2002; Whitfield 2002; Wang et al. 2015). This species has a typical vertebrate inner ear at the cellular level, and its anatomy and development have been intensively described (Bever and Fekete 2002; Whitfield 2002). The rapid and accessible embryogenesis in zebrafish combined with the available genetic and genomic tools for systematic gene discovery made possible to create several mutant lines with impaired development and function of the inner ear, including models for human deafness and vestibular disorders (Whitfield 2002; Nicolson 2005; Varshney et al. 2016). Besides, this model organism has been widely used to investigate the molecular mechanisms of hair cell regeneration and for drug screening of ototoxic agents (Chiu et al. 2008; Ou et al. 2010; Varshney et al. 2016).

Zebrafish is an ostariophysan species with Weberian ossicles linking the swim bladder to the inner ear, enhancing hearing especially at higher frequencies (Bang et al. 2001). Even though this species has become a reference model in hearing research, limited information is available on its adaptation to the natural soundscape. Moreover, this species is typically raised in housing facilities with elevated noise levels generated by aerators, air and water pumps, water circulation, and feeding systems, which may compromise its well-being and breeding conditions.

This chapter focuses on zebrafish as a model in bioacoustics, covering the existing knowledge on (1) species adaptation to its natural soundscape, (2) acoustic conditions in typical zebrafish housing systems, and (3) noise-induced changes in the structure-function of the auditory system.

## Adaptation to Natural Soundscapes

By listening to the aquatic soundscape or acoustic scene, fish can extract relevant biotic information about the presence of conspecifics and potential predators and prey (Popper and Fay 1993; Ladich 2013). In addition, they can also gather

information on abiotic parameters such as wind, cavitation, and water currents going through different types of substrates (Popper and Fay 1993; Lagardère et al. 1994), which is important for orientation and selection of appropriate habitats. Indeed, the soundscape provides a wider three-dimensional perspective on the surrounding world than is possible using vision, olfaction, or electroreception. Since fish can detect and process both sound pressure and particle motion, localize sound sources, and perform auditory scene analysis, the underwater acoustic scene can be extremely rich in information compared to terrestrial habitats (Fay and Popper 2000; Fay 2009; Bretschneider et al. 2013; Popper and Hawkins 2018).

Zebrafish is a small cyprinid native to the Indian subcontinent, where it is typically found in shallow, slow-flowing, or stagnant waters in Northeastern and Southwestern India, Nepal, Bangladesh, South Pakistan, and Northern Myanmar (Engeszer et al. 2007; Arunachalam et al. 2013). This species tolerates a variable range of altitudes, temperatures, and water conditions, living in water bodies with widely different substrates and levels of vegetation, from natural wetlands to man-made ditches (Engeszer et al. 2007; Spence et al. 2008; Sundin et al. 2019).

The only study that evaluated the acoustic features of zebrafish natural habitats was conducted by Lara and Vasconcelos (2019). In this study, the environmental soundscape was recorded in five distinct habitats in Karnataka, Southwest India, from stagnant or low-flow water courses to the margins of a main river. The substrate found in these locations varied from sand, lime, silt, and/or bedrock, and zebrafish were either stationary swimming in compact shoals countering the water current or disperse in loose groups.

Lara and Vasconcelos (2019) reported a considerable variation in Sound Pressure Level (SPL) from 103 dB in a low-flow small stream, up to 126 dB re 1 μPa in a main river course (Tunga river). The spectral energy was overall concentrated bellow 3000 Hz, but energy peaks varied considerably between habitats mostly due to differences in abiotic sources (volume and speed of water flow, cavitation, and sediment composition and transportation). Similarly, other studies have reported comparable variability in ambient noise spectral profiles in freshwater ecosystems (Crawford et al. 1997; Lugli et al. 2003; Wysocki et al. 2007).

Moreover, Lara and Vasconcelos (2019) identified a quiet noise window (100–2000 Hz) within the noisiest habitat (Tunga river), which matched the species best hearing range between 600 and 1000 Hz (Higgs et al. 2003, Wang et al. 2015; Monroe et al. 2016) – see Fig. 1. A quiet window within the soundscape has been identified in other freshwater habitats (Crawford et al. 1997; Lugli et al. 2003; Lugli 2010), and it is most likely generated by propagation loss and absorption of sound waves in shallow environments (Erbe et al. 2022). Interestingly, the frequency range of such quiet window exhibited the highest variability across all acoustic environments (Lara and Vasconcelos 2019). Such findings strongly suggest that zebrafish may rely on environmental acoustic cues for orientation in the wild, which should be further investigated.

Indeed, the natural history of zebrafish may lend support to the sensory drive framework proposed by Endler (1992). According to this view, perception systems should evolve to be effective within local environmental constraints (e.g., Amorim et al. 2018). Although zebrafish are not known to produce sounds, contrary to other

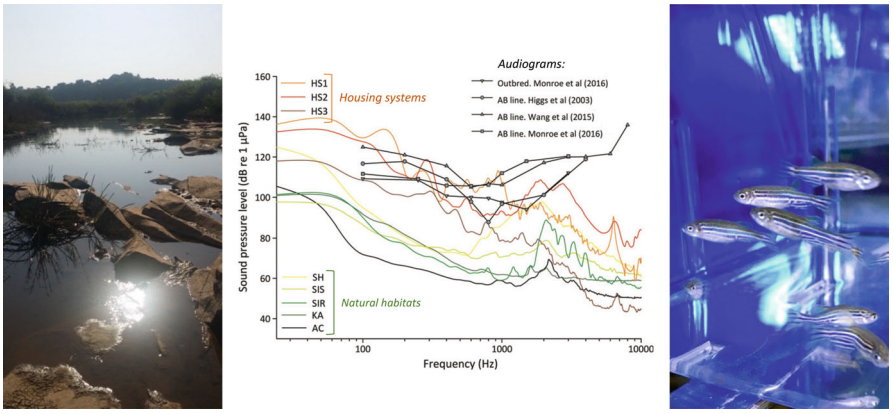



**Fig. 1** **Left**: Typical zebrafish habitat near Kaveri river, Southwest India (Karnataka state). **Middle**: sound spectra from both natural and captive noise conditions compared with zebrafish audiograms (gray bulleted lines). Mean auditory thresholds indicated are from AB wild-type line and wild-type line from Liles Tropical Fish, Inc. (Ruskin, FL). Natural habitats: SH–Shringeri, fast-flow Tunga River; SIS–Sidi Halla (sandy), low-flow stream; SIR–Sidi Halla (rocky), medium-flow stream; KA–Kallahalli, natural wells connected to Kaveri river; AC–Achacanni, shallow low-flow stream; and Zebrafish Housing Systems: HS1 and HS2 (stand-alone systems from Yakos 65, Taiwan) and HS3 (multilinking system with external pumping/filtering units from ZebTEC, Techniplast). Right: Zebrafish in typical laboratory tanks. (Adapted from Lara & Vasconcelos (2019))

cyprinids (Britz et al. 2021), they may rely on acoustic cues to learn about their surroundings (Popper and Fay 1997; Popper and Sisneros 2022). Future studies should consider playback experiments using sound recordings from different natural habitats to evaluate zebrafish preference for specific auditory cues.

Since Lara and Vasconcelos (2019) is the only study available on the zebrafish natural soundscapes, many questions remain open. For instance, particle motion is key for understanding how fish perceive their acoustic environment, including zebrafish (Popper and Sisneros 2022). Particle motion can be calculated indirectly from sound pressure measurements under certain conditions, but not in shallow waters where zebrafish and most fish species and invertebrates are typically found (Nedelec et al. 2016). Hence, future work should include direct measurements using a particle-motion sensor in the field and a comparison between pressure and particle motion across different zebrafish habitats, from low-flow courses and pools in mudflats to hilly watercourses.

Moreover, a comparison between wet and dry seasons would certainly be useful to obtain a more comprehensive characterization of the acoustic variation within a natural habitat since the information available is only based on the dry season – weak wind and no rain (Lara and Vasconcelos 2019). Both wind and rain are known to be important factors that alter underwater acoustics (Ma et al. 2005).

Finally, to investigate sensory adaptation to the natural soundscapes, upcoming research should measure auditory sensitivity of zebrafish caught in the natural

habitats. The audiometry studies available are based on animals that have been raised in captivity for multiple generations and are most likely different from the original wild strand (Harish et al. 2019, Milla et al. 2021). This is probably one of the reasons for the high variability in the auditory sensitivity curves reported in prior investigations – see Fig. 1. Fish-rearing systems imply phenotypic selection based on survivability and breeding rates under artificial environments, which leads to inevitable genetic and phenotypic changes (Fu and Yuna 2022). For instance, a recent study by Pantoja et al. (2020) showed that artificial selection of behavioral differences in acoustic startle habituation in larval zebrafish modifies whole-brain dopaminergic activity within just two generations, and that behavioral differences were carried on to adulthood with potential fitness implications.

Besides, the research on zebrafish hearing should consider not only the issue related to artificial selection in captivity that may have shaped auditory sensitivity, but also the chronic noise conditions that are typically found in housing facilities, which are discussed next.

## Coping with Noisy Housing Systems

Elevated background noise is a common problem in intensive fish farming systems, particularly recirculating systems, which are equipped with aerators, air and water pumps, blowers, filtration systems, harvesters, and feeding and maintenance machinery. Altogether, this equipment generates low-frequency noise mainly under 1000 Hz (Lagardère 1982; Bart et al. 2001; Wysocki et al. 2007) that typically falls within the best sensitivity ranges of most cultured species. However, only a few studies investigated the effects of aquaculture noise on animals, and results are not clear. Although some studies point to reduced egg viability, reproduction, and growth rates (e.g., killifish, Banner and Hyatt 1973; brown shrimp, Lagardère 1982, Regnault and Lagardère 1983; salmon, Terhune et al. 1990), others reported the absence of growth and physiological effects (rainbow trout, Wysocki et al. 2007, Davidson et al. 2009).

Zebrafish are commonly maintained in large-scale fish facilities before being used for research. Knowing that considerable number of studies focus on questions related to hearing, development, and ecotoxicology, a major concern becomes the potential noise effects of captive conditions on zebrafish well-being and underlying physiological implications (Lara and Vasconcelos 2021). Lara and Vasconcelos (2019) also characterized the acoustic properties of three distinct zebrafish housing systems, from two stand-alone systems with multiple tanks, pump, and filters integrated into a single rack to a multilinking system with multiple racks containing a water deposit connected to an external module enclosing all pumps and filters. The overall noise levels varied from 120 to 147 dB with most energy below 1000 Hz. The simpler systems with integrated filtering/pumps had higher SPL of 133–146 dB and spectral energy up to 22 dB above the species auditory thresholds. This is likely to induce significant masking effects and maybe even hearing loss, besides overall physiological stress and behavioral disruption. Contrastingly, the system with an

external water treatment module (containing filters/pump) presented lower noise levels of 120–126 dB re 1 μPa. Although the spectral energy was still well above the natural levels within the best hearing range of zebrafish (see Fig. 1), this housing system seems to be a considerable improvement to minimize noise transmission into tanks, following the recommendations by Lawrence and Mason (2012).

Building on the abovementioned, future research should test the potential implications of raising zebrafish under distinct noise environments. This may imply using playback setups to evaluate the exact contribution of the acoustic component for zebrafish well-being throughout the life cycle. This species undergoes auditory sensitivity changes with growth and aging (Zeng et al. 2021), and this might be a critical point when assessing sensitivity to noise exposure.

Furthermore, since particle motion is such a key component of fish auditory sensing, it becomes critical to consider this parameter in future research.

## Noise-Induced Hearing Loss

Building on this background, an obvious step to understand how zebrafish perceive sound and are affected by exposure to noise conditions was to rely on the Auditory Evoked Potential (AEP) recording technique to measure auditory sensitivity in this species under different acoustic conditions (Fig. 2). Breitzler et al. (2020) investigated for the first time Noise-Induced Hearing Loss (NIHL) at different amplitude levels and the underlying morphological damage of the inner ear saccular hair cells in zebrafish adults. After exposure to continuous white noise for 24 h, AEP recordings revealed noise level-dependent Temporary Threshold Shifts (TTS) up to 33 dB paralleled by hair cell loss at the highest level investigated (150 dB re 1 μPa). Recovery of the hearing function was dependent on the noise level, varying from 7 days (130–140 dB) up to 14 days (150 dB). At the loudest sound level, regeneration of the sensory tissue was observed within 7 days, which was prior to functional recovery. This study established zebrafish as an effective model to evaluate noise-related sensory loss and underlying cellular damage and recovery process.

Besides noise amplitude, temporal patterns in noise exposure are critical when assessing the impact of acoustic disturbances on organisms. Hence, a following study (Wong et al. 2022) evaluated the impact of different temporal patterns – continuous, fast and slow regular intermittent, and irregular intermittent noise on hearing loss (Fig. 2). The noise treatments consisted of short 1-s pulses interspersed with varying silent intervals similar to impulsive sounds generated in construction activities (such as pile driving) that require bottom-mounted structures (Sabet et al. 2015). Based on AEP recordings, Wong et al. (2022) identified highest threshold shifts in continuous and fast intermittent treatments, and no differences were found between regular and irregular regimes. Such sensory loss was accompanied by decreased presynaptic activity (Ribeye b protein). This study suggested that overall acoustic energy is more important than regularity when assessing the negative effects of noise on the auditory system.

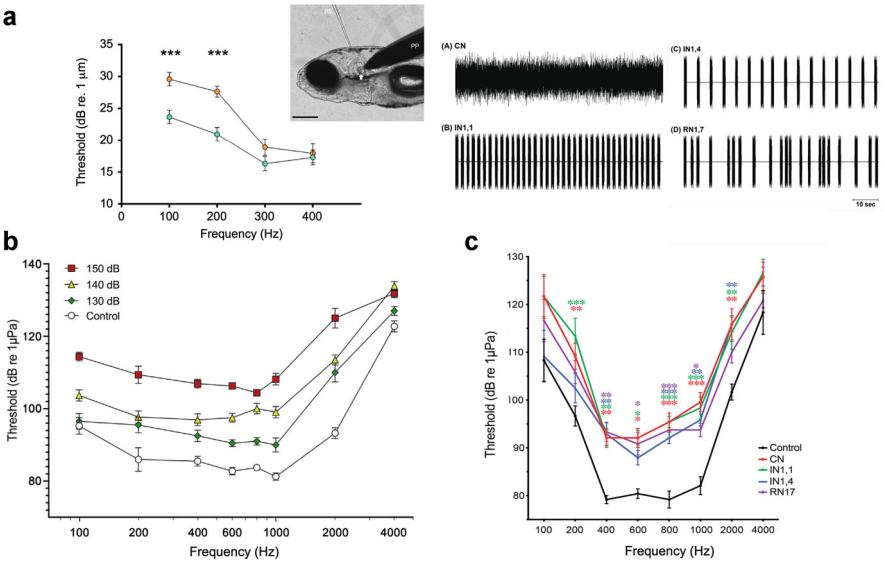



**Fig. 2** (**a**) **Top**: detail of a 5-day postfertilization zebrafish mounted in agarose for microphonic potential recordings, showing recording electrode (RE) and the stimulus probe (PP) touching the otolith. Scale bar: 200 µm. **Bottom**: microphonic thresholds from 5 dpf zebrafish from the control- and noise-treated group, showing significant differences at 100 Hz ($F1,22 = 17.60$, ***$P < 0.001$) and 200 Hz ($F1,27 = 23.84$, ***$P < 0.001$). Values are Means± Standard Error of the Mean (SEM). Based on Lara et al. (2022); (**b**) mean auditory thresholds ± SEM of zebrafish after 24 h of either lab silent conditions (control, c. 105 dB re. 1 mPa) or white noise exposure (at 130, 140, and 150 dB). Noise induced significant changes in the auditory thresholds ($F3, 27 = 73.91$, $p < 0.001$) (Adapted from Breitzler et al. (2020)); (**c**) **top:** acoustic treatments with variable temporal patterns of white noise: (**a**) continuous noise (CN); (**b**) intermittent noise with 1 s noise and 1 s silent interval (IN1,1); (**c**) intermittent noise with 1 s noise and 4 s silent interval (IN1,4); and (**d**) intermittent random noise with 1 s noise and 1–7 s silent (on average 4 s, RN1,7). **Bottom:** mean auditory thresholds ± SEM of zebrafish after 24 h of either silent lab conditions (control, c. 105 dB re 1 µPa) or white noise of different temporal patterns. *$P < 0.05$, **$P < 0.01$, and ***$P < 0.001$. (Adapted from Wong et al. (2022))

These studies were based on zebrafish adults, but auditory sensitivity is known to vary across the life cycle in this species (Zeng et al. 2021). Hence, an additional study evaluated the effects of noise exposure on larval zebrafish (Fig. 2). Lara et al. (2022) provided the first evidence that increased noise levels can impact auditory sensitivity and the amount of auditory hair cell receptors in a larval fish, and that such sensory loss is correlated with behavioral hypersensitization. The authors tested the effects of chronic exposure to white noise (150 dB dB re 1 µPa) on saccular sensitivity and morphology at 3 and 5 days postfertilization (dpf), as well as on auditory-evoked swimming responses using the Prepulse Inhibition (PPI) at 5 dpf. The method used to determine auditory sensitivity was based on field potential recordings (microphonics) from populations of saccular hair cells. Contrary to adult zebrafish that are difficult to measure saccular potentials due to the position

of the sensory epithelia, larval zebrafish are transparent, and their inner ear saccule is more accessible for electrophysiology recordings. Hence, Lara et al. (2022) relied on this advantage to measure saccular sensitivity from noise-treated larva and showed sensitivity loss at both 100 and 200 Hz compared to control, while the PPI tests revealed hypersensitization and a similar threshold shift at 200 Hz.

Bhandiwad et al. (2018) have previously tested the impact of noise on auditory sensitivity in larval zebrafish at 5–7 dpf but did not find significant changes in absolute PPI thresholds. Nevertheless, comparing results is challenging because these authors used lower stimulus frequencies of 30 and 90 Hz, yet the noise-treated group exhibited slightly higher auditory thresholds.

The findings from Lara et al. (2022) further showed that sensory loss in larval zebrafish was accompanied by circa 19–21% decrease in saccular hair cell number. This means that the acoustic environment affected inner ear development, when accessory hearing structures are not yet present and sound reception comes mostly from particle motion (Grande and Young 2004). Similarly, Uribe et al. (2018) also found saccular hair cell damage in 6 dpf zebrafish induced by acoustic trauma system using underwater cavitation that generates high-intensity broadband sounds. The noise treatment was about 186 dB re. 1 μPa RMS and applied for 120 min, which resulted in 14% decrease in saccular hair cells. This study, however, did not measure the actual changes in auditory sensitivity in larval zebrafish.

Only a few studies evaluated the effect of chronic noise in early ontogeny (Bureš et al. 2017; Dorado-Correa et al. 2018), which is a critical time for establishing phenotypic traits (Mueller 2018). Larval fish are known to rely on acoustic cues to detect suitable habitats for settlement, and the presence of elevated background noise may impair their hearing and habitat identification (Simpson et al. 2004; Leis and Lockett 2005; Montgomery et al. 2006; Parmentier et al. 2015). Further research should rely on the technical advantages of zebrafish, such as easy breeding and short life cycle, to investigate whether the acoustic environment in early ontogeny has carryover effects to subsequent life stages and transgenerational consequences.

## Final Remarks

The present chapter provides a review of the existing knowledge on the natural soundscape of zebrafish, establishing an important ground for future studies on the role of environmental noise shaping the hearing sense of wild zebrafish. It has also highlighted the importance of controlling noise conditions in laboratory housing facilities, as they have potential to mask hearing and induce hearing loss and physiological stress. Upcoming work needs to address important questions related to potential effects of chronic noise from housing systems on the zebrafish auditory system and physiological stress, among other questions that were abovementioned.

This review also focused on noise-induced hearing loss and inner ear damage in zebrafish based on different noise levels, temporal patterns, and developmental stage. The studies explained showed that the amount of acoustic energy is more important than regularity when predicting noise effects, and that auditory threshold

shifts and sensory-structural recovery are level dependent. Sensory loss was accompanied by reduced saccular hair cell number in both adult and larval zebrafish, and decreased presynaptic activity in adults.

Some measurements to improve acoustic conditions in housing systems were described, such as using a separate water treatment unit, but additional steps in experimental work with acoustic isolation periods before testing should be considered. The duration of the isolation period would then depend on the overall noise level registered in the housing system. Since continuous noise seems to cause higher impact on zebrafish hearing, a possible intermittent regime in operational protocols of aeration, pumping, and filtering systems should be evaluated.

Furthermore, there remains a scarcity of studies exploring how sensory systems respond to environmental pressures through genetic adaptation and sensory plasticity. The timescale over which such adaptations might take place remains largely unexplored (Kelley et al. 2018). Zebrafish should be considered as a promising platform for testing transgenerational effects and evolutionary consequences of altering soundscapes. By investigating their interaction with intrinsic factors such as aging (Vasconcelos et al. 2023), and key environmental parameters associated with climate change (e.g., temperature, acidification, and oxygen levels), zebrafish could yield valuable insights into the broader impact of these changes on aquatic organisms and ecosystems.

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# Section VI

# Hearing and Auditory Assessment: Marine Mammals

# Using Playback Experiments to Estimate the Hearing Range and Sensitivity in Humpback Whales

43

Rebecca A. Dunlop, Michael J. Noad, and Dorian Houser

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**Abstract**

Despite increasing evidence that underwater anthropogenic noise has negative consequences on baleen whale hearing and behavior, there are no direct data available on mysticete hearing in noise. Here, a behavioral response study design is used to quantify humpback whale hearing in natural ocean noise. A suite of tonal signals, at frequencies ranging from 62 Hz to 22 kHz, will be used as the stimuli. A change in humpback group behavior is assumed to indicate that the whales heard, and responded to, the signal. The intended sample size is between eight and ten independent humpback whale groups per frequency. Individual whale and group behavior will be quantified using both land-based data (where a team will track the group from a land-based station) and acoustic tag data (deployed on individual whales to record fine-scale 3D movement underwater).

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The signal-to-noise ratio will be estimated at the response position of the whale, assuming that this will be consistent between groups for each frequency regardless of the level of background noise. Preliminary results confirm that humpback whales are likely to respond to signals at noise at levels comparable to other marine mammals and suggest these whales can hear the 16 kHz signals. Not only does this provide early data on hearing to high frequencies in a mysticete, but results indicate their ability to hear signals in noise is likely better than expected.

**Keywords**

Audiogram · Impact · Baleen

## Introduction

Increasing anthropogenic underwater noise is of global concern given evidence of adverse physiological and behavioral effects on marine mammal populations (Cox et al. 2006; Erbe et al. 2016; Finneran 2015; Southall et al. 2016). Potential impacts on individual marine mammals include changes in their hearing threshold in high received levels, as well as disruptive behavioral changes, which can occur in a range of received levels. In some species, subtle but potentially more widespread behavioral disturbance likely depends only on the detection of anthropogenic signals in noise. Currently, we have no direct data on mysticete audiograms, yet this information is needed by a range of stakeholders (e.g., Navy, the oil and gas industry, industry regulators) who wish to predict how underwater anthropogenic sound might impact whales.

Baleen whale hearing can be assessed in several ways: (1) assessment of the frequency of their vocalizations, (2) functional models based upon auditory anatomy, (3) the performance of electrophysiological hearing tests, and (4) using behavioral responses to sound exposures. The first, and simplest, method assumes baleen whales can hear the frequencies at which they produce vocalizations. Since vocalizations are often associated with social and breeding activities, it is generally assumed that vocalization frequencies overlap with the range of hearing. However, empirically measured hearing ranges typically extend above and below the frequencies of vocalizations, indicating that the use of vocalization frequencies for predicting the frequency range of hearing is limited to a specific frequency band within the range of hearing. Additionally, no information about hearing sensitivity can be derived from the source level of vocalizations without additional information on signal detection capabilities of the intended receivers.

A second approach relies on anatomical studies of the ear, including finite element modeling (FEM). Here, some studies have focused on mysticetes. Houser et al. (2001) used cochlear membrane measurements of humpback whales (*Megaptera novaeangliae*) to develop a band-pass filter model of auditory sensitivity. The model suggested that peak sensitivity was between 2 and 6 kHz, with good sensitivity from 700 Hz to 10 kHz, although it did not consider external and middle ear

transfer functions. Recently, Tubelli et al. (2018) used FEM to develop transfer functions in humpback whales for different components of the peripheral auditory system. Their tympanic membrane (TM) model suggested best sensitivity at 1–3 kHz with good sensitivity (to −40 dB) from 15 Hz to 3 kHz. A second tympanic bone (TB) model suggested best sensitivity at 2–3 kHz, with good sensitivity from 200 Hz to 9 kHz. In fin whales (*Balaenoptera physalus*), FEM modeling of bone conduction predicts good hearing sensitivity from 20 Hz to 10 kHz (to −40 dB relative to peak) compared to 500 Hz to 10 kHz when not considering bone conduction (Cranford & Krysl 2015). Model results are therefore inconsistent, and audiometry data are required for validation.

The third approach uses audiometry (e.g., hearing tests), which have culminated in a reasonable understanding of hearing in toothed whales. Most audiometric studies focus on determining the minimum level of sound that can be "detected" by an animal for a particular frequency, often in the presence of natural or experimentally derived noise. There are two common audiometric approaches. The first is behavioral audiometry, which involves training an animal to respond if it hears a sound (e.g., Johnson 1967; Awbrey et al. 1988; Finneran et al. 2005; Kastelein et al. 2010; Houser et al. 2013a, b; Branstetter et al. 2017). The second is electrophysiological audiometry, which measures neural activity generated by the auditory system when it is stimulated by sound (Ruser et al. 2016; Houser & Finneran 2006; Szymanski et al. 1999). Electrophysiological studies are now regularly performed in small, toothed whales (dolphins and porpoises, e.g., Houser et al. 2007, 2008; Finneran et al. 2009; Castellote et al. 2014) since these species can be maintained or rehabilitated (if stranded) by humans.

While these techniques have been used successfully on odontocetes and pinnipeds, neither technique has been successfully applied to any species of baleen whale; none are kept in captivity, and their size largely precludes them from electrophysiological work. Only one electrophysiological experiment in a baleen whale, a gray whale calf (*Eschrichtius robustus*), has ever been undertaken (Ridgway & Carder 2001), and the results were reported as "equivocal."

The fourth and final method is the use of behavioral response studies. There have been a multitude of published behavioral response studies where the animals are subjected to various sources of noise (mainly anthropogenic) and their behavioral response (e.g., changes in dive, movement, and feeding behavior) is documented. The simple assumption here is that, in the absence of any other potential stimuli, if a whale responds to a noise source, then it must have heard it. Few studies, however, have used behavioral response data to make inferences about whale hearing sensitivity. This requires a more nuanced approach and many subjects to avoid pseudoreplication. In addition, the behavioral measurement of *absolute* hearing threshold requires an environment where the background noise is lower than the threshold by an amount that is more than the critical ratio (CR); otherwise, any sensitivity assessment must be assumed to be masked.

While it is possible to achieve low noise in studies with animals in tanks (e.g., Kastelein et al. 2009), background noise levels in the ocean will generally be high enough to potentially mask signals over much of the presumed frequency range of

hearing, even in what are generally considered quiet conditions. This is particularly the case at low frequencies (e.g., < 500 Hz) where traffic noise, the noise of distant shipping (Wenz 1962), contributes substantially to the prevailing background noise levels in many parts of the ocean. Mysticete hearing is expected to extend well below 500 Hz, where most of the energy in their vocalizations occurs.

Although a true audiogram (unmasked hearing threshold) will provide stakeholders with the sensitivity of hearing in mysticetes, this information is of limited use in a practical sense. Although estimates of absolute hearing thresholds can provide the basis for a weighting function (e.g., Finneran 2016; National Marine Fisheries Service 2016), which will more realistically estimate the whale's perception of loudness, these values would strictly apply for signals near the threshold of hearing where noise is unlikely to limit detection. However, at frequencies of good hearing, an audiogram will not be useful for determining threshold of detection of sounds in noise, as this will be limited by the noise in a band around that frequency and not by absolute sensitivity. It is only at the upper and lower ends of the audiogram, and in conditions of low background noise, that a measurement of absolute threshold would be achievable. In terms of determining the detection of sounds in the natural environment, and therefore predicting behavioral disturbance, frequency-dependent masked hearing thresholds, or measurements of the CR, are likely to be more useful to stakeholders that must address impacts and mitigation in potentially noisy, real-world environments. Ideally, both the absolute sensitivity and CR would be known.

The intent is to use behavioral response experiments to estimate hearing sensitivity in humpback whales under ambient noise conditions, like the behavioral observation audiometry (BOA) used in human infants. To do this, tones will be played to migrating humpback whales at frequencies across their expected hearing range. The received level of the tone at the whale when a statistically significant change in behavior occurs defines the minimum response level (MRL) for that frequency. Where the ambient noise level affects the threshold of detection, presumably in the region of best hearing sensitivity, the MRL will also be masked. At lower and higher frequencies, it is expected ambient noise will be sufficiently low relative to the threshold of hearing, and the behavioral response will occur at the MRL in the absence of masking.

## Methodology

Low-frequency ambient noise levels in the ocean are generally lower around Australia than in the Northern Hemisphere because of lower shipping densities. The proposed site at Peregian Beach, which is on the eastern coastline, is inshore and relatively sheltered from distant shipping, and low-frequency noise levels at low wind speeds (6–10 knots) are 10–15 dB quieter than would be predicted by Wenz' curves (Wenz 1962; Cato 2012).

Work will be conducted at Peregian Beach, 100 km north of Brisbane, Queensland, Australia, during September and October 2021–2023, when whales are migrating from breeding to feeding grounds (Fig. 1). The field site is unusual in that it is a

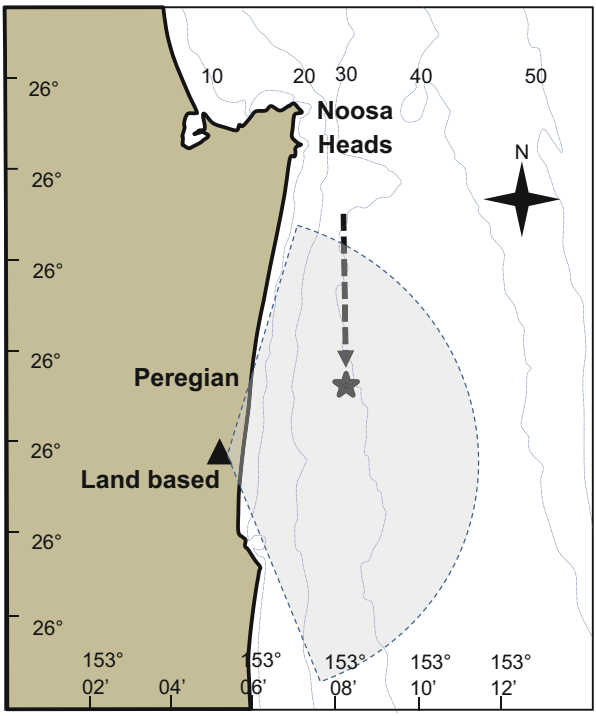



**Fig. 1** A representative drawing of the Peregian study area. An indicative pathway taken by migrating groups is shown by a dashed line, and an indicative position of the source vessel is shown as a star (but will vary based on the path of the group). The field of view of the land station is shown by the shaded area

near-shore migratory corridor for a large population of humpback whales (>30,000), at least half of which pass within 10 km of shore.

Major benefits of using this site are as follows: (1) much lower noise levels than in many ocean sites; (2) the predictable movement of the whales allows targeting and tagging whales relatively easily, as well as detecting responses reliably; (3) group responses can be measured from land leading to a robust sample size with minimal interference; (4) the social context of the target group (location and behaviors of whales within several kilometers of the focal group) can also be measured from land as social groupings have been demonstrated to affect behavior; (5) new groups will be available every day (low risk of resampling); (6) 11 previous field seasons at this site provide a wealth of background data on whale movements, behavior, and abundance; and (7) the study area has been extensively characterized acoustically leading to validated empirical models.

## Field Observational Setup

Land-based data will be collected by a visual team operating from a 73-m-high hill. A "focal follow" team will identify candidate groups as they move southward along the coast. Groups will be assessed for suitability for each experiment based on

having a relatively constant direction and travel velocity and therefore a predictable path. After selecting a candidate group, they will focally follow the group using a theodolite to capture every surface behavior and surface position of the group throughout the experiment.

*Tagging phase* – Attempts will be made to tag one of the whales in the group using a suction cup acoustic tag. During a tagging attempt, the vessel accelerates to within ~5 m of the group, and the tag is deployed from the end of a 6 m carbon-fiber pole.

*"Before" phase* – Even if tagging is unsuccessful, the experiment on the group will be continued using the land-based focal follow data. The group will be given ~20 min to "recover" to pre-tagging behavior before the beginning of the "before" phase. Target groups will then be tracked from land as well as from a small research vessel (which will maintain a distance of at least 300 m from the group) for at least 20 min before and then during the entire experimental treatment. The land-based observers will remain blind to the treatment (silent control or frequency upsweep transmission).

As the whales are migrating, their movement behavior is broadly predictable in that groups tend to travel at a reasonably consistent speed and course. The path of the experimental group can be predicted well enough during the post-tagging and "before" phase data from land-based tracking to position the source vessel so that the target group passes by. The experimental group will be approximately 4.5 km to the north of the source vessel at the start of the "during" phase.

*"During" phase* – Once the source vessel is anchored in a suitable position, it will deploy an acoustic transducer and await instructions on when to initiate the sound exposure. At least 20 min of "before" observations is required before exposure starts, and it is important that whales are not first exposed to the signal at a level where it will be detectable. Rather, we intend to start transmission of the exposure signal at a distance that is not detectable but will become detectable as the whales approach the source vessel.

Signal transmission will continue until the group has passed the source vessel and the distance between the group and source is increasing. Efficacy of the approach will be confirmed by control experiments in which no transmission occurs. Each experiment will target one group of whales and use one treatment to produce a balanced and independent sample of all treatments.

## Acoustic Measurement and Tag Recovery Phase

Once the group has cleared the area, there will be a series of boat-based hydrophone measurements of the frequency upsweep at the location at which the behavior of the group appeared to change (see collection of acoustic data below), as well as at points between the observed change and the source vessel. If tagging was successful, the tagging vessel will then follow the experimental group until the tag detaches from the whale, floats to the surface, and is retrieved. The tags will be fitted with VHF transponders so that they can be easily detected and found once at the surface.

### Tone Stimulus Characteristics

There will be seven different treatments: a silent control plus six frequency upsweep treatments. Frequency upsweeps will be one-third-octave upsweeps with starting frequencies of 62 Hz, 250 Hz, 1 kHz, 4 kHz, 16 kHz, and 22 kHz. These frequencies have been chosen to provide data for areas of likely best sensitivity (models suggest 1–6 kHz) as well as frequencies where significant sensitivity roll-off is expected (62 Hz; predicted by Houser et al.'s 2001 cochlear model and Tubelli et al.'s 2018 TB model). Targeting the lower end of the expected frequency range of humpback hearing will provide information on the slope (dB/decade) of the low-frequency roll-off, which is critically important to the development of LF weighting functions.

The upsweeps will be 2 s in duration and repeated every 4 s (hence a 50% duty cycle). Although the upsweep reduces the precision of the test in terms of it being a discreet frequency, the one-third-octave bandwidth is much less than the spacing between upsweeps. Upsweeps will be transmitted using a Lubell VC2C, Lubell LL916, or GeoSpectrum M18C-4, depending on the frequency of stimulation.

The source level used in each experiment will depend on the frequency of the tone used and the background noise (ambient noise in the absence of vessels – experiments will not be conducted in the presence of significant vessel noise). Using the results of test transmissions conducted during a pilot season (2020), source levels for each frequency will be tailored so that the SNR at 3–4 km from the source will be well below the anticipated masked hearing threshold based on measured critical ratios of odontocetes, pinnipeds, and sirenians (Erbe et al., 2016), as well as the responses of humpback whales to a 2 kHz tone in a previous experiment at this site (Dunlop et al. 2013). Note that many of the responses in the Dunlop et al. (2013) experiments occurred at stimulus onset and only a few samples likely captured the response threshold where the whale swam into the sound field. The source level for the proposed experiments has therefore been adjusted to capture the likely response threshold at approximately 3 km from the source vessel, based on results of Dunlop et al. (2013) and predictions from Erbe et al. (2016).

## Data Collection and Analysis

Land-based observers will conduct observations of groups of whales using a surveyor's theodolite linked to a purpose-designed tracking and behavioral recording program (S2SS; developed by Eric Kniest, University of Newcastle). The focal follow team will track and collect high-resolution data including every surface behavior and location of the experimental group during each experiment as the group moves southward through the study site (Dunlop et al., 2015, 2016a, b, 2017a, b). The position of the group relative to the source vessel, as well as the behavior of the group, is captured as a time series before and during transmission of the test tone.

Land-based focal follow data will be extracted and processed post-field. Here, five measurements will be extracted related to each group's movement and dive behavior:

- Group course
- Group speed
- Group dive distance
- Group dive time
- Group surface interval time

Group course and speed will be estimated using one position of the group each time it surfaces, usually the first or second position (if multiple positions are available during the surfacing event). Humpback whale dive behavior consists of a bout of "surfacing dives" (the short, shallow dives that occur during respiration bouts which are usually tens of seconds in duration) followed by a "deep dive" in which the group submerges for a longer period (usually a few minutes). A "deep dive" period is defined as the time from when the last group member disappears to when the first group member reappears, and the "surface interval" is defined as the time spent on or just under the surface between deep dives, incorporating all brief "surfacing dives." To separate deep dives from surface interval dives, for each group, the lag time between sightings will be calculated. This will give a time series of short (less than 1 min) lag times related to the group being on the surface and long lag times (greater than 1.5 min) related to the group being in a deep dive. Dive distance and dive time will then be estimated using the position and time of the group when last sighted before a deep dive and the position and time of the group when first sighted after the dive. Group surface interval will be estimated using the time the group first reappeared after a dive and the time the last member of the group disappeared before a deep dive.

Tags will provide data on high-resolution movement (3D pitch, roll, heading, and dive depth of the tagged animal), potential startle responses, and fluking patterns. The tag also records acoustics; it will potentially record the received frequency upsweeps, although it is very unlikely that the tone will be audible on the tags at the point of response due to flow noise on the tags. Resolution of the received level will be informed by the site-specific propagation model and additional acoustic recordings made by the tag vessel. Tag sensor data will be calibrated for its position on the whale. Dives, defined according to a minimum depth threshold and values such as bottom depth, points of ascent/descent, maximum depth, bottom time, etc., will then be extracted. Overall dynamic body acceleration will be calculated using the accelerometer data, with a rate of change in acceleration used to detect "quick" body movements such as a startle response. Fluke strokes will be identified as a cyclic variation in pitch. Absolute heading will be included as sine and cosine components, as well as heading variability (variance in heading over a 1 min window) and speed (vertical velocity divided by the sine of the body pitch angle) for ascents and descents correlated to flow noise.

Integration of the data streams will increase the power of statistical approaches to detecting behavioral changes. Potential behavioral responses will be quantified from the two datasets by extracting the following variables:

1. Fine-scale dive responses: Dive variables including descent rate and duration, ascent rate and duration, total dive duration, maximum depth, surface interval, fluke rate, body acceleration, and heading variance will be extracted for each dive on the tag.
2. Group-level dive and movement responses: Group travel speed, travel direction, dive time, dive distance, and surface interval time will be quantified for successive dive cycles throughout the "before" and "during" phases (i.e., starting from approximately 20 min after tagging attempts had ceased).

The analysis will follow an analytical procedure published by De Ruiter et al. (2013). Each group's dataset will be considered as a multivariate time series of dive, respiration, and movement parameters. These multivariate parameters will then be collapsed into four univariate time series, one related to group-level movement behavior (using course, speed, and dive distance), the second related to group-level dive behavior (using dive time, surface interval), the third related to individual (tagged) dive responses (dive time, number of respirations per surfacing, ascent and descent rate), and the fourth related to individual (tagged) movement responses (speed, overall dynamic body acceleration, roll, fluke rate).

First, using only (pre-exposure) baseline data, the mean, variance, and covariance of all measured variables will be estimated. These will then be used to calculate a Mahalanobis distance for each time (tag data) and dive (focal follow data) window during transmission. A plot of time along the x-axis and Mahalanobis distance on the y-axis will be created for each group, representing a time series of univariate data points per experiment. The change point, where the calculated Mahalanobis distance significantly deviates from baseline, will then be determined. The threshold for a change-point detection will be set at the 95th percentile of the baseline Mahalanobis distance, following previous studies using this technique (e.g., Stimpert et al. 2014). This would be, for example, a dive which was substantially shorter than baseline dives, an extended surface interval, a significant deviation in course, and/or a significant change in travel speed. The dive number or time window at which this point occurred is assumed to be the point at which the group first responded to the tone signal. For the control dataset, there should be no substantial deviation from baseline for any dive or within any time window and therefore no change point detected. If a change point is not detected after groups and whales were exposed to the tone, it will be assumed the individual or group did not respond.

The distance between the experimental group and the source corresponding to the "change point" will then be determined using the theodolite focal follow data. This will allow for the received level and signal-to-noise ratio at the change point to be estimated from recordings made at the time of the experiment (using bottom-mounted loggers as well as boat-based hydrophone recordings); see acoustics section below.

Acoustic data will be collected using a combination of deployed autonomous recorders, boat-based hydrophone recordings, and tag recordings. All experiments will occur within a well-defined study area, and an autonomous acoustic logger (Wildlife Acoustics SM2M) will be deployed for the duration of the field season to provide a continuous record of noise for context.

Recordings will also be collected during each experiment. After anchoring, the source vessel will record continuously using HTI-96-MIN hydrophone (High Tech, Inc., MS) into a H4n Pro digital recorder (Zoom Corporation, Tokyo) to provide a record of the signal, to provide a measure of noise, and to assess the presence of various undesirable noise sources such as nearby singing whales (which could also change the behavior of the focal group) and nearby vessels. During the transmission period, the tag vessel will continue to follow the whale group slowly at a distance of more than 300 m. When a change of behavior is observed by the focal group, the tag vessel will deploy a SoundTrap 300 STD recorder (Ocean Instruments, Auckland) on a light mooring at the location of the change in behavior to capture the signal and noise as soon as possible after the putative response. After the end of the experiment, the tag vessel will make a series of 2 min recordings (High Tech HTI-96-MIN hydrophone, H4 Zoom digital recorder) approximately 200, 500, 1000, and 2000 m from the source vessel on a line between the source vessel and the location of the putative response. During these recordings, the tag vessel will shut down and, if necessary, deploy one or more sea anchors to limit drift and reduce unwanted flow noise. Acoustic tag recordings will also be obtained from the subset of tagged animals.

The recorder deployed at the site of the putative behavior change and the tag boat recordings will accurately measure relevant noise levels and allow the development of a specific and accurate empirical model of received level as a function of distance from the source for each experimental group. Although the deployed SoundTrap may capture the transmitted signal, this modeling is necessary as signal received levels are expected to be very low and may not be detectable or measurable as they will be very close to noise floor of recorder. Additionally, subsequent analysis of the behavioral data may demonstrate that behavior changed at a different distance from the source than where the SoundTrap was deployed and the ability to accurately estimate received level and noise at those locations will be required.

The tag boat recordings will be processed in several steps. First, they will be assessed for the presence of the signal by spectrographic examination. The highest-level examples of the signal in each recording will be saved as clips. Analysis will measure the raw signal level as well as noise in the same third-octave band and then provide a noise-corrected estimate of the true received level of the signal. These will then be fit to a frequency-specific propagation model to predict the received level at the behavioral change point.

Noise in the relevant one-third octave will also be measured over the entire ~2 min recording of the tag boat recordings as well as over a few minutes at the start of the deployed SoundTrap recording. The SoundTrap recording will provide the best estimate of true noise at the change point, while the tag boat recordings will provide useful additional measures. Noise in all recordings may fluctuate temporally

over a few seconds (especially due to singing whales), and it is important that our measure of noise represents the quieter parts of the recording where the signal may have been most detectable. Lowest reasonable third-octave noise will be used to also calculate the spectral noise level by making the appropriate bandwidth correction for the relevant one-third octave. Together these estimates of received level and noise at the change point will be used to provide a conservative (highest likely) estimate of third-octave SNR as well as a conservative estimate of spectral SNR, effectively a measure of the critical ratio of responsiveness (CRR), for each experiment.

Tag recordings will also be used, but it is unlikely that the tones will be audible on the tags at the point of response due to the relatively high flow noise recorded on the tags and the low received levels expected. In this study, tags are most useful for detecting cryptic changes in behavior rather than recordings of the sound exposure.

Planned sample sizes are eight to ten focal groups per treatment (depending on the response variance), of which, based on previous experience with tagging, three are likely to include tagged animals. This sample size is based on a power analysis for the detection of a significant behavioral change in response to tones at a population level (Dunlop et al. 2013). The analysis outlined below involves estimating behavioral changes at an individual level and then using these results to determine the "most sensitive" group – in effect, defining the MRL. A sample size of eight to ten provides an adequate distribution of responses with which to do this. The three tagged animals per treatment will inform these results, meaning a much more robust sample size can be achieved within the time frame.

The analyses above will produce a series of SNRs at the change point and response level for each frequency upsweep from both tag dive data and group-level movement data. Each group will have either one measure of the response level (from focal follow movement data) or two (where one individual will be tagged). The $SNR_0$ per frequency will be used as the MRL for that frequency.

## Results and Discussion

The following presents preliminary results from the first field season (2021) to demonstrate proof of concept. A total of 15 experiments were successfully completed. Due to tag failures, data analysis relied only on land-based focal follow data.

Preliminary results for four experiments are presented: a control, a 1 kHz tone, a 4 kHz tone, and a 16 kHz tone experiment. The Mahalanobis distance was calculated using the land-based data as described above. Two Mahalanobis distance measures were calculated from the group-based data: one that encompassed three movement variables (group travel speed, course, and the distance traveled underwater in between surfacing events) and one that encompassed two dive variables (group long dive time and group surface interval).

Figure 2 presents the results for each experiment. The control experiment was carried out on a mother-calf-escort group. As this group was logging and only two dive cycles and three sightings were available in the pre-exposure period, the dive Mahalanobis distance was not calculated. The start of the transmission phase

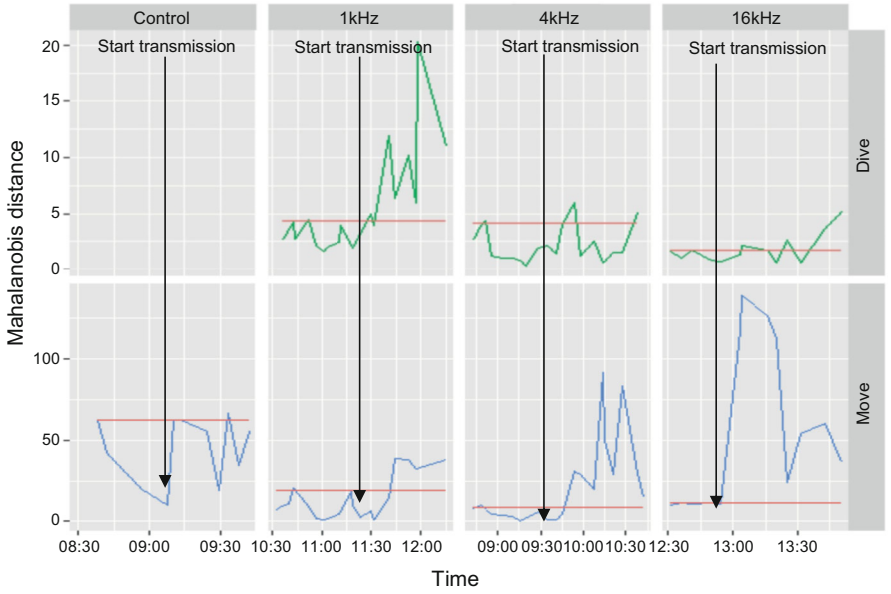



**Fig. 2** Results of a control and 1 kHz, 4 kHz, and 16 kHz experiment with the Mahalanobis distance for dive behavior in green and movement behavior in blue. The red line is the 95th percentile for the baseline period. The start of transmission is indicated by the black arrow

(equipment deployed, no signal) was taken as when the group was within 3 km of the source vessel as per the other trials. The calculated Mahalanobis distance estimated during the simulated transmission phase did not fall outside of the 95th percentile distances estimated during the baseline period.

For the 1 kHz experiment, transmission started when the mother-calf-escort group was approximately 3.7 km from the source vessel. The timepoint at which a movement response occurred was the surfacing event at which the calculated Mahalanobis distance for movement was greater than the 95th percentile calculated during the baseline period. Parameters were measured from the previous surfacing event to this surfacing event, meaning the change in behavior likely occurred at some point during the dive cycle. This change in movement was due to the group changing course to a westerly (inshore) direction, an increase in their dive distance between surfacing events, and an increase in their travel speed. A dive response also occurred at the same surfacing event (and distance from the source vessel) due to an increase in dive time. The group was 2.9 km from the source vessel when surfacing after the change in dive occurred and 2.7 km from the source vessel when surfacing after the change in movement occurred. At 2.9 km, the received level of the signal at the group was estimated to be 86 dB re 1 μPa, and the spectral SNR (or CRR) was estimated to be 20 dB.

The mother and calf groups used for the 4 kHz and 16 kHz experiments both responded by changing their movement (changing to an inshore, westerly,

direction, slowing their travel speed, and decreasing their dive distance). For both experiments, this change in behavior occurred at signal onset which was approximately 2.9 km from the source vessel for the 4 kHz experiment and approximately 3.4 km from the source vessel for the 16 kHz experiment. The received level at signal onset (and behavior change) was estimated to be 85 dB re 1 μPa for the 4 kHz experiment and 83 dB re 1 μPa for the 16 kHz experiment with spectral SNRs (CRRs) estimated to be 26 and 38 dB, respectively. Like the 1 kHz experiment, changes in behavior likely occurred at some point during the dive cycle; however, measurements could only be taken after the group had surfaced. Future experiments will therefore attempt to capture the response threshold by extending the distance at which the during phase begins and/or lowering the source vessel and using tags.

In summary, these preliminary experiments demonstrate proof of concept in that targeted playback experiments can be used to inform the hearing range and sensitivity of humpback whales. Out of the 15 experiments completed, a significant change in movement and/or dive behavior (as estimated using the Mahalanobis distance) occurred in all groups apart from the control group. The changes in behavior occurred at SNRs that are consistent with CRs from other marine mammal studies and studies of mammalian hearing more broadly. However, for approximately half of these experiments, the response occurred at signal onset, meaning the experiment likely did not capture the response threshold. In addition, due to tag failure, the response distance had to be estimated using the position of the group on the surfacing after detection had occurred. Future experiments will therefore extend the distance of the during phase to at least 4.5 km from the source vessel and/or lower the source level of the signal to ensure that the group swims into the sound field. Tags will also be deployed to enable the point at which the group changes behavior, even if it occurs during a dive cycle, to be more accurately captured.

Although in its early stages, the current study demonstrates that behavioral response studies can be useful for more than determining responses to anthropogenic sound. Behavioral response studies on hearing in free-swimming, wild cetaceans are possible under certain conditions and are similar in principle to the methods used in BOA. Already, this work has confirmed that humpback whales are likely to have CRs comparable to other marine mammals and suggests that humpbacks can hear 16 kHz signals. The whales routinely appeared to detect signals that were not evident using single hydrophone recordings (audibly or spectrographically) and in conditions of high conspecific noise (song chorusing) showing, for the first time, that mysticetes likely have considerable processing gains in hearing as demonstrated in some other marine mammals (Branstetter et al. 2010). Not only does this provide early data on hearing to high frequencies in a mysticete but also shows that their ability to hear signals in noise is potentially superior to our own ability to detect underwater signals, at least using single-sensor systems. In both frequency range of hearing and hearing sensitivity, we are therefore likely to be underestimating the acoustic information available to humpback whales under natural conditions.

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# Anatomical Measurements of Dugong Auditory Systems for Evoked Potential Studies

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Darlene R. Ketten, Klaus Lucke, and Janet M. Lanyon

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**Abstract**

Dugongs (*Dugong dugon*) are listed globally as "Vulnerable to Extinction," raising substantial concerns for their welfare. Underwater sounds are potentially critical survival cues for dugongs, but there are few data on dugong hearing, much less on how ambient sounds, natural or anthropogenic, may affect dugongs.

A project was undertaken in 2022 to obtain auditory evoked potential (AEP) responses in wild Australian dugongs during health assessments. In support of that study, a parallel effort investigated cranial and auditory system anatomy of

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dugongs to determine optimal placements of hydrophones for delivering sound stimuli and of the recording electrodes for AEP response measurements. Relevant anatomical measurements were obtained from computerized tomography (CT) imaging and dissection of three dugong specimens collected in a previous study of stranded dugongs from northern Australia in collaboration with Dr. Helene Marsh and Dr. Donna Kwan of James Cook University.

The study demonstrated three external landmarks that allow triangulation of surface points closest to the middle ear, inner ear, auditory nerve, and brainstem of dugongs. These anatomical landmarks are proportionately spaced across animals, allowing placement calculations for animals tested in field studies regardless of body mass, age, and sex.

**Keywords**

Dugongs · Hearing · AEP · Auditory anatomy · Marine mammal · CT imaging

## Introduction

Sirenians (manatees and dugongs) are found throughout the tropical and subtropical waters of the world, inhabiting fresh water, estuarine, and marine coastal regions. These relatively docile herbivores are air-breathing obligate aquatic mammals. Although they are part of the general group called marine mammals, their closest taxonomic, evolutionary group is the Proboscidea, which includes elephants (Domning et al. 1982). This somewhat surprising lineage raises several interesting questions regarding their sensory adaptations; e.g., are their sparse body hairs a dominant, tactile sensory system overshadowing hearing and/or vision; how acute is their vision and hearing; are their auditory abilities driven primarily by adaptation to water-borne sounds; do they share the exceptional low frequency hearing of elephants; and do they have any echolocation ability similar to those of odontocetes? Further, how do any of these sensory abilities vary across extant sirenian species living in different habitats? Some sirenians, such as the Amazonian manatee (*Trichechus inunguis*), live exclusively in shallow fresh water environments with essentially no predators (except humans). By contrast, dugongs live in marine coastal waters, with substantial risks from multiple predators.

Dugongs are found in the Indian and Pacific Oceans in warm shallow seas in areas of abundant seagrasses. In Australia, dugongs occur in the coastal waters of northern Australia with a substantial population in the Torres Straits. Dugongs are listed worldwide as "Vulnerable to Extinction" (IUCN 2021) and under Australian legislative acts are listed also as "Endangered, Vulnerable, and Other Protected Fauna." Despite these protections, there are multiple threats to dugong survival, particularly loss of seagrasses on which they depend. Because seagrasses are primarily found in shallow nearshore waters, dugongs may be subject to ship strikes, net entanglements, and behavioral disturbances from commercial and private vessel traffic and coastal construction, particularly near urban centers (Hodgson and Marsh 2007; Marsh et al. 2018). This again raises the question of what dugongs can hear and whether sounds from vessels and other human activities may impact dugong hearing and impair

sound-dependent activities such as intraspecific communication, detection of predators, and navigation.

Behavioral, auditory evoked potential (AEP), and anatomical studies have been conducted to investigate hearing in manatees, another group of sirenians (Bullock et al. 1980; Supin et al. 2001; Klishin et al. 1990; Gerstein et al. 1999; Ketten et al. 1993). These studies demonstrated the advantage of combining physiological and anatomical approaches and provide data for comparison of results across sirenian species. However, it is premature to assume it is possible to extrapolate dugong hearing capabilities from manatee data. It is important to keep in mind that despite their taxonomic ties, manatees evolved in environments with different adaptive pressures including a different sound profile from that of their marine relatives. Dugong ears may consequently have different hearing ranges and sensitivities and localization abilities than those of manatees.

Currently, there is a serious lack of auditory data in any form for dugongs. Frequencies of underwater calls range from 0.15 to 18 kHz for dugongs, implying a hearing range of at least seven octaves (Southall et al. 2019). One pilot AEP study was conducted on a single older captive male (Ketten et al. 2000). The test frequencies ranged between 2 kHz and 40 kHz. Best sensitivity was found at 10 kHz; however, there was a noticeable a region of diminished sensitivity in the hearing response beginning at 4 kHz and peaking at 8 kHz with a return to a typical hearing curve form at 10 kHz. This phenomenon is commonly referred to as "notch," which is characteristic of frequency loss in older individuals and typically occurs near the point of peak sensitivity. The observed notch in the study animal suggests the individual had diminished hearing at least at 4-8 kHz and may potentially have had diminished hearing at other frequencies, particularly at higher frequencies of his hearing range, and therefore does not represent average, normal hearing abilities for this species. A second consideration for the Ketten et al. (2000) study is that, in the absence of any related anatomical data, electrode placement was arbitrarily based on a line drawn between the external opening of the auditory canal (meatus) and the rear margin of the eye. This may have affected the absolute levels of the responses but is unlikely to have affected the shape of the total response curve (Fig. 1).

AEP measurements are based on playing sound stimuli at different frequencies and recording neural responses evoked by each stimulus. The precise placement of stimulus sources (loudspeakers, suction cup hydrophone, freestanding hydrophones, etc.) and response detectors (surface or needle electrodes) may differ by species based on anatomical differences in the location of auditory structures and sources employed. The results noted above underscore the need for AEP data from individuals of both genders and multiple age group in order to have a clear understanding of hearing in any species.

## Dugong Auditory System Anatomical Project

To determine the feasibility of measuring hearing in wild dugongs, a pilot project was conducted of AEP measures of hearing in wild dugongs in Australia in 2022. The AEP data were to be collected from multiple wild dugongs undergoing routine health assessments (Lanyon et al. 2010). Prior to the AEP studies, a preliminary project was

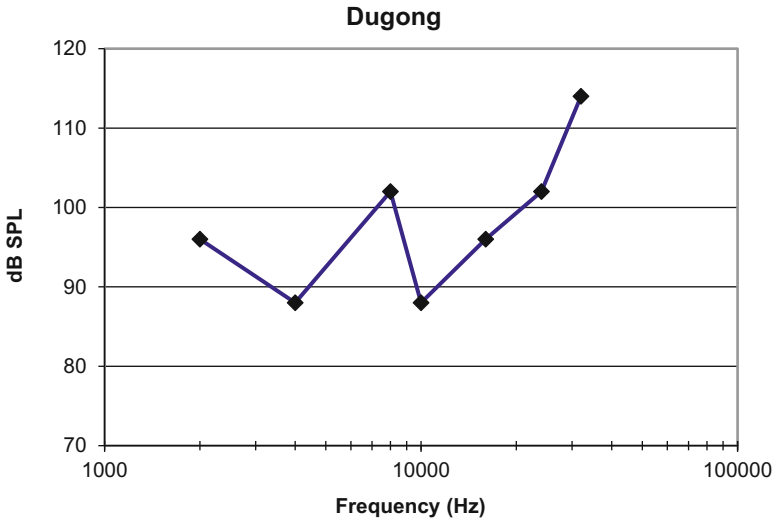



**Fig. 1** AEP-derived audiogram from a captive male dugong obtained in a single session at Sea World, Queensland. Note the marked decrease in sensitivity between 4 and 8 kHz. (Data courtesy of William Dolphin, personal communication, non-published data. Used with permission, copyright D. R. Ketten. All rights reserved)

undertaken to characterize the cranial and auditory anatomy of dugongs to determine locations for placement of surface hydrophones to deliver sound stimuli to the test subjects and to assess multiple sites for placement of recording electrodes that would optimize AEP responses based on proximity to auditory neurologic structures and would be accessible in a live, prone dugong undergoing examination and testing for AEP measurements.

To accomplish this, dugong skull architecture, bone thicknesses, head musculature, and the proximity of multiple surface locations were analyzed to determine the lowest impedance path and shortest distances from head surface to the middle ear, inner ear (cochlea), auditory nerve fibers, and auditory brainstem. In addition to determining optimal locations, it was necessary to formulate a means of establishing the electrode location on any individual based on its relationship to consistent, readily identifiable surface landmarks. Therefore, the main objective was to find surface features from which it was possible to triangulate the best locations to optimize the sound input and electrode placement to detect the AEP neural response signal.

## Methods

### Specimens

Dugong heads were obtained postmortem from male and female stranded animals under permits for collection and export of specimens issued to Dr. Helene Marsh (James Cook University, Queensland, Australia) letters of authorization and

importation and permits for importation of specimens from endangered species issued to D. R. Ketten. Three specimens were selected to span the potential range of body size/age and sex of animals that may be tested in the AEP project: an adult male (ID D-dug 02), an adult female (ID D-dug 08), and a young calf male (ID D-dug 04). Postmortem condition varied, but all specimens had intact brain and auditory system tissues. The specimens were held chilled at 4 °C until received and prepared for gross examination and imaging.

Each specimen was first examined directly to assess external tissue integrity and to document and measure positions of multiple external landmarks and to explore what internal, underlying features could be identified by palpation, such as the external opening of the external meatus (ear canal), the mandible (lower jaw), and the zygomatic bone (cheekbone) (see Figs. 2 and 3).



**Fig. 2** Adult female dugong head, left lateral view, showing surface distances from snout edge (at 99 cm mark on tape) to eye (below 109 cm mark on tape) to meatus (dark spot inside white circle located at 116.5 cm). (Used with permission, copyright D. R. Ketten. All rights reserved)

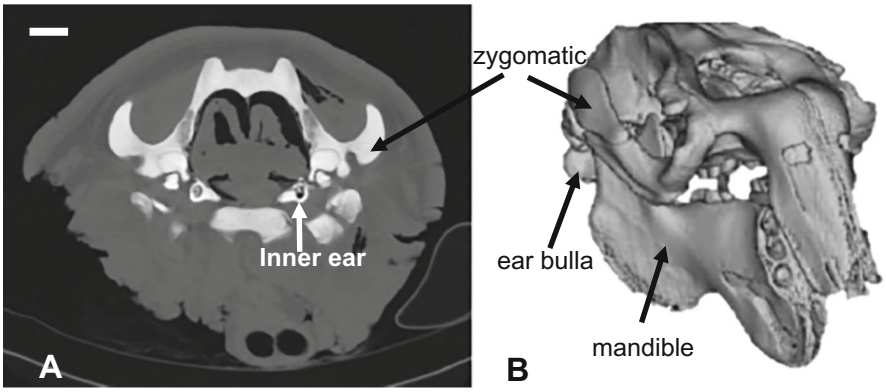



**Fig. 3** (**a**) CT scan 3 mm slice cross-section at the level of the middle and inner ear in an adult male dugong. The arrow points to the periotic bone, which is the part of the ear bulla that contains the cochlea, which is the dark spiral in each periotic bone. The scale bar = 7 mm. (**b**) Three-dimensional CT reconstruction of the skull of the same specimen demonstrating the position of the zygomatic flange and the mandible to the ear bulla which contains the middle and inner ear

## Imaging

The intact dugong heads were examined using a Siemens Volume Zoom helical computed tomography (CT) scanner at the Woods Hole Oceanographic Institution Computerized Scanning and Imaging Facility (http://csi.whoi.edu). The specimens were scanned using an imaging protocol of 0.5 mm and 3 mm acquisitions, 0.5 to 3 mm table speed; KV and effective mAS varied according to the mass of tissue being imaged. Data were acquired and reconstructed in soft tissue (U40) and bone kernels (U70), 200 FOV for whole heads. Bone and soft tissue windows at standard and extended scales were used in image reconstructions because of the extreme densities of some bony tissues and the need to differentiate small gradations in soft tissues of the auditory system. All data were archived as both raw acquisition data and DICOM formatted image files. Primary images were formatted at 0.1, 1.0, and 3.0 mm slice thickness in the transaxial plane. Raw acquisition data were employed for imaging at smaller fields of view and for multiplanar reconstructions in sagittal and coronal planes and to digitally realign the slice plane to match a mid-modiolar cochlear axis.

All image data were further processed and reconstructed into 3D and video images using Siemens VRT software, Amira 5.4, RadiAnt version 2020.1.1, and Osirix 12.0 software programs.

# Results

## Measurements

Three-dimensional reconstructions of the CT scans provided visualizations of the relationship and digital measures of distances for anatomical landmarks to critical ear features in each of the three dugong heads. Distances were obtained from specimens by direct surface measures and on CT scans to determine nearest distances to the inner ear from each of three external landmarks: anterior dorsal bridge of the snout/facial disk, midpoint of the eye, and the external meatus of the ear canal, which is a small dark opening, approximately 3 mm in diameter in adults (Fig. 2).

Scans and dissections were used also to determine distances in additional internal structures palpable on the surface. Internal structures that could be detected readily by palpation included the posterior edge of the mandible and the zygomatic bone, a prominent elongated ovoid bony flange beneath the eye that forms part of the eye socket in sirenians (Fig. 3). These two structures are located just dorsal and anterior to the ear bullae, which are the bones that contain the middle ear cavity and ossicles (tympanic bulla) and cochlea (periotic bulla).

For the two adult dugongs (Fig. 3), measurements were as follows:

| Measurements | Female | Male(cm) |
|---|---|---|
| Snout tip/facial disk to mid-eye | 10.4 | 12 |
| Mid-eye to meatus | 6.8 | 7.6 |
| Meatus to middle ear bulla | 4.5 | 5.2 |
| Rear edge of mandible to middle ear | 2.5 | 3.0 |

Because mammalian ears are fully functional *in utero* in late stages of pregnancy, and because it was uncertain what age animals may be tested in future studies, it was decided to determine whether there were significant differences in the landmark and ear locations and proportionalities according to age. Therefore, similar measures were obtained from the one young specimen available, a very young male calf.

On the juvenile (Fig. 4), the measures are the following:

| Measurements (cm) | Juvenile male |
|---|---|
| Snout tip/facial disk to mid-eye | 6.5 |
| Mid-eye to meatus | 4.4 |
| Meatus to middle ear bulla | 2.07 |
| Rear edge of mandible to middle ear | 1.2 |

## Placement Determinations

Based on these measurements, two options were proposed for determining where to place electrodes on each individual subject during the AEP experiments. One option is to palpate the jaw and locate the rear edge of the mandible, then locate the midpoint of the zygomatic bone. The point at which lines drawn posterior from the mandible and ventral from the zygomatic cross is the approximate location of the entry to the middle ear and also the closest lateral point to the auditory nerve

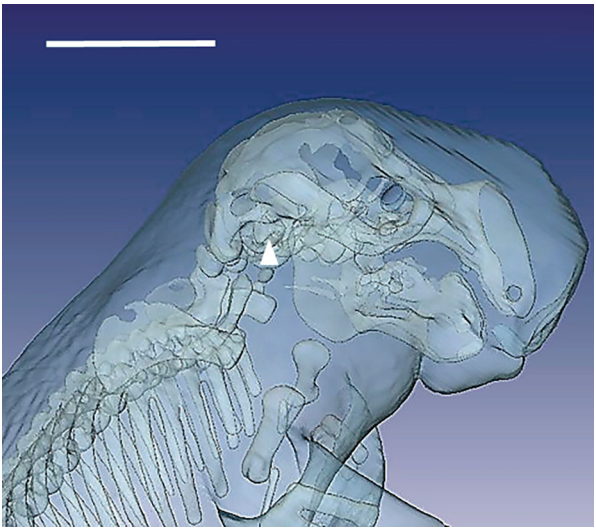


**Fig. 4** Three-dimensional reconstruction of scans of young male dugong calf. The skin has been rendered translucent to show the underlying structures of the head. The white arrow is pointing to the tympanic bone/bulla housing the middle ear. The inner ear is located in the periotic bone/bulla located medial to the tympanic bone. Based on the described triangulation, the tip of the arrow is approximately the surface location proposed for the AEP sensory signal receiving electrode location. Scale bar = 5 cm. (Used with permission, copyright D. R. Ketten. All rights reserved)

and brain stem. A second option proposed was to employ four strings or strips. By calculating the proportionalities for the measures listed, the same may be expected for the subject being tested. Therefore, triangulating the intersection of the described projections from the surface landmarks provides a location closest to the position of the cochlea.

It was recommended also to place the hydrophone as the input source over the meatus. This was based on observations during a dissection on one of the heads in which the meatus was injected with a blue dye in order to trace the path of the external auditory canal. Although the canal was quite narrow, it appeared patent and the dye was seen to end near the surface of the tympanic bulla, close to the eardrum.

## Discussion

Based on the research described in this chapter, it was decided the meatus should be the default location for the sound source input. However, the optimal path for sound to the ear remains uncertain. The dugong has a defined external meatus and a narrow canal that is patent for at least part of its length. It was not possible to establish a definitive connection in the one dissected specimen for the external canal onto the tympanic membrane (eardrum) because of tissue degradation. Therefore, it is uncertain if dugong sound reception is primarily via the meatus and external canal or via an alternative route. One intriguing possibility is the unique sirenian feature of a hypertrophied lipid filled zygomatic bone which lies adjacent to the ear bones and has been proposed in manatees as a possible underwater reception path. This is a feature that may be explored in future studies by measuring differences in receptivity for sound source placement at each of these possible paths.

For the location of the receiving electrode it was determined that the optimal position would be a point that is as close as possible to the middle and inner ear bulla. The scans showed this position can be found by locating the intersection of a line drawn approximately 5 cm ventrally from the meatus with a point 2–3 cm posterior to the rear edge of the lower jaw. That location should also coincide with the bottom ridge of the zygomatic bone which, if palpable, is an equally reliable landmark. As is shown in the reconstructions, this point lies directly over the bulla just below the long arm of the malleus, which is one of the three well-developed middle ear bones and is the closest accessible minimally invasive point to the neural bundle for a lateral probe (Fig. 4).

The head measurements showed that the critical ear features lie at positions that are predictable and proportional in juvenile and adult dugongs. Additional specimens will show us whether there is a general ratio electrode placement template that can be employed for each animal based on a few external measures. The main difference observed is in the interaural separation of the inner ear structures which is significantly larger in the adult. That may be relevant for localization abilities for younger versus adult animals, but this was not a critical element for the study described here.

The anatomical work in this combined project focused on measures of surface features related to sound reception and CT-scans of adult and juvenile heads to locate the shortest paths surface to inner ear and auditory nerve locations. Surface measures were taken of visible and palpable structures to guide electrode placement on live animals. Scans of three heads were uploaded to create 3D reconstructions that visualize and measure distances between surface features and relevant skull, ears, blubber, and musculature within each dugong head.

## Conclusions

The AEP study employing the anatomical study proposed approaches to locating electrode and stimulus source obtained good results with an appreciable signal-to-noise ratio. This suggests that analyses of the fundamental skull, musculature, and auditory anatomy of dugong juvenile and adult heads provided useful guidance for optimal hydrophone placement for transmitting acoustic stimuli transmission to the ear and, even more important, determining locations for placement of electrodes for detection of auditory nerve response signals.

This was accomplished primarily by examining CT images of three existing dugong specimens using surface measures, CT scanning of the intact specimens, and dissection of the auditory regions. Head measurements showed critical ear features lie at predictable and proportional positions in juvenile and adult dugongs. These measurements lay the foundation for developing a "bespoke" template for electrode placement for each animal based on a few external measures.

The value of CT imaging in this study parallels the diagnostic and anatomical value of scan data in for human and veterinary medicine. Clearly, this is an approach to that can provide undisturbed anatomical geometry *in situ* for many larger and rare marine species.

Preliminary additional observations suggest there are significant differences in interaural distances of dugong versus manatee ears; that the dugong eardrum and middle ear structures are stiffer; and that the auditory nerve fiber number is greater in dugongs. The preliminary conclusion is that the dugong ear is fundamentally sirenian but may have a higher frequency of peak sensitivity and potentially better underwater localization than that of the manatee. These are speculations on limited data but they do demonstrate that there remains a wealth of questions to investigate about the sensory capabilities of dugongs.

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# Incorporating Particle Motion in Fish Communication and Listening Space Models

45

Ian T. Jones, S. Bruce Martin, and Jennifer L. Miksis-Olds

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**Abstract**

Noise from vessel activity is a major contributor to underwater soundscapes throughout many parts of the world. Vessel noise often induces physiological and behavioral changes in marine animals and can mask ecologically important signals, i.e., limit animals' abilities to detect these signals. Quantitative study of in situ masking in marine environments is often challenged by limited understanding of noise characteristics, sound propagation, and hearing physiology of animals. As a result, studies often make assumptions about model parameters based on limited data. Masking studies in marine environments traditionally use sound pressure, which is detected by mammals and some fishes, to describe in situ acoustic data and animals' hearing thresholds. Few masking assessments have incorporated particle motion, the back-and-forth vibratory part of sound which is the primary stimulus for fish and invertebrate hearing, limiting application of models to these abundant taxa. In this chapter, models previously employed to estimate masking effects of noise on communication and listening space in aquatic environments are summarized, followed by an example and discussion

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of how particle motion data can be incorporated into these assessments. Challenges and needs for more accurate estimation of masking experienced by fish and invertebrate taxa are discussed.

**Keywords**

Masking · Fish · Invertebrates · Ship · Vessel · Particle motion · Listening space · Communication space

## Introduction

Vessel noise is frequently present in marine environments and often greatly exceeds natural ambient levels of underwater soundscapes. Historical estimates in several regions report increases in the contribution of noise from shipping traffic to marine soundscapes between 1950 and 2012 (Frisk 2012; Miksis-Olds et al. 2013). Vessel noise has a multitude of adverse physiological and behavioral impacts on marine taxa. Studies of these impacts are numerous for marine mammals, fewer for fishes, and even fewer for aquatic invertebrates (Hawkins et al. 2015). Behaviorally, species across these broad taxa may exhibit startle responses; spend less time foraging; alter swimming, schooling, or social behaviors; and alter the rate and other characteristics of their calls (Magnhagen et al. 2017). Vessel noise may also lead to reduced reproductive success (de Jong et al. 2018). Further, noise from shipping (and other anthropogenic activities) can mask, i.e., inhibit, animal's ability to detect ecologically important signals in their environment (Slabbekoorn et al. 2010; Erbe et al. 2016). These natural signals may include mating or other social calls from conspecifics, sounds revealing the presence of predators or prey, navigation cues, and other signals within animals' ambient soundscapes. Accordingly, it is critical to monitor and model vessel noise to understand how it may inhibit marine animals' abilities to communicate and detect sounds in their habitats and to assess the effectiveness of noise mitigation practices.

Models estimating masking in marine environments typically predict reductions, due to noise, in the range over which animals can communicate with conspecifics or more generally the range over which they can detect (or "listen in" on) natural sound signals. Communication range estimates are applicable to species known to call for specific ecological functions, such as to attract mates. Models for reductions in listening range can be applied to any species that detects sound and whose hearing thresholds (minimum amplitudes the animal can detect pure tones at given frequencies) have been measured. Several excellent papers have applied models of communication and listening space reduction to estimate masking effects of vessel and seismic survey noise on a variety of marine mammal species and a few fish species (e.g., Clark et al. 2009; Stanley et al. 2017; Putland et al. 2018).

Few masking studies have included acoustic particle motion, which is the primary stimulus for hearing in fishes and aquatic invertebrates (Popper and Hawkins 2018; Rogers et al. 2020). A recent lab study reported raised hearing thresholds due to

masking noise for both sound pressure and particle motion in oyster toadfish, *Opsanus tau* (Rogers et al. 2020). However, to the authors' knowledge, no published studies modelling communication or listening space changes due to masking have incorporated particle motion data. "Particle motion" refers to the back-and-forth oscillatory motion of particles in a sound field, whose collective motion (compression and rarefaction) leads to net propagation of sound pressure waves. This chapter summarizes recent methods applied to estimate changes in communication and listening space for marine taxa during anthropogenic noise, followed by an example and discussion of how particle motion data can be, and why it ought to be, incorporated into masking studies focusing on fish and invertebrate taxa.

## Methods Assessing Masking Potential of Anthropogenic Noise

Table 1 summarizes metrics used by recent studies to estimate masking effects of sound pressure level associated with underwater anthropogenic noise on communication and listening by marine animals. Most of these studies focused on vessel noise, though some addressed masking by air gun noise from seismic surveys. These studies have estimated sound pressure level masking effects on communication in multiple fish and mammal species including New Zealand bigeye (*Pempheris adspersa*), Atlantic cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), harbor seals (*Phoca vitulina richardii*), bottlenose dolphins (*Tursiops truncatus*), Bryde's whales (*Balaenoptera edeni*), fin whales (*Balaenoptera physalus*), humpback whales (*Megaptera novaeangliae*), and right whales (*Eubalaena glacialis*). Studies on masking of listening have focused on Atlantic cod, bowhead whales (*Balaena mysticetus*), belugas (*Delphinapterus leucas*), bearded seals (*Erignathus barbatus*), ringed seals (*Pusa hispida*), and, more generally, categorical low-, mid-, and high-frequency cetaceans. The reader is referred to Erbe et al. (2016) for a thorough review of communication masking studies on marine mammals and related terminology and definitions that may be extended to other animal taxa.

Beyond the parameters listed in Table 1, frequency bands and time windows of analysis for signal (e.g., conspecific call) levels, ambient noise levels, and masking noise levels must be carefully chosen to represent the sound perception abilities (and sound production characteristics, if assessing communication) of species of interest. The frequency window chosen should be equal to an animal's critical bandwidth, which is defined based on critical ratios (Fletcher 1940; Erbe et al. 2016). A critical ratio is the difference between a sound level (usually sound pressure level) of a pure tone that is just audible during white noise and the sound level of the white noise. Notably, this definition has been expanded to include other continuous broadband noise in place of artificial white noise (Erbe et al. 2016). The critical bandwidth is the bandwidth of white noise at which the detection threshold of a tone at the center of the noise band stops increasing with increasing width of the noise band; this bandwidth is considered to estimate the bandwidth of an animals' auditory filter at the same center frequency (Fletcher 1940). Critical bandwidths have been directly measured for some marine mammals, including California sea lions (*Zalophus*

**Table 1** Metrics used to model absolute (e.g., in meters) or relative (e.g., as percent difference) spatial extents of masking for communication or listening, from recent studies focusing on marine animals and underwater noise sources. All of these studies focused on sound pressure levels

| Metric name | Definition | Equation | References |
|---|---|---|---|
| Signal excess (SE) | 50% probability of detection of signal if the signal's presence is random in space and time | $SE = SL - PL - NL - DT$,<br>where:<br>SL = source level<br>PL = propagation loss[a]<br>NL = noise level<br>DT = detection threshold | Clark et al. (2009), Hatch et al. (2012), Matthews et al. (2015)[b], Pine et al. (2016, 2020, 2021), Stanley et al. (2017), Cholewiak et al. (2018), Putland et al. (2018) |
| Communication range (CR)[c] | Range or radius from the caller over which call can be detected by a conspecific. (Absolute quantity) | SE is set to zero to solve for range $r$ in the PL term. The exact equation for CR depends on how PL is specified. For example,<br>if: $PL = N\mathrm{Log}_{10}(r)$,<br>then: $CR = 10^{(SL - NL - DT)/N}$ | Clark et al. (2009), Matthews et al. (2015), Pine et al. (2016, 2020, 2021), Stanley et al. (2017), Cholewiak et al. (2018), Putland et al. (2018) |
| Communication space (CS) | Area, centered at the caller, over which call can be detected by a conspecific. (Absolute quantity) | If assuming equivalent propagation conditions at all azimuths:<br>$CS = \pi * CR^2$ | Clark et al. (2009), Cholewiak et al. (2018), Gabriele et al. (2018) |
| Communication range reduction (CRR) | Percent difference in the communication range during noise and the maximum communication range under quiet conditions. (Relative quantity) | $CRR(\%) = 100\left(\frac{r_1 - r_2}{r_1}\right)$,<br>where:<br>$r_1 = 10^{(SL - NL_1 - DT)/N}$<br>$r_2 = 10^{(SL - NL_2 - DT)/N}$<br>$NL_1$ = mean or percentile of sound level of the environment without masking noise in the animal's critical bandwidth (CB) or of the animal's audiogram in the CB, whichever is higher<br>$NL_2$ = mean or percentile of sound level during masking noise, in CB<br>N = spreading coefficient | Hatch et al. (2012), Pine et al. (2020) |
| Listening range reduction (LRR) | Percent difference in linear range over which a receiver can detect a sound during noise and maximum range under quiet conditions. (Relative quantity) | $LRR(\%) = 100(1 - 10^{-\Delta/N})$,<br>where:<br>$\Delta = NL_2 - NL_1$ | Pine et al. (2020) |

(continued)

**Table 1** (continued)

| Metric name | Definition | Equation | References |
|---|---|---|---|
| Listening space reduction (LSR) | Percent difference in area over which a receiver (centered in area) can detect a sound during noise and maximum range under quiet conditions. (Relative quantity) | $LSR(\%) = 100(1 - 10^{-2(\Delta/N)})$ | Matthews et al. (2015), Pine et al. (2018, 2020) |

[a]Propagation loss can include multiple terms depending on the level of complexity and accuracy deemed appropriate for an environment. Simpler models often include a single term, $N\text{Log}_{10}(r)$, where $N$ is a spreading coefficient and r is the range from the sound source

[b]Matthews et al. (2015) used two additional terms, + DI and + SG, where DI is the directivity index, representing an animal's ability to discriminate sounds coming from a particular direction (spatial release from masking), and SG is the signal gain, representing an animal's ability to use knowledge of the time-frequency structure of a call to differentiate it from background noise

[c]Some studies use the terms "communication space" and "communication range" interchangeably, but these are separated here to distinguish between explicitly reported area (e.g., $km^2$) and linear range (e.g., km)

*californianus*), harbor seals, and northern elephant seals (*Mirounga angustirostris*), and indirect methods to estimate auditory filters through critical ratios have been used for many marine mammal species including whales, seals, and dolphins (Erbe et al. 2016). Decidecade (approximately one-third octave) bands are often used as a conservative approximation of marine mammal critical bandwidths (Erbe et al. 2016). Critical bandwidth estimates are only available for a few fish species, including Atlantic cod (Hawkins and Chapman 1975) and goldfish (*Carassius auratus*; Fay 1992). To the author's knowledge, no published data exist on auditory filter bandwidths of aquatic invertebrates. One-octave bands have been used to approximate the critical bandwidth for fishes, based on estimated critical bandwidths measured for Atlantic cod (Hawkins and Chapman 1975; Stanley et al. 2017; Pine et al. 2021). Call characteristics, including source level which is a key input parameter for communication space models, are undescribed for most fish and invertebrate species as well. Because data on call characteristics, critical bandwidths, and detection thresholds are limited for fishes and invertebrates, studies on these taxa often assume these values based on taxonomically related species.

There are multiple mechanisms by which animals can compensate for or obtain "release" from masking effects to improve signal detection during noise. "Spatial release from masking" (SRM) can occur when the noise and signal are received from different directions. "Comodulation masking release" (CMR) can occur when the noise has amplitude modulation in a wider frequency range than the auditory filter bandwidth around a signal, i.e., it is coherently modulated (Hall et al. 1984). Animals can also exhibit "vocal plasticity," altering call characteristics to increase detectability of calls by conspecifics during noise. Call alterations may include increasing the amplitude of calls (also known as the "Lombard effect"), increasing

call rate, or changing frequencies of calls. These three types of "anti-masking" strategies have been widely studied in terrestrial mammals (including humans), and evidence for them exists in multiple marine mammal species (Hall et al. 1984; Brumm and Zollinger 2011; Erbe et al. 2016). Terms accounting for SRM and CMR have been included in equations for signal excess in some marine mammal communication space models (Clark et al. 2009; Matthews et al. 2015).

Published data on anti-masking mechanisms are currently available for only a few species of fishes and are lacking for aquatic invertebrates (Hawkins et al. 2015). The CRM strategy has been quantified in goldfish (Fay 2011). Fishes have been shown to have different sensitivities to sound along different axes (Hawkins and Popper 2018). How fishes might incorporate directional cues to attain spatial release from masking is not well understood (Popper and Hawkins 2018), though some evidence exists for this ability in Atlantic cod (Hawkins and Popper 2018). Vocal plasticity in amplitude, call rate, or frequency has been observed in several fish species including damselfish (Pomacentridae), Lusitanian toadfish (*Halobatrachus didactylus*), and freshwater blacktail shiner (*Cyprinella venusta*; Holt and Johnston 2014; Radford et al. 2014). Plainfin midshipman have also been observed increasing the amplitude of their mating calls when played artificial tones (Brown et al. 2021). To the authors' knowledge, reduced signal-to-noise ratios (SNR) of signal detection by the listener or increased SNR of a caller's vocalizations via these mechanisms have only been quantified for sound pressure and not particle motion. Owing to limited data on anti-masking strategies of aquatic animals, including mammals, such strategies are often excluded from listening and communication space models (but see Clark et al. 2009; Matthews et al. 2015), likely leading to over-estimation of masking for species that employ these strategies (Erbe et al. 2016).

## Incorporating Particle Motion in Modelling Masking Effects of Vessel Noise

To apply sound pressure level methods reviewed in the previous section to particle motion data, particle acceleration, velocity, or displacement levels can be used in place of sound pressure levels. Of these, particle acceleration is typically considered the most relevant to fish and invertebrate hearing (Popper and Hawkins 2018). However, there are some additional considerations to be made. Unlike sound pressure which has a single magnitude in all directions at a given point in space, particle motion is a vector quantity with a magnitude and a direction. Particle motion can be reported for individual axes or for vector sums of two or three orthogonally spaced axes to give a 2D or 3D direction and magnitude. Thus, masking in terms of particle motion could be described for multiple directions with respect to an animal. Although there is much to be discovered about how mechanistically fishes and invertebrates utilize particle motion cues to distinguish signals during noise, given the known importance of particle motion to their sound detection, it is prudent to incorporate it into assessments of masking effects of noise on these taxa (Popper and Hawkins 2018; Nedelec et al. 2021).

Potential masking of vessel noise was investigated for one fish species, red drum (*Sciaenops ocellatus*). Red drum produce courtship calls that are tied to reproductive success (Montie et al. 2016) and inhabit continental shelf waters in the US Mid- and South Atlantic and Gulf of Mexico (Lowerre-Barbieri et al. 2008). Audiograms for red drum have been published for pressure and particle motion (Horodysky et al. 2008). Because source levels of red drum calls were not available in peer-reviewed literature, this analysis focused on listening range reduction (LRR). LRR is the percent difference in linear range over which a receiver can detect a sound during noise and the maximum range under quiet conditions. It is calculated as follows:

$$\mathrm{LRR}(\%) = 100\left(1 - 10^{-\Delta/\mathrm{N}}\right), \tag{1}$$

where $\Delta = \mathrm{NL}_2 - \mathrm{NL}_1$ and N is a geometric spreading coefficient. $\mathrm{NL}_1$ is defined here as the median root-mean-square (RMS) sound level of the environment without masking noise in the animal's critical bandwidth or level of the animal's audiogram at the center of the critical bandwidth, whichever is greater (i.e., the "baseline" level). $\mathrm{NL}_2$ is the median sound level during masking noise, in the critical bandwidth. One-octave bands were chosen to represent critical bandwidths of red drum, based on estimated critical bandwidths measured for Atlantic cod, and following recent communication space studies on fishes (Hawkins and Chapman 1975; Stanley et al. 2017; Pine et al. 2021).

To explore the use of in situ particle motion data in masking models, passive acoustic data were leveraged from five sites of the Atlantic Deepwater Ecosystem Observatory Network (ADEON) located throughout the US Mid- and South Atlantic Outer Continental Shelf (Tripathy et al. 2021; Kowarski et al. 2022). Passive acoustic data were collected at ADEON sites from compact four-channel hydrophone arrays (0.5 m, orthogonal spacing) on bottom landers during November 2017 to December 2020; the data from November 2017 to June 2018 were considered here. RMS pressure and particle acceleration values were calculated in 60 s time windows. The average RMS pressure value between the four hydrophones for each 60 s window was taken to report sound pressure level. The finite-difference approximation (Gray et al. 2016) was used to calculate particle acceleration for three orthogonal axes, and for simplicity, the vector norm (square root of the sum of squares of linear acceleration of each axis) was calculated to give scalar magnitudes (as in Jones et al. 2022). Times of vessel closest points of approach (CPAs) were identified with an automated detector (Martin 2013) and manually checked to remove time points with confounding hydrodynamic flow signal and broadband transient signals (of unknown origin). Ambient (baseline) soundscape pressure and particle acceleration levels were defined for time points outside of when vessels were detected. Site locations, depths, total number of vessel CPAs, and counts of vessel CPAs with RMS sound pressure level and particle acceleration values above red drum's hearing thresholds (in terms of sound pressure and particle acceleration, respectively) are summarized in Table 2.

**Table 2** Study site locations, depths, and counts of vessel closest points of approach (CPAs) including totals detected per site; numbers of CPAs with 60 s RMS levels ($NL_2$), calculated in one-octave bands centered at 128 Hz, above red drum's audiogram at 128 Hz in terms of sound pressure; and those in terms of particle acceleration

| Site name | Site abbreviation | Latitude; longitude | Depth (m) | Total num. CPAs | Num. CPAs with $NL_2$ > hearing threshold (pressure) | Num. CPAs with $NL_2$ > hearing threshold (particle acceleration) |
|---|---|---|---|---|---|---|
| Hatteras South | HAT | 35.200° N; 75.020° W | 296 | 255 | 97 | 118 |
| Wilmington | WIL | 33.585° N; 76.451° W | 461 | 272 | 48 | 55 |
| Savannah Deep | SAV | 32.042° N; 77.348° W | 790 | 101 | 34 | 47 |
| Jacksonville | JAX | 30.493° N; 80.003° W | 317 | 93 | 34 | 45 |
| Blake Escarpment | BLE | 29.251° N; 78.351° W | 872 | 200 | 40 | 54 |

Soundscape and vessel noise data were limited to a one-octave band centered at 128 Hz, a frequency at which red drum's hearing thresholds are relatively low (more sensitive) compared to other frequencies. The vessel noise values ($NL_2$) and $NL_1$ values were calculated with sound pressure levels and particle acceleration levels, to obtain separate sound pressure- and particle acceleration-derived LRR values. The coefficient of propagation loss was chosen to be N = 15 (for both sound pressure and particle acceleration), which is intermediate between cylindrical and spherical spreading. For both sound pressure and particle acceleration, red drum's hearing thresholds were higher than median ambient soundscape RMS levels at each site; thus, the fish's hearing threshold (at 128 Hz, linearly interpolated from measured thresholds at 100 and 200 Hz) was used as $NL_1$.

A greater number of CPAs were above hearing thresholds in terms of particle acceleration than in sound pressure at each site, though these differences were not significant ($p > 0.05$, Mann-Whitney U test, $n = 5$ sites). RMS pressure and particle acceleration of vessel noise were strongly linearly correlated (HAT, $R^2 = 0.81$; WIL, $R^2 = 0.85$; SAV, $R^2 = 0.92$; JAX, $R^2 = 0.83$; BLE, $R^2 = 0.84$), suggesting that vessel noise was being received (by the hydrophone array) in the far field (Nedelec et al. 2021).

Taking the differences between particle acceleration-derived LRR and sound pressure-derived LRR for each individual vessel CPA revealed that particle acceleration LRRs were greater than sound pressure LRRs for most vessel passes (Fig. 1a).

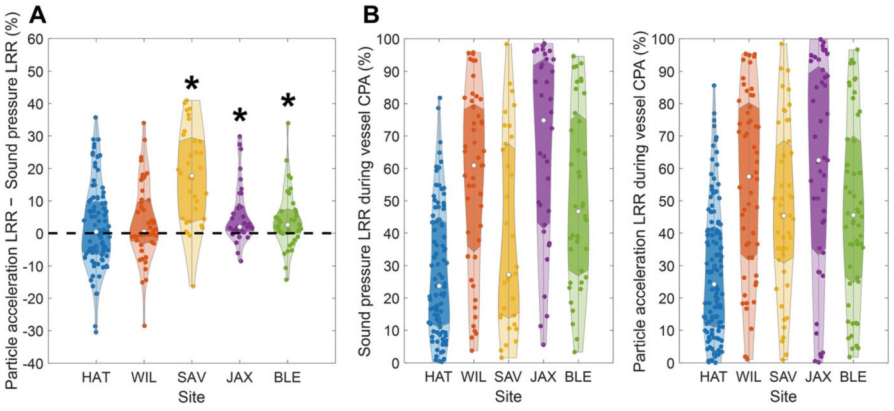



**Fig. 1** (**a**) Differences between percent listening range reduction (LRR) in terms of particle acceleration and that in terms of sound pressure, for each vessel's closest point of approach (CPA). Asterisks indicate sites with significantly different medians from zero ($p < 0.01$, Wilcoxon rank sum tests). Sample sizes for each site in this figure differ from those for CPA counts in Table 2, as the RMS sound pressure level and RMS particle acceleration level of the vessel had to both be above hearing thresholds to be included here: HAT, $n = 89$; WIL, $n = 48$; SAV, $n = 34$; JAX, $n = 34$; and BLE, $n = 39$. (**b**) Distributions of sound pressure-derived LRR (left) and particle acceleration-derived LRR (right) values for all vessel passes with sound levels above hearing thresholds in terms of sound pressure and particle acceleration, respectively. Sample sizes are the number of CPAs above hearing thresholds in Table 2 for pressure and particle acceleration. Filled circles are individual data points, open (white) circles are medians, and darker shaded regions are inter-quartile ranges (25th percentile to 75th percentile)

For three sites, SAV, JAX, and BLE, median differences (particle acceleration LRR minus sound pressure LLR) were significantly greater than zero ($p < 0.01$, two-tailed Wilcoxon rank sum tests). Conversely, considering all CPAs above thresholds, the distributions of sound pressure-derived LRR values and particle acceleration-derived LRR values were statistically similar to each other for each site ($p > 0.05$, Mann-Whitney U tests), ranging from near 0% to between 82% and 100% (Fig. 1b).

In this scenario, when taking the ratios of vessel noise RMS levels to hearing thresholds, median predicted LRRs were greater for particle acceleration than for sound pressure (Fig. 1a). However, combining vessel CPAs over the study period for each site, the distributions of LRR values for each site were similar in terms of sound pressure and particle acceleration (Fig. 1b), and octave band levels in each metric correlated well with each other, suggesting vessel noise approximated plane waves received in the far field (Nedelec et al. 2021). This meant that the variability in LRR among vessel passes was greater than the variability in the difference of sound pressure-derived LRR and particle acceleration-derived LRR for individual vessels. Thus, sound pressure and particle acceleration appeared to give redundant results (for this species, noise source, and region) over the long term, but for most sites, the median LRR values were systematically different between the metrics for individual vessel passes. Audiogram curves may have different shapes in terms of pressure and particle motion for given species, i.e., they may differ in relative sensitivities for each metric across frequencies (Radford et al. 2012), which could lead to different LRRs for each metric.

Masking assessments for fishes and invertebrates would be more accurate and useful and could be applied to more species, with audiogram data on more species in terms of both pressure and particle motion, and ideally using widely agreed-upon methodological standards. Masking models are sensitive to uncertainties in animals' audiograms (Pine et al. 2020). There are a lack of audiogram data in terms of particle motion and lack of standardized, consistent methodology for obtaining audiograms for non-mammal taxa. Among the fish species with available audiogram data, most were only obtained in terms of pressure. Further, for given fish species, wide ranges (over 40 dB for goldfish) in thresholds have been published, due to different data collection and threshold analysis methods, environmental variables (e.g., differences in temperature), and equipment used across studies (Halvorsen and Ainslie 2018). Fishes' and invertebrates' audiograms are often higher than the median ambient soundscape levels of their habitats (Tricas and Webb 2016; Jones et al. 2022); thus, listening range estimates for these taxa often will be directly weighted by their audiograms rather than by ambient soundscape levels.

Though not investigated in detail here, particle motion should also be considered in communication range and CRR estimates for fishes and invertebrates. Modelling communication range is more complex than modelling listening range because it requires more *a priori* knowledge including source levels of biological signals and anthropogenic noise sources (Pine et al. 2020). Source level estimates for known sound signals produced by fishes and invertebrates are limited, and those in terms of particle motion are even more scant. Many of these taxa communicate with

conspecifics in the near field, where particle motion and pressure will not predictably scale with one another and must both be measured directly (Nedelec et al. 2016). Additionally, particle motion will drop off with distance more rapidly than sound pressure in the near field (Kalmijn 1988). Thus, even if the masking noise is received in the far field, communication ranges of ecologically relevant signals in terms of particle acceleration may often be more limited than those in terms of sound pressure.

## Conclusions

Measuring and reporting both sound pressure and particle acceleration is recommended for models of changes in listening and communication space during noise for fishes and aquatic invertebrates. These models also require accurately collected (ideally in a standardized manner) data on sound levels and propagation of anthropogenic noise, natural soundscapes, and hearing thresholds of study animals. More co-located measurements of pressure and particle motion of soundscapes and noise sources are needed because (1) pressure and particle motion will not scale predictably with one another in all environments and (2) incorporation of particle motion data allows analyses in terms of the primary sound-detection stimulus for these taxa. Even when sound pressure and particle motion correlate well, as for vessel noise in the present example study, ratios of noise levels to hearing thresholds for individual noise events may differ for sound pressure and particle motion, and one cannot assume equivalent audiogram curves (e.g., same shape) for each metric. Further, even when studies focus on species that likely do not sense sound pressure at all (e.g., elasmobranch fishes and many aquatic invertebrates), including sound pressure data is still useful to improve understanding of the relationships between sound pressure and particle motion in different environments and for different types of noise and to allow applicability of datasets to sound pressure-sensitive taxa. Although the present example was in relatively deep water (300–900 m), in shallower, nearshore environments such as harbors and bays, particle motion is less likely to scale predictably with pressure (Nedelec et al. 2016). Thus, its measurement and inclusion in masking studies of particle motion-sensitive taxa are of heightened importance in shallower environments.

Higher ratios of received vessel noise to hearing thresholds led to higher particle motion-derived LRR values in the present example using red drum as the study species, compared to pressure-derived LRR values. In this scenario, estimating LRRs just in terms of sound pressure would have under-predicted LRR values for red drum. Again, LRR estimates can be strongly influenced by audiograms, and this is especially true for many fish and invertebrate species whose thresholds are below median (and often higher percentiles) of ambient soundscape levels. Development of standards in hearing measurements and particle motion measurements for such taxa will increase precision and confidence of LRR models. Potential differences in estimated CRR should be explored in future studies as well.

With limited masking studies using particle motion at present, it is difficult to predict how sound pressure- and particle motion-derived LRR values will differ from each other for other species, regions, and noise sources. Where individual vessel passes were concerned in the present example, particle motion-derived LRRs tended to be higher for red drum than pressure-derived ones, indicating a potential for LRRs to differ for each metric. However, in this deepwater environment where vessel noise was likely received in the far field, variability in source levels across multiple vessel passes led to similar LRR distributions. The extent that LRR values will differ for pressure and particle motion will likely depend on depth, bathymetry, and other factors affecting sound propagation, species, and characteristics of the noise source (e.g., vessel type) specific to each study scenario. It would be interesting in future studies to assess whether specific mitigation methods recently applied, such as vessel slowdowns and engineering solutions that quiet sound emitted by engines and propellers, equivalently lead to reductions in sound levels and LRR estimates for both particle motion and sound pressure. One cannot expect them at present to be identical for any given scenario. The authors hope this discussion will encourage more researchers to incorporate particle motion data in masking models for fishes and invertebrates. To facilitate these analyses, more particle motion measurements should be made in underwater soundscape and noise monitoring projects.

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# Measuring Hearing Sensitivity of Wild Dugongs in Moreton Bay, Australia

46

Klaus Lucke, Janet M. Lanyon, and Darlene R. Ketten

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**Abstract**

A pilot study to measure hearing capabilities in wild dugongs (*Dugong dugon*) was conducted in Moreton Bay, Australia. To successfully obtain hearing measurements, an approach for measuring auditory responses in wild dugongs using neurophysiological measures was developed. Preparatory to the measurements, basic head anatomy of dugongs was investigated to optimize placements of the acoustic transmitting and receiving neuronal sensors. Three dugongs were selected as suitable candidates for the auditory measurements based on the results of a preliminary health assessment. Following the absence of observable responses by the first animal, click and sinusoidally amplitude modulated signals were used as acoustic stimuli for the second and third animals, eventually

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resulting in reproduceable and scalable neuronal responses. The recorded neuronal signals represent a proof of concept for the first auditory measurement in wild dugongs. Valuable insights were gathered during the pilot study allowing optimization of the procedure for a planned follow-up study which will test a larger number of wild dugongs.

**Keywords**

AEP · Health · Anatomy

## Introduction

Increasing levels of human-made noise in the world's oceans have prompted concerns about physiological impacts on the hearing apparatus and functioning of marine wildlife species and their disruption to behaviors including alterations in habitat use (NRC 2003, 2005). Despite these concerns, there are comparatively few measures of hearing ranges and sensitivities of wild marine mammals. Thus, it is difficult to determine the potential effects of noise on these animals in natural habitats.

In Australia, dugongs (*Dugong dugon*) occur in the coastal waters of northern Australia from Moreton Bay in the east to Shark Bay on the Western Australian coast. Dugongs are listed as *Vulnerable to Extinction* at a global scale (IUCN 2021) and listed and protected under the EPBC Act in Australia. Major environmental threats to dugongs vary with location but the most important threat globally is considered to be degradation and loss of their seagrass habitat. Furthermore, as a consequence of their habit of grazing in shallow nearshore areas, boat strike and incidental entanglement in fishing nets and in swimmer protection nets present mortal risks for dugongs (Hodgson et al. 2007).

As fully aquatic animals, underwater sound is considered a critical survival cue for dugongs, potentially important for detection of predators, awareness of underwater features such as reefs and shoal areas, and intraspecific communication (Ichikawa et al. 2012). How dugongs perceive and respond to underwater sound may also affect their potential to avoid collisions with vessels (Hodgson and Marsh 2007), evade underwater construction zones, and detect nets equipped with acoustic alarms. Understanding the auditory capabilities of dugongs has important consequences for threat mitigation and conservation planning. At present, there is a lack of auditory response data for dugongs.

Dugongs are members of the Sirenia, a mammalian order of herbivorous aquatic mammals that includes manatees. Studies on the phylogenetically related manatees (West Indian *Trichechus manatus*, Amazonian *T. inunguis*, and West African *T. senegalensis*) suggest that there are substantial differences among sirenians in terms of their hearing abilities. One study on a captive West Indian manatee found responses to acoustic stimuli ranging from 400 Hz to approximately 45 kHz with best sensitivity near 25 kHz, but also indicated some responses at signals as low as 15 Hz (Gerstein et al. 1999). It has been suggested that the perception of some

signals, especially around 20 Hz, were likely vibrotactile rather than auditory (Gaspard et al. 2013).

Auditory evoked potential (AEP) studies have also been conducted on all three manatee species (Bullock et al. 1980; Klishin et al. 1990; Supin et al. 2001). The use of AEPs in marine mammal audiometry is a recognized means of determining animal hearing quickly and safely in a variety of taxa (Supin et al. 2001; Reichmuth et al. 2013; Finneran 2015). The AEP method for auditory threshold determination compares well to traditional behavioral methods (Yuen et al. 2005) and has advantages in allowing rapid threshold measurements from untrained subjects (Nachtigall et al. 2005, 2007).

A comparison of the AEP data across these studies suggested that each of the three manatee species has narrower hearing ranges than the behavioral work reported, with best sensitivity typically in the 10–15 kHz range. These studies also demonstrated that the AEP approach is feasible on sirenians of most sizes and has the advantage of being able to test multiple individuals from each species, across sexes and all age cohorts.

One brief study conducted on a single captive male dugong indicated a probable low frequency cutoff near 300 Hz and high frequency cutoff near 40 kHz (Ketten et al. 2000). However, there was also a noticeable "notch" in the hearing response at 8–10 kHz suggesting that this individual had diminished hearing in at least that range and possibly overall (▶ Chap. 44, "Anatomical Measurements of Dugong Auditory Systems for Evoked Potential Studies").

Vocalizations may also be used to provide some (limited) indications on the potential range of "best" hearing, because in most mammals sounds are produced near but not at the frequency of peak sensitivity of hearing (Wartzok and Ketten 1999). However, thus far, studies on the acoustic repertoire of sirenians are few and often insufficiently controlled to yield information on a full functional range. Sound production data are available for three of four sirenian species: frequencies of underwater calls range from 0.4–22 kHz for West Indian manatees compared to 0.15–18 kHz for dugongs (Southall et al. 2019). However, the extent of their hearing range and auditory sensitivity remains unclear.

This chapter describes a pilot study to measure hearing capabilities of wild dugongs through the auditory evoked potential (AEP) approach. An initial study examined the morphology of the dugong head and hearing apparatus to determine appropriate placement of each of the acoustic transmitting and neuronal receiving sensors (see ▶ Chap. 44, "Anatomical Measurements of Dugong Auditory Systems for Evoked Potential Studies"). The hearing tests conducted in the field were the first to measure auditory capacity of wild dugongs.

## Methods

A pilot study to test hearing sensitivity was successfully conducted on three wild dugongs in Moreton Bay, Southern Queensland, Australia, in May 2022, as part of an annual health assessment program (Lanyon et al. 2010). This study was conducted under The University of Queensland Animal Ethics Approval Certificate

2021/AE000821, Scientific Research Permit WA0019236, and Moreton Bay Marine Park Permit MPP18-001119-1.

The technical approach for the proposed measurements mainly follows the relevant ANSI/ASA standard (S3/SC1.6-2018) for obtaining audiograms from toothed whales but is partially modified to account for the aim of achieving basic auditory responses from a dugong. In the absence of direct measurements, the first measurements were intended to establish baseline neuronal responses.

AEP measurements are based on playing sound stimuli at different frequencies and recording neural responses evoked by each stimulus. The precise placement of stimulus sources (loudspeakers, suction cup hydrophone, jawphones, free standing hydrophones, etc.) and response detectors (surface or needle electrodes) may differ by species based on the anatomical differences in the location of auditory structures and sources employed.

## Dugong Head Morphology

Optimizing jawphone and electrode placement was accomplished by analyses of the fundamental skull, musculature, and auditory morphology of the dugong head (see ► Chap. 44, "Anatomical Measurements of Dugong Auditory Systems for Evoked Potential Studies"). These data provided the necessary guidance for optimal placement for transmitting acoustic stimuli to the ear and, equally or more important, determining locations for placement of electrodes for best detection of neural response signals.

## Subjects

The study was conducted over the Eastern Banks of Moreton Bay, Queensland, in May 2022. Several dugongs were captured using the open water method (Lanyon et al. 2006) and craned onto the back deck of a research vessel for medical examination as part of an annual health assessment program (Lanyon et al. 2010).

Each dugong was held unrestrained and in a recumbent prone position on a closed-cell foam mattress under shade on the rear deck of the research vessel. Vital signs were monitored throughout the restraint period including during health assessment prior to the hearing tests. Vital signs included body temperature measured through use of an oral temperature probe, heart rate measured at 5 min intervals using a Doppler ultrasound unit, and respiratory rate measured continually in real time. Vital signs for each dugong remained within the normal ranges for apparently healthy dugongs (Lanyon et al. 2010).

Three dugongs were selected as suitable candidates for the pilot study to measure hearing sensitivity through auditory evoked potentials (AEP), based on their apparently healthy state, excellent body condition, normal vital signs, docile demeanor, and the clinical sampling having proceeded in a timely manner so that the dugong could be released within the 60-min period for handling the animals as stipulated by the permit.

The first dugong sampled was a pubertal male (MB22207), 250 cm straight body length (BL) with unerupted tusks (previously captured for health assessment as an immature in June 2015, BL 225 cm). The second dugong (MB22211) was an immature female (210.5 cm BL), while the third (MB22212) was a mature female (286 cm BL).

## Hearing Tests

Stimulus generation and data acquisition were conducted using the evoked response study tool (EVREST) electrophysiological monitoring system (Finneran 2009). To control the position of the sound source and stimulus level received by the animal, the sound projector (a 2.5 cm diameter spherical hydrophone, ITC 1042) was embedded in a silicon suction cup which was placed over the left external auditory meatus (blue arrow in Fig. 1).

Three 28-gauge needle electrodes (inverting, noninverting, and ground), which were long enough to penetrate the hypodermal blubber layer, were used to acquire the AEPs (red arrows in Fig. 1). The noninverting electrode was placed along the dorsal midline and over the assumed location of the brainstem, i.e., at the animal's neck. The inverting electrode was positioned over the ipsilateral meatus and the ground electrode was located on the back of the animal.

Test stimuli consisted of sinusoidally amplitude modulated (SAM) sounds of 32 ms duration and a modulation rate of 600 Hz. The stimuli were presented at a rate of 24.8 presentations per second (following the results for manatees, Mann et al. 2005). The signals were presented at low to moderate sound pressure levels (SPLs) relative to the published hearing sensitivity of the manatees tested in previous studies and gradually varied to obtain responses. Sounds were generated via a laptop computer and SPLs were controlled by a digital attenuator.

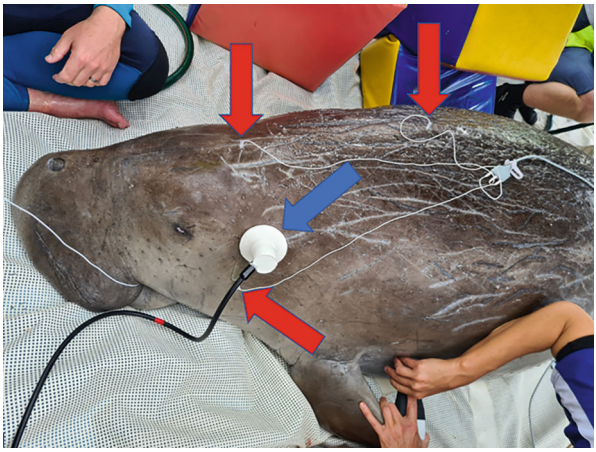


**Fig. 1** Dugong lying on a mattress showing the placement of the sound transmitter (blue arrow) and the three receiving AEP electrodes (red arrows) during hearing tests. (Copyright: Janet Lanyon)

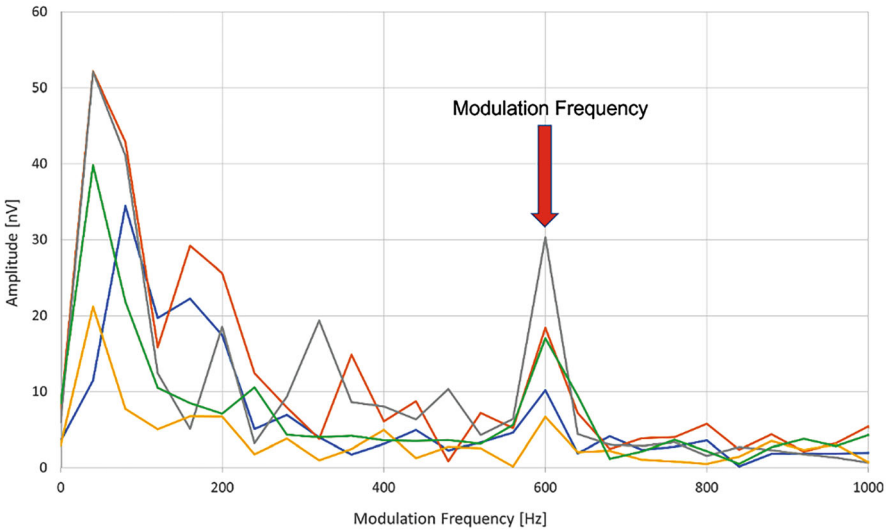



**Fig. 2** Spectral analysis of the AEP responses to acoustic stimulation at 10 kHz which was amplitude modulated at 600 Hz; red arrow indicating a neuronal response at the modulation frequency. (Copyright: Klaus Lucke)

Evoked potentials were collected in response to up to 1024 presentations of the stimulus at each sound level. Evoked potential magnitudes were calculated by performing a Fast Fourier Transformation (FFT) to identify and quantify the neuronal response at the modulation frequency (indicated by the red arrow in Fig. 2). The potential difference between the inverting and noninverting electrodes was amplified (94 dB) and bandpass filtered (30 Hz to 3 kHz) with a biopotential amplifier (ICP511; Grass Technologies) before being digitized using a multifunction data acquisition card (NI PCI-6251, National Instruments, USA) and stored on a laptop computer. Evoked responses were digitized at a rate of 40 kHz with 16-bit resolution over a 40.3-ms sweep duration. The differential electrode signal was synchronously averaged using a weighted averaging method (Elberling and Wahlgreen 1985). During the measurements, the boat's engine and all accessory electrical devices had to be switched off to reduce the ambient noise onboard and to eliminate electrical artifacts in the AEP measurements.

## Results

Two dugongs tested in this study showed detectable neuronal responses to presentation of sound stimuli. Neuronal responses at the modulation frequency were found at repeated measurements of 10 and 16 kHz at varying received stimulus levels; a clearly identifiable neuronal response was also identified at a single measurement conducted at 32 kHz.

Emitted sound levels were varied during successive measurements at 10 and 16 kHz over a range of up to 12 dB. These revealed that the neuronal responses increased proportionately with increases in signal level. Absolute received levels were not calculated as the sound source had not been recently calibrated.

## Discussion

This study is the first attempt to investigate hearing in wild dugongs. The results of the AEP measurements conducted during the pilot study provide a proof of concept for using this approach for systematic auditory studies on wild dugongs that are temporarily captured and restrained. The analysis of the recorded data revealed that neuronal responses scale with the received level, thereby providing a first approximation for determining frequency range of hearing, frequency of best hearing, and, potentially, hearing thresholds in these animals. While these auditory parameters for this species were not established in this pilot study, valuable insights and experience were gained which will be invaluable for planning and conducting a more comprehensive and definitive auditory study.

The animals were sufficiently docile to successfully conduct AEP measurements and, what is more important, showed no signs of discomfort nor increased stress due to the procedures involved in the hearing test.

The anatomical results provided invaluable guidance for the best placement of the sound source and AEP electrodes. The precise locations will be optimized during the follow-on study but results already show a good signal strength (i.e., a good signal-to-noise ratio).

The conclusion of the pilot study, apart from a proof of concept, is that under appropriate environmental conditions it will be possible to sample a larger number of dugongs including testing at least two frequencies per animal within the limited time frame currently permitted for holding wild dugongs out of water. By testing three basic frequencies per animal (e.g., a low, high, and a random frequency), it should be possible to create a composite audiogram representing the hearing range and frequencies of best sensitivity for Moreton Bay dugongs. A detailed investigation of the dugong hearing system and anatomical comparison with manatees will be essential to assess the robustness of the auditory results.

**Acknowledgments** Funding for this pilot study was provided by the Department of Agriculture, Water, and the Environment, Canberra. This pilot project was a collaborative effort by The University of Queensland Dugong Team, Sea World Australia, Woods Hole Oceanographic Institution, and JASCO Applied Sciences.

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# Section VII

# Hearing and Auditory Assessment: Other Vertebrates

# Calculating Underwater Auditory Thresholds in the Freshwater Turtle *Trachemys scripta elegans*

47

Andria K. Salas, Alyssa M. Capuano, Craig A. Harms, Wendy E. D. Piniak, and T. Aran Mooney

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**Abstract**

Knowledge of an animal's hearing sensitivity is critical to understanding how species perceive their auditory world and their susceptibility to hearing loss when exposed to anthropogenic noise. Auditory evoked potentials (AEPs) are commonly used to measure auditory responses, but it can be challenging to compare thresholds across studies. A tractable source of inter-study variation is the effect of the threshold determination method. Using control data collected for a study investigating underwater hearing loss in the freshwater turtle *Trachemys scripta elegans* (red-eared slider; n = 3), a subset of 400 Hz underwater thresholds was

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analyzed using three methods: regression, visual, and F-statistic. The regression method consistently calculated lower thresholds compared to the visual and F-statistic methods, with a mean difference of 6.2 and 7.3 dB re 1 μPa, respectively. The mean difference between the visual and F-statistic methods was 1.1 dB, with the F-statistic method producing the highest thresholds. Regression-determined thresholds were also lower than visual at 175, 600, 800, and 1000 Hz by a mean of 4.3–8.2 dB; relative sensitivity across frequencies was consistent between the two methods. Elucidating how thresholds may vary depending on analytical approach improves our ability to compare data across Testudine species, a highly threatened taxon.

**Keywords**

Hearing · Auditory evoked potentials · Ear · Reptiles · Sound · Noise pollution · Soundscape

## Introduction

Noise pollution is a prevalent anthropogenic component of the sensory world of many marine and freshwater animals (Gedamke et al. 2016; Rountree et al. 2020). There are numerous potential impacts of this noise on both vertebrates and invertebrates (Duarte et al. 2021), including a temporary or permanent reduction in hearing abilities (Smith et al. 2004; Popper et al. 2014; Finneran 2015). Knowledge of an animal's hearing sensitivity, which varies by species, individual, and age, is critical to understanding the susceptibility of a species to this noise-induced hearing loss (Saunders and Dooling 2018). Auditory sensitivity is described using hearing thresholds, which are the minimum amplitudes of a sound that are detectable to an animal. Hearing abilities can be evaluated using either behavioral (e.g., psychophysical) or electrophysiological methods, although it should be noted that these two approaches are not equivalent, as the former also incorporates perception (Popper and Hawkins 2021). In both cases, an auditory stimulus is presented, and either the animal's electrophysiological or behavioral response is observed. Behavioral techniques require an animal to produce a trained, conditioned, or natural response upon detection (Chapman and Hawkins 1973; Bhandiwad et al. 2013; Lavender et al. 2014). Electrophysiological methods acquire responses using electrodes on (often sedated) animals to measure the neurological voltages produced in response to the auditory stimulus (Corwin et al. 1982). In general, psychophysical methods are considered to produce lower thresholds (i.e., greater sensitivity) than electrophysiological methods (Yuen et al. 2005; Finneran and Houser 2006; Lavender et al. 2014; Sisneros et al. 2016), as behavioral responses reflect integration and perception throughout the auditory system. However, for many species and life stages (e.g., cephalopods, crustaceans, larval fishes) or when working with wild animals, training is not feasible. In these cases, electrophysiological methods are needed to measure auditory sensitivity at the level of neural responses by the ear.

Auditory thresholds are obtained using auditory evoked potentials (AEPs), a minimally invasive and relatively efficient method of measuring sensitivity to sound (Burkard et al. 2007). One consideration is that the experimental setup (e.g., study methodology, equipment, background sound conditions) can affect the calculated thresholds, thus influencing the comparison of thresholds measured in different studies (Ladich and Fay 2013; Sisneros et al. 2016). While the effects of some sources of inter-study variation can be difficult to quantify (e.g., electrode depth), we can improve our understanding of methods that are more tractable to enable a better interpretation of thresholds measured across studies.

Here, three different analytical methods of interpreting AEP data to generate auditory thresholds were compared: a regression-based method, visual discrimination of the responses, and an F-statistic method. This comparison was made using AEP measurements collected from the freshwater turtle, *Trachemys scripta elegans* (red-eared slider). Despite the prevalence of anthropogenic noise and the threatened status of many Testudine species (Rhodin et al. 2018; Stanford et al. 2020), there are no data informing the occurrence or magnitude of noise-induced temporary threshold shifts (TTS) in turtles in either air or water. To address this data gap, underwater auditory thresholds were collected from three *T. scripta elegans* individuals after control conditions and after exposure to continuous white noise (50–1000 Hz) that varied in amplitude and duration. Applying different analytical methods to these control data provided an opportunity to understand how thresholds may vary depending on the method chosen. While this comparison is useful to consider hearing studies across taxa, it is especially helpful for Testudine species, a threatened group for which there are limited auditory data. An improved ability to compare data between studies increases our predictive power to understand the susceptibility of both freshwater and sea turtles to underwater noise pollution.

## Methods

As part of a study investigating TTS in Testudines, auditory thresholds were measured repeatedly at 400 Hz in *T. scripta elegans*. The study was approved by the Massachusetts Division of Fisheries and Wildlife (Permit #: 075.20LP) and the Institutional Animal Care and Use Committee (WHOI ID#s: 25252.01 and 25999). Three adult females (referred to as RES01, RES04, and RES05) were used to remove the confounding variables of age and sex (Wang et al. 2019; Wang et al. 2021). Turtles ranged from 18.5 to 18.8 cm in carapace length, 17.7 to 18.5 cm in carapace width, and 865 to 1100 g in weight. Control measurements were used to define each animal's baseline auditory abilities to enable the determination of the occurrence and magnitude of TTS. Hearing at 400 Hz was studied because this frequency is in the range of their highest auditory sensitivity (Fig. 1). Over the course of the study, 76 thresholds for RES01 and 94 thresholds for both RES04 and RES05 were measured.

A combination of sedation and physical restraint was used to enable the collection of underwater AEP measurements (Harms et al. 2014; Piniak et al. 2016). Turtles were sedated with intramuscular midazolam (1 mg/kg) and dexmedetomidine (3–90 μg/kg), administered by a veterinarian. RES01 required the addition of ketamine (2–3 mg/kg).

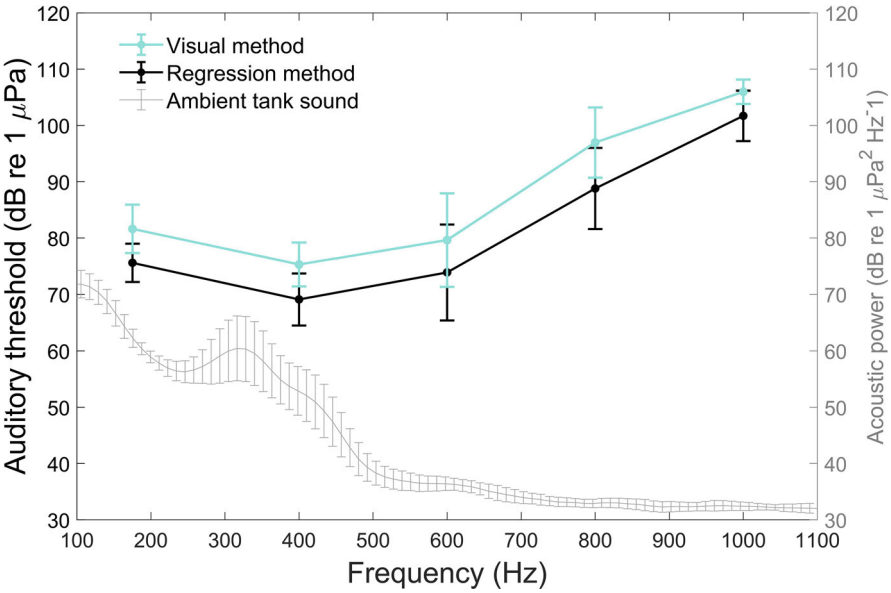



**Fig. 1** Comparing *Trachemys scripta elegans* underwater auditory thresholds calculated using the visual and regression methods. Audiograms represent the mean (1 SD error bars) of thresholds collected for three individuals for the test frequencies of 175, 400, 600, 800, and 1000 Hz. The 400 Hz threshold determined using the regression method represents the mean of the 75 thresholds selected to compare to the visual method; the mean of all 400 Hz thresholds collected across the study for the three individuals is 70.3 dB re 1 μPa (n = 264). Ambient tank sound (mean ± 1 SD) was calculated using recordings from 10 randomly selected experimental days during the testing period, with hydrophone at the position of the turtle's head without the animal present

After the desired plane of sedation had been reached, turtles were wrapped with an elastic bandage to both limit movement and for securing to a plastic platform, which could slide vertically along a PVC frame to raise the turtle to breathe. This platform and frame also maintained a consistent placement of the turtle's head underwater. Testing took place in a PVC tank (0.84 L × 0.38 H × 0.53 W), and the turtle's ear was approximately 13 cm below the waterline and 45 cm away from an underwater speaker (UW-30; Lubell Labs Inc., OH, USA), which transduced the auditory test tones. A recording electrode (Rochester Electromedical Inc., FL, USA) was inserted 2–3 mm dorsal to the right tympanic membrane, in an anterior-to-caudal direction. The reference electrode was inserted into the subcutaneous tissue to the left or right of the tail and the ground electrode hung in the water. The generation of the stimulus tones and collection of the electrophysiological responses of the turtles to these tones were controlled through custom LabVIEW software (National Instruments, TX, USA). The equipment and general setup for the AEP measurements were similar to that used in previous auditory studies [e.g., Stanley et al. 2020].

Five sequential 400 Hz thresholds were collected during each session for each turtle. Test tone stimulus amplitudes were measured at the beginning and end of each experimental day, and the average of the two measurements was used to determine the

thresholds measured that day. All three individuals were tested on the same day. Each threshold trial started at the same sound level (no attenuation). The stimulus amplitude was then attenuated in 5 dB steps until no response was observed in the response waveform or its fast Fourier transform (FFT) spectrum. The greater the attenuation, the lower the received level by the turtle. An average of 250–500 responses were collected for each attenuation level, depending on how quickly the FFT stabilized. These averaged waveforms were exported from LabVIEW for subsequent threshold determination in MATLAB (v R2019a, MathWorks, MA, USA).

An FFT (1024 points) was calculated of the averaged AEP waveform collected for each attenuation level (e.g., Fig. 2). The FFT has a peak at approximately twice the test frequency (Piniak et al. 2016), and the specific frequency used for the response was dependent on the individual and the parameters of the FFT; in the example in Fig. 2, the peak occurred at 781 Hz. As in other taxa (Pacini et al. 2010; Mooney et al. 2020a; b; Dinh and Radford 2021), this peak decreased with decreasing amplitude of the auditory test stimulus (Fig. 2). These FFTs were used in each of the three threshold analysis methods investigated.

Initially, thresholds were determined using the regression method, as in Mooney et al. (2010). This method regresses the FFT peak by the amplitude of the stimulus that elicited each response (Fig. 3). Points were removed from this regression analysis for three reasons. First, points that represented suprathreshold responses were removed, as these responses often had a wider frequency bandwidth and a more variable amplitude at twice the test frequency. Second, while the first attenuation level that produced no AEP response was included in the regression, any additional

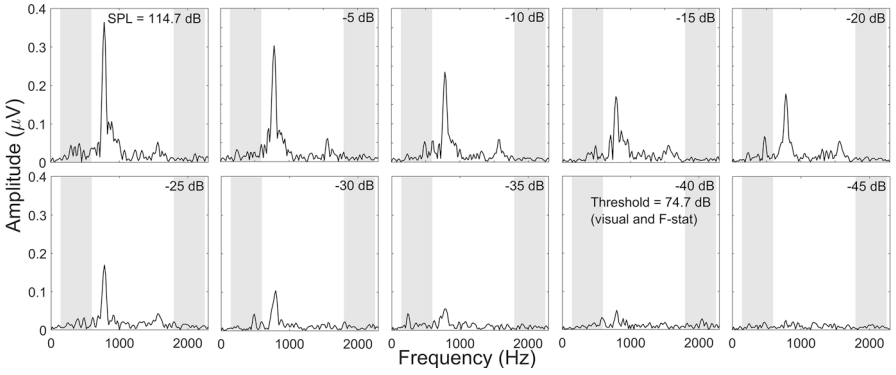



**Fig. 2** Example fast Fourier transforms (FFTs) of auditory evoked potentials (AEPs) collected from *Trachemys scripta elegans* underwater in response to a series of 400 Hz tone pips. The AEPs were measured from RES05 in response to decreasing sound levels. The stimulus was first presented with no attenuation (root-mean-square sound pressure level = 114.7 dB re 1 μPa). The peak of the turtle's auditory response was at 781 Hz, and this response was determined to be indistinguishable at the attenuation level of −40 dB (using visual, regression, and F-statistic methods). The corresponding auditory threshold for these data using the visual and F-statistic method was 74.7 dB re 1 μPa. The frequencies shaded in gray represent those selected to calculate the average background noise for the F-statistic method

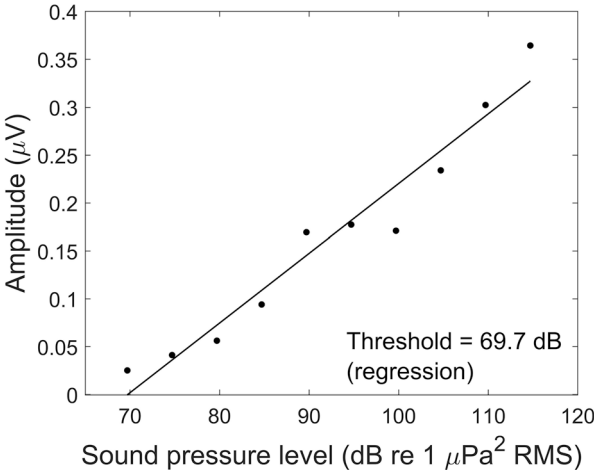



**Fig. 3** Auditory threshold determined using the regression method. Here, the auditory response peaks of RES05 in Fig. 2 are regressed against the sound pressure levels of the test tone stimulus that produced those responses. The auditory threshold using the regression method was 69.7 dB re 1 μPa, which is where the regression line crosses the y-axis at $y = 0$ (theoretical no response)

points representing no response were excluded as these would otherwise artificially lower the threshold estimation. No response was determined if there was no distinguishable peak at twice the test frequency; if there was a small peak at this frequency, it was required to be greater than approximately 0.02 μV, the typical noise floor at surrounding frequencies. This cutoff threshold for distinguishing a peak from electrophysiological and/or environmental noise varied slightly across trials. In the example in Fig. 2, the last response was determined at −40 dB re 1 μPa attenuation. Third, while response amplitudes typically decreased relatively monotonically with stimulus amplitude (as in Fig. 2), there were cases when points deviated significantly from this line (often due to variation in peak frequency) and were excluded. The r-square goal of these regression lines was typically >0.9, with variation by trial and turtle. This regression line was used to determine the auditory threshold: the stimulus amplitude on the x-axis at which the regression line crossed $y = 0$ was considered the theoretical no response and the threshold (Supin et al. 2001). In the example demonstrated in Figs. 2 and 3, the threshold was 69.7 dB re 1 μPa (Fig. 3).

To compare this regression method to alternate threshold determination methods, five control trials were randomly selected for each of the three turtles to reanalyze using visual and F-statistic methods, for a total of 25 thresholds at 400 Hz per individual. The visual method, often used to analyze evoked potential recordings in fishes (Kenyon et al. 1998; Wright et al. 2010; Stanley et al. 2020), involves visually inspecting the waveform and/or FFT peak and determining the minimal sound level at which there is no detectable response. This sound level is taken as the threshold. It is unlike the regression method because it does not directly use a sound level with no detectable response and does not extrapolate below the background physiological and environmental noise level. Thus, in the example in Fig. 2, no response was first seen at −40 dB, and the threshold using the visual method was determined by subtracting 40 dB from the test tone amplitude of 114.7 dB re 1 μPa, resulting in a threshold of 74.7 dB re 1 μPa. In this study, one person analyzed all data for consistency.

The F-statistic method, often applied in odontocetes and humans (Dobie and Wilson 1993; Finneran et al. 2007), is considered to reduce the subjectivity in determining when no responses occur in the FFTs of the AEP waveforms. This method compares the amplitude at the frequency selected in the FFT to represent the response to an average of the electrophysiological noise at frequencies both less and greater than the response frequencies. For example, in Fig. 2, the frequencies used to determine the mean background noise level are shaded in gray. These frequencies were selected because they are outside the widest response bandwidth and could be used consistently across all three individuals. These frequencies also excluded the first harmonic of the response (at approximately 1600 Hz) seen at the higher stimulus sound levels (0 through −25 dB attenuation; Fig. 2). These windows were selected to be equivalent on both sides of the response (30 frequency points) to achieve 60 total points, resulting in degrees of freedom of F(2, 120) (Finneran et al. 2007). These degrees of freedom were used to identify the appropriate critical value using an F-table with alpha = 0.05. The F-statistic was computed by dividing the amplitude of the FFT response by the average noise over the selected bandwidths. An AEP response was determined to be present if this F-statistic was greater than the critical value. In the example in Fig. 2, the F-statistic method was in agreement with the visual and regression methods, as the F-statistic fell below the critical value after the −40 dB attenuation level. Like the visual method, the F-statistic method also determines the threshold by subtracting that first no-response attenuation level from the non-attenuated amplitude of the test tone. Thus, here the visual and F-statistic methods both produced a threshold of 74.7 dB re 1 μPa.

To compare the three threshold determination methods, the mean and range were calculated of the 25 thresholds determined using each method for each individual, in addition to the 75 thresholds combined across individuals. For each threshold, the difference in threshold value between each method combination was also calculated.

The visual method has been commonly used in turtle auditory studies (Bartol et al. 1999; Dow Piniak et al. 2012; Martin et al. 2012; Piniak et al. 2016; Wang et al. 2019). Therefore, the regression-determined *T. scripta elegans* audiogram collected in preparation for the TTS study was also reanalyzed using the visual method. Mean auditory thresholds at 175, 600, 800, and 1000 Hz were determined, in addition to the 400 Hz control data (as above). Two to eight thresholds were collected per frequency (other than 400 Hz) for each turtle. As for the 400 Hz thresholds, the stimulus amplitude was decreased in 5 dB steps to determine thresholds at 175 and 600 Hz, substituting as needed smaller step sizes at 800 and 1000 Hz due to lower sensitivity at these higher frequencies. The presence of a response was also more variable at 1000 Hz.

## Results

Auditory thresholds of the three *T. scripta elegans* individuals depended on the method chosen to analyze the AEP data. At both the individual and species level, data analyzed using the regression method led to lower thresholds compared to the visual and F-statistic methods (Fig. 4, Tables 1 and 2). The F-statistic method

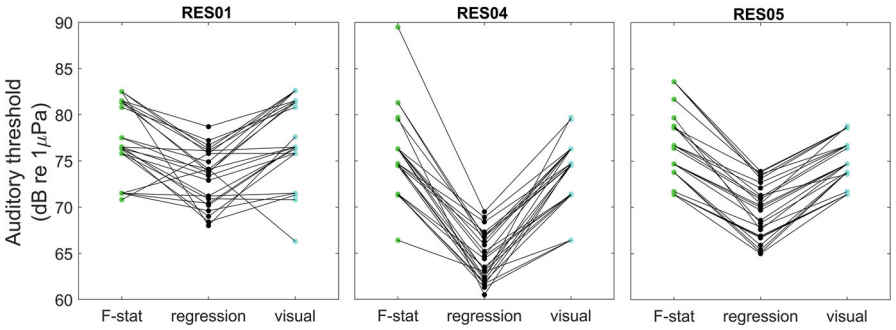



**Fig. 4** Comparison of 400 Hz auditory thresholds calculated using the F-statistic, regression, and visual methods. Seventy-five thresholds measured from three *Trachemys scripta elegans* individuals were calculated using the three different methods; lines connect thresholds determined using the same auditory evoked potential data to show how threshold values change depending on method

**Table 1** Mean (± SD) and range of 400 Hz auditory thresholds analyzed using regression, visual, and F-statistic methods (in dB re 1 μPa)

| Threshold determination method | RES01 (n = 25) | RES04 (n = 25) | RES05 (n = 25) | Individuals combined (n = 75) |
|---|---|---|---|---|
| Regression | 73.3 (3.0)<br>68–78.7 | 64.5 (2.6)<br>60.5–69.5 | 69.6 (2.9)<br>65–73.9 | 69.1 (4.6)<br>60.5–78.7 |
| Visual | 77.3 (4.5)<br>66.3–82.5 | 73.6 (3.6)<br>66.4–79.7 | 75.0 (2.5)<br>71.4–78.8 | 75.3 (3.9)<br>66.3–82.5 |
| F-statistic | 77.3 (3.9)<br>70.8–82.5 | 75.6 (4.4)<br>66.4–89.5 | 76.4 (3.9)<br>71.4–83.6 | 76.5 (4.1)<br>66.4–89.5 |

**Table 2** Mean (± SD) and range of differences in 400 Hz auditory thresholds analyzed using regression, visual, and F-statistic methods (in dB re 1 μPa)

| Methods compared | RES01 (n = 25) | RES04 (n = 25) | RES05 (n = 25) | Individuals combined (n = 75) |
|---|---|---|---|---|
| Regression minus visual | −4.0 (3.7)<br>−9.5–7.8 | −9.1 (3.3)<br>−17.4–−3.4 | −5.4 (1.6)<br>−8.7–−2.3 | −6.2 (3.7)<br>−17.4–7.8 |
| Regression minus F-stat | −4.0 (3.8)<br>−14.5–5.0 | −11.1 (3.8)<br>−20–−3.4 | −6.8 (2.5)<br>−12.1–−3.8 | −7.3 (4.5)<br>−20.0–5.0 |
| Visual minus F-stat | 0 (3.5)<br>−10.0–5.0 | −2.0 (3.2)<br>−10.0–5.0 | −1.4 (2.3)<br>−5.0–0 | −1.1 (3.1)<br>−10–5.0 |

produced the highest mean 400 Hz thresholds for RES04, RES05, and across the 3 individuals and was equal to the visual threshold mean for RES01 (Table 1). When directly comparing the methods, the visual and F-statistic thresholds were the most similar, with a mean difference of only 1.1 dB across the 3 turtles (Table 2). There were greater differences when comparing the regression thresholds to visual and F-statistic thresholds, with a mean difference of −6.2 and −7.3 dB re 1 μPa,

respectively. The regression method led to thresholds up to 15–20 dB lower than the same data analyzed using the visual or F-statistic methods (Table 2). Overall, RES04 showed the greatest differences in threshold value when comparing the three analytical methods; RES01 showed the smallest differences (Table 2, Fig. 4). RES01 was the only turtle with thresholds that decreased when using the visual or F-statistic methods when compared to the regression method. Two (of 25) visual thresholds were lower than the regression thresholds (by −0.1 and −7.8 dB re 1 μPa; Fig. 4), and the value of one threshold was equivalent between methods. Two F-statistic thresholds were lower compared to the regression method(−2.6 and −5.0 dB; Fig. 4).

In general, the range and variation were similar between the three analytical methods (Table 1). However, the number of unique values varied depending on the method chosen, considering thresholds rounded to the nearest tenth for all three methods. The regression method provided 20, 23, and 25 unique values (out of 25 per animal) for RES01, RES04, and RES05, respectively. Visual thresholds provided 12, 8, and 9 unique values for RES01, RES04, and RES05, respectively. The F-statistic method generated 10, 10, and 11 unique values for RES01, RES04, and RES05, respectively. The thresholds determined using the visual and F-statistic methods showed a more clustered distribution compared to the more continuous range of regression-determined thresholds (Fig. 4).

Using the regression method, there was also greater individual variation in mean 400 Hz threshold among the three turtles. RES04 was the most sensitive turtle (lowest threshold) and RES01 the least sensitive (highest threshold), regardless of which of the three methods used. Using the regression method, the difference in mean threshold between RES04 and RES05 was 5.1 dB re 1 μPa and the difference between RES05 and RES01 was 3.7 dB. Using the visual method, this difference was 1.4 dB (RES04 and RES05) and 2.3 dB (RES05 and RES01), and using the F-statistic method, this difference was 0.8 dB (RES04 and RES05) and 0.9 dB (RES05 and RES01).

The regression method also produced lower thresholds at the other four frequencies investigated, compared to those calculated using the visual method (Fig. 1, Table 3). Combining individuals, the mean of the 75-threshold subset for 400 Hz is shown in Fig. 1 and Table 3 to compare to these same data analyzed using the visual method; the 400 Hz threshold using the mean of all data collected (n = 264) was 70.3 dB re 1 μPa (SD = 5.6). The mean difference between the visual and regression methods across the five frequencies was 6.1 (SD = 1.4) dB re 1 μPa.

**Table 3** Mean (± SD) auditory thresholds analyzed using regression and visual methods, individuals combined (in dB re 1 μPa)

| Frequency (Hz) | Regression | Visual |
|---|---|---|
| 175 | 75.6 (3.4) | 81.6 (4.3) |
| 400 | 69.1 (4.6) | 75.3 (3.9) |
| 600 | 73.9 (8.5) | 79.6 (8.3) |
| 800 | 88.8 (7.2) | 97.0 (6.2) |
| 1000 | 101.7 (4.5) | 106.0 (2.2) |

## Discussion

This study shows that the determination of auditory thresholds collected from *T. scripta elegans* can be dependent on the method chosen to analyze the AEP data. In reality, all threshold determination methods require some degree of data interpolation and assumptions. However, considering how the predicted thresholds vary among the three methods can enable a better comparison between studies that use different analytical approaches. The potential advantages and disadvantages of these methods are also considered.

Auditory evoked potential data collected from three turtles was first analyzed using the regression method, and the visual and F-statistic methods were selected to compare to the original thresholds. These three methods each start with an FFT (itself dependent on selected parameters) of the animal's response waveform and consider at which received sound level this response is no longer distinguishable from the background electrophysiological and environmental noise (e.g., electromagnetic interference or EMI). The F-statistic method addresses this question by dividing the average amplitude at frequencies surrounding the response frequencies to the response amplitude and compares the quotient to a critical value (Finneran et al. 2007). The visual method instead makes this determination by eye and by using a cutoff threshold to help determine when a response peak is indistinguishable from the noise. For example, Piniak et al. (2016) used a 6 dB cutoff. Here, a similar approach to determine no responses for the regression method was employed by using a 0.02 μV response threshold, which varied slightly by session depending on the noise floor. The regression method uses the regression model to quantitatively assess how the response amplitude changes with received level and estimates the threshold from that model, which naturally falls below the measurement noise floor. A better model fit can instill higher confidence in the threshold estimate. An advantage of the F-statistic method is that it inherently integrates electrophysiological noise level and can be the most automated of the three methods; however, by the same turn, it also sacrifices the opportunity for the human eye to distinguish an auditory response in an FFT. Human pattern recognition can be excellent and is often used in other bioacoustic methods such as whistle discrimination [e.g., Silva et al. 2016]. Elevated background noise, contributed across the selected noise bandwidths or from isolated peaks (e.g., from EMI), does not always negate a real auditory signal, especially if that response peak matches, in frequency and shape, the responses observed at higher received levels. Indeed, there were instances when the F-statistic fell below the critical value despite the presence of what was considered a response peak using the regression and visual methods. This may have contributed to higher 400 Hz thresholds when using the F-statistic method (considering the mean and individual results for RES04 and RES05; Table 1).

The regression and F-statistic methods require selecting a response frequency to regress the amplitude at that frequency by sound level or to calculate the F-statistic, respectively. While this peak was often consistent through attenuation levels [at approximately twice the test frequency (Piniak et al. 2016)], there were cases where this peak varied, such as in suprathreshold or bimodal responses. When using

the F-statistic, one might predict the greatest quotient to occur at the highest stimulus sound levels, but this did not always occur. Suprathreshold responses resulted in energy distributed across a greater response bandwidth and a lower amplitude at the chosen response frequency, compared to response amplitudes at lower received sound levels. In these cases, determining a response using only the criterion of an F-statistic greater than the critical value could lead to falsely determining no response at the highest stimulus sound levels. This issue could be addressed by visually confirming the response measurements, as was done here. In the regression method, these suprathreshold responses were typically removed from the regression.

The frequency that provided the most consistent peak throughout the attenuation levels was selected for the regression method, and this was straightforward in most cases. However, in the case of RES01, the response peak was often bifurcated between two frequencies, and the best frequency for the regression (as determined by the $r^2$ value and maximizing number of points used) was not always the frequency where the last distinguishable response was seen. In contrast, the F-statistic and visual methods allow more flexibility in the frequency selected to determine a response, and this led to the only four instances, all using data collected from RES01, where the regression method produced a higher threshold compared to the other two methods (Fig. 4 and Table 2). This also likely contributed to the smallest threshold differences occurring in RES01 when comparing thresholds calculated using the regression method to the F-statistic and visual methods (Table 2).

The visual and F-statistic methods produced very similar 400 Hz thresholds (Fig. 4 and Table 2), similar to that observed in grey seals [*Halichoerus grypus* (Ruser et al. 2014)], while the regression method produced the lowest thresholds (Table 1). In studies comparing regression to F-statistic [*Delphinapterus leucas* (Mooney et al. 2020a)] and visual methods [*Loligo pealeii* (Mooney et al. 2010)] across several frequencies, there was overall a high correspondence between the methods compared, but this outcome varied by frequency. These comparisons (regression and F-statistic, regression and visual) showed nearly identical thresholds at some frequencies, but thresholds at other frequencies showed differences similar to those observed here (6–7 dB). Counter to Mooney et al. (2010), visual thresholds were observed to be consistently higher than regression thresholds across all frequencies tested (Fig. 1).

Visual thresholds are often determined using the sound level of the last detectable response (Kenyon et al. 1998). F-statistic methods may involve a certain number of reversals, essentially averaging sound levels producing last detectable responses and no responses [e.g., Finneran et al. 2007], which are also used in some behavioral threshold studies (Kastelein et al. 2009; Finneran and Schlundt 2010). However, given that detection methods may be somewhat masked by noise, it is likely that the threshold is lower than the predictions of the visual and F-statistic methods. The regression method assumes neuronal responses that may be below the noise floor and are not observable given the methodology employed. This method addresses this uncertainty by interpolating a theoretical no response by using in the regression the first attenuation level (i.e., highest received level) at which no response was detected. This explains why the regression method predicted lower thresholds than the other

two methods, which may allow this electrophysiological method to more closely reflect findings of lower thresholds in behavioral studies where both detection and perception are integrated (Popper and Hawkins 2021).

Interestingly, greater individual variation was observed in the mean 400 Hz thresholds using the regression method compared to the visual or F-statistic methods. This finding may be partially driven by the limited number of values a threshold can take when using the visual and F-statistic methods. Test tone levels were attenuated in 5 dB increments, which define the potential threshold values, whereas the regression method interpolates between these increments. Non-biological variation across methods is introduced by the dB increments of attenuation and calibration precision, yet the regression method may allow for more biological variation to be observed. If the final detectable response is fairly strong or marginal, the visual and F-statistic methods calculate the same threshold. The regression method, however, would interpolate the theoretical no response of the points in those two cases differently, leading to different thresholds. Further, since the regression method determines thresholds using response data from several received levels versus just the lowest level that produced a response, this method may allow for a better understanding of auditory responses. However, the regression method requires a sufficient number of points for the model. This inherently requires testing a series of received levels, and potentially more than may be necessary to obtain a threshold using the visual method. This additional testing may be prohibitive under some conditions (e.g., limited sedation time, testing in the field) if it requires more time to determine a threshold.

Compared to the visual method, the regression method produced lower thresholds and a more sensitive audiogram across all tested frequencies (Fig. 1, Table 3). The mean visual thresholds were within the 1 SD error bars for the 600 and 1000 Hz regression thresholds. Further, the error bars for both methods overlapped for all test frequencies reflecting a general correspondence. Importantly, the shape of the audiogram (relative sensitivity across frequencies) was consistent using either method. Despite this general similarity, the visual method consistently produced thresholds 4–8 dB re 1 μPa greater than the regression method across these five test frequencies for *T. scripta elegans*. This knowledge may be valuable as we seek to understand thresholds measured across turtle species, particularly endangered or threatened species. Such information would help managers better compare the auditory sensitivity and TTS results in these freshwater turtles to other species for which visually determined thresholds are available.

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# Underwater Hearing and Noise Exposure Concerns in Humans

48

Brandon M. Casper

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**Abstract**

Human divers can be exposed to high-energy underwater noise sources as part of their jobs. Establishing safety guidance for noise exposure is critical to protect divers from injury. However, in-air guidance cannot be used because humans detect sound underwater using a different pathway (bone conduction) than the traditional air-conducted pathway. To date, the body of literature on what humans can hear and how we respond to noise exposure underwater is limited, although interest in the topic is growing. This chapter will provide a thorough review of the existing underwater hearing experiments and evaluate which studies have collected quality hearing threshold data that can be used to establish what humans

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are capable of hearing when submerged. This chapter will also review the studies that have worked to establish auditory injury thresholds by exposing humans to underwater sound.

**Keywords**

Diving · Threshold · Noise exposure · Hearing damage · Threshold shift · Audiogram · Underwater hearing

## Introduction

Underwater noise has long been a concern for aquatic animals (Popper and Hawkins 2016). For over a century, researchers have studied the effect of sound on the behavior and physiology of marine animals, primarily mammals and fishes. Much less emphasis has been placed on the potential impact of underwater noise on humans, specifically human divers (Casper and Babina 2022). Recreational divers are not usually at risk of being exposed to high-energy noise sources, but military and commercial divers often operate in noisy environments as part of their regular duties. Examples of high-energy noise sources include underwater construction tools, large vessel traffic, sonar, and explosives. This population of divers is at risk of potential hearing damage, as well as other non-auditory injuries. Establishing safe exposure guidance for divers operating around active acoustic sources is critical to protect them from harm while also allowing them to complete their jobs.

A major challenge to developing effective underwater noise guidance is that, when submerged, evidence indicates that humans do not hear through the traditional air-conducted pathway. Instead, humans hear through the mechanism of bone conduction. For a detailed description of this alternate underwater sound pathway, please refer to Casper and Babina (2022). In short, there is experimental evidence that humans detect sound underwater that is delivered directly to the inner ear through bone conduction via the skull, bypassing the outer and middle ear. The research that supports this phenomenon will be discussed in further detail in the "Underwater Hearing Threshold Studies" section of this chapter. The challenge, however, is that most of what we know about noise safety guidance comes from in-air sound studies, and this in-air safety guidance cannot be used for underwater noise exposure. Therefore, researchers must determine what humans can hear underwater and what sound pressure levels will lead to injury, whether in the form of a hearing threshold shift or other non-auditory injury.

The goal of this chapter is to review the existing literature to illustrate the wide variations in results of hearing thresholds measured in humans. Additionally, the scant information available on human response to underwater noise exposure, including behavioral and physiological responses, will be discussed to emphasize the challenges associated with developing safety guidance to underwater noise exposure.

## Literature Review Approach

A review of the literature was conducted using PubMed, Google Scholar, and Defense Technical Information Center (DTIC) to identify articles related to underwater human hearing and noise exposure studies published between 1900 and 2022. Articles, conference proceedings, book chapters, and technical reports were only used if they were either publicly available through a journal website or, in the case of technical reports, downloadable through DTIC. Search terms included "human underwater hearing," "underwater noise," "diving noise," "underwater tool noise," "noise exposure," "human hearing thresholds," "hearing threshold shift," "diver hearing," and "diver noise exposure." Conference abstracts were not included. It should be noted that the author is aware of several additional international reports that exist but are not available for public distribution and were therefore not included in this chapter.

This underwater hearing literature review is organized in chronological order by decade, to show the reader how the exploratory research in this field has been inconsistent, with the bulk of efforts occurring between the late 1960s and mid-1980s.

## Underwater Hearing Threshold Studies

### Pre-1960s Experiments

The first documented experiment of underwater hearing in humans was the seminal work performed by Stetter (1929). The goal of the study was to compare hearing sensitivity at a single frequency between humans and minnows (*Phoxinus laevis* L.). A research subject was placed in a transparent water-filled box. The experimenter would walk up and down a hallway blowing a whistle at 662 Hz. The subject would signal to another experimenter if they could hear the stimulus while submerged. There was no calibration of the signal level, so Stetter was only able to conclude that minnows were more sensitive to the 662 Hz signal than humans. Even this basic conclusion should be made with caution as the small, enclosed testing environment would have been subjected to numerous surface reflections. Regardless of the limitations, this study established that humans are capable of hearing tones underwater.

Ide (1944) of the US Naval Research Laboratory performed underwater threshold measurements on several divers. This document was not available for this review, so any assessment made is based on critiques from Smith (1969). Smith's assessment (p. 4) was "Ide failed to report many important methodological details." No information is given on how the sound pressure level (SPL) at the diver's head was estimated and controlled or on the specifics of the test procedure. Underwater ambient noise levels were not reported. Air-conducted hearing levels of subjects were also not reported. Smith reported that the tests were conducted at depths of 0.9 to 1.8 m in water that was 10.7 to 12.2 m deep. Furthermore, Smith reported

"audiometer-type tests" were used, and the subjects were three men listening underwater with "the unaided ear." Measurements were collected from 100 to 6000 Hz. Overall, threshold results (using "audiometer-type tests") below 1000 Hz were higher than any other study reported in this chapter, raising questions about ambient noise levels during testing. With so many unknowns, the author agrees with Smith (1969) that it is not appropriate to include this data in the "Analysis" section of this chapter.

Sivian (1947) reported on an earlier study in which subjects were tested in a swimming pool. Initially, their ears were just above the water to collect in-air hearing thresholds at 1000 and 3000 Hz. The method of obtaining thresholds was not reported in this study. The subjects were then submerged either 22.8 or 38.1 cm, and thresholds were collected from the same airborne source. The subjects were found to have a 44–55 dB decrement in sensitivity from underwater to in air. No measurements of the sound levels were made. The measurements in air are suspect due to the uncontrolled environment and suspected high levels of ambient noise (Smith 1969). Sivian, however, was the first to suggest that human underwater sound detection could be through bone conduction.

Reysenbach de Haan (1956) was the first to attempt to measure signal level underwater. In an 18-m-deep pool, subjects (number not reported) were positioned 25 m away from the sound source with their heads "just below" the surface. Thresholds were measured between 1000 and 16,000 Hz. The signal level was lowered until the subjects reported that they could no longer detect the sound. Sound pressure level was not measured at the subjects' heads; rather, it was calculated based on measurements from a different location in the test pool. Additionally, ambient noise was not measured (or reported). The depth of the subjects could also have affected the results. With their heads just below the surface, sound transmission due to reflections could interfere with the results. An interesting variable that was explored was the difference in thresholds obtained if the diver had air trapped in the ear canal. How the air was removed was not documented, but thresholds were found to be lower at most frequencies. It should be noted that this finding was not observed in more recent studies. Reysenbach de Haan also promoted the theory of bone-conducted hearing underwater.

Hamilton (1957) was the first scientist to measure sound pressure levels at the location of the divers' heads as well as the ambient noise of the environment. Two subjects using scuba would sit on either end of a support beam with a hydrophone placed between them at the level of their heads. The divers were submerged 4 m below the surface of a 9.1-m-deep lake. During signal transmission, divers would hold their breath to reduce breathing and bubble noise. This was also the first study to test in-air and underwater hearing using the same subjects, a trend that would continue moving forward with most studies. Tones between 250 and 4000 Hz were tested in both environments. To establish thresholds, the sound levels were decreased by 1 dB steps until the subjects could not hear a signal and then increased until the subjects could reliably hear again. This was repeated 20 times at each frequency resulting in a thorough, rigorous data set.

To further test the bone conduction theory, Hamilton (1957) had the divers plug their ears with their fingers, with subjects reporting that they did not perceive a difference in loudness of the signal. In addition, the underwater hearing thresholds compared to air-conducted thresholds were offset by about 40 dB, which matched the difference between air-conducted and bone-conducted thresholds reported in the literature at that time.

The only concerns raised by this study were the potential influence of the second diver in proximity to the first. It was possible that the presence of another diver could disrupt the sound field. However, Hamilton (1957) reported seeing no difference in signals from the hydrophone whether there were one, two, or no divers. The other concern is that one diver could be influenced by the behavioral responses of the other diver. Regardless of these concerns, this study was well designed and controlled, providing confidence in the results.

Wainwright (1958) obtained underwater hearing thresholds in two subjects using the method of limits at a depth of 3 m in a 15.2-m-deep lake. Thresholds were obtained between 125 and 4000 Hz. The transducer was calibrated prior to testing, but there was no mention of measuring the received level at the divers' heads or measurements of ambient noise (though this facility is known for its quiet environment, as the author has also used it for testing). Additionally, Wainwright measured in-air thresholds in both subjects. Results from this study suggest higher hearing thresholds than many other studies conducted around this time period (e.g., 15–30 dB higher than Hamilton (1957)). Smith (1969) suggested these subjects may have had some hearing impairments, and therefore these results would not be generalizable to the overall human population.

One method that Wainwright (1958) employed was to have the subjects plug their ears with their fingers. The subjects reported no change in perceived loudness, consistent with Hamilton's (1957) testing. The subjects used a closed-system (rebreather) scuba system to minimize the noise from air bubbles released during breathing. Wainwright made claims that the sound made from the rebreather system is below the threshold of human hearing, but it is unclear if the measurement of these sounds was done in air or underwater or how they were conducted.

A summary of the above pre-1960s studies reveals numerous approaches to measuring underwater hearing in humans. These studies established that humans are able to hear sounds underwater, with the likely pathway occurring via bone conduction rather than the traditional air-conducted pathway through the outer and middle ears. In general, most studies had unreliable methods, whether it was a lack of underwater measurements or subjects with questionable in-air hearing thresholds. With so many discrepancies, it appears that Hamilton's (1957) underwater threshold study is the only reliable study that can be carried forward for additional analysis.

## 1960s Experiments

Montague and Strickland (1961) tested seven subjects using similar methodologies to Hamilton (1957) and Wainwright (1958), except for using the von Békésy

procedure for measuring hearing thresholds. Divers were equipped with open-stage scuba gear. The thresholds were established between 250 and 6000 Hz. Four of the divers had normal in-air hearing, and three divers showed signs of high-frequency hearing loss. Overall, the underwater thresholds showed high variability between the subjects at most frequencies. The sound transducer was calibrated prior to testing, but no measurements were made at the divers' head or of the ambient noise. With these variables missing, it leaves many questions about the validity of the data.

Of interest, the researchers also had the divers wear neoprene hoods of differing thickness. All divers wore a 3/16″ hood, and some tried other thicknesses. In general, it appeared that the hood resulted in a 20 dB attenuation of frequencies at 1000 Hz and greater. There was no difference in thresholds at frequencies below 1000 Hz. When divers were asked to pull the hood back from their foreheads and ears, the hearing was restored.

Brandt and Hollien (1967) presented the first of a series of underwater hearing tests that were conducted by the University of Florida Communication Sciences Laboratory. Most of the studies were conducted at the Bugg Spring Test Facility in Florida in a 53.3-m-deep spring that was known for its quiet environment. The experimenters built an elaborate setup such that the divers would always be at a known location from the transducers and the signals received by the subjects could be easily measured and calibrated. Eight subjects (male and female) had their thresholds measured between 125 and 8000 Hz using the von Békésy procedure. The divers used open-circuit scuba equipment and held their breaths during acoustic signal transmission to reduce the bubble noise. Testing was done at two depths (3.7 m and 10.7 m) with minimal differences in thresholds between the two depths or between males and females at most frequencies.

Given the care taken with data collection, the underwater thresholds from this study are considered to be accurate and representative of human underwater hearing. There was a trend of reduced hearing at higher frequencies based on air-conducted thresholds; however, it is unclear if this is due to hearing loss or the noise levels in the environment where the air-conducted thresholds were obtained.

Two additional studies were conducted by the team during this decade. One examined the effects of air trapped in the outer auditory canal on underwater hearing thresholds (Hollien and Brandt 1969), and the other explored the effect of depth on thresholds (Brandt and Hollien 1969). In the first study (Hollien and Brandt 1969), subjects wore specialized ear plugs as they descended to the test depth. Once pressure was equalized, they would remove the ear plugs, and the test was administered. This method resulted in testing with the air trapped in the ear canals. These subjects were also tested with their ears irrigated with water to remove any trapped air bubbles. In both cases, the divers had similar hearing thresholds, further implying that the traditional air-conducted pathway of hearing is not the primary path of underwater hearing detection.

In the second study (Brandt and Hollien 1969), divers were tested at 21.3 and 32 m to further explore if depth had any effect on underwater hearing thresholds. There were minimal observed differences in hearing at these depths and with those previously tested (Brandt and Hollien 1967).

Smith (1969) measured underwater hearing thresholds in 16 male divers in a 22.9-m-deep quarry with the subjects 4.6 m below the surface. A reference hydrophone was used to measure the sound field at the divers' heads. Unique to this study, air conduction and bone conduction thresholds were measured (using standard audiometric methods) to compare to the underwater thresholds from 1000 to 8000 Hz. Smith provided the most compelling evidence for underwater hearing via bone conduction by finding that divers with suppressed air-conducted thresholds did not show suppressed underwater hearing thresholds. Only divers with suppressed bone-conducted thresholds had suppressed underwater thresholds.

In this same study, eight of the subjects had hearing thresholds measured with and without neoprene hoods. Results showed reduced sensitivity when wearing the hoods, consistent with Montague and Strickland (1961). Smith followed this study with another in a deeper environment, looking at the protective effects of hoods with depths. There was no observed difference in attenuation between hoods at 10.1 and 30.2 m, suggesting that the hoods provide equal protection at least to 30.2 m. There is an expectation that at some "crush" depth, the air trapped in neoprene will be reduced enough to minimize the sound reduction capabilities.

Smith (1969) conducted one further study to look at underwater threshold variability within subjects. He ran eight subjects three to four times, with at least 24 h separating each test. He concluded that intra-subject variability was within the range that is normally observed in air-conducted hearing testing, though his results did show higher variability with increasing frequency.

Underwater hearing testing in the 1960s provided a wealth of data that were collected in quiet, controlled testing environments with proper characterization of the sound fields. Specifically, the tests conducted by Brandt and Hollien (1967) and Smith (1969) qualify as data sets that can be carried forward for further analysis. Brandt and Hollien (1967) recognized the importance of testing in a low ambient noise environment, specifically choosing their location based on this variable. They also constructed an apparatus that would give them precise knowledge of the divers' heads and the ability to calibrate their sound measurements. Smith (1969) followed this same path while also recognizing the value of measuring subjects' bone-conducted thresholds to provide additional evidence that underwater hearing is via bone conduction.

## 1970s Experiments

Norman (1971) measured underwater hearing in three divers at three different frequencies, 250, 1000, and 2000 Hz, using the up-down parameter estimation by sequential testing method. Thresholds were measured using four different conditions: bareheaded, fully hooded, bareheaded with neoprene patches covering the ears, and hooded with holes cut so that the ears were fully exposed. As in previous studies, Norman found that hearing thresholds at 1000 and 2000 Hz were increased in all cases where the skull was covered (hood) or mostly covered (hood with ear

holes). The hood provided minimal attenuation at 250 Hz. The neoprene patches that only covered the ears had no impact on hearing thresholds.

Unfortunately, Norman (1971) did not report any of the thresholds measured; rather, he described changes in thresholds relative to the bareheaded condition. In addition, the study was conducted in a 2.6-m-deep swimming pool with sounds produced by an in-air speaker that was floating in a plexiglass box. There was no mention of how sound levels were calibrated.

Hollien and Feinstein (1975) conducted a study similar to Norman (1971), except that they used a hood that had ear tubes that extended into the outer ear canal. The study was conducted at the same Bugg Spring Test Facility as their previous work (Brandt and Hollien 1967). Seven subjects were tested from 250 Hz to 8000 Hz in the bareheaded, hooded, and ear-tube conditions at a depth of 9.1 m. They observed no difference in thresholds between hooded and hood with tubes, suggesting that the air-conducted pathway does not play a role in underwater hearing. However, at frequencies above 500 Hz, there were significantly lower hearing thresholds in the bareheaded condition when compared with either hooded condition, indicating that neoprene hoods attenuate sound only at higher frequencies. When comparing measured thresholds between this study and the previous one (Brandt and Hollien 1967), the latter study yielded lower average thresholds and will be used as the representative data from this research team.

The 1970s were largely devoid of underwater hearing threshold studies, although these two experiments are worthy of mention (Norman et al. 1971; Hollien and Feinstein 1975). Both continue to further the notion that underwater hearing is accomplished through bone conduction. The data from these studies will not be carried over for final analysis.

## 1990s Experiments

There were no hearing threshold studies conducted during the 1980s. Al-Masri et al. (1993) initiated a series of UK studies that would set new standards in terms of the number of subjects tested and optimal testing environments. Al-Masri et al. (1993) measured the underwater free field of hearing (also known as minimum audible field [MAF]) in 54 normal-hearing sport divers. The testing was conducted in an acoustically isolated steel box resulting in an ambient noise average of 44 dB re 1 μPA across most frequencies. While the depth of the tank was only 1.5 m, the sound field around the subjects was calibrated. Subjects held their breath during testing. The sounds tested were 1/3 band octave noise frequencies between 250 Hz and 8000 Hz, resulting in lower underwater hearing thresholds (20–35 dB) at most frequencies compared to any other human studies up to this point. Only the average thresholds were reported, however, and the authors do not provide error bars or raw data. In-air hearing thresholds were also collected.

Parvin and Nedwell (1995) conducted a similar test with 21 subjects. They used a slightly larger acoustically isolated steel box that was 2 m deep, but still had significantly reduced ambient noise levels, though not quite as low as Al-Masri

et al. (1993). Parvin and Nedwell used a combination of two underwater speakers to collect 1/3 band octave noise thresholds between 25 Hz and 16,000 Hz (von Békésy procedure), representing the widest range of frequencies tested by any underwater study. They also made comprehensive calibration measurements to ensure the accuracy of the sound field around the divers.

While the thresholds were not as low as Al-Masri et al. (1993), Parvin and Nedwell's data were still lower than most previous studies, and their expansion at the low- and high-frequency ends represents the most comprehensive measurements to date. Their data also includes error bars to illustrate the variability between subjects.

These two 1990s studies resulted in the largest data sets and lowest measured hearing thresholds of any underwater experiment. They also worked in optimal testing environments with very low ambient noise levels compared to previous studies as well as careful calibration of the sound fields. Both of these data sets will be carried on for final analysis.

## 2000–Present Experiments

Fothergill et al. (2004) sought to narrow down the exact frequency in which neoprene hoods begin to effectively reduce measured hearing thresholds in humans. The researchers measured the underwater hearing thresholds of 20 subjects from 100 to 500 Hz with and without 3 mm hoods using a two-interval forced-choice adaptive tracking method. This study was conducted in an 11.6-m-deep Navy test pool known for its low ambient noise. Divers were 1 m below the surface in an acoustically transparent, temperature-controlled tank. The sound field was carefully calibrated. Thresholds were identical with and without hoods at frequencies below 400 Hz. There was a significant difference (around 8–10 dB) between thresholds at 400 and 500 Hz. This study established the point at which the presence of a neoprene hood starts to affect underwater hearing in divers.

Fothergill et al. (2018) further explored the attenuation properties of neoprene hoods by testing diver hearing at numerous depths. Thirteen divers wore a 7-mm-thick neoprene hood while having their hearing thresholds measured between 100 and 12,000 Hz using a modified von Békésy procedure. Depth conditions ranged from 1.2 to 40.2 m, with all depths achieved by having the diver submerged in a water-filled container within a hyperbaric chamber. Data were reported as the amount of hood attenuation relative to bareheaded thresholds which were not specifically reported but were collected at each simulated depth. At 500 Hz and greater, there was a general pattern of reduction of attenuation with increased depth, with differences of 4–25 dB between the 1.2 m depth and the 40.2 m depth, depending on the frequency.

Sørensen et al. (2022) investigated other possible underwater hearing mechanisms in humans. They measured in-air and underwater thresholds between 125 and 12,000 Hz in seven subjects using the method of limits approach. In addition to the traditional metric of sound pressure, they also collected particle motion thresholds.

Sørensen et al. were also careful to test in an environment with low ambient noise. Overall, their sound pressure level thresholds were similar to many of the previous studies with the exception of 125 and 250 Hz which were the highest of any reviewed in this chapter. No explanation was given for these higher thresholds, as the ambient noise levels at these low frequencies were well below the human responses.

Sørensen et al. hypothesize that at frequencies below 1000 Hz, skull motion would be moving in unison with the particle motion of the water due to both having the same impedance, thereby removing the possibility of bone conduction at lower frequencies. They offer an alternative hypothesis relating to the resonance (around 600 Hz) of the air cavity in the middle ear which might explain humans' underwater hearing sensitivity being greatest around 500 Hz.

These first two of these 2000s studies provide valuable information on the attenuation properties of neoprene hoods which is critical for detection as well as protection from underwater sound sources. The hearing thresholds of Fothergill et al. (2004) and Sørensen et al. (2022) were measured in a well-controlled environment, and the quality of the data is high and will be carried on for final analysis.

## Analysis

We are including six underwater hearing threshold studies from the literature review for analysis (Hamilton 1957; Brandt and Hollien 1967; Smith 1969; Al-Masri et al. 1993; Parvin and Nedwell 1995; Fothergill et al. 2004). These experiments paired underwater hearing thresholds with in-air thresholds from the same subjects, and the researchers carefully measured the sound field that the divers were exposed to with precise, calibrated techniques. They also strived to conduct the experiments in relatively quiet, controlled environments to reduce the risk of signals being masked by ambient noise.

As seen in Fig. 1, the six experiments report different results, with as high as 45 dB separating hearing sensitivity at 500 Hz. As most studies did not report the raw data for individual subjects, or standard deviations, we can only make assumptions about how much overlap might exist between experiments. Regardless, there is significant variability between underwater thresholds obtained from the different experiments. Given that the UK studies (Al-Masri et al. 1993; Parvin and Nedwell 1995) were conducted in underwater anechoic and vibration-isolated environments and have the lowest thresholds at most frequencies, these tests are considered to be the gold standard of underwater hearing.

Another conclusion that can be drawn from these studies is that humans' most sensitive hearing range while underwater is from 400 to 1000 Hz. This represents a shift toward lower-frequency sensitivity when compared to in-air hearing, which is usually centered around 3000–4000 Hz.

When comparing human underwater hearing thresholds to other aquatic animals (Fig. 2), humans seem to fall in the relative middle. Humans are more sensitive than

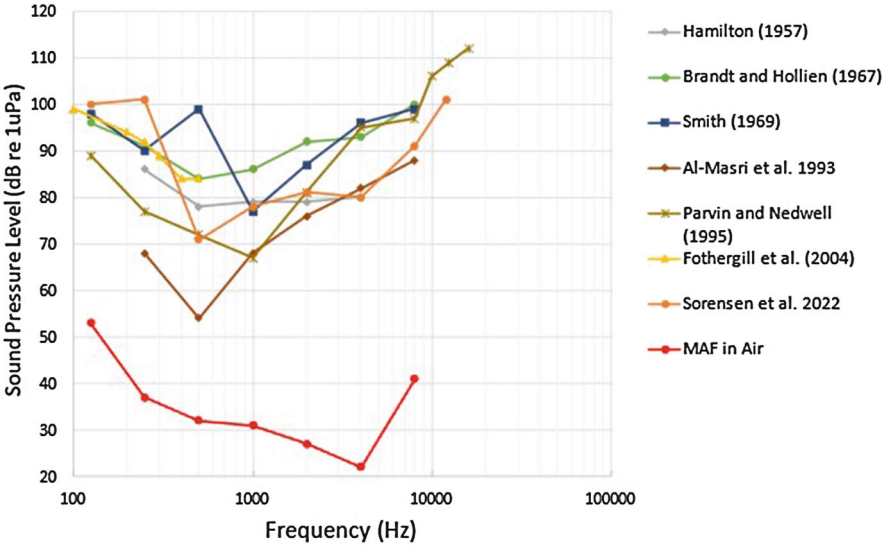



**Fig. 1** Human underwater hearing thresholds. These six studies were chosen based on methodological approaches that included calibration of the acoustic signals, quiet testing environments, and the collection of in-air hearing thresholds in addition to underwater thresholds. For comparison, MAF in air data is shown. (From Al-Masri et al. 1993)

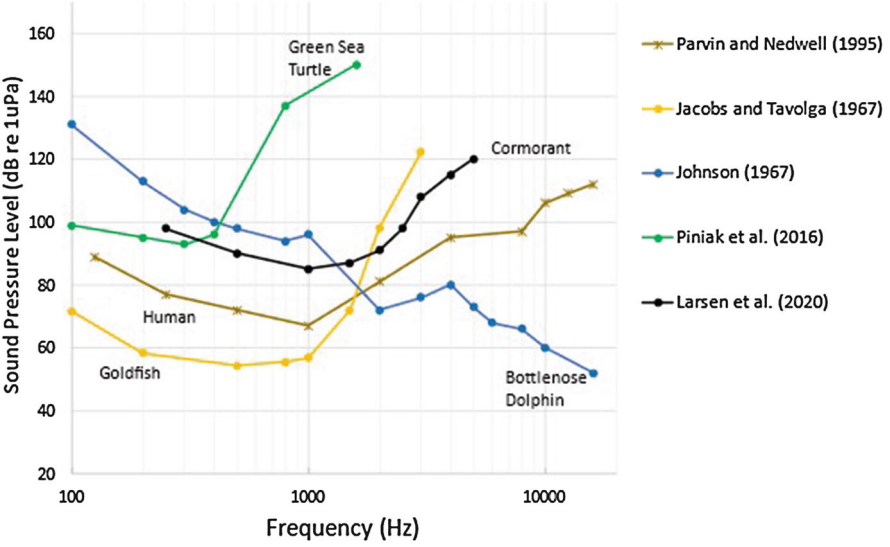



**Fig. 2** Human underwater hearing thresholds compared with aquatic animals

turtles (Piniak et al. 2016) and cormorants (Larsen et al. 2020). They are also more sensitive at lower frequencies than dolphins (Johnson 1967), but this is unsurprising, given that dolphins produce high-frequency whistles and echolocation clicks. Dolphins have more sensitive hearing at all frequencies of 2000 Hz and greater. In contrast, goldfish (Jacobs and Tavolga 1967), known for their low-frequency sensitivities, are more sensitive than humans until around 2000 Hz, after which humans are more sensitive.

## Underwater Noise Exposure Studies

Providing safety guidance to protect people from noise-induced injury is a large, diverse field of study. There is a wealth of data sets that have been used to set in-air hearing standards. However, as previously mentioned, the pathway for sound transmission to the inner ear underwater is different than in air (Casper and Babina 2022), meaning that in-air standards do not apply to underwater noise exposure. Developing safety guidance for underwater noise exposure is an ongoing effort that has been hindered by the lack of data for noise-induced injury, as reviewed in this section.

Montague and Strickland (1961) tested a hypothesis that if underwater thresholds are 40–70 dB higher underwater than in air, then the most tolerable loudness level for a given frequency underwater should be 40–70 dB higher than it would be in air as well. To test this, they exposed 17 divers to 1500 Hz, 1 s tones in an open water trial at a depth of 7.6 m. The researchers increased the sound pressure level in 1 dB steps until the divers deemed the sound level too unpleasant to continue. The tolerance level for these divers was set at 200 dB re 1 μPa. In the same study, while wearing hoods, divers were able to exceed the maximum output of 206 dB re 1 μPa without stopping the experiment. All divers reported a "rotational movement of the visual field" when exposed while unhooded beginning at 191 dB re 1 μPa that became more noticeable as the sound pressure level was increased. One diver also reported dizziness. Post-dive audiograms revealed no significant shifts in hearing thresholds.

Smith (1965) exposed two hooded divers to 1900 Hz, 700 ms signals with a 5% duty cycle at 195 dB re 1 μPa for 30 min at a depth of 5.2 m. Both divers reported that the signal was quite loud, but post-exposure audiograms revealed no threshold shifts at any frequencies.

Smith et al. (1970) built on what Montague and Strickland (1961) investigated by testing whether underwater signals that induced a temporary threshold shift (TTS) were 40–70 dB higher than those that produced an equivalent TTS in air. Six unhooded divers were exposed to a 3500 Hz signal that was 1250 ms in length every 2.5 s for a total of 15 min at 194–204 dB re 1 μPa. Five different subjects were exposed in air to 164–179 dB re 20 μPa. A TTS of 10–22 dB was observed in the divers. The difference between in-air and underwater levels for inducing equivalent TTS was approximately 68 dB, which is within the difference of in-air and underwater hearing thresholds. In addition, one diver reported a visual distortion sensation similar to what Montague and Strickland (1961) reported.

Molvaer and Gjestland (1981) measured the noise levels of underwater pneumatic rock drills and water lances as perceived by divers wearing fully enclosed dive helmets. Three divers were tested, the first two wearing a different type of helmet and the third a 5 mm neoprene hood. Their hearing was measured during a control period and pre- and post-exposure to one of the tools. There were notable threshold shifts in both helmeted divers, with shifts as high as 30–40 dB at 5 and 8 kHz. Threshold shifts in the hooded diver were less prominent, averaging 10 dB or less.

The water lance had the highest sound levels outside of the helmets at 170 dB re 1 μPa. The two helmeted divers had a microphone inside their helmets to measure the internal noise levels produced by the tools. Sounds were attenuated between 30 and 50 dB depending on frequency relative to in-water measurements. There was a general trend of increasing attenuation of the signal (as it passed through the helmet) as the frequency increased.

Smith and Wojtowicz (1985) exposed four bareheaded divers for 25 min to tones at 700, 1400, and 5600 Hz, at sound levels between 169 and 191 dB re 1 μPa. The authors suggested that this would be similar to frequency ranges and durations that divers might be exposed to with underwater tools. TTS measured at 2 min post-exposure were between 23 and 55 dB. Recovery time for these shifts lasted from 24 to over 50 h until divers returned to baseline hearing thresholds. With such high levels of TTS, the authors were not able to make a comparison between underwater and in-air sounds that would yield the same levels of TTS. Non-auditory effects that were reported included the sensation of head vibration, a tickling sensation in the middle ear, ringing in the ears, and pain and blood in ears.

Smith et al. (1988) expanded on Smith and Wojtowicz (1985) by running the same experiment but with lower sound pressure levels to determine what levels would reliably induce noticeable TTS. Again, four bareheaded divers were exposed for 10 min to 700, 1400, and 5600 Hz at sound levels between 151 and 176 dB re 1 μPa. Threshold shifts between 3 and 21 dB were observed among the subjects with increasing TTS following increases in sound pressure levels. No non-auditory symptoms were reported. Sound levels that yielded this same level of TTS in air were roughly similar to those observed in water, with the largest difference being at 1400 Hz where levels needed to be 5–15 dB higher underwater.

Smith (1988) exposed eight hooded divers to 3500 Hz tones (250 and 500 ms in length) at sound levels up to 218 dB re 1 μPa for up to 1 h. Hour-long exposures of 217 dB re 1 μPa yielded 40 dB of TTS, while shorter durations (5 and 20 min) at equal or lower sound pressure levels yielded TTS of 0 to 28 dB. Non-auditory effects that were observed included force felt on their face mask, water within the facemask bursting into spray, and some possible ocular effects.

Smith et al. (1996) conducted a series of noise exposure tests using warble tones with center frequencies between 125 and 6000 Hz. In the first test, seven unhooded divers were exposed for 15 min to warble tones of 250 Hz at 141 dB re 1 μPa; 500, 1000, 2000, and 3000 Hz at 161 dB re 1 μPa; and 6000 Hz at 171 dB re 1 μPa. TTS was less than 5 dB at all frequencies except for 500 Hz which was up to 15.5 dB. In the second test, 12 unhooded divers were exposed to 4 min of warble tones with center frequencies between 125 and 6000 Hz between 121 and 161 dB re

1 μPa. TTS did not exceed 10 dB for any of the subjects following exposures. It was concluded that divers could be exposed to 15 min of warble tones at 151 dB re 1 μPa for 500–1000 Hz, 161 dB re 1 μPa for 2000–3000 Hz, and 171 dB re 1 μPa for 4000–6000 Hz with no risk of auditory injury. Also, at lower frequencies, divers could be exposed to 4 min of 125–1000 Hz at 161 dB re 1 μPa with no risk of injury. There were no reports of non-auditory symptoms.

Steevens et al. (1997) tried to identify if an open water noise source exposure would yield different results than the more confined studies that have been reviewed so far. Utilizing the Bugg Spring Test Facility (also used by Brandt and Hollien (1967)), signals were produced using two speakers to create a plane progressive traveling acoustic wave. Six divers were exposed to a warble tone with a center frequency at 240 Hz and a slow sweep signal from 230 to 320 Hz, both at 160 dB re 1 μPa for 15 min. Divers were tested in both a helmeted scenario and bareheaded. Change in thresholds following exposure was 5 dB or less and could have been due to other non-experimental factors rather than the sound exposure.

Fothergill et al. (2001) explored the aversiveness of low-frequency sonar in recreational divers (no US Navy training). Twenty-six unhooded divers were exposed to pure tones, 30 Hz hyperbolic sweep up, and 30 Hz hyperbolic sweep down, each presented at six center frequencies between 100 and 500 Hz and between 130 and 157 dB re 1 μPa. Each signal was 7 s in duration for a total of 12.6 min of exposure. There was a general trend of increased aversiveness and subjects dropping out of the procedure with higher sound pressure levels. The frequencies that divers found most aversive were 100 and 500 Hz, with 250 Hz being the least aversive. Their final recommendations were that recreational divers could be exposed to 100–500 Hz at up to 145 dB re 1 μPa with minimal impact.

## Analysis

Many of these noise exposure studies were motivated by the need to establish a human response to a specific frequency, likely within the range of a sonar system of interest at the time. Because of this, the frequency range of sounds that have been tested is limited. Threshold shifts of varying amounts were observed at some frequencies between 700 and 6000 Hz, but the amount of TTS and the levels needed to induce these shifts were frequency dependent. Data were also collected over much of this same frequency range in which TTS was not observed at lower levels. Unfortunately, there is not enough overlap between these experiments to establish an onset of TTS, which is critical for establishing underwater hearing guidance.

Understanding the human behavioral responses (or perception) of these high-energy signals is also valuable. A few studies looked at aversion (Fothergill et al. 2001) or maximum tolerable level (Montague and Strickland 1961), but these were limited in scope. While a signal may not be loud enough to cause physical injury, it could still be perceived as uncomfortable or disturbing enough that it could disrupt a diver's ability to safely complete a dive. To some extent, this phenomenon was illustrated in these studies as subjects sometimes experienced perceptual aberrations

that presumably could affect performance and would drop out of the study once they reached a sound level that was uncomfortable enough to avoid continued exposure. But in both studies, there was no evidence of TTS from these exposures.

Expanding on the limited body of literature on underwater sound exposure is important to improve upon existing noise exposure guidance, but it is hard to recommend this line of study as it puts subjects at risk of temporary or permanent hearing damage or other non-auditory symptoms that could put the diver at risk. While it is not likely that additional data will be introduced into this limited field of study, any opportunities to better understand human response to underwater noise exposure would be valuable. Regardless, the existence of the experiments that have been conducted provides enough insight to provide recommendations as needed for this frequency range.

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# Obtaining Underwater Hearing Data for the Common Murre (*Uria aalge*)

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Kirstin Anderson Hansen, T. Aran Mooney, and Magnus Wahlberg

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**Abstract**

Many species of seabirds dive to great depths and for prolonged intervals in pursuit of food, and they may therefore be affected by anthropogenic underwater noise. The paucity of seabird hearing data precludes the current understanding of noise impacts and limits the abilities to enact with informed mitigation efforts. One interesting family of birds in this respect is the auks (Alcidae). A previous study showed that common murres, *Uria aalge*, react to underwater sound, but no underwater hearing thresholds have so far been derived from any auk species. Here the underwater hearing abilities of a male common murre were measured using psychophysical methods. The results indicate that common murres, a deep diving seabird, respond to underwater sound at a received level of 130 dB re

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1 μPa, underscoring the potential that this species, and perhaps other auks, may be affected by man-made underwater noise.

**Keywords**

Aquatic birds · Behavior · Psychophysics

## Introduction

Hundreds of species of marine birds make their living by performing extensive dives pursuing fish and invertebrates. Marine birds are considered especially vulnerable to climate change, bycatch, and other human-induced factors, making them among the most imperiled of all avian taxa (Lascelles et al. 2016; IUCN 2017). Additionally, there are recent concerns that anthropogenic noise may affect marine birds while diving, similar to how noise can affect marine mammals and fish (Popper et al. 2008; Popper and Hawkins 2012, 2016).

A key to understanding how marine birds are influenced by underwater noise is to discern which senses they use while diving. They are thought to rely on visual cues, just like most birds do in air. However, some marine birds can hunt under poor light conditions and in murky waters; they use tactile and perhaps also gustatory cues while diving (Enstipp et al. 2007; Grémillet et al. 2005; Thiebault et al. 2019; Wenzel 1980). Underwater sound is efficiently propagated through the environment and is a major sensory cue for diverse groups of marine animals, such as whales, dolphins, seals, fish, and arthropods, as well as for semiaquatic animals such as turtles and crocodiles (Thewissen and Nummela 2008). Given these sensory conditions, seabirds are also likely to make use of widely available underwater acoustic cues.

Both great cormorants and black ducks can detect underwater sounds (Anderson Hansen et al. 2017; Larsen et al. 2020; McGrew et al. 2022). Furthermore, common murres and gentoo penguins react negatively to intense underwater noise pulses, and gentoo penguins can be conditioned to pair underwater sound cues with food (Anderson Hansen et al. 2020; Sørensen et al. 2020; Rasmussen et al. 2022). Thus, there is some evidence that underwater sound cues can be utilized by marine birds while diving. But more information is needed to determine their sensitivity to underwater sound, and how they make use of underwater sound cues.

An interesting species in this respect is the common murre (*Uria aalge*), a common North Atlantic species of auk. Auks are found throughout the Northern Boreal and Arctic regions and are known for their extreme foraging abilities, diving to great depths for prolonged time and staying for months and even years at sea before gathering on land in large colonies to breed (Barrett and Furness 1990; Croll et al. 1992; Tuck 1961). There is physiological evidence that Atlantic puffins (*Fratercula arctica*) and common murres hear well in air (Smith et al. 2023; Mooney et al. 2019, 2020). They acoustically communicate in air and live in raucous seabird colonies, underscoring the importance of aerial hearing (Hansen 2018; Smith et al. 2023). Finally, as noted above, a previous study showed that common murres are

sensitive to sound and react negatively to underwater sound pulses (Anderson Hansen et al. 2020). However, their underwater sound sensitivity has not yet been determined.

Here, a common murre was trained for psychophysical measurements to begin to address underwater hearing thresholds, and the equipment and steps used to derive the bird's abilities to hear a narrowband tone of high intensity are outlined. The experimental setup offers a reliable method to obtain hearing data from an animal not easily worked with standardized psychophysical instrumentation.

## Materials and Methods

### Experimental Subject

A male captive-born common murre was tested at the University of Southern Denmark's Marine Biological Research Center in Kerteminde, Denmark. Upon his arrival in November 2017 (3 years old), he received extensive training in operant and classical conditioning for psychophysics, as well as husbandry and enrichment behaviors.

### Training

Positive reinforcement methods were used to train a yes/no testing paradigm. The murre was trained to swim through an underwater hoop, which then triggered the start of the trial via an optical sensor. The bird then swam to one of the two response targets, indicating if it heard the stimulus or not (Fig. 1). For a *YES* trial to be considered correct, the murre touched the left (red) response target with his beak after a stimulus was presented. For a *NO* trial to be considered correct, the murre touched the right (yellow) response target after no stimulus was presented. A correct response was marked with a conditioned reinforcer, a dog whistle, followed by an

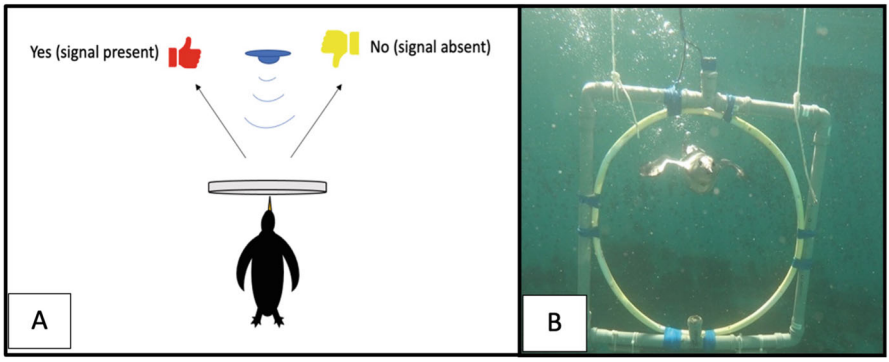



**Fig. 1** (**a**) Diagram of the experimental setup showing the general trial design; (**b**) the bird swimming through the experimental hoop, from the view of the speaker and response targets

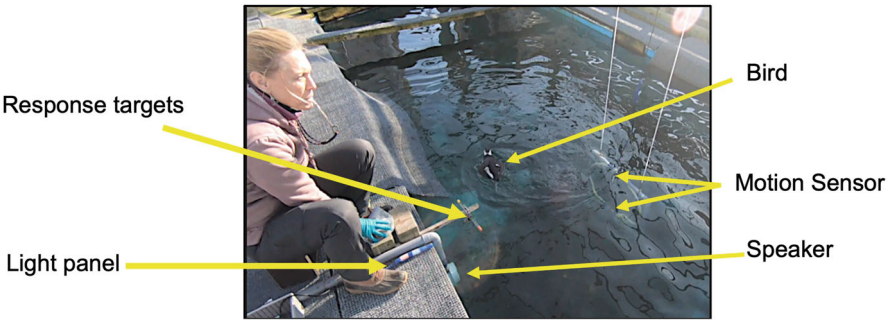



**Fig. 2** Experimental setup showing the various components as seen from above water

unconditioned reinforcer, a piece of fish, which was given by the trainer. If the bird responded incorrectly, no conditioned reinforcer was presented and no unconditioned reinforcer was given. The bird was then given the cue to start a new trial (Fig. 2).

## Apparatus

The setup was made to accommodate the bird's natural swimming behavior. A transmitter and sensor (Automation24, Essen, Germany, models OGS102 and OGE108) on the top and bottom of the hoop triggered the start of each trial, with a small light panel next to the trainer, indicating if the trial was a stimulus-present trial or stimulus-absent trial. The light panel secured impartiality of the trainer by providing a double-blind procedure as the trainer did know the type of trial (stimulus present or absent) until the bird passed through the hoop and gave its response. The trainer used a computer mouse to record the response of the bird, left click for correct and right click for incorrect. The total number of trials and the number of correct and incorrect responses, as well as the relative intensity of the sound stimulus for each stimulus-present trial, were registered and compiled at the completion of each session. For a given session, the animal received a constant sound level (for signal present trials), or no sound, following a method-of-constants paradigm. All sounds were generated, and data were recorded using a custom LabView program implemented on the nearby laptop connected with USB to a DAQPAD (National Instruments USB-9162, digital to analog converter working as sampling rate 48 kHz with 16 bits). The output of the DAQPAD was connected to a Basetech AP-2100 amplifier and to an underwater loudspeaker (University Sound UW-30).

## Stimulus

The stimulus was a 1 kHz 1 s pure tone with a received level of 130 dB re 1 μPa rms. During calibration, the stimulus was measured in different locations in and around

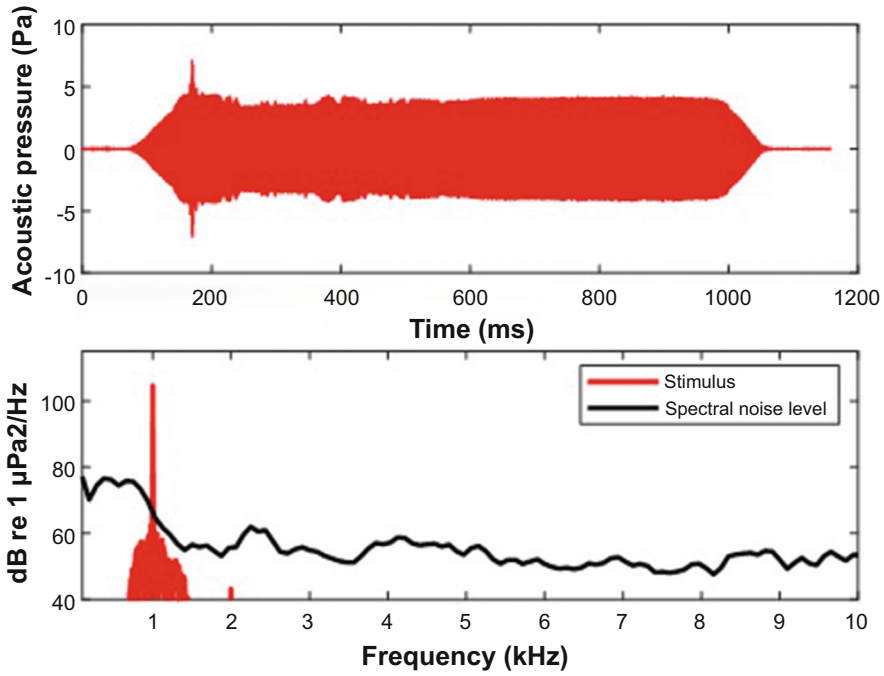



**Fig. 3** (Top) Stimulus waveform and (bottom) frequency analysis of stimulus and background noise, both in units of spectral levels (analysis bandwidth 1 Hz)

the hoop with a calibrated Soundtrap HF300 and a Reson 4032 hydrophone connected to an Olympus LS100 recorder (sample rate 48 kHz, 16 bits). The signal level varied by less than 3 dB within the hoop and ±10 cm in front and behind the hoop. Ambient noise levels were measured for 2 min before and after sessions at regular intervals with the same equipment used for stimulus measurements (Fig. 3).

## Testing Procedure

The session was run by the trainer using a custom-made Labview (National Instruments version 2018) program on a laptop. Once the session was started, a preprogrammed randomized sequence following a Gellerman protocol initiated either a stimulus-present or a stimulus-absent trial (Gellermann 1933). Both trial types were initiated in the same way with the only difference being that no stimulus was presented in signal-absent trials. For signal-present trials, the stimulus was presented when the murre swam through the hoop and triggered the motion sensor. It was not possible for the trainer to hear the underwater stimulus during testing. Each session consisted of 25 trials, out of which 10 were catch trials, with no signal present.

## Results

The murre's performance during the psychophysical tests were highly variable; however, the correct detection rate of 74% was consistently and significantly higher than the false alarm rate of 54% (Fig. 4; Wilcoxon sign rank test, $p < 0.05$). The stimulus level was believed to be clearly audible for the murre, from the following reasoning. Comparing in-water and in-air hearing thresholds has only been achieved in one species of sea birds, the great cormorant, indicating that the thresholds are comparable in units of pressure (Larsen et al. 2020). In-air hearing data on common measured by Smith et al. (2023). Extrapolating from in-air hearing data to underwater hearing thresholds under the assumption of equal pressure sensitivity indicates that the stimulus level used in the experiments were more than 60 dB higher than the presumed hearing threshold of common murres at 1 kHz.

## Discussion

Even though the response of the murre was variable between different sessions, the data clearly shows it could hear the underwater stimulus (Fig. 4). This is corroborated by the fact that the signal level was more than 60 dB above the presumed hearing threshold underwater at 1 kHz (Fig. 5). The stimulus intensity is similar to

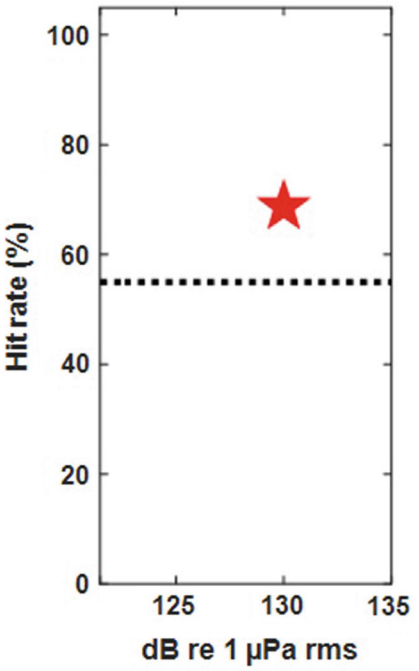



**Fig. 4** The correct detection rate at 130 dB re 1 µPa, indicated by a star, compared to the false alarm rate, indicated by a dotted line

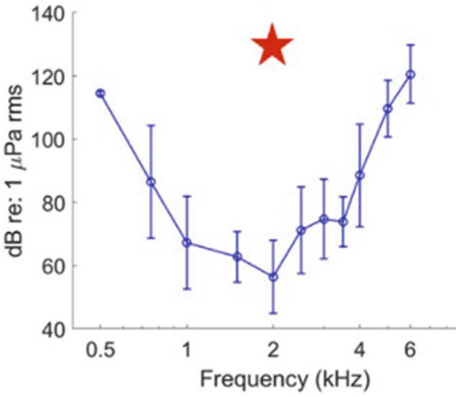



**Fig. 5** Stimulus level in underwater psychophysical experiments, compared with the estimated under water audiogram of a common murre, under the assumption of equal pressure thresholds in air and under water. (Modified from in-air ABR data from Smith et al. Smith et al. 2023)

the sound levels of signals (white noise and midfrequency sonar signals), which induced unconditioned flight responses from a feeding station (Anderson Hansen et al. 2020). The results indicate that murres can hear the stimulus level of 130 dB re 1 μPa were therefore as expected.

Notably, this animal reliably responded to the presence of the tone. With a correct detection rate of 74%, significantly above a false alarm rate of near chance, the murre certainly was conditioned to the task. This suggests that murres can be conditioned for underwater hearing tests and that it may be possible to determine auditory thresholds using psychophysical methods. The next step is to address correct detection rates at lower sound levels.

As the bird could clearly hear the stimulus, the reason for its variable performance in the psychophysical sessions cannot be explained by the bird having problems hearing the stimulus. Previous studies have shown that seabirds are extremely challenging to train for psychophysical work (Anderson Hansen et al. 2017, Maxwell et al. 2017, Rössler et al. 2022). From prolonged work with training marine birds for psychophysical work, the murres seemed less inquisitory and motivated to participate in operant conditioning tasks than, e.g., cormorants. Further work is needed to understand if the lack of motivation is caused by a lifestyle with a rather predictable environment and social behavior, or possibly some other factor.

The outlined here could be used to derive underwater hearing thresholds of the murre by reducing the stimulus level until the correct detection rate reaches the false alarm rate. The derived thresholds would be for a moving bird, which may be higher than a stationary bird due to noise created by moving through the water. As murres are dynamic swimmers, constantly moving while diving, such a threshold would more realistically indicate the hearing abilities of the bird in natural situations. Additional work could leverage these methods to determine if murres (or other seabirds) functionally use sound underwater, perhaps via auditory directionality and sound source localization experiments. Given their sensitivity to underwater sound, additional studies on noise impacts, in this and related taxa, are also warranted.

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# Section VIII

# Noise Effects: General

# Implications of Multimodal Integration and Masking on Acoustic Deterrent Efficacy

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Philip S. Ericsson, Paul R. White, and Paul S. Kemp

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**Abstract**

Acoustic deterrents are popular form of behavioral guidance measure used to manage fish movements in rivers. However, despite their popularity, the reported efficacy of these systems is often quite variable. One potential reason for this is that peripheral sensory inputs can influence how fish respond to deterrent stimuli. While little research has examined these effects in relation to deterrent function, they have been well studied for many decades in different contexts. Therefore, this chapter provides a brief review of literature pertaining to how multimodal integration and masking influence fish behavior and suggests how this vast body of literature may inform the design and implementation of acoustic deterrents in the future. Highlighted is the need to understand the sensory background against which deterrents operate, as peripheral sensory information may impact the perception of and thus behavioral response to a given deterrent stimulus. Furthermore, acoustic masking may impair a fish's ability to properly distinguish acoustic deterrents from background noise. Improving acoustic deterrent efficiency will

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require complimentary approaches, utilizing fieldwork to quantify the sensory environment present against which acoustic deterrents must function, and controlled experiments to elucidate the relationships between these conditions and fish behavioral responses to deterrent stimuli.

**Keywords**

Multimodal integration · Perception · Masking · Acoustic deterrents

## Introduction

The development of infrastructure for energy production, agriculture, and other purposes creates dams and other barriers which reduce river connectivity and disrupt fish movement, blocking access to important nursery, feeding, and spawning grounds (Baumgartner and Wibowo 2018; Conallin et al. 2019; Delavan et al. 2017). While fish passes are being installed and retrofitted to improve fishes' ability to safely bypass these barriers, migration delays still occur when fish encounter such infrastructure, which can lead to increased predation, reduction in energy stores, and loss of mating opportunities (Caudill et al. 2007, 2013; Fjeldstad et al. 2018; Gowans et al. 2003). Furthermore, impingement upon and entrainment into infrastructure is also common, resulting in physical damage and mortality for fish that are unable to avoid these barriers (Taft 2000). Additionally, there is a need to limit the spreading threat of invasive species which can disrupt ecosystem services and negatively impact the status of native species in river systems where they have been introduced (Cupp et al. 2021; Murchy et al. 2017).

These threats have led to the development of a range of both physical barriers and non-physical, behavioral deterrents to guide fish movements in rivers. Non-physical deterrents have become an increasingly popular option for guiding fish movements in rivers (Jesus et al. 2019; Noatch and Suski 2012; Popper and Carlson 1998). Such deterrents function by triggering the existing sensory biases to elicit avoidance behaviors in target species and are often preferred over physical barriers as they are less costly to construct and maintain and do not interfere with the normal flow of the river (Noatch and Suski 2012). Behavioral deterrents utilize a wide range of abiotic stimuli, including light (Kim and Mandrak 2017), $CO_2$ (Tix et al. 2018), electricity (Miller et al. 2022), air bubbles (Flores Martin et al. 2019), and acoustic stimuli (Deleau et al. 2020; Heath et al. 2021). While there is no "one size fits all" solution (Noatch and Suski 2012), and each of these deterrent types has their own applications, acoustic stimuli have several advantages that make them particularly suited for managing fish movements in rivers. For example, all fish are capable of detecting acoustic stimuli (Popper and Fay 2011), and many fish species have unique specially adapted hearing frequency ranges, meaning it may be possible to fine-tune and target acoustic stimuli toward a specific species or subset of species (Vetter et al. 2018). Furthermore, acoustic stimuli spread rapidly underwater due to the physical properties of the medium, and their propagation is independent of water clarity,

meaning that these stimuli can be detected at a great distance from the source in the often highly turbid environment of many rivers (Popper et al. 2020).

Despite the advantages of using acoustic stimuli as non-physical deterrents, their efficacy is often highly variable, both between sites and between species (Maes et al. 2004; Putland and Mensinger 2019). One potential reason for this variation is that they have been deployed in a wide range of environmental conditions (Jesus et al. 2019; Noatch and Suski 2012). Non-physical deterrents have historically been constructed using trial-and-error approaches, with a lack of detailed understanding of how fish respond to specific sensory stimuli (Popper et al. 2020). Studies examining behavioral responses to acoustic deterrents often focus on studying the effects of stimulus presentation such as frequency characteristics (Currie 2021; Vetter et al. 2018), inter-species differences (Deleau et al. 2020; Matsuda 2020), or temporal characteristics (Currie et al. 2020) but don't consider how other sensory information might influence how individuals respond to these stimuli. However, it is important to realize that behavior can be influenced by information received across different sensory channels (Fetsch et al. 2013), and the perception of a specific stimulus might vary depending on the presence of other sensory stimuli (Chan and Blumstein 2011; Halfwerk and Slabbekoorn 2015; Paglianti et al. 2010). So far however there has been little consideration of how the wider sensory environment influences the perception of behavioral deterrents, and it is therefore useful to provide a brief overview of how such peripheral stimuli may influence fish behavior and to discuss how these effects may influence the efficacy of acoustic deterrents.

## Multimodal Integration

Multimodal integration is a process by which animals incorporate information received across different sensory modalities into decision-making and behavioral processes (Partan and Marler 1999). Different sensory channels are capable of providing independent estimates of the environment (Driver and Spence 1998); thus integrating stimuli from different sensory channels can function to decrease environmental uncertainty, which allows the individual to exhibit more accurate behavioral responses (Munoz and Blumstein 2012, 2020). Indeed, (Partan and Marler 1999) set out a framework to explain how the integration of multimodal information influences behavior. Here, individual component stimuli are first classified as either "redundant" or "non-redundant" based on the type of behavioral response they each elicit in isolation. Redundant stimuli are those that elicit the same type of behavioral response from an individual, while non-redundant stimuli are those that elicit different responses from each other. Then, depending on if the stimuli are redundant or non-redundant, a set of potential responses to the combined multimodal stimulus can be categorized as examples of different processes relative to the behaviors elicited by each individual stimulus (redundant stimuli: equivalence, enhancement, or antagonism; non-redundant stimuli: independence, dominance, modulation, or emergence) (Fig. 1). These potential outcomes highlight the varied effect that integration can have on behavior and show that behavioral response to an

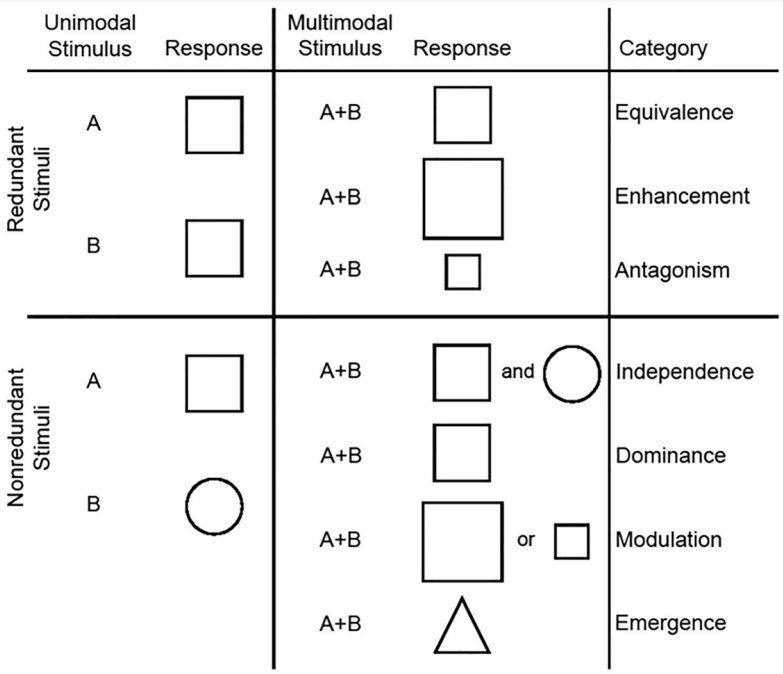



**Fig. 1** Potential behavioral outcomes from multimodal integration. Depending on how the individual responds to each component stimulus in isolation (response to unimodal stimulus), different potential responses to multimodal stimuli are possible (Adapted from Munoz and Blumstein (2012))

acoustic deterrent may differ quite significantly depending on the information available to other modalities.

Studies of multimodal integration in fish have largely focused either on elucidating the organization and function of the nervous system (as in the case of studies examining the function of the Mauthner cells under different sensory conditions (Medan et al. 2018; Otero Coronel et al. 2020) or on understanding how the environment and anthropogenic impacts influence fitness-related behaviors, such as those in predator-prey interactions (Hasan et al. 2018; Montgomery et al. 2002) and mating displays and communication (de Jong et al. 2018; Halfwerk and Slabbekoorn 2015; van der Sluijs et al. 2011).

Mauthner cells are large reticulospinal command neurons that integrate information from different sensory channels (Eaton et al. 2001) and initiate fast-start startle response behaviors. As such, these cells have been extensively studied as a model system for understanding how multiple sensory inputs inform startle response behavior in fish (Domenici and Hale 2019). Multimodal integration has been shown to enhance this behavior, with recent research showing that the probability of a startle response increases when fish are exposed to a multimodal combination of visual and acoustic stimuli (Martorell and Medan 2022; McIntyre and Preuss 2019). Specifically, these studies show that startle response behavior is more likely when

fish are exposed to pulsed tonal stimuli in combination with a concurrent looming visual stimulus in comparison to fish exposed to only one of the individual stimuli. The potential application of this research has been recognized, and acoustic deterrents that incorporate a complimentary multimodal component (usually either stroboscopic lights (Bui et al. 2013) or bubble curtains (Flores Martin et al. 2021)) are currently being designed and implemented. It has been suggested that fish may be less likely to habituate toward such multimodal deterrents, due to the presence of multiple aversive stimuli (Lecker et al. 2015); however research into the effectiveness of the systems over traditional, unimodal deterrents show that their efficacy can still vary (Bzonek et al. 2020, 2021).

Connections like these between fundamental research focused on understanding the integration of different sensory inputs within an individual and more applied work focused on designing improved behavioral guidance measures highlight the importance of multidisciplinary approaches to problem-solving. In this context, fundamental research should be used to identify stimulus-response relationships and how they change under different multimodal environments, while more applied work should focus on taking this knowledge into the field and testing to see whether or not these relationships hold in more naturalistic settings.

The effect of multiple different stimuli on startle response behavior has also been shown to depend on species-specific sensory capabilities and on the direction from which stimuli are detected. For example, Canfield (2006) showed that visual stimuli detected in the rostral visual field of *Haplochromis* depolarized Mauthner cells 1.5 times more than did the same stimuli presented from a caudal direction. Additionally, behavioral testing showed that rostrally projected visual stimuli in combination with an acoustic stimulus produced startle responses with significantly larger turning angles and turning speeds than for multimodal stimuli where the visual stimulus was projected from a caudal direction. If a deterrent stimulus is only effective from certain directions relative to the individual, then its performance will likely be severely impacted due to variations in individual behavior and approach direction. As such the specific physiology and sensory sensitivities of target species should be considered not just when determining what stimuli to use but also in the installation of the system. Transducers and strobe lights should be placed in locations where they will have the maximum chance of eliciting the desired behavior from target species.

The role of multimodal integration in guiding predator avoidance behaviors is a particularly interesting point of comparison when considering how responses to acoustic deterrents might vary with ancillary sensory stimuli. Behaviors in these two contexts are similar to each other, as deterrents seek to elicit avoidance behaviors from target species. In this sense behavioral deterrents can be thought of as predators, alerting target species to their presence with the stimuli they emit. As such, studies examining how predator avoidance behaviors vary with different sensory inputs could provide interesting hypotheses to explain why responses to behavioral deterrents might change under different sensory conditions. Paglianti et al. (2010) demonstrated that the behavioral response of Rosy barb (*Barbus conchonius*) to conspecific chemical alarm cues varied depending on the availability of visual cues. The authors found that fish did not exhibit typical startle behaviors in response to the

cue in dark conditions, suggesting that this avoidance behavior functions as a form of conspecific communication that is only beneficial in the presence of visual cues. In terms of behavioral deterrent function then, it would be interesting to explore whether or not the type of avoidance response exhibited in response to a deterrent varies with light levels or the availability of visual cues. If indeed avoidance responses vary with different light levels, then the efficacy of a deterrent system would likely suffer greatly. It is therefore vital to test the function of these systems under different conditions in order to minimize any potential negative effects.

Multimodal integration has been shown to influence behavior in other contexts as well. For example Vowles and Kemp (2012) and Vowles et al. (2014) considered how multimodal sensory inputs influence downstream migration behavior, showing that the behavioral response of brown trout (*Salmo trutta*) and chinook salmon (*Oncorhynchus tshawytscha*) is influenced by both hydrodynamic and visual cues. Juveniles of both these species avoided an increasing velocity gradient more often when visual stimuli were present than they did when visual stimuli were absent. Thus, multimodal integration appears to also play an important role in guiding migratory and movement behavior in fish. Indeed, multimodal integration can impact behavior in any context where an individual is presented with multiple sensory inputs (Bzonek et al. 2022; Kasurak et al. 2012). It is therefore likely that such integration influences how fish respond to acoustic deterrents. For example, the perceived risk associated with a deterrent stimulus might depend on the quality of sensory information across different modalities, which in turn could impact the behavior expressed by an individual in response to it. However, to the authors' knowledge, little work has been done to examine these effects, save for Currie (2021) who found that roach (*Rutilus rutilus*) avoided an insonified route during upstream migration more often under higher flow speeds. Thus, there is a pressing need to determine if the behavior expressed in response to acoustic deterrents is dependent on other sensory stimuli. Determining these relationships will help explain why the efficacy of acoustic deterrents is so often variable and may provide insights useful for developing improved acoustic deterrents in the future.

## Masking

While the theory of multimodal integration has been used to explain how multiple sensory inputs can improve behavioral accuracy, peripheral sensory information can affect behavior in other ways as well. Paying attention to sensory information received from different modalities is not always advantageous. Animals have a limited attentional budget, meaning they do not possess the capacity to process all the sensory information they receive (Dukas 2002), so splitting attention between multiple sensory inputs can also impair behavioral responses to relevant stimuli (Dominoni et al. 2020). In fact, the distracted prey hypothesis states that any perceivable stimulus is capable of distracting an individual by re-allocating some of its finite attention (Chan and Blumstein 2011). Thus, information detected by sensory modalities other than the auditory sense may also influence the perception of

acoustic deterrents. Such masking and distraction effects have been studied under various contexts for many decades. However, as with studies of multimodal integration, rarely in regard to behavioral deterrent performance, thus there is a use in providing a short introduction to these topics and how they may relate to acoustic deterrent efficacy.

Acoustic masking occurs when the presence of one sound (noise) raises the threshold required for detection of another sound (signal) above what it would be in the absence of noise (Fay 2012). An important aspect of this phenomenon is that masking can occur when noise levels are lower than those of a given signal. Studies in fish have shown that increased broadband white noise, natural ambient noise, and anthropogenic noise mask the detection of pure tones, conspecific calls, and other sounds by increasing the sound level required to detect them (Amoser and Ladich 2005; Fay 1974; Ladich and Schulz-Mirbach 2013; Wysocki and Ladich 2005). The authors are not aware of any studies examining the masking effects of noise on the detection of acoustic deterrents. However, knowledge developed from studies elucidating the structure and function of the fish ear as well as the masking effects of anthropogenic noise can serve to guide this research.

In fish and other vertebrates, the auditory system can be thought of as a series of bandpass filters, referred to as critical bands (Fletcher 1940), covering the extent of a given species hearing range, with each band only receptive to a certain frequency bandwidth. Thus, in order for noise to mask the detection of a given signal, they must share (at least partly) the same critical band. Increasing noise levels without increasing signal levels leads to a decrease in the signal-to-noise ratio (SNR). Early studies of acoustic masking in fish showed that masking effects are strongest when the frequency bands of the masker and the signal are more similar and that masking effects decrease the more dissimilar the two frequency bands are (Tavolga 1974). One important consideration for improving deterrent efficacy then could be to tune deterrent stimuli so as to minimize the overlap with frequency bands present in the ambient environment as much as possible. However, it's unlikely that a deterrent stimulus would be able to be tuned in such a way as to completely minimize such overlap with ambient noises. Freshwater soundscapes vary widely (Amoser and Ladich 2010), and the presence of transient or temporary changes in background noise levels and spectra due to anthropogenic activity such as shipping or the operation of infrastructure could make it difficult to sufficiently tune a given stimulus.

Interestingly, masking effects induced by ambient noise have been shown to decrease when that noise is coherently modulated over the time domain (Hall et al. 1984). Little work has been done to examine these effects in fish; however Fay (2011) found evidence of such comodulation masking release in goldfish. If ambient noise varies coherently in amplitude across its frequency range, then the strength of any potential masking of acoustic deterrent stimuli may be reduced in comparison to the effect of a continuous white noise masker. The author calculated this reduction to be approximately 5 dB for goldfish. Thus, determining the temporal consistency of ambient noise is important when considering the potential masking effect of ambient noise on deterrent detection.

Masker efficacy may also depend on the type of signal being detected. For example, Belanger et al. (2010) showed that the round goby (*Neogobius melanostomus*) showed lower detection thresholds for a pulsed conspecific call than they had for tone burst stimuli ranging from 100 to 600 Hz. The authors state that using pure tone stimuli to test auditory capabilities in fish may provide a biased view of sensitivity, as the fish ear evolved to detect complex acoustic stimuli as opposed to tones (Belanger et al. 2010). If increased detection thresholds for artificial stimuli are indeed common across fish species, then basing predictions about the effects of masking stimuli on deterrent signal detection may be misguided. Comparing behavioral sensitivity thresholds to artificial and naturally occurring stimuli would provide useful insights into the validity of such approaches and could help identify knowledge gaps pertaining to how fish perceive deterrent signals.

Importantly, simple detection thresholds may not be the most relevant metrics for understanding how masking impacts the perception of acoustic deterrents. Dooling et al. (2015) classifies different auditory behaviors, each of which requires increasingly large signal-to-noise ratios (SNR) to elicit. Signal detection is the least complex of these behaviors (requiring the lowest SNR), as it simply defines the threshold at which an individual can detect that a signal is present in the environment, while the more complex behaviors of discrimination and recognition each require increasingly higher SNR. Fay (2011) was the first to examine differences in SNR thresholds required for signal detection and discrimination in fish, finding that discrimination thresholds of a 400 Hz pure tone against a background of broadband noise in the goldfish (*Carassius auratus*) were around 4 dB re 1 μPa higher than those for signal detection. The author points out that this difference in threshold level could translate to a significantly reduced range at which an individual would be able to acquire sufficient information about the signal and respond appropriately. If this relationship holds for other species in more natural environments, then the effective range of acoustic deterrents in noisy locations could be significantly reduced compared to what would be assumed from examining signal detection thresholds alone.

The effects of acoustic noise on hearing sensitivity thresholds are also dependent on species-specific hearing abilities. Fish species with specialized adaptations for hearing, such as the Otophysi, possess structures which connect their swim bladder and inner ear, often through mechanical linkages or protrusions of the swim bladder (Popper and Fay 2011). Species with these adaptations are sensitive to both the sound pressure and particle motion component of sound, have lower hearing thresholds, and are more sensitive to acoustic masking effects than species that do not have such specialized adaptations (Ladich and Schulz-Mirbach 2016). Indeed, comparative studies have shown that the hearing thresholds of species lacking any hearing specializations are less affected by noise than species that have them. In similar lab experiments, both Wysocki and Ladich (2005) and Ladich and Schulz-Mirbach (2013) found not only that AEP response thresholds for species possessing specialized hearing adaptations were masked by lower noise levels but also that the magnitude of these effects is stronger for these species than for species lacking any such specializations, such that hearing thresholds for pressure-sensitive species increased more. In contrast however, Ramcharitar and Popper (2004) found that black drum (*Pogonias cromis*), a species lacking any specialized connections

between their inner ear and swim bladder, was more affected by masking than was Atlantic croaker (*Micropogonias undulatus*), a species possessing specialized swim bladder projections that terminate near their inner ear. In this study the black drum exhibited a loss of frequency sensitivity at the upper extent of their hearing range under the loudest masking level (136 dB re 1 μPa) and exhibited larger threshold shifts across much of its hearing range at that same masking level than did the Atlantic croaker. While these results appear to conflict with each other, it is worthwhile to consider if indeed masking effects are on average stronger for species that possess specialized hearing adaptations. Acoustic deterrents should be most efficacious for species that possess strong hearing abilities, as it is expected that these species will be more reliant on acoustic cues in general and should be better able to detect acoustic stimuli than species without such adaptations. Thus, any potential masking effects imposed by ambient noise on a given deterrent may result in larger-than-expected decreases in performance. It would be worthwhile for future research to determine if species selected for management with acoustic deterrents are particularly sensitive to masking effects. Determining such relationships would benefit the development of improved acoustic deterrents and minimize environmental effects on their performance.

Perhaps it is not surprising then that research has shown that hearing thresholds also vary with ambient environmental sound levels. In an early example, Chapman and Hawkins (1973) suggested that the hearing thresholds of cod (*Gadus morhua*) were masked in all but the most calm sea conditions tested in their experiment. More recently, Amoser and Ladich (2005) directly examined how the hearing thresholds of two fish species differ in their ability to detect sound pressure in different levels of environmentally generated background noise. Here, the auditory thresholds of common carp (*Cyprinus carpio*) were highly masked by river-generated noise (hearing range threshold increases up to 49 dB) but less so by lake environment noise (maximum threshold increases of 12 dB across the frequencies tested). Conversely, the hearing thresholds of European perch (*Perca fluviatilis*) did not increase by more than 13 dB in the loudest background noise treatment. Thus, the hearing thresholds of this pressure-sensitive species appear to be masked by ambient noise levels.

Ambient noise levels differ across different freshwater habitat types (Vračar and Mijić 2011; Wysocki et al. 2007), with levels in large rivers such as the Danube reported as high as 137.8 dB re 1 μPa (Amoser and Ladich 2010); furthermore, anthropogenic activities can generate noise levels greater than 150 dB re 1 μPa (Wang et al. 2020). Not only are these levels higher than the hearing sensitivity thresholds reported for many species in quiet conditions (Ladich and Schulz-Mirbach 2013), but they are also similar to the SPL levels of many acoustic deterrents reported in the literature ((Putland and Mensinger 2019) supplementary material). Thus, these noise sources may mask acoustic deterrent stimuli if their sound level is not sufficiently high enough (of course, however, any potential masking ability of ambient environmental noise will depend also on its spectral profile).

It seems likely then that ambient and anthropogenic noise sources influence fish hearing sensitivity by increasing the threshold levels required to elicit behavioral

responses and altering the frequency sensitivity of different species (Ramcharitar and Popper 2004). Therefore, it is important that ambient noise levels are quantified when determining what sound level should be used for a given acoustic deterrent. However, in a recent review of acoustic deterrent performance, Putland and Mensinger (2019) show that ambient sound levels are rarely reported in the literature, with around 10% of their cited papers having reported ambient noise levels. The authors argue that sound levels should be defined with regard to both species-specific hearing sensitivities and local ambient conditions so that appropriate signal-to-noise levels are maintained. Indeed, without an understanding of background noise levels, it is not possible to adequately assess whether or not the sound levels of an acoustic deterrent stimulus would be sufficient to elicit the appropriate behavioral response from target species.

## Conclusions

It is clear that the wider sensory environment may influence a fish's ability to perceive acoustic deterrents, with potentially important consequences for the efficacy of these systems. While significant research effort has been devoted to examining the effects of both multimodal integration and masking in fish, there is almost no consideration of how these interactions may impact the detection of and response to behavioral deterrents. Specifically studying how sensory perception in this context is dependent, environmental conditions may provide much needed answers to improve the efficacy of these systems. Testing whether or not changes in behavioral responses to artificial deterrent stimuli follow the same rules as responses to natural biotic stimuli (such as acoustic predator cues) under different conditions could help connect the more applied focus of behavioral deterrent design to fundamental biological theory. This perspective would benefit a more systematic approach to the testing of acoustic deterrents and behavioral guidance measures more widely. Indeed, efforts to improve the efficacy of acoustic deterrents will be well served by complimentary and interdisciplinary approaches. Field studies focusing on quantifying the sensory environment present upon which acoustic deterrents must operate can inform controlled, experimental approaches aimed at disentangling the complex relationships between specific sensory inputs and behavior. In turn, such studies should be validated by field experiments using more "natural" conditions. As the need for behavioral guidance systems becomes more widespread, adopting more systematic approaches to their design will not only save precious funding but will also lead to the implementation of more robust designs in the future.

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# Harmonized Shipping Sound Test Signals to Assess Effects on Aquatic Animals

51

Christ A. F. de Jong

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**Abstract**

One of the main aims of the EU Horizon 2020 research program SATURN (developing solutions to underwater radiated noise, https://www.saturnh2020.eu/) is to determine the relevant metrics of ship-radiated underwater sounds that can be related to effects on the physiology and behavior of aquatic animals. Where possible, these metrics will then be related to the corresponding ship design and operational parameters, to enable the development of mitigation solutions. As part of the approach, controlled exposure experiments will be conducted on representative fish and invertebrate species, involving playback of these representative test signals.

The relevant temporal and spectral features that can be attributed to ship-radiated sound characteristics have been characterized, based on data available in the literature. A suite of artificially synthesized ship sound signals has been developed. These signals are considered suitable for the assessment of harmful

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underwater noise characteristics in the sense that they are relevant for the impact on marine life as well as subject to technically achievable mitigation measures. Using harmonized test signals in different bioacoustic playback studies facilitates the comparability of results. These signals are offered as a standard reference for external stakeholders and future projects.

**Keywords**

Shipping sound · Impact · Marine life · Playback signals · Harmonization

## Introduction

Assessment of the impact of underwater sound on aquatic animals is complex because of the variety of taxa involved, each with their own spectral and temporal sensitivity to sound and their behavioral and physiological sensitivity to acoustic disturbance. The three large taxonomic groups in aquatic ecosystems are invertebrates, fishes, and marine mammals. Fishes and invertebrates are considered to be particularly sensitive to the particle motion components of sound and hear sounds predominantly in the lower-frequency ranges, usually below 1000 Hz (Hawkins et al. 2015). Invertebrates' low capacity or inability to escape sound sources makes them potentially more vulnerable to a continuous sound source because they are likely to receive a higher dose of sound compared with species that can escape a noisy area like fish or marine mammals. Because field studies on fish and invertebrates are complex, the SATURN project studies impact of sound on these species in the laboratory. Laboratory conditions and captive animals are considered critical for progress and essential for revealing fundamental aspects of the processes underlying negative effects of noise pollution. Controlled exposure experiments are conducted on representative species using playback of harmonized test signals and state-of-the-art analysis to determine the physiological, pathological, and behavioral effects of both particle motion and acoustic pressure on the exposed animals.

Playing back ship sound for controlled exposure experiments is not a trivial task (Erbe et al. 2019). Recordings of ship sound are affected by the characteristics of the recording equipment and by the environment in which they are made. Playbacks of these signals are affected by the characteristics of the playback equipment and by the environment. Consequently, relevant sound features may get lost in translation.

Using harmonized test signals in different bioacoustic playback studies facilitates the comparability of results. Therefore, a suite of artificially synthesized ship sound signals has been developed that are considered suitable for the assessment of harmful underwater noise characteristics in the sense that they are relevant for the impact on marine life as well as subject to technically achievable mitigation measures.

## Ship-Radiated Underwater Sound

Underwater-radiated sound has been used by navies since World War I as a means for detecting ships (passive sonar); see, e.g., Ainslie 2010. Hence, the main sound source mechanisms on ships are generally well-known (e.g., see Baudin and Mumm 2015). These are generally the propulsors (propellers, waterjets, or other) and the engines that drive these propulsors, but also other onboard machinery (such as generators, pumps, etc.) and the hydrodynamic flow around the ship hull can contribute to the underwater-radiated sound; see, e.g., Urick 1983; Ross 1976.

Propeller sound sources include inhomogeneous wake flow, turbulence at leading edge (inflow turbulence), trailing edge and blade tip, and cavitation on blades (sheet cavitation), blade root, at the hub, and in the tip vortex. These source mechanisms create tonal sound at harmonics of the propeller "blade rate" (i.e., the frequency at which the propeller blades pass one by one through the nonuniform wake of the hull, given by the product of the propeller shaft rotation rate and the number of blades) and broadband sound from cavitation and from turbulence. The amplitude of the broadband propeller cavitation sound is typically modulated at the blade rate, because the dominant cavitation dynamics occurs when the propeller blade passes through the wake peak, typically located near the 12 o'clock position. The broadband sound from turbulence can be amplified in some frequency bands by propeller blade resonances. In some cases, interaction between blade resonance and vortex shedding at the trailing edge can cause strong tonal sound known as "propeller singing." This can generally be mitigated by small modifications of the shape of the trailing edge; see, e.g., Carlton 2012.

Rotating and reciprocating machines produce tonal sound at harmonics (positive integer multiples of the fundamental frequency) of the rotation (or oscillation) rate of the machines, including, for example, gear mesh frequencies. Combustion engines produce tonal sound at harmonics of their firing rate.

These characteristics are for regular ship transit sounds. Maneuvering, switching machines on and off, and the operation of thrusters for dynamic positioning can generate additional transient sounds.

## Sound Features

The sound signals used for controlled exposure experiments in the laboratory must represent the relevant temporal and spectral features that can be attributed to ship-radiated sound characteristics. Spatial features of the playback sound field are relevant as well, but these are mainly related to the geometry of the laboratory test setup and playback devices. This chapter aims at the design of harmonized digital test signals for playback studies. The focus is on the temporal and spectral features of the sound signals. It is noted that the design of the playback system to convert these (electronic) signals into the appropriate sound pressure and sound particle motion field is an essential next task in the design of the individual controlled exposure studies for the different animal species.

## Spectral Features

During regular ship transits, the ship-radiated sound spectrum contains both broadband sounds, typically generated by propeller cavitation, and tonal components, associated with the propeller and with rotating and reciprocating machinery.

**Proportional bandwidth spectra**, for a relevant auditory bandwidth, provide the relevant information about the spectral distribution of the sound energy in the signal in relation to detection and to possible behavioral response and noise-induced hearing threshold shifts.

**Narrowband spectra** provide the relevant information about the presence of dominant tonal sounds in the signal. Humans exposed to sounds with a prominent tonal character experience higher annoyance than when exposed to broadband sounds at the same sound pressure level. Consequently, the airborne environmental noise assessment standard (ISO 1996-1, 2016) suggests adding an adjustment ("penalty factor") of 3 to 6 dB to the measured sound level of sounds with a prominent tonal character. For this purpose, ISO/PAS 20065 (2016) describes an objective method for assessing the audibility of tones in noise, in relation to human hearing. The tonal character of sound may also be relevant for behavioral response of aquatic animals to sound exposure, but this has not yet been investigated. Therefore, signals with and without tonal content are developed that allow investigating the effect of the tonality of ship sound signals in controlled exposure studies. In opportunistic acoustic measurements of ships transiting offshore California (McKenna et al. 2013), dominant narrowband tones were present in 10% of the measured ships.

## Temporal Features

The audibility of sound signals for animals is affected by signal duration; see, e.g., Kastelein et al. 2010. The integration time of harbor porpoise hearing was found to be similar to that of other mammals, including humans. Because the "ears" of fishes and invertebrates are different from those of mammals, it is well possible that these have different integration times, but data are largely lacking. One study by Art Popper (1972) did not show any dependence on signal duration, but a later study by Dick Fay and Shelley Coombs (Fay and Coombs 1983) does suggest an integration time of some hundred milliseconds in goldfish.

The sound radiated by a ship due to propeller and machinery, if the ship is sailing at a constant speed and course, can be considered "stationary" (meaning that statistical parameters remain constant) at timescales larger than the rotation period (the inverse of the rotation rate) of the main noise sources. Typically, these rotation periods are of the order of 1 s or less.

The amplitude of broadband propeller cavitation sound is **modulated at the blade rate.** Defense and security organizations use analysis methods such as DEMON (Detection of Envelope Modulation on Noise), to estimate the shaft rotation frequency of detected vessels, as a means for classification; see, e.g.,

Chung et al. 2011; Fillinger et al. 2011. Animals with a hearing time constant $\tau \approx 125$ ms can likely perceive the amplitude modulation for blade rates up to about 8 Hz ($=1/\tau$) as loudness fluctuations. In human psychoacoustics, faster (>12 Hz) amplitude modulations are qualified as "roughness" of the sound (Daniel 2008). It is likely that this perception occurs in other species as well, but this has not been investigated. Geometric differences between individual blades or shaft eccentricity may also lead to sound modulation at the shaft rotation rate.

The sound to which a static animal is exposed when a ship passes **varies in time due to the decreasing and increasing distance** to the ship, in combination with the interference effects due to reflections of the ship sound at the water surface (the Lloyd mirror effect in underwater acoustics). For typical ship speeds (<20 m/s) and observation distances (>100 m), these variations occur over time periods of the order of 5 s or more.

The relative motion between ship and observer will also cause a variation of the observed tonal frequencies due to the Doppler effect. Due to the large difference between the ship speed and the sound speed in water, this is a subtle effect which is tentatively ignored in this chapter.

In addition, propeller tip-vortex cavitation produces low-frequency broadband noise related to variations between blade passages, hence variations in time that have a timescale of the order of blade rate frequency or smaller; see Bosschers 2018. These may have various origins and may include ship motions, variations in rpm, turbulence in the wake, and variations in nuclei. These variations in low-frequency sound radiations are not well-described and hence tentatively ignored in this chapter.

## Signal Representation

Digital signals representing field quantities such as sound pressure or sound particle displacement, velocity or acceleration, characterize the time-varying amplitude of a sound signal $x(t)$ at discrete, uniformly distributed times $t_n$, with $\delta t$ the time step between the samples, so that $t_n = t_1 + (n - 1)\delta t$, with $n = 1, 2, \ldots, N$; see, e.g., Ainslie et al. 2018. The time step corresponds with a sample rate (or sample frequency) $f_s = 1/\delta t$. The number of samples $N$ that describe a signal of duration $T$ equals $N = T/\delta t$. The maximum sound frequency that can be represented by a digital signal without distortion (aliasing) is limited by the Nyquist frequency $f_s/2$. The statistics of time series $x(t_n)$ characterize the signal.

The short-time Fourier transform (STFT) of the signal quantifies the relevant temporal and spectral features of the sound signal. The frequency resolution $\delta f = 1/\Delta t$ in the STFT is inversely proportional to the duration $\Delta t$ of the subsequently analyzed sections of the signal. The STFT reveals the tonal and broadband sound components and how these vary with time.

Another statistic of the sound signal that may be relevant for predicting the risk of hearing loss or other adverse effects of sound on aquatic animals is kurtosis (the moment of order 4 in the standardized probability distribution of a random variable) (ISO 3534-1 2006); see also Müller et al. 2020. Kurtosis is recommended for

quantifying "impulsiveness," by Martin et al. 2020. They note: "Sounds from vessels are normally considered non-impulsive; however, 66% of vessels analyzed were impulsive when weighted for very-high frequency mammal hearing . . . vessel sounds that generate high kurtosis when VHF-C auditory frequency weighted are a mix of rubbing and rasping sounds, cavitation sounds, and mechanical knocking." Although there is no generally accepted definition of "impulsive" or "non-impulsive" sounds, it makes sense to include the kurtosis as a feature in studies of the effect of sound on marine animals.

## Ship Sound Signals

In order to be able to systematically investigate which features of ship sound are the most harmful to aquatic animals, it is proposed to apply synthesized sound signals with independent variation of the relevant sound features. An earlier example of synthesizing ship sound was presented by (Yağımlı and Sertlek 2013).

Selection of a harmonized set of representative test signals depends on the intended application of these signals. The following assumptions were made:

- In SATURN. the playback signals are mainly required for controlled exposure experiments in the laboratory on fishes and invertebrates.
- The main frequency range of interest for these experiments is between approximately 10 Hz and 1 kHz.
- The primary interest is in the effects of stationary "continuous" sound from ships transiting at constant speed and course.
- The main sound feature of interest is the *broadband sound level*. Here "sound level" can refer to either sound pressure level or the level of sound particle displacement, velocity or acceleration, in a specified frequency bandwidth. The sound level to which animals are exposed in playback tests depends on the design of the laboratory setup and the playback system.
- The next feature of interest is the *spectrum* (frequency content) of the signal. This depends on the combination of test signal and playback system. Where appropriate, the test signal can be adapted to correct for the characteristics of the playback system. Design of the appropriate spectrum for the test signals depends on information about the frequency sensitivity of the species that is being investigated as well as on the characteristics of the sound source and environment. The presence of dominant "tonal sounds" in the spectrum may be relevant for behavioral response studies.
- Another potentially relevant sound feature for behavioral response studies is the temporal variation of the exposure (*amplitude modulation*) and the corresponding perception of "fluctuations" and "roughness" of the sound.

The spectrum and the temporal structure have been included in the design of the test signals.

## Broadband Ship Source-Level Spectra

Average broadband source-level spectra of different classes of ships (transiting at constant speed and course) are predicted by the JOMOPANS-ECHO ship source-level model (MacGillivray and de Jong 2021). This model calculates the source-level spectrum $L_S(f)$ of ship as a function of speed, length, and (AIS) ship type. The model is based on the statistics of a large database of ship source-level measurements from the Vancouver Fraser Port Authority-led Enhancing Cetacean Habitat and Observation (ECHO) program.

In a subsequent vessel noise correlation study, again carried out in the ECHO program, MacGillivray et al. 2020 confirmed that "speed through water and actual vessel draft (i.e., the two primary operational parameters) were generally the most influential predictors for underwater noise," while "vessel size (represented via length overall) was ranked as the design parameter with the strongest correlation to underwater radiated noise." An important feature is that "the loud vessels exhibited a distinct cavitation noise hump near 50 Hz, as e.g. discussed by (Wittekind and Schuster 2016), whereas the quiet vessels exhibited a flatter spectrum below 100 Hz." Such a "hump" is tentatively included in the JOMOPANS-ECHO model for all cargo vessels (container ships, vehicle carriers, bulkers, tankers). The origin of the hump in the radiated noise spectrum is still unexplained. It may be due to propeller vortex cavitation, but the Lloyd mirror effect and cavitation excited hull-plate vibrations also have an effect.

The JOMOPANS-ECHO ship source-level model calculates the ship source-level spectrum, for an assumed fixed source depth of 6 m below the water surface. The received sound in a given environment will depend on the propagation conditions. These will affect the shape of the sound spectrum. For the synthesis of ship sound signals, a deepwater Lloyd mirror propagation loss correction such as suggested by ISO 17208-2 has been incorporated, to avoid overemphasizing the lower frequencies in the playback spectrum.

Two widely different source-level spectra have been taken to construct the following two synthetic ship signals:

A. Broadband continuous sound (10 Hz–4 kHz) for a typical fishing boat of 50 m length at a speed of 11 knots (5.7 m/s).
B. Broadband continuous sound (10 Hz–4 kHz) of a typical large containership of 300 m length at a speed of 18 knots (9.3 m/s)

## Ship Signal Synthesis

To generate the corresponding time signal, the following steps are taken:

1. Select duration ($T = 10$ s) and sample frequency ($f_s = 2^{13}$ Hz $= 8192$ Hz ). This also determines the following parameters:
   (a) Time resolution: $\delta t = 1/f_s \approx 0.122$ ms.
   (b) Number of samples: $N = T \cdot f_s = 81920$
   (c) Nyquist frequency: $f_{\max} = f_s/2 = 4096$ Hz
2. Evaluate the source-level spectrum at discrete frequencies $f_k = k \cdot \delta f$, with frequency resolution $\delta f = 1/T = 0.1$ Hz and $k = 1..N/2$.
3. Convert to spectral density:

$$L_{S,f}(f_k) = L_{S,\mathrm{ddec}}(f_k) - 10\log_{10}\left(\left(10^{\frac{1}{20}} - 10^{-\frac{1}{20}}\right) \cdot \frac{f_k}{1\ \mathrm{Hz}}\right)\ \mathrm{dB}$$

4. Convert to root-source factor with amplitude $Y(f_k)$:

$$Y(f_k) = 10^{\frac{L_{S,f}(f_k)}{20\ \mathrm{dB}}} \cdot 1\ (\mu\mathrm{Pa\ m})^2/\mathrm{Hz}$$

5. Convert this one-sided spectrum to a full double-sided Fourier spectrum:

$$Y'(f_k) = \begin{cases} 0, & \text{for } k = 0 \\ Y(f_k), & \text{for } k = 1..\left(\frac{N}{2}+1\right) \\ Y(f_{N-k}), & \text{for } k = \left(\frac{N}{2}+2\right)..(N-1) \end{cases}$$

6. Next, add a random phase $\phi_k$ (uniformly distributed in the range $0..2\pi$, with $\phi_0 = 0$) to each element of the spectrum, to represent random noise:

$$Y_k = Y'(f_k) \cdot e^{i\phi_k}$$

7. Apply the inverse Fourier transform and keep the real part:

$$y_n = \mathrm{Re}\left\{\frac{1}{N}\sum_{k=0}^{N-1} Y_k e^{\frac{i2\pi kn}{N}}\right\}$$

8. Normalize to the maximum amplitude in the resulting time series to generate the sound signal for playback: $y(t_n) = y_n/\max(|y_n|)$, so that the amplitude in the wav file is in the range $[-1\ ..\ 1]$. The amplitude at which the signal is played back will depend on test configuration, equipment, and settings.

### Amplitude Modulation

The temporal structure ("roughness") associated with blade rate amplitude modulation is added in the next two signals:

A. Fishing boat signal A enhanced with blade rate amplitude modulation, representing a single four-bladed propeller rotating at 180 rotations per minute (rpm). This corresponds with a blade passing frequency ($f_b$) of 12 Hz.
B. Containership signal B enhanced with blade rate amplitude modulation, representing a single five-bladed propeller rotating at 90 rotations per minute (rpm). This corresponds with a blade passing frequency of 7.5 Hz.

The amplitude modulation is incorporated in the sound signal by multiplying the times series $y(t_n)$ with an amplitude modulation function:

$$y_{\mathrm{am}}(t_n) = y(t_n) \cdot [1 + A_{\mathrm{am}} \sin(2\pi f_b t_n)]$$

The modulation amplitude $A_{\mathrm{am}}$ is not generally known. The test signals are synthesized with a tentatively assumed $A_{\mathrm{am}} = 0.25$.

## Tonals

Next, the spectral structure is modified by adding tonals in the following signals:

A. Fishing boat signal A enhanced with tonals at propeller blade rate (12 Hz) harmonics and at harmonics of a ship service diesel generator (30 Hz and harmonics).
B. Containership signal B enhanced with tonals at propeller blade rate (7.5 Hz) harmonics and with two dominant tonals at 350 Hz and 580 Hz, as observed by McKenna et al. 2013.

The following signals represent the combination of amplitude modulation and tonal structure:

A. Fishing boat signal C enhanced with tonals at propeller blade rate (12 Hz) harmonics and at harmonics of a ship service diesel generator (30 Hz and harmonics).
B. Containership signal B enhanced with tonals at propeller blade rate (7.5 Hz) harmonics and with two dominant tonals at 350 Hz and 580 Hz.

The tones are added via the following:

$$y_{\mathrm{t}}(t_n) = y(t_n) + \sum_{m=1}^{M} A_{\mathrm{t},m} \sin(2\pi f_m + \phi_m)$$

The phase angles $\phi_m$ of the individual tonals are randomly drawn from a uniform distribution between 0 and $2\pi$.

**Table 1** Statistical properties of the audio signals for the synthesized ship sound signals. Different spectral features are represented by the two ship types and the signals with and without tonals; different temporal features are represented by the signals with and without amplitude modulation. The zero-to-peak levels are equal to 0 dB, indicating that the signals in the wav files are normalized relative to the maximum amplitude in the 10-s signal. In playback studies, the properties will be modified by the playback system

| | Signal | $L_{pk}$ [dB] | $L_{rms}$ [dB] | Kurtosis $\beta$ |
|---|---|---|---|---|
| A | FishingBoat_broadband.wav | 0 | −13.4 | 3.01 |
| B | ContainerShip_broadband.wav | 0 | −13.2 | 3.06 |
| C | FishingBoat_broadband_modulated.wav | 0 | −14.1 | 3.34 |
| D | ContainerShip_broadband_modulated.wav | 0 | −13.2 | 3.42 |
| E | FishingBoat_broadband_tonals.wav | 0 | −12.3 | 2.93 |
| F | ContainerShip_broadband_tonals.wav | 0 | −12.3 | 2.89 |
| G | FishingBoat_broadband_modulated_tonals.wav | 0 | −14.7 | 3.27 |
| H | ContainerShip_broadband_modulated_tonals.wav | 0 | −12.0 | 2.96 |

### Ship Signal Spectra

The statistical properties of the signals are given in Table 1. The peak level and the kurtosis are calculated over the 10 s signal. The scaled root-mean-square level ($L_{rms}$) is calculated over 1-s segments of the signal, with 50% overlap and a Hann window, and then averaged. Figures 1 and 2 show corresponding waveforms and spectra.

## Conclusions

This chapter describes an approach for selecting and synthesizing ship underwater-radiated sound signals for playback studies. This approach has been applied to generate eight test signals for laboratory studies in the EU Horizon 2020 project SATURN, representing various aspects of the radiated sound of a large containership as well as a typical fishing vessel. Signals are available from https://www.saturnh2020.eu/resources proposed in this chapter and can easily be extended to synthesize signals that are independent of the average spectra of ships but focus on specific sound features, such as band-limited noise, transient signals, or tonal sounds, if required by the test hypotheses for the controlled exposure studies. These could, for example, represent the transient sound due to switching of engine or gear or tonal sound from propeller singing such signals.

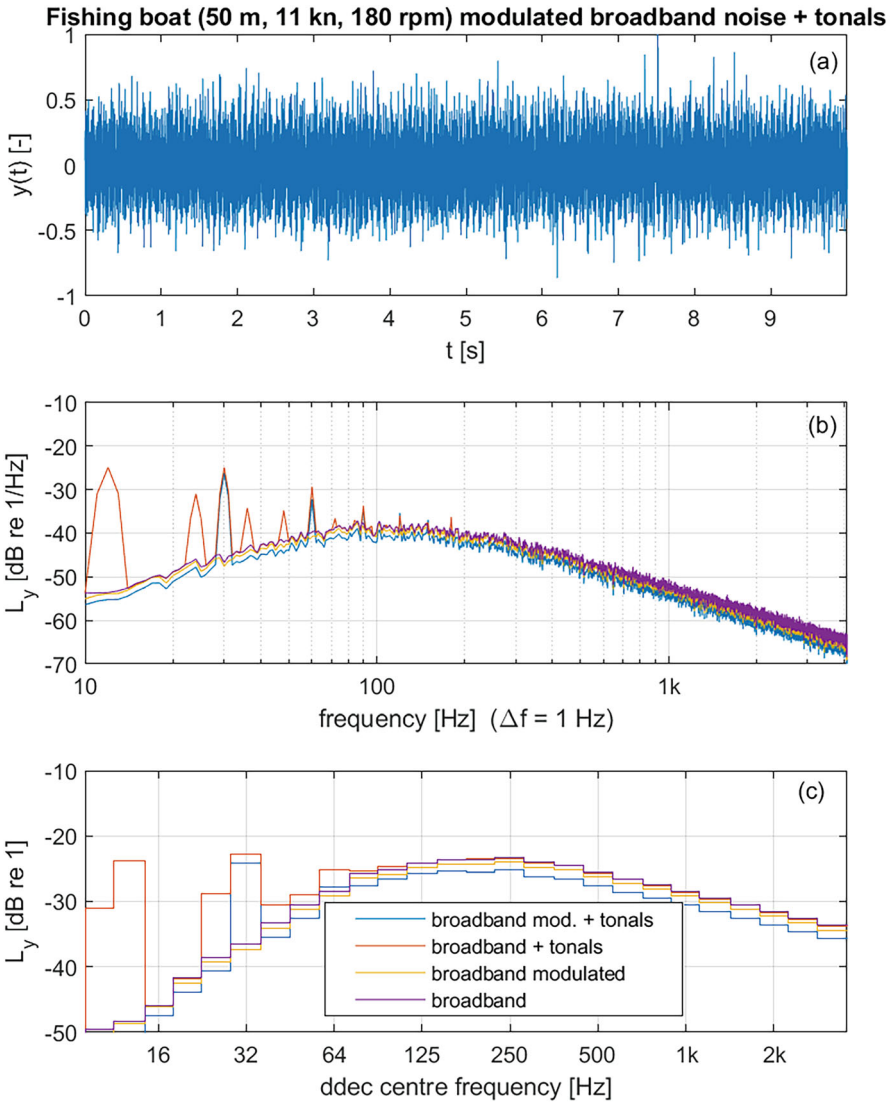



**Fig. 1** Waveform (**a**), narrowband spectrum (**b**), and decidecade spectrum (**c**) of the fishing boat sound signal G that represents the blade rate-modulated broadband noise plus tonals (in blue). The spectra are compared with those of the sound signals A, C, and E, without tonals and without modulation, scaled so that the total power in the spectra is equal for the three signals

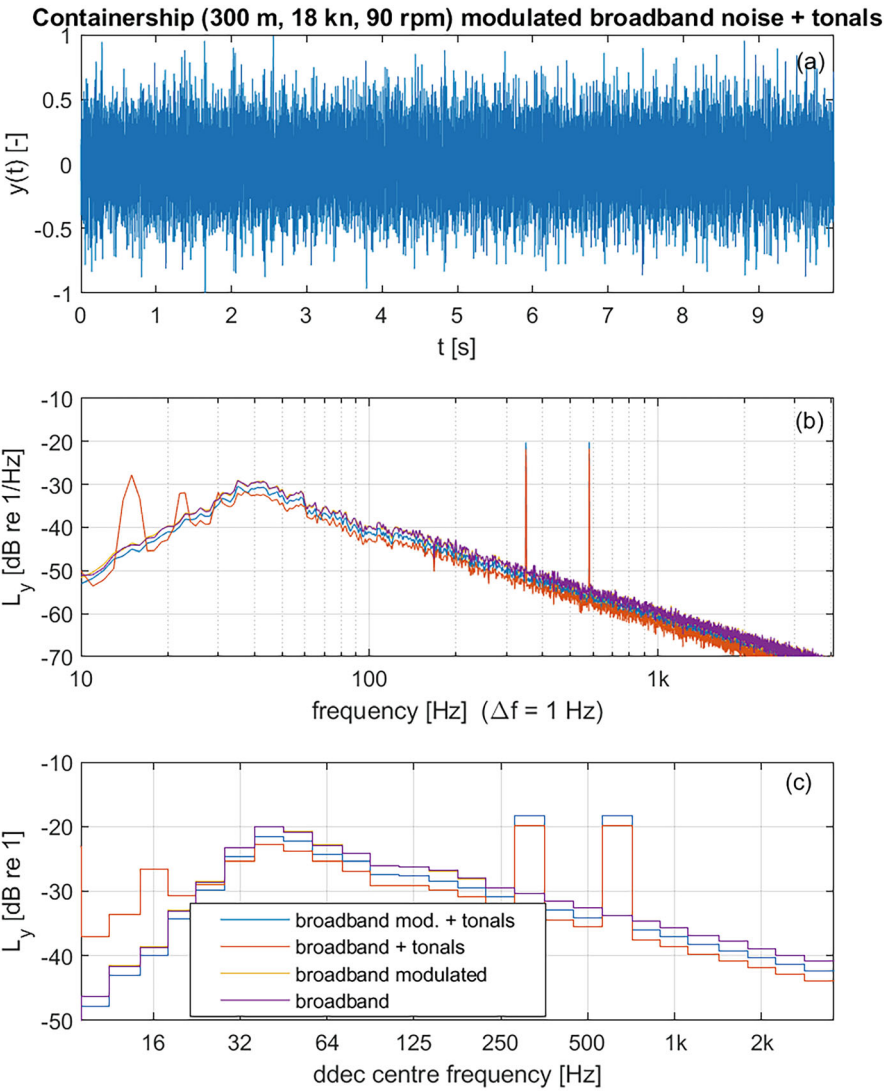



**Fig. 2** Waveform (**a**), narrowband spectrum (**b**), and decidecade spectrum (**c**) of the containership sound signal H, that represents the blade rate-modulated broadband noise plus tonals (in blue). The spectra are compared with those of the sound signals B, D, and F, without tonals and without modulation, scaled so that the total power in the spectra is equal for the three signals

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# Underwater Noise Levels in the Coastal Waters of Poland and Malta: Reviewing the Known Impact on Organisms

52

Wenesa K. Dylewska, Adam Gauci, Alan Deidun, and Jarosław Tęgowski

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**Abstract**

Underwater noise is legally recognized in the European Union as a pollutant of the marine environment. An increasing number of countries are initiating monitoring programs to study the levels of underwater noise. There is a need to compare these levels as well as share experience in order to forecast possible monitoring difficulties. This chapter shows that levels of underwater noise in the Baltic Sea (Poland) and in the Mediterranean Sea (Malta) were similar in the given monitoring periods as were the issues met during work. The article also presents a summary of the studied impact of underwater noise on organisms found in both localities. Based on the analysis, it can be deduced that species living in both localities can be negatively affected by the present noise. Actions are required to control and minimize underwater noise in both places. Although there are studies showing impacts of noise on aquatic organisms, there is still a lot of data missing. Dedicated programs should be created to make it

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possible for stakeholders (e.g. scientists, students) to get involved in addressing this issue. A database on the underwater noise levels and the resulting influence on the marine environment could be a useful tool to improve regulations and work practice.

**Keywords**

Underwater noise · Baltic Sea · Mediterranean Sea · Noise monitoring · Sound pressure level · Impact · Marine environment · Fish · Marine mammals

## Introduction, Motivation, and Aim of the Study

This is a perspective of an oceanography student who happened to study both in Poland and Malta as well as of scientists from both countries. Monitoring was carried out in these countries to evaluate the levels of noise in the coastal waters. The aim of this work is not only to compare the data but also to highlight how common the problem of underwater noise is, how important it is for future studies to cooperate, share knowledge and monitoring experience. Scientists from other countries (e.g. Morocco) are also expressing their interest in the problem (Chahouri et al. 2021) and might as well attempt to measure levels of underwater noise.

Noise is recognized as a pollutant of the marine environment by law according to the European Union Marine Strategy Framework Directive (MSFD) Descriptor 11 as well as within Regional Sea Conventions (e.g. the Barcelona Convention, the Helsinki Convention). Nevertheless, the problem has not been sufficiently explored and addressed. Another part of this work will present information gathered from published studies on the impacts of underwater noise for the organisms observed in the Baltic Sea in Poland and in the Mediterranean Sea in Malta. The aim of the summary is to emphasize the seriousness of this issue as well as to point out the need for more studies. A great complexity of the phenomenon of underwater noise pollution requires combined efforts of many people (e.g. policymakers, scientists, etc.).

## Monitoring in Malta and Poland

Data in Malta were obtained from the preliminary deployment of an underwater noise meter. Monitoring took place over 44 days. In this analysis, recordings collected consecutively over 15.2 days were used (30/06/2020 to 16/07/2020). The data were collected using the "SoundTrap ST300" by "Ocean Instruments" instrument model (Fig. 1). This was deployed at a depth of 30 m just outside Valetta's Grand Harbor.

Data in Poland were obtained from the Baltic Sea Information on the Acoustic Soundscape (BIAS 2023) program monitoring (Sigray et al. 2016) – one out of five stations has been chosen (located in the Gulf of Gdańsk) for the aim of the comparison and the monitoring was done throughout the year 2014. The data were collected using WildLife Acoustics SM2M (Fig. 2). It was deployed at a depth of 71 m. As the noise logger was in an area in which trawling takes place, a protective cage was used.

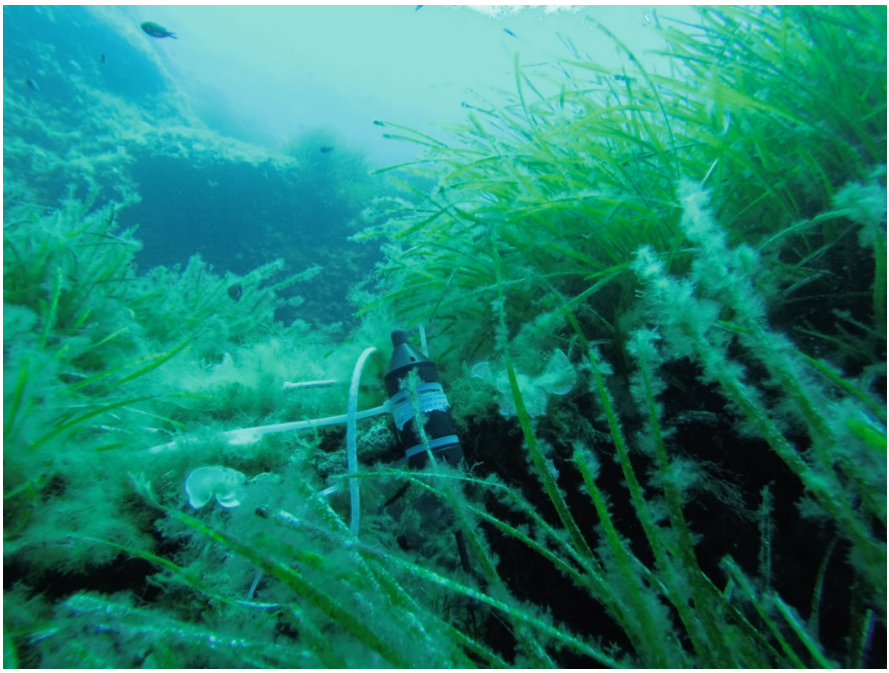


**Fig. 1** Noise logger in Malta – Ocean Instruments SoundTrap ST300

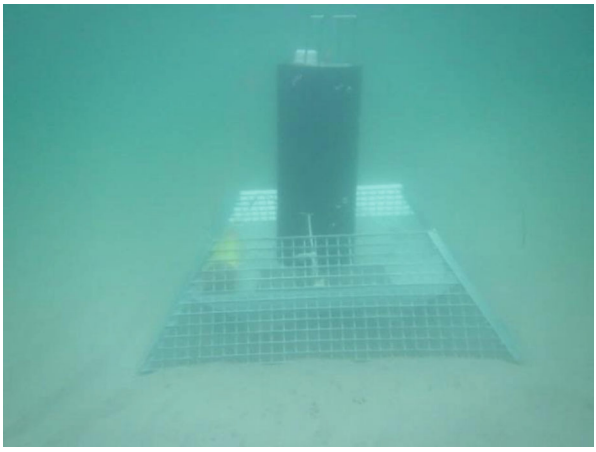


**Fig. 2** Noise logger in Poland – WildLife Acoustics SM2M secured with a metal cage

The comparison of data collected in Poland and Malta was focused on the frequencies 63 Hz and 125 Hz, which are usually the indicator of the low-frequency sound produced mostly by vessels (Mustonen et al. 2019).

Sound pressure levels (SPL) were obtained and compared as percentage exceedance levels. The 90% exceedance level of 77.1 dB re 1 μPa meant that for 90% of time the sound recorded was exceeding 77.1 dB. (For more information regarding general underwater noise monitoring, see Dekeling et al. 2014.) In both locations 90% of the time SPL were exceeding 70 dB re 1 μPa. As a result of comparison, although Poland had slightly lower SPL, both locations are characterized by very similar and high SPL for low-frequency sounds (63 Hz, 125 Hz).

Although Malta and Poland are far away from each other, and their coastal waters differ greatly in many characteristics, they both share the same problem of underwater noise pollution.

The monitoring locations were associated with shipping traffic, which explains the high levels of underwater noise. It is worth to notice that the monitoring of underwater noise should be focused on regions of particular importance for marine organisms. Nevertheless, due to the ubiquity of shipping traffic, the resulting underwater noise can impact multiple species in various areas. It can particularly affect organisms that require large areas to live, leading to noise pollution, chemical pollution, ship strikes, and habitat fragmentation (Pirotta et al. 2019) (Figs. 3 and 4).

## Different Basins, Same Challenges – On the Importance of Sharing Experience

Both research groups had to deal with a displacement of the monitoring equipment. In Poland, it was caused by trawling nets. Therefore, noise loggers were further secured using protective cages. In Malta one of the loggers was lost due to strong currents or fishing nets. This example clearly shows that sharing information about

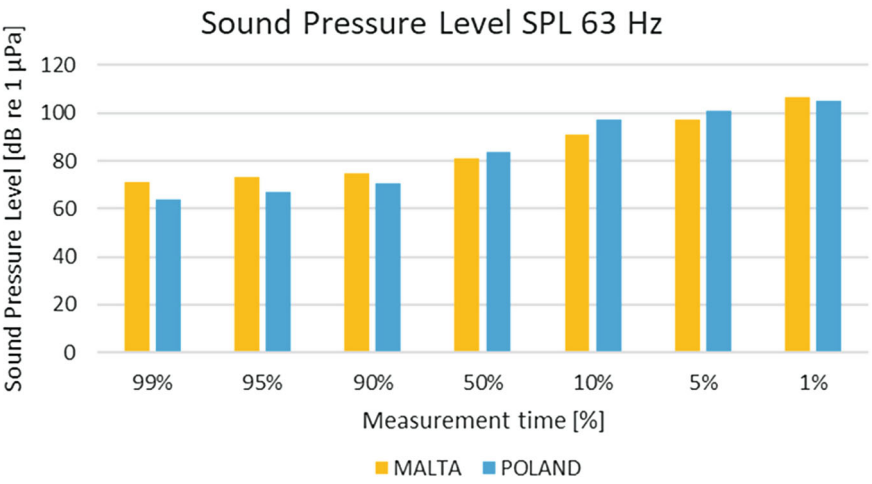



**Fig. 3** Comparison of the exceedance levels of SPL (63 Hz) between Poland and Malta

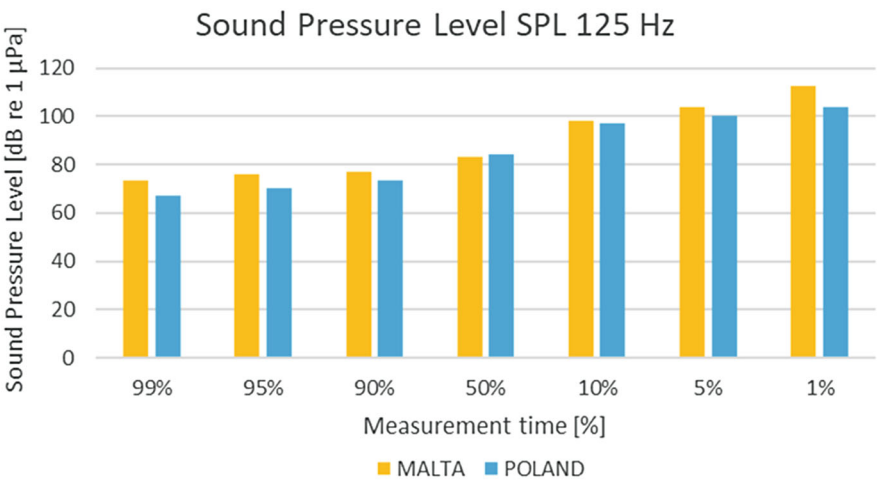



**Fig. 4** Comparison of the exceedance levels of SPL (125 Hz) between Poland and Malta

the problems met and ways to deal with them can help researchers avoid disturbance of the monitoring and additional costs. It also indicates that common protocols for data collection need to be presented.

## The Known Impact of Noise on Aquatic Organisms Found in the Baltic Sea (Poland) and the Mediterranean Sea (Malta)

Monitoring of underwater noise is not enough. Data on how such noise can influence organisms that live in the region is necessary in order to properly address the issue. Hence, information on how noise can influence some of the organisms that exist in the Baltic and Mediterranean Sea has been summarized in Tables 1, 2, 3, and 4.

This preliminary study intends to emphasize the real threat imposed by underwater noise levels. Studies already show the negative impacts of underwater noise on organisms. However, more studies are required to properly assess and address this issue.

The summary indicates that boat noise is a cause of multiple negative effects for organisms found both in the Baltic and in the Mediterranean Sea (Tables 1, 2, 3, and 4). Based on the monitoring data from Malta and Poland that has shown high levels of noise for the frequencies 63 Hz and 125 Hz that can be used as indicators of boat noise it can be assumed that species living in both localities could be negatively affected by the present noise. Thus, actions are required to control and minimize underwater noise pollution in both places.

It is worth adding that noise caused by explosions and other naval activities can be a cause of some animals being stranded (Table 3). Conflicts and wars are known for causing cetaceans strandings (Mahreen 2022). An example of this can be how the

**Table 1** Species of mammals that occur in the Baltic Sea (Poland)

| Species | Observed impact of noise | Type of noise | Reference | The IUCN Red List Category[a] |
|---|---|---|---|---|
| **Harbor Porpoise** (Phocoena phocoena) | Disruption of foraging | Vessel | Wisniewska et al. (2018) | CR (Baltic Sea subpopulation) |
| | Behavioral reaction | Vessel | Dyndo et al. (2015) | |
| | Moving away from the source of noise | Airgun | van Beest et al. (2018) | |
| | Disruption of activity patterns | Seismic survey noise | Pirotta et al. (2014) | |
| | Temporary shift in masked hearing thresholds | Seismic airgun | Lucke et al. (2009) | |
| | Temporary threshold shifts | Octave-band noise at 4 kHz (alike pile driving) | Kastelein et al. (2012) | |
| | Acoustic activity Reduced by 100% for 1 h after pile driving | Pile driving | Brandt et al. (2011) | |
| | A strong avoidance reaction within 20 km distance from the source of noise | Pile driving | Dähne et al. (2013) | |
| | Significant decline in echolocation activity | Offshore wind farm | Teilmann and Carstensen (2012) | |
| **Baltic grey seal** (Halichoerus grypus) | Potential masking of vocalizations | Ship noise | Bagočius (2014) | LC (Baltic Sea subpopulation) |
| | Behavioral response | High-frequency sonar | Hastie et al. (2014) | |
| **Harbor Seal** (Phoca vitulina) | Ability to detect quiet tones in a noisy background | Broad band noise in both air and water | Turnbull and Terhune (1990) | LC |
| | Avoidance | Tidal turbine noise | Hastie et al. (2018) | |
| | Not frightened away | Acoustic harassment devices | Jacobs and Terhune (2002) | |
| | Permanent threshold shift | 4.1 kHz pure tone fatiguing stimulus. Sound levels and | Kastak et al. (2008) | |

(continued)

**Table 1** (continued)

| Species | Observed impact of noise | Type of noise | Reference | The IUCN Red List Category[a] |
|---|---|---|---|---|
| | | durations were gradually increased to a maximum received sound pressure of 184 dB re 1 μPa | | |
| | Temporary threshold shift | Octave-band noise centered at 2500 Hz at two sound pressure levels: 80 and 95 dB SL | Kastak et al. (2005) | |

[a]*CR* critically endangered, *(EN)* endangered, *VU* vulnerable, *NT* near threatened, *LC* least concern, *DD* data deficient

Russia's invasion of Ukraine and the naval activities involved in the war have been recently mentioned in grey literature as a possible cause of increased numbers of stranded cetaceans along the coast of the Black Sea (Kroeger 2023).

Some of the species that are negatively influenced by underwater noise pollution have been assessed as critically endangered (e.g., harbor porpoise and european eel) (IUCN 2022). Furthermore, underwater noise is not the only problem that is affecting them. Chemical pollution, overfishing, and global warming are also a major threat. Together these can cause cumulative detrimental effects. With echolocation, harbor porpoises already find it difficult to detect fishing nets in time to avoid collision (Kastelein et al. 2000). Anthropogenic noise can add to their problems. On the other hand, acoustic deterrents might help cetaceans and other animals not to fall victim to bycatch. It can be also a challenge to use acoustic deterrents that would be effective but not harmful for the marine environment.

Some studies on the influence of noise on organisms are outdated and there is still a lot of data missing. It could be investigated how noise affects marine organisms based on the studies that are available and the similarities between different species, e.g. fish that have a swim bladder usually have higher hearing sensitivity than fish that do not have it (Wiernicki et al. 2020). At the same time, fish, as Friedrich Ladich observed, can be called "insects" among vertebrates, giving their variety that includes variety of their hearing organs (Ladich 2014). For that reason, one has to be cautious when making assumptions about how noise influences fish in general based on the studies on some of their species. More data is needed to introduce and enforce legislation. As the European Union Marine Strategy Framework Directive states, one of the descriptors of good environmental status of the marine environment is introduction of noise at levels that do not adversely affect the marine environment. Therefore, there is a need to assess these levels and control them.

**Table 2** Fish species that occur in the Baltic Sea (Poland)

| Species | Observed impact of noise | Type of noise | Reference | The IUCN Red List Category[a] |
|---|---|---|---|---|
| **Atlantic Cod** (Gadus morhua) | Stress response (increased cortisol levels);<br>Freeze reaction;<br>Lower fertilization rate (lower spawning performance);<br>Lower viability | Noise generated in the laboratory with a frequency range of 100–1000 Hz | Sierra-Flores et al. (2015) | VU |
| | More frequent startle reaction of newly hatched fish when exposed to noise;<br>Larvae exposed to regular noise used their yolk sacs faster after 2 days of exposure to noise;<br>Survival-related factors during development can be influenced by the timing of noise | Ship noise recordings | Nedelec et al. (2015) | |
| | Flight reaction | Trawling | Ona (1988) | |
| | Masking | Broad-band noise | Hawkins and Chapman (1975) | |
| | Lowering abundance | Seismic shooting | Engås et al. (1996) | |
| **European Eel** (Anguilla anguilla) | Compromised antipredator behavior (becoming easier prey);<br>Diminished spatial performance;<br>Increased ventilation and metabolic rates (stress) | Recordings of ships passing through harbors | Simpson et al. (2015) | CR |
| **Burbot** (Lota lota) | Data deficient | N/A | N/A | LC |
| **Baltic herring** (Clupea harengus membras) | Appeared to move to greater depths after the use of an airgun | Airgun | Slotte et al. (2004) | DD |
| | Data deficient<br>(One of the two Baltic species emitting sounds, next to cod, so could be potentially affected) | N/A | Klusek and Lisimenka (2016) | |
| **European Sprat** (Sprattus sprattus) | Fish schools were more likely to disperse when exposed to noise | Pile driver<br>Airgun | Hawkins et al. (2014) | LC |

[a]*CR* critically endangered, *(EN)* endangered, *VU* vulnerable, *NT* near threatened, *LC* least concern, *DD* data deficient

**Table 3** Various species that occur in the Mediterranean Sea (Malta)

| Species | Observed impact of noise | Type of noise | Reference | The IUCN Red List Category[a] |
|---|---|---|---|---|
| **Loggerhead Turtle** (Caretta Caretta) | Avoidance | Low-frequency sound | O'hara and Wilcox (1990) | VU |
| | Diving | Airgun | DeRuiter and Doukara (2012) | |
| **Leatherback** (Dermochelys coriacea) | Potential masking acoustic cues in sea-finding of hatchlings | Human conversation/ vehicle traffic noise | Holtz et al. (2021) | VU |
| **Kemp's Ridley (Lepidochelys kempii)** | Mass strandings | Explosives (used to remove petroleum platforms) | Klima et al. (1988) | CR |
| **Green Turtle** (Chelonia mydas) | Data deficient | N/A | N/A | EN |
| **Hawksbill Turtle** (Eretmochelys imbricata) | Data deficient | N/A | N/A | CR |
| **Fin Whale** (Balaenoptera physalus) | Modification of Fin Whale songs and behavior; Temporal and spectral song features decreased; Leaving the area for an extended period; Modified communication is to compensate for noise | Ship noise, airgun noise | Castellote et al. (2012) | VU |
| | Potential negative effects in the reproduction and survival | Ship noise, Airgun noise | Castellote et al. (2010) | |
| | Altered migration movements | Seismic exploration noise | Castellote et al. (2009) | |
| | Decrease in call detections with increasing shipping noise | Shipping noise | Ramesh et al. (2021) | |
| **Common Bottlenose Dolphin** (Tursiops truncatus) | Mass strandings | Explosives (used to remove petroleum platforms) | Klima et al. (1988) | LC |
| | Seasonal displacements from noisy areas | Leisure boating | Rako et al. (2013) | |
| | Auditory masking | Natural, anthropogenic, and synthesized noise | Branstetter et al. (2013) Branstetter and | |

(continued)

**Table 3** (continued)

| Species | Observed impact of noise | Type of noise | Reference | The IUCN Red List Category[a] |
|---|---|---|---|---|
| | | | Finneran (2008) | |
| | Increased whistle repetition | Watercraft noise | Buckstaff (2004) | |
| | Lombard response – increased vocal effort | Vessel noise | Kragh et al. (2019) | |
| | Temporary threshold shifts | Loud low- and mid-frequency noise | Nachtigall et al. (2003) Nachtigall et al. (2004) Mooney et al. (2009) Finneran et al. (2005) | |
| | Decrease in vigilant behavior; Decrease in target detection through echolocation | Pile drivers | Branstetter et al. (2018) | |
| | Affecting echolocation | Wide band noise generators | Romanenko and Kitain (1992) | |
| | Abandoning the area; In the case of trawlers, dolphins changed the acoustic behavior to compensate for the masking noise | Different kinds of boats | La Manna et al. (2013) | |
| | Affecting whistling (but not as a single factor) | Boat noise | La Manna et al. (2019) | |
| **Mediterranean Monk Seal** (Monachus monachus) | Potential stressor, stopped from leaving the cave to feed; Probably one of the causes of stranding | Data deficient | Danyer et al. (2014) | EN |
| **Neptune Grass** (Posidonia oceanica) | Alterations in Posidonia oceanica root and rhizome statocysts; Decrease in the number of rhizome starch grains (which have a crucial role in energy storage); Degradation in the specific fungal symbionts of P. oceanica roots | Low frequency noise (50–400 Hz) | Solé et al. (2021) | LC |

[a]*CR* critically endangered, *(EN)* endangered, *VU* vulnerable, *NT* near threatened, *LC* least concern, *DD* data deficient

**Table 4** Fish species that occur in the Mediterranean Sea (Malta)

| Species | Observed impact of noise | Type of noise | Reference | The IUCN Red List Category[a] |
|---|---|---|---|---|
| **Dusky Grouper** (Epinephelus marginatus) | No studies on the effect of noise but produces sound (calls) therefore sound could affect communication and spawning | N/A | Bertucci et al. (2015) | VU |
| **European Eel** (Anguilla anguilla) | Compromised antipredator behavior (becoming easier prey);<br>Diminished spatial performance;<br>Increased ventilation and metabolic rates (stress) | Ships | Simpson et al. (2015) | CR |
| **Atlantic Bluefin Tuna** (Thunnus thynnus) | Behavioral deviations in tuna schools | Boat noise | Sarà et al. (2007) | EN in the Mediterranean Sea |
| | Changes in swimming patterns | Ship and offshore wind farm operational noises | Puig-Pons et al. (2021) | |
| **Long-snouted Seahorse** (Hippocampus guttulatus) | A significant increase in the respiratory rate (stress);<br>In site abandonment | Motor boat | Palma et al. (2019) | NT in the Mediterranean Sea |
| **Gilt-head Seabream** (Sparus aurata) | Increase in motility;<br>Increase in lactate and haematocrit levels;<br>Decrease of glucose | 0.1–1 kHz linear sweep (150 dB rms re 1 μPa) | Buscaino et al. (2010) | LC in the Mediterranean Sea |
| **Brown Meagre** (Sciaena umbra) | Increased vocalization rate (possible vocal compensation) | Boat noise | Picciulin et al. (2012)<br>Picciulin et al. (2021) | VU in the Mediterranean Sea |
| | No changes in activity levels;<br>No change in sound emission frequency;<br>Increased flight and hiding behavior | Boat noise | La Manna et al. (2016) | |
| | Potentially affects acoustic communication | Boat noise | Codarin et al. (2009) | |
| **Red-mouthed Goby** (Gobius cruentatus) | Increased amount of time spent in the shelter | Boat noise | Picciulin et al. (2010) | LC in the Mediterranean Sea |
| | Diminished ability to maintain territory (more submissive behavior, won less encounters than in the silent condition) | Recreational boat noise | Sebastianutto et al. (2011) | |
| | Potentially affects acoustic communication | Boat noise | Codarin et al. (2009) | |

(continued)

**Table 4** (continued)

| Species | Observed impact of noise | Type of noise | Reference | The IUCN Red List Category[a] |
|---|---|---|---|---|
| **Damselfish** (Chromis chromis) | Potentially affects acoustic communication | Boat noise | Codarin et al. (2009) | LC |
| | A significant decrease of the time spent caring nests | Boat noise | Picciulin et al. (2010) | |
| | In the presence of greater traffic volume there were lower feeding frequencies (modification of foraging behavior);<br>The escape took longer when a boat passed nearby | Boat noise | Bracciali et al. (2012) | |
| | Stress<br>(elicits hematological stress parameters) | Simulated boat noise | Vazzana et al. (2017) | |

[a]*CR* critically endangered, *(EN)* endangered, *VU* vulnerable, *NT* near threatened, *LC* least concern, *DD* data deficient

## Ideas to Implement

At the moment, studies are done ad-hoc. Common standards need to be applied (e.g. Adeon project, ADEON 2023, Ainslie et al. 2018, Warren et al. 2018) so that all data collected can be more easily compared. A common and updated base of data on the underwater noise levels as well as the resulting influence on the marine environment and its organisms could prove to be a useful tool to improve regulations and work practice when it comes to activities that can introduce noise to the marine environment.

Finally, it is important to popularize knowledge about this issue both among the general audience, ship owners as well as scientists and students. Dedicated programs should be created to make it possible for them to get involved in addressing this problem (e.g. citizen science programs, study programs, internships, training programs).

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# Responses of Oregon Demersal Species to Seismic Survey Noise: Evaluating Behavior and Movement

53

Lauren K. Borland, Scott A. Heppell, Taylor K. Chapple, Kaustubha Raghukumar, and Sarah K. Henkel

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**Abstract**

Previous investigations into the effects of impulsive anthropogenic noise in the marine environment have focused on mammals, while few have focused on how harvested demersal species may respond behaviorally to this sound. This chapter assesses the impacts of noise borne from a geophysical seismic survey of the Cascadia Subduction Zone along the Oregon coast on the movement and

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behavior of three fish – lingcod, black rockfish, and China rockfish – and Dungeness crab using telemetry data from an in situ tagging effort in summer 2021 before, during, and after the geophysical seismic survey. Received sound levels were obtained using a vector sensor array and are presented in Raghukumar et al. (this issue). To understand whether the soundscape influences short-term behavior, patterns in detected acceleration and depth were analyzed. Preliminary results presented in this chapter suggest that there are differences in black rockfish instantaneous acceleration as well as distance from the surface in the defined "during" and "after" periods of time as the research vessel passed near the area as compared to the "before" period. There were also differences in China rockfish acceleration during and after as compared to before the period of noise, but there was little difference in acceleration values after the noise event compared to during.

**Keywords**

Sound · Behavior · Seismic survey · Rockfish

## Introduction

Sound is a crucial aspect of the underwater environment for fishes – various species use sound to communicate, identify predators, navigate, and many other activities needed for survival in their habitat (Montgomery et al. 2006, Carroll et al. 2017). Disruptions and disturbances in the natural soundscape can have important impacts on all these activities and likely have important implications for the functioning of the ecosystem at large (Rossi et al. 2016). Human-generated noise has become increasingly prevalent in the past few decades (Slabbekoorn et al. 2010; Duarte et al. 2021) as humans explore more of the oceans, extract resources, and increase global commerce. In the northeast Pacific, marine species are impacted by ever-increasing levels of acute and chronic anthropogenic noise (Hildebrand 2009), as human activities offshore are increasing in frequency and magnitude; pile driving, offshore renewable energy, and seismic surveying are all becoming more commonplace (Frisk 2012). Many large-scale activities in the ocean are accompanied by regulation and required discussion on the impact that noise generated by such activities may have on the ecosystem using best-available science. However, most of the available research is focused on charismatic and well-studied marine mammals (Hawkins and Popper 2017). Current regulatory decisions regarding the impacts of noise-generating activity are being made with relatively little research that focuses on noise impacts on fish and other demersal organisms. This has been identified as a significant issue for the conservation of aquatic life, as noise can affect myriad aspects of the ecosystem. For example, noise can mask acoustic communication in fish (Codarin et al. 2009). This paucity of data on nonmarine mammal effects has been a point of concern for recreational anglers, commercial fishers, as well as resource managers. There is clearly a significant need for further research and attention on this issue to protect fisheries and conserve ecosystems in the northeast Pacific.

In the summer of 2021, a geophysical seismic survey was conducted along the northwest coast of the United States, mapping the Cascadia Subduction Zone off the shores of Oregon and Washington with an array of 36 air guns. This event was a rare opportunity to observe the behaviors of demersal fish in situ to an infrequent energetic noise stimulus. In this chapter, behavioral responses were examined in two rockfish species – black rockfish, *Sebastes melanops*, and China rockfish, *S. nebulosus*. The study was completed within the Redfish Rocks Marine Reserve (RRMR) in Southern Oregon as the geophysical survey research vessel made three passes near the area, allowing for replicate observations, and previous data shows site fidelity of rockfish within this marine protected area (Calvanese 2016). Previous *s*tudies of rockfish (genus *Sebastes*) responses to seismic survey noise used animals in field enclosures with only a single air gun (Pearson et al. 1992), so little is known about how they may respond to the differing aspects of seismic survey noise in situ and how habitat use of specific species will influence their behavioral responses to the noise. The study aims to assess how behavioral metrics differed between the two rockfish species in response to a changing soundscape, and this chapter includes preliminary results of the analyses.

## Methods

A vector sensor array of hydrophones (the NoiseSpotter®, Raghukumar et al. 2020) was deployed in the Redfish Rocks Marine Reserve (RRMR) near Port Orford, Oregon, USA, at 36-meter depth near the rocky reef structures within the site to measure the received acoustic levels corresponding to the three passes of the seismic survey research vessel. Further methods and data on changes in the soundscape before, during, and after the seismic survey passes are described in Raghukumar et al. (this issue). Three days with detectable air gun pulses were identified – June 11, 17, and 18 – within the 52-day study period (May 20 – July 11). Periods of the study are defined as before (prior to June 11), during (June 11, 17, and 18), and after (after June 18). Raghukumar et al. (this issue) report cumulative sound exposure levels (cSEL) were highest on June 11 but also note an increase in cSEL in the after period as compared to before due to high wind speeds.

An array of 45 Vemco VR2W acoustic receivers (Innovasea Systems, Nova Scotia) was also deployed within the RRMR for overlapping coverage of the site and to enable triangulation of acoustically tagged animal positions (Fig. 1). Three fish species were captured – lingcod, *Ophiodon elongatus*, black rockfish, and China rockfish (using hook and line gear) – and one invertebrate, Dungeness crab, *Cancer magister*, (using crab pots) for tagging. Fifteen individuals of each species were tagged with Vemco acoustic tags (V13A) to capture acceleration and position as well as depth (using V13AP tags) for the semi-pelagic black rockfish.

Most lingcod and Dungeness crab individuals tagged in this study exited the acoustic array either before or during the period of the seismic survey, although some lingcod returned after the survey (Fig. 2). Due to low sample sizes for these two species during the periods of the seismic survey noise, it is not possible to ascertain

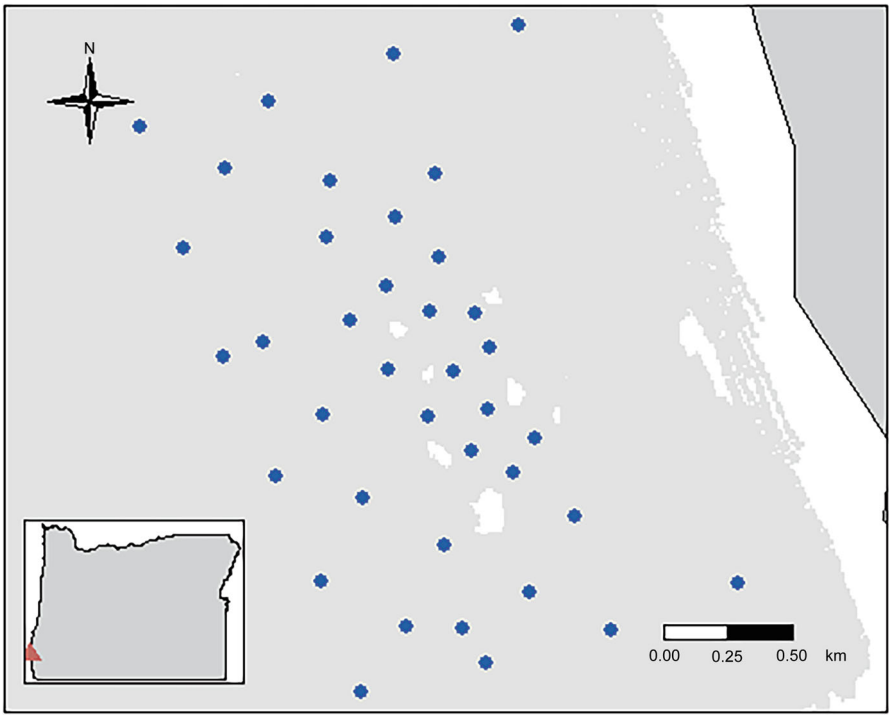



**Fig. 1** Acoustic receivers deployed in the RRMR in Southern Oregon. White polygons within the array depict emergent rocks in the marine reserve

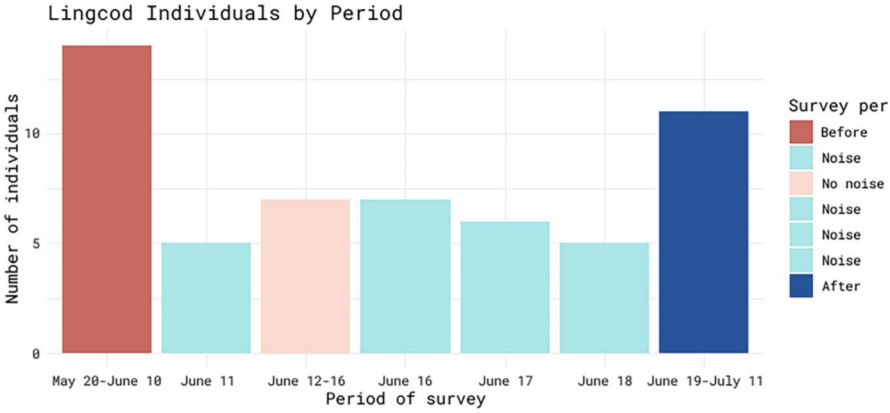



**Fig. 2** Number of lingcod individuals detected in each of the study periods

whether there are any behavioral changes due to the noise, so no further analyses for Dungeness or lingcod are presented in this chapter.

To standardize the total number of observations for comparisons of behavioral patterns, 3 full days within each of the before and after periods (6 total) were randomly

chosen to compare to the 3 days with the seismic survey vessel presence (defined as the during period). These same days were used for analyses of both rockfish species. Violin plots with means were used to visually assess differences in acceleration values for each rockfish species. Differences in acceleration values (and depth for black rockfish) among periods were tested with a mixed-effects model, with study period as a fixed effect and transmitter (i.e., individual), water temperature (degrees Celsius), and salinity as random effects. As a proxy for spatial use, the number of unique acoustic receivers that each rockfish individual was detected on for each period of the study and a repeated measures ANOVA were used to compare between periods. All analyses were conducted using RStudio; important packages included lubridate for working with dates and times (Grolemund and Wickham 2011), ggplot2 for data visualization (Wickham 2016), and lme4 for analyzing mixed-effects models (Bates et al. 2015).

## Results and Discussion

Black rockfish had more incidences of higher acceleration (Fig. 3) in the survey periods with detected air gun pulse noise (mixed-effects model, $p < 0.001$ for survey period) as compared to before. Post hoc Tukey tests showed a significant difference for before to June 11 ($p = 0.0002$, coefficient estimate $= 0.547$ m/s$^2$), before to June 17 ($p = 0.0096$, coefficient estimate $= 0.378$ m/s$^2$), and before to June 18 ($p = 0.0036$, coefficient estimate $= 0.374$ m/s$^2$). Potential habituation over the three passes of the seismic survey vessel was observed as the coefficient of the change in acceleration values decreased for each of the dates with detected air gun pulse noise.

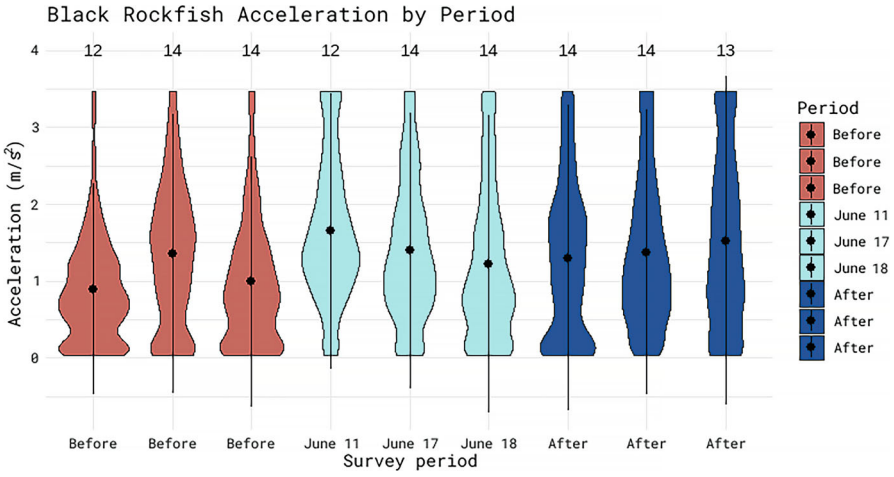



**Fig. 3** Distributions of black rockfish acceleration values within each of the selected study days. Numbers above the plots depict the number of fish included in each date. Each of the study dates with noise from the seismic survey had higher acceleration values than the average of the before days. The average of the after days was also higher than before

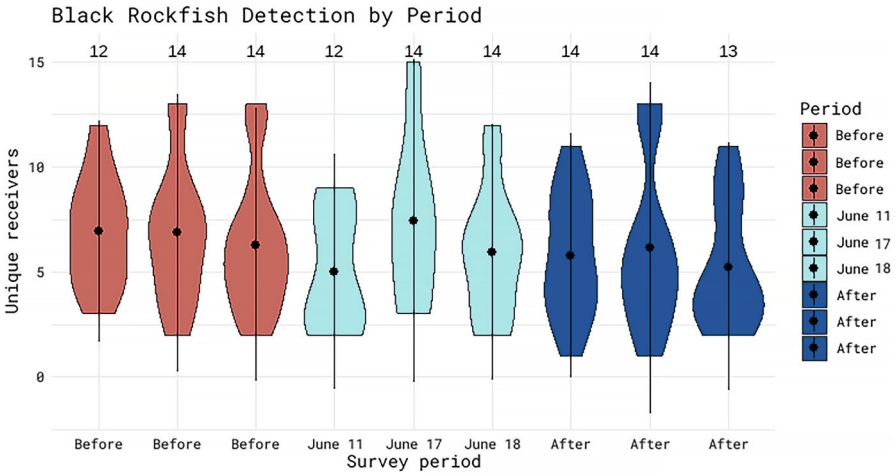



**Fig. 4** Number of unique receivers black rockfish individuals were detected on for each of the selected study days

The acceleration in the after period was also significantly greater than the before period ($p < 0.0001$, coefficient estimate $= 0.34$ m/s$^2$), but the after period was not different than any of the during dates. This behavioral response aligns with what is known about black rockfish behavior (Love et al. 2002, Hannah and Rankin 2011); individuals of this species are more mobile than other rockfish species, and startle responses are what we might expect in response to a noise disturbance. Black rockfish appear to traverse less area on two of the dates in the during period, as they were detected on fewer receivers (Fig. 4) as compared to the before period. A repeated measures ANOVA assessing the number of receivers an individual was detected on over each period (average of 3 randomly chosen days) showed that the date of the survey was significant for the number of receivers for individual black rockfish ($p = 0.043$). Post hoc tests further show a significant difference between the before and after period ($p = 0.03$). The difference in number of receivers between the before and during periods also appears interesting, although not statistically significant ($p = 0.17$), as there is a large amount of variability in the number of unique receivers fishes were detected on in the during period. Overall survey period was not significant for variability in black rockfish depth positions, although post hoc tests show significant differences in pairwise comparisons between before and June 11 ($p = 0.0186$, coefficient estimate $= -2.24$ m) and June 11 and after ($p = 0.0058$, coefficient estimate $= 2.25$ m). These estimates show that black rockfish traveled up in the water column (negative depth indicates closer to the surface) on the first day of the pass of the survey vessel and then further down in the water column in the after period as compared to the initial survey period. However, further exploration into the bathymetry of the site should be conducted to convert distance from surface to height off the bottom, as the distance from the seafloor may be more ecologically relevant than distance from the water's surface due to the highly rugose seafloor, including emergent rocks, within the study area.

China rockfish acceleration values were higher in the first day of seismic survey noise and in the after period with high wind-generated noise as compared to the before period (Fig. 5) (mixed-effects model, $p = 0.0093$ for survey period). Post hoc Tukey tests showed a significant difference for before to June 11 ($p = 0.0022$, coefficient estimate $= 0.413$ m/s$^2$) and before to after ($p < 0.0001$, coefficient estimate $= 0.171$ m/s$^2$). Overall, this species appears to have a smaller use area within the RRMR than the black rockfish (Fig. 6); this is expected as this species is

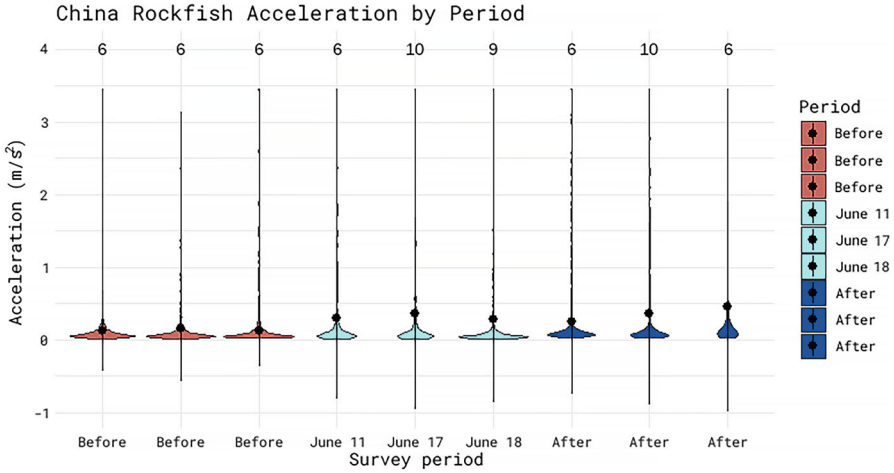



**Fig. 5** Distributions of China rockfish acceleration values within each of the selected study days. Numbers above the plots depict the number of fish included in each date. The first date with noise from the seismic survey (June 11) had higher acceleration values than the average of the before days. The average acceleration across the after days was also higher than before

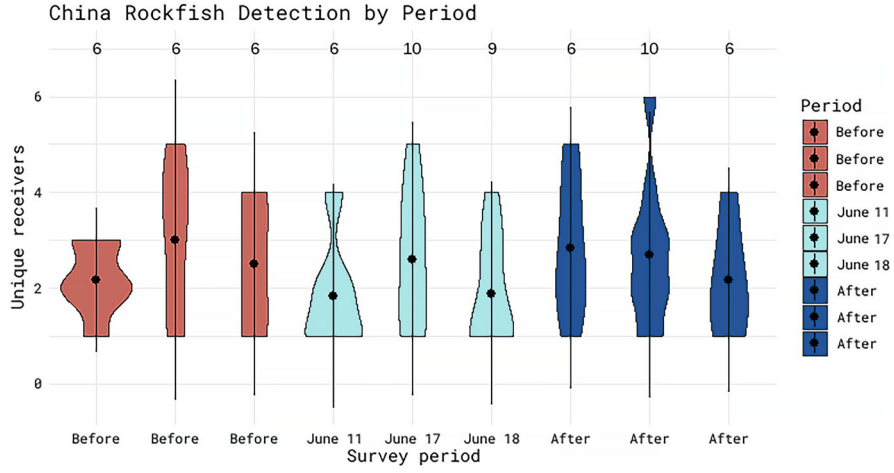



**Fig. 6** Number of unique receivers China rockfish individuals were detected on for each of the selected study days

known to move around less than black rockfish. A repeated measures ANOVA for the average number of receivers each individual was detected on over three periods (before, during, after) shows no significant difference between periods ($p = 0.279$). All post hoc pairwise tests show no significant differences in comparisons. China rockfish may be staying near high-quality rocky habitat at all times, not just in times of noise disturbance.

## Conclusions

An understanding of how anthropogenic noise impacts the soundscape is important to understanding potential effects on species that exist along the Oregon coast. The study in this chapter will contribute to an understanding of Oregon's nearshore, characterizing how fish will likely respond to increasing human activity and noise in the future, and is broadly applicable to the NE Pacific shelf as the study species range from California to the Aleutian Islands. Anthropogenic noise can influence fish movement which can impact other economic activities. Changing distributions of animals or increasing stress in the long term, for example, can affect fishing success. As Bejder et al. (2009) note, a decrease in a specific behavioral response (as our data indicate for black rockfish acceleration) or changes in tolerance may not be a habituation response, as there could be less visible physiological effects. Improper assumptions about fish response to chronic noise over time may have harmful impacts to species if a decrease in response is noted without further investigation, and approval for projects that generate chronic, constant noise in the ocean is granted. Noise impacts on lower trophic level species can also have bottom-up effects on top predators and/or highly migratory species with high economic value. Overall, there is a clear need for further research as well as attention by regulators and managers to the impact of anthropogenic noise on fish behavior, ecology, and physiology to ensure resilient nearshore ecosystems.

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# Development of Underwater Acoustic Deterrents for Invasive Species Control

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Christa M. Woodley, Nicholas A. Barkowski, Aaron C. Urbanczyk, and Jeffery K. Zuercher

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**Abstract**

Globally, invasive non-native species (INNS), due to anthropogenic activities, are a form of aquatic pollution that severely affects the environment, ecological diversity, and economics. INNS are often unconstrained by competition and predation, exploit resources throughout their range, and thus continuously expand into vulnerable environments. Isolation attempts, targeted removal, and intentional fragmentation involve the use of deterrents and barriers in freshwater systems to control species movement and population expansion. Underwater acoustics is currently the only non-structural deterrent or attractant that can be agnostic or specific to a group of fish based on their hearing range, thereby reducing ecosystem impacts and risk to humans compared to other non-structural technologies. In addition, the costs to deploy and maintain underwater acoustics systems can be significantly lower than other options. An

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underwater acoustics deterrent system (uADS), along with an electrical deterrent, flushing, automated barge clearing deterrent, and dispersal bubble curtain, is proposed as a multi-deterrent system for Brandon Road Lock and Dam (BRLD; Joliet, Illinois, USA) on the Des Plaines River to control the spread of invasive carp. Herein, we will provide updates on the current deployments of uADS. Results of ongoing underwater acoustic research will help managers and stakeholders understand the effectiveness of uADS and, most importantly, provide considerations for introducing noise into an ecosystem and characterizing impacts on native species.

**Keywords**

Invasive species · Underwater acoustic · Deterrent · Hearing · Species management · Carp

## Introduction

The United States Army Corps of Engineers (USACE) manages over 48,500 $km^2$ of public lands, 40,230 km of coastal and interior waterways and channels, 250 water resources projects, and globally supports other countries through technology exchange and subject matter expert (SME) knowledge access. Due to the complexity of habitats, structures, and missions, management of threatened and endangered (T&E) species and invasive non-native species (INNS) species are an ever-present issue for the USACE. Species management is further compounded by changes in the need for training operations, coastal development, aging infrastructure, removal of infrastructure, new navigation criteria, growing coastal and waterway human populations, and climate change. Land and aquatic species management, however, is conducted in such a manner that it complements the mission of the USACE and the surrounding areas (e.g., residential, urban, industrial). For this reason, the USACE often balances economic, social, and environmental needs by managing the impacts of the INNS, commercial and recreational activities on the environment, on navigation (and thus the economy), on humans and human safety, and on costs (near-term installation, long-term operation, and maintenance). As a result, the USACE has invested in and supported research and development of structural and non-structural deterrents (NSDs) and management alternatives for over two decades to reduce effects on T&E species or maximize impacts on INNS (Structural deterrents or barriers: They may have significantly different primary functions, such as water flow control structures, that result in a secondary impact, such as blocking or constraining species movements or passage. There are some physical barriers whose primary function is fish deterrent or removal, such as exclusions screens.) (Non-structural deterrents: They include any stimulus or non-solid obstruction that discourages or prevents a targeted species from passing through a region of interest. Non-structural deterrence systems [typically] do not constrain water flow or restrict navigation.). One of the better examples of long-term INNS efforts conducted by the

USACE is the electric barrier installed in the Chicago Area Waterway System (CAWS) (Fig. 1).

In the early 2000s, the USACE explored options to prevent the transfer of INNS between the Mississippi River and Great Lakes basins through the CAWS. The CAWS consists of five hydrologic pathways; however, only one, the Chicago Sanitary and Shipping Canal (CSSC) is an open connection between the Mississippi River and Great Lakes basins via the Chicago Harbor Lock at the confluence of the Chicago River and Lake Michigan in Chicago, Illinois, USA. The CSSC has previously allowed the movement of aquatic species between the basins. After the Eurasian Ruffe (*Gymnocephalus cernua) and the* Round Goby (*Neogobius melanostomus*) became established in the Great Lakes, there was sufficient concern that these species could spread into the Mississippi River Basin through the CSSC. The construction of a demonstration electric barrier in the CSSC was proposed and approved (Sparks et al. 2011) to reduce the likelihood of interbasin transfer of these two species. Subsequent barriers (Barrier IIA, IIB) were constructed in 2009 and 2011, respectively. In 2021 the demonstration electric barrier was upgraded to increase electrical output and is referred to as Barrier 1. The three barriers now operate to deter the spread of INNS, particularly invasive carp, from the Mississippi River to the Great Lakes (Veraldi et al. 2011; Parker et al. 2015). The operational parameters of the electric barriers were established based on tests with subadult Bighead and Silver Carps (Holliman et al. 2015). Currently, the adult population front of invasive carp is located approximately 75 km from Lake Michigan, with three locks and an electric barrier in between the carp and Lake Michigan (Fig. 1).

Even though electrical barriers allow for hydrological connectivity and navigation between the Illinois River and the Great Lakes, they have two critical disadvantages. The first disadvantage is the severe risk to humans and human safety, which has not resulted in a loss of human life. Secondly, the electrical barriers limit the migration of many native and desirable fish (estimated 80–100%) (Maceina et al. 1999; Holliman et al. 2015; Parker et al. 2015; Weber et al. 2016). The CAWS electric barriers are non-selective and, depending on settings, water quality, and the presence of commercial navigation, can result in the incapacitation, injury, and even mortality of any fish attempting to migrate through this corridor regardless of direction. This significant limitation to native species migration is a strong motivation for resource managers to continue to push for "selective" non-structural deterrents that function on desired INNS, or groups of fish, rather than all species.

## Invasive Carp

The spread of invasive carps (*Hypophthalmichthys molitrix*, *H. nobilis*, *Ctenopharyngodon idella*, and *Mylopharyngodon piceus*) continues to threaten aquatic ecosystems (Pegg et al. 2002; Solomon et al. 2016; Sullivan et al. 2018) across the United States (US). Since the 1960s, the importation and subsequent introductions of invasive carp have led to broad-range expansion and population increases (Sullivan et al. 2018; Altenritter et al. 2019) in the Mississippi River Basin

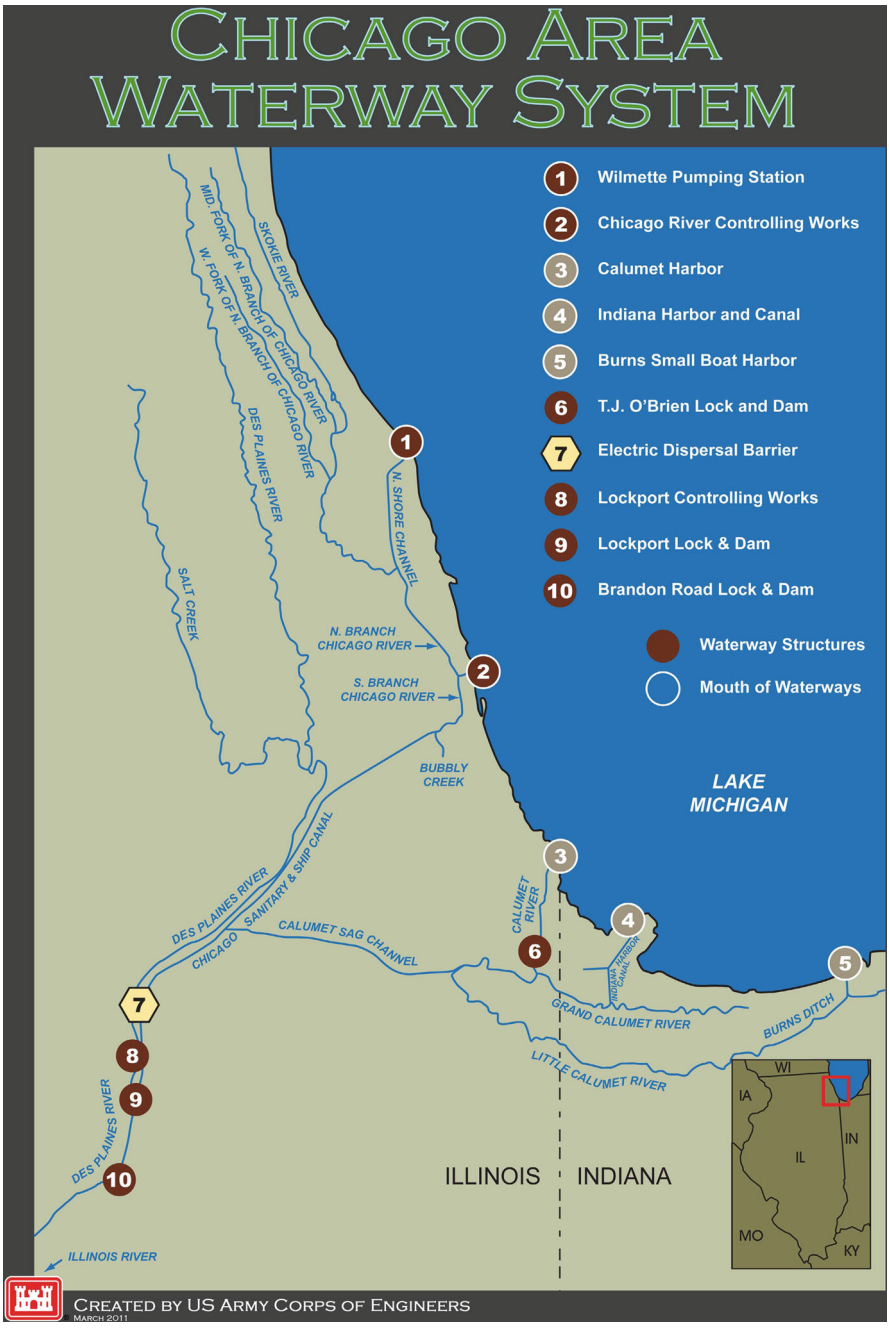



**Fig. 1** Area of interest within the United States. (Inset) The inset shows the location of Brandon Road Lock and Dam (BRLD) is noted by a "1," and the Chicago Area Waterway System (CAWS) is noted by a "2" (Image Credit: USACE Chicago District)

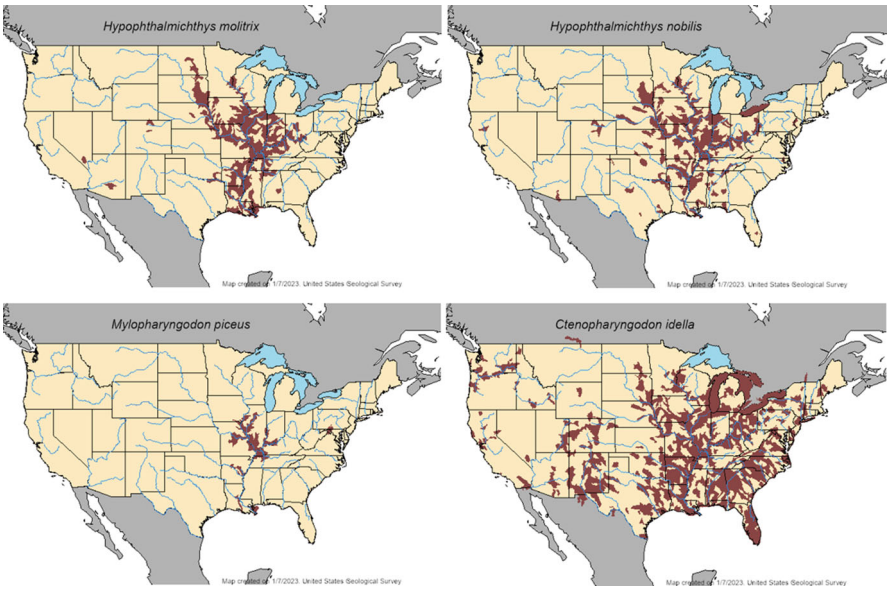



**Fig. 2** Distribution maps of the four invasive carp species within the United States. (TOP LEFT) Silver carp, *Hypophthalmichthys molitrix*. (TOP RIGHT) Bighead carp, *H. nobilis*. (LOWER LEFT) Black carp, *Mylopharyngodon piceus*. (LOWER RIGHT) Grass carp, *Ctenopharyngodon idella*. (U.S. Geological Survey. 2023. Nonindigenous Aquatic Species Database, Gainesville, FL. http://nas.er.usgs.gov. Accessed 13 Jan 2023)

and smaller water bodies across the United States (Fig. 2). These large-bodied planktivores (*sans M. piceus*, a molluscivore) compete for resources with native planktivores and the early life stages of other native fishes resulting in adverse effects on aquatic species populations, fish communities as well as impacting water quality directly and indirectly (Solomon et al. 2016; Phelps et al. 2017; Sullivan et al. 2018; Altenritter et al. 2019). Furthermore, invasive carp can negatively affect recreational boating, commercial navigation, and fishing, resulting in negative economic impacts (Solomon et al. 2016; Chick et al. 2020). The establishment of invasive carp in the Great Lakes may jeopardize fisheries valued at USD 7B per year (Lauber et al. 2020).

An international effort has been underway to address the density, dispersal, and adverse impacts of invasive carp through research and development of various methods, technologies, and management approaches. Population reduction efforts have focused on commercial harvest programs (Love et al. 2018), marketing for human consumption (Bouska et al. 2020), and the development of novel equipment and techniques (Butler et al. 2019; Ridgway et al. 2021). New prototype technologies to manage the spread of invasive carp include non-structural deterrents such as the electric barrier (Clarkson 2004), light deterrents (Kim et al. 2019), overhead attractants (Ribeiro et al. 2022), carbon dioxide (Cupp et al. 2021), air bubbles (Zielinski and Sorensen 2016), underwater acoustics deterrent system (uADS)

(Bzonek et al. 2021), pheromone treatments, or combinations of these (Dennis et al. 2019). Placement of non-structural deterrents at strategic pinch-points (e.g., locks and dams, and narrow corridors where fish passage is reduced to a confined area) impedes the movement of invasive carps while reducing costs of managing large river reaches and aids in achieving the management objective (Noatch and Suski 2012; Cupp et al. 2021; Jones et al. 2021).

## Single and Multiple Non-structural Deterrents

Once an INNS is established, it is often challenging to eliminate the invading species; thus, mitigation and management efforts focus on limiting the spread of the INNS (Van der Zanden and Olden 2008). Physical barriers (e.g., concrete structures, weirs, exclusion screens) can be effective measures for limiting or blocking INNS; however, they may result in a hydraulic disconnect between the upstream and downstream corridors (Jones et al. 2021). Physical barriers prevent the movement of non-target biota, including invertebrates (Grill et al. 2015), aquatic vegetation (Jones et al. 2020), mammals, reptiles, amphibians, and fishes. Additionally, physical barriers can influence hydrological flow, accrue debris, and thus impact navigation (Van der Zanden and Olden 2008; Grill et al. 2015).

In recent years, exclusion and/or attraction have been increasingly used as a management strategy to control the spread of aquatic INNS species in the United States and globally. Generally, these technologies are often not 100% effective at blocking INNS. The efficacy of deterrence for NSDs typically ranges between 60% and 99% due to a wide range of deployment equipment, locations, and site-specific challenges (Noatch and Suski 2012; Jones et al. 2021). The efficacy of NSDs varies with the sensory capabilities of fish species – much like with humans – changes with species, age, and life history experiences (Noatch and Suski 2012). NSDs offer the distinct advantage of maintaining hydrological flow and low to no impact on humans, human safety, and commercial and recreational vessels (*sans* the electrical and chemical deterrents). Scientists, resource managers, and stakeholders must be diligent and evaluate how to maximize the efficacy of a NSD. This requires a technical understanding of the strengths and weaknesses of each NSD technology, how the NSD will be deployed, what is expected of the NSD within the variabilities of the environment, and how the NSD elicits a chosen behavior of a species at a given life stage, and other biological aspects.

Where non-structural deterrents are a feasible option to provide selective fish passage, managers can mitigate impacts of habitat fragmentation and impacts on T&E and other desirable fishes. There is growing interest to further develop the use of physiological and behavioral stimulants as a powerful approach to control and influence INNS while limiting impacts on T&E and native species. This requires that technology be developed to elicit sensory or physiological responses shared by an INNS species or a group of fish (Noatch and Suski 2012). Since non-structural technologies tend to target altered behavior, the primary response variable often is movement-based. A successful research and development effort for a NSD is based

on the ability of the research and manager team to support lab studies, mesoscale field studies, and large-scale field studies to allow for feedback on lessons learned to improve the next generation of full-scale NSDs. Underwater acoustics is one technology that can target the hearing capabilities of INNS, can be engineered to be moderate to highly effective in both attraction and deterrence scenarios, assessed through movement and performance metrics, and ultimately, can be finessed through collaborations between lab and field studies.

## Underwater Acoustics as a Selective NSD?

In 2018, the USACE stepped in to further the uADS progress, explicitly focusing on navigation lock approaches. The effort started by examining several variables such as the bathymetry, water quality, lock operations, ambient sound and noise associated with a typical shallow-water lock (i.e., BRLD), and the 100-horsepower signal recommended in the literature as a possible noise source for modulating invasive carp behavior (Vetter et al. 2017). At BRLD, sound was collected using a passive underwater acoustic monitoring system of eight-channels, 24-bit Autonomous Multichannel Acoustic Recorders (AMARs, JASCO Applied Sciences). Each AMAR was fitted with three M36-100 hydrophones and one M20 vector sensor (GeoSpectrum Technologies Inc). The spectral noise floor, then, was 20 dB re 1 μPa2Hz-1 and the nominal ceiling is 171 dB re 1 μPa. Next, we examined capabilities of commercial off-the-shelf technologies for underwater transducers that were tolerant of shallow-water locks and capable of handling commercial navigation prop wash and debris that passes through the USACE locks (USACE-ERDC 2022). Much of the vessel noise generated within the BRLD approach was contained in the 100–1000 Hz range (Fig. 3). In addition, many of the noise events, that accounted for the sound energy, were relatively short-duration events (vessel traffic, lock drainage, biological calls, and vehicle/bridge traffic) and primarily under 700 Hz.

From the BRLD campaign, the development of an ambient sound map and a three-dimensional propagation model, using range-dependent parameters and out-of-plan reflections (Marine Operations Noise Model [MONM, PE 3D, JASCO Applied Jasco Sciences]), suggested that the sound generated by a uADS should focus on generating deterrence signals with a frequency content greater than 500 Hz to be above the majority of the ambient sound levels (<700 Hz) and outside the hearing range of most native Mississippi-basin fish species (estimated 800 Hz) (Table 1), yet below the estimated 5 kHz of the upper limit of invasive carp (Fig. 4).

The results confirmed that the underwater acoustics production would propagate through the BRLD downstream approach, that care should be taken when developing signals due to the reverberant environment, and that lock approaches could provide infrastructure to protect a uADS from navigation strikes or objects caught in downstream flows. In addition, the variability in river stage, seasons, flood pulses, sediment fluxes, and temperature are dampened within the lock approaches. Thus, the operational parameters of a NSD would require fewer continual adjustments during a long-term deployment.

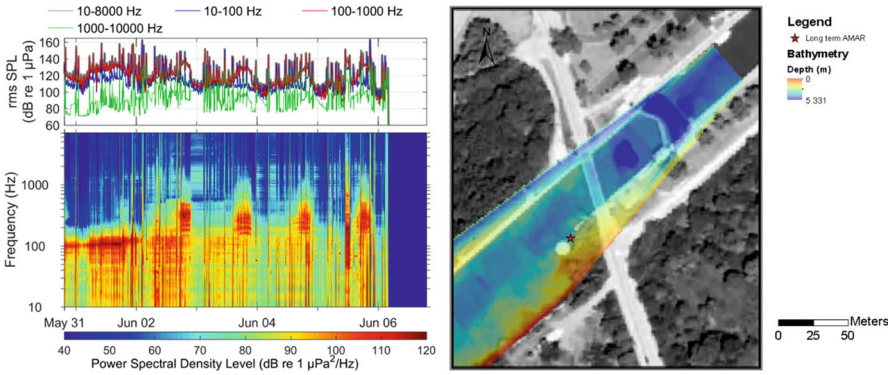



**Fig. 3** (LEFT) Band limited spectral levels and long-term power spectral density from an Autonomous Multichannel Acoustic Recorder (AMAR, JASCO Sciences). The 100–1000 Hz band (red) is approximately the same as the total energy band 10–8000 Hz (grey), indicating almost all sound energy. The lowest sound energy levels were in the 1000–8000 Hz band (green). (RIGHT) The location of the long-term AMAR set between the mooring cell and lock wall in the approach channel of Brandon Road Lock and Dam. The overlay is the bathymetry of the approach channel

**Table 1** Examples of hearing capabilities based on auditory evoked potentials (AEPs) of native and T&E species associated with the Mississippi River and Des Plaines rivers where uADS could be deployed for invasive carp deterrence

| Species | Hearing sensitivities (#Hz: # dB re 1 µPa) | | Reference |
|---|---|---|---|
| Northern Pike *Esox lucius* | 100 Hz: 99 ± 3.5 | 200 Hz: 87 ± 3.6 | Mann et al. 2007 |
| Channel Catfish *Ictalurus punctatus* | 100 Hz: 96.44 ± 1.53 | 200 Hz: 89.11 ± 0.92 | Wysocki et al. 2009 |
| Paddlefish *Polyodon spathula* | 250 Hz: 131 | 500 Hz: 143 | Lovell et al. 2005 |
| Lake Sturgeon *Acipenser fulvescens* | 200 Hz: 133 | 500 Hz: 144 | Lovell et al. 2005 |

In March 2021, an experimental uADS was deployed in the downstream lock approach of Lock No. 19 (Mississippi River, Keokuk, Iowa). Lock No. 19 was identified by stakeholders as a viable location for assessing the impacts of underwater acoustics on the behavior of invasive carp and had a current population of invasive carp; thus, INNS were not introduced, nor surrogate species needed to examine the large-scale field assessment (USACE 2021). A prior study of the fish community near and in Lock No. 19 indicated that invasive carps and native fishes commonly use the lock for upstream passage (Fritts et al. 2021).

For this system, the USACE conducted several acoustic analyses before installation to establish a sound propagation loss model (using a geometric spreading loss), examine design configurations for ensonification of the lock approach, and assess the ambient noise pollution. Similar to the campaign conducted at BRLD, this campaign was conducted using HP 1424 (Lubell Labs, Whitehall, Ohio) positioned on a

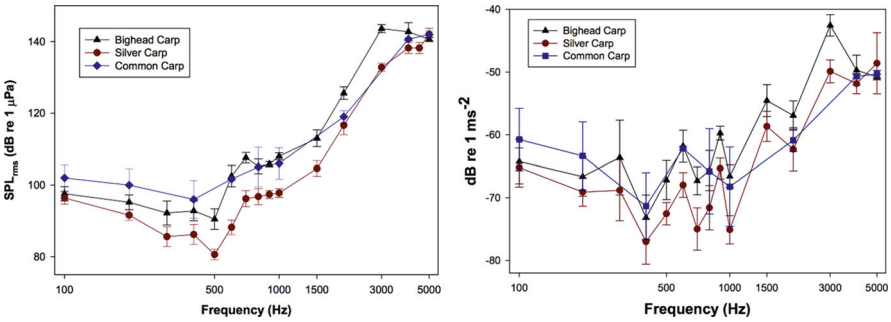



**Fig. 4** (LEFT) Audiograms for Bighead, Silver, and Common carp. Each data point represents the minimum sound pressure level (SPL; dB re 1 µPa $SPL_{rms}$) necessary to invoke an *auditory* evoked potential (AEP) response (tested range: 100 Hz to 5000 Hz). (RIGHT) Particle acceleration thresholds (dB re 1 µPa $ms^{-2}$) for the Bighead, Silver, and Common carp. Data are reported as mean (±S.D.). (Accessed 2023 Jan 13; Vetter et al. 2018)

projector vessel for signal transmission, and the signal was recorded on a receiver vessel using an AMAR, set to 24-bit, with three M36-100 hydrophones (GeoSpectrum Technologies Inc., Dartmouth, Canada) and one M20 vector sensor (GeoSpectrum Technologies Inc., Dartmouth, Canada). With these components, the spectral noise floor is 20 dB re 1 $\mu Pa^2$/Hz and the nominal ceiling is 171 dB re 1 µPa. To reduce vessel generated noise on the receiver and projector vessels, the sound production (CDi 2000, Crown Audio, Lake Mary Florida), communications (i.e., radios), and data collection systems (i.e., water quality, GPS, chart plotters, laptops, etc.) were powered by uninterruptable power supplies (SMT2200C, American Power Conversion, West Kingston, Rhode Island). Campaign data was used on four projector locations along the top of the 5th discharge lateral and one location within the 5th discharge lateral and recorded the signals at 28 locations 1 m off the lock floor, and at 28 locations mid-water column.

The final deployed uADS consists of a "soundbar" in which 16 transducers were co-located in a discharge channel (patent pending COE-789 [USACE-ERDC 2022]) (Fig. 5). The projector cables are placed in a series of cuts to a control station. The noise generated by the soundbar is recoded using a series of 8104 hydrophones, Lan-XI Data Acquisition System, and Tescia 2022.0 (Hottinger, Brüel and Kjær [HBK], Virum, Denmark). The Lock No. 19 evaluation is ongoing and primarily conducted between typical navigation season (March through early December) to examine navigation and seasonal impacts on the operation and effectiveness of the uADS (Brey et al. 2023).

## Brandon Road Interbasin Project (BRIP)

Currently, the USACE, federal and state partners, and stakeholders are developing the Brandon Road Interbasin Project (BRIP, Fig. 6). The project is a complex ecosystem protection effort to prevent invasive carp and other aquatic nuisance




**Fig. 5** (TOP) Deployment of the soundbar (which hangs over in the photo) in the 5th discharge lateral at Lock No. 19 in the Mississippi River (Keokuk, IA, Photo credit: Mark Cornish, USACE). (LEFT) Photo of silver carp over the uADS (powered off) at Lock No. 19 with the miter gates closed (Photo credit: Christa M. Woodley, USACE). (RIGHT) Deployment of two types of acoustic receiving systems at Lock No. 19 to study fish movements (Hydroacoustic Technology Incorporated [HTI; 307 kHz]; Innovasea Systems Inc., Boston, MA, USA) and sound events (8104, low noise shield cable, Hottinger, Brüel and Kjær [HBK], Virum, Denmark)

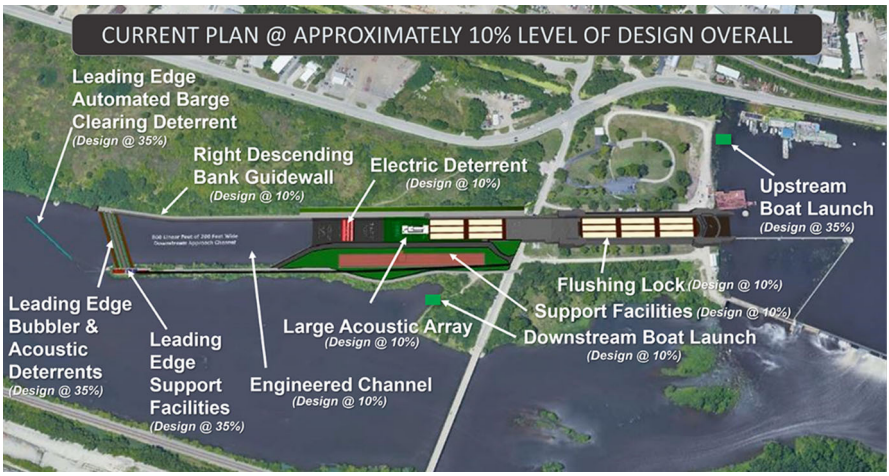



**Fig. 6** Brandon Road Interbasin Project (BRIP) Proposed 10% Design. This is the current proposed design of the multi-deterrent system proposed for Brandon Road Lock and Dam (BRLD; Joliet, IL, USA) as of August 2022. The BRIP proposal includes an automated barge clearing deterrent, two underwater acoustics deterrent systems, an electric deterrent, and a flushing lock to prevent the passage of adults, juveniles, and larvae of invasive species. (Accessed 13 Jan 2023; http://www.mvr.usace.army.mil/Missions/Environmental-Stewardship/BR-Interbasin-Project/)

species movement into the Great Lakes via the Illinois Waterway (USACE 2018). Bandon Road Lock and Dam (Joliet, Illinois, USA) has been identified as the critical pinch-point where layered technologies could be used to prevent the movement of invasive carp populations into the Great Lakes. The pre-construction engineering design phase of the BRIP was initiated in December 2020 when the state of Illinois signed an agreement with the USACE. The BRIP approach involves a layered system of deterrents that include flushing the locks with water, an engineered channel to reduce hiding places for species while providing infrastructure for an electric deterrent, underwater acoustics, dispersal bubble curtain, and an automated barge clearing deterrent. In addition, the project supports public education and outreach, regional fish monitoring, integrated pest management, state-regulated pesticide treatment, and manual removal (USACE 2018). The conceptual idea for the multi-deterrent system proposed is to prevent INNS from passing through BRLD. However, much like the electric barrier in the CSSC, the BRIP will allow for hydrological connection, but significantly limit migration through the lock by native and desirable species.

## Next Steps

To date, there are many assumptions on the potential impacts of NSDs, in particular, the impacts of underwater acoustic deterrents on the environment, community, and even infrastructures. In terms of the most needed unknown, yet least studied, are the potential

long-term impacts of NSDs and the impacts of underwater acoustics. Most long-term studies have focused on near-term impacts, such as ontogenetic developments, physiological stress, audio impairments, and behavioral alteration or disruptions in movement-migration patterns. Without understanding the long-term consequences of underwater acoustic deterrents, there are unrevealed and potentially insidious risks to T&E and native species, community ecology (including fishes, turtles, mammals, invertebrates, plants, and microbiology), biodiversity, and ultimately on, ecosystems health (Hawkins et al. 2015; Popper and Hawkins 2019; Popper et al. 2020). The cumulative effects of long-term exposures to underwater acoustic deterrents may even force evolutionary pressure (e.g., ontogenetic shifts, hearing shifts, hair loss, behavioral responses). Furthermore, most research on underwater acoustic noise or impacts on biota tends to be coastal and ocean-based studies (Murchy et al. 2019; Popper and Hawkins 2019; Popper et al. 2020), resulting in inferences on impacts for freshwater systems. While this does not detract from the quality of the studies, it leaves a significant gap in understanding the impacts on riverine and shallow-water environments.

Hand in hand with biology-based assessments are needed within rivers and lakes that focus on understanding soundscapes, propagation of sound and particle motion through locks, commercial navigation waterways (i.e., areas dredged, or receiving greater than expected flushing flows), and riverine systems. Similarly, the production of sounds from underwater acoustic NSDs is also needed to estimate the impacts of the sound production and refine the sounds for the desired impact or targeted species. Ultimately, the considerations of more complex multi-deterrent deployments, efficacies, and impacts both near- and long-term are critical and should include large science groups of multi-disciplines from regional and global points of view.

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# Section IX

# Noise Effects: Fishes

# Investigating Behavioral Responses of Marine Fish to Anthropogenic Noise: Use of Acoustic Telemetry

55

Kate McQueen and Lise Doksæter Sivle

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**Abstract**

Fish commonly respond to anthropogenic noise through behavioral changes. There are limitations to replicating noise and observing behavioral responses in controlled lab or cage experiments, which make observing behaviors in natural environments a preferred option. Acoustic telemetry offers advantages in providing the ability to monitor behavior of wild, individual fish in their natural environments while exposed to realistic sound from anthropogenic sources. In recent years, acoustic telemetry has been used in several studies to address questions about the behavioral responses of marine fish to various types of anthropogenic noise. Despite several benefits to the use of acoustic telemetry, there are logistical and analytical challenges to consider, and for some research questions, combining field studies using acoustic telemetry with experiments in more controlled environments may be most appropriate.

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**Keywords**

Remote monitoring · Impact · Sound · Tagging · Field study

## Introduction

Anthropogenic noise is increasingly being recognized as an important pollutant of marine environments, with the potential to impact a wide variety of marine taxa in diverse ways (Williams et al. 2015; Carroll et al. 2017; Duarte et al. 2021). Sound can be produced by many human activities, including seismic surveys, pile driving, offshore wind farms, sonars, fishing vessels, and shipping (Duarte et al. 2021). These sounds contribute to the "acoustic scene" or "soundscape" of marine environments, which are composed of sounds of biotic and abiotic origin related to specific times and locations (Duarte et al. 2021).

Marine vertebrates detect and analyze the acoustic scene to gather important information about their environment (Popper and Hawkins 2019). Hearing is thus an important sense for fish, which use acoustic information for a variety of purposes (Popper and Hawkins 2019). For example, fish use sound to gather cues about their environment, to detect prey and predators, and to communicate with conspecifics (Popper and Hawkins 2019). Increased anthropogenic noise levels can interfere with the ability of fish to detect and respond to biologically relevant sounds, impacting this large, diverse group of marine organisms in several ways (Slabbekoorn et al. 2010; Popper and Hawkins 2019).

Reported effects of anthropogenic noise on fish have ranged from physical injury caused by loud, impulsive sounds at close ranges (McCauley et al. 2003; Popper et al. 2005; Halvorsen et al. 2012; Smith et al. 2022) to no apparent response to noisy human activities such as seismic surveys (Peña et al. 2013; Meekan et al. 2021). Fish are often reported to exhibit behavioral responses to anthropogenic noise (Carroll et al. 2017), for example, changes in swimming depth and activity levels in response to seismic surveys (Davidsen et al. 2019; van der Knaap et al. 2021), changes in small-scale swimming behaviors in response to pile driving (van der Knaap et al. 2022), and changes in schooling behavior and swimming depth in response to playback of impulsive sounds (Hawkins et al. 2014).

Studying behavioral responses of fish is relevant for understanding large-scale impacts of noisy human activities at sea, as behavioral responses are likely to be the most widespread and may affect fitness-related behaviors such as feeding, reproduction, or migration (Popper and Hawkins 2019). Therefore, although not resulting directly in injury or death, behavioral changes caused by anthropogenic disturbance can lead to population level consequences (Pirotta et al. 2018), though this has so far been difficult to quantify for sound disturbance of fish due to lack of necessary data (Slabbekoorn et al. 2019).

Detecting realistic behavioral responses of marine fish to anthropogenic noise is not straightforward. It is perhaps easiest to observe fish behavior in a laboratory setting, but problems associated with exposing fish to realistic sound in tanks are well documented (Duncan et al. 2016; Popper and Hawkins 2019). Net pen or cage

experiments, where fish are contained within, for example, a floating net pen in the sea, largely avoid the problems associated with achieving realistic sound exposures that hamper tank experiments. However, fish confined to a net pen have limited scope for displaying realistic behavioral responses, such as dispersal away from the area (Hawkins et al. 2020; Slabbekoorn 2016). Additionally, fish reared in captivity can respond differently to stimuli than wild fish (e.g., Meager et al. 2011). Monitoring free-ranging, wild fish exposed to real sound sources provides the highest chance of observing realistic behavioral reactions. However, observing and recording behaviors of free-ranging fish within marine environments can be challenging.

In this mini-review, various methods for observing responses of free-ranging fish to anthropogenic noise are presented. Acoustic telemetry is introduced as a relatively novel method for observing behavior of free-ranging fish that avoids some of the limitations of the other methods. An overview of published studies that have applied acoustic telemetry is presented, and advantages and limitations of this method are discussed.

## Methods for Observing Behavioral Responses of Free-Ranging Fish

Available methods for detecting behavioral responses of free-ranging fish to anthropogenic noise include video observation, direct visual census using divers, active acoustic methods to detect fish and schools, analysis of catch data, and tagging. Each method has some strengths and limitations.

### Visual Observation

Visual observation of fish responses to anthropogenic noise can be achieved through video recordings (Wardle et al. 2001; Paxton et al. 2017; Meekan et al. 2021) or direct observations made by divers (Miller and Cripps 2013). Visual observation can be used to directly observe instantaneous individual behavioral reactions to noise (Wardle et al. 2001) or changes in abundances and species distributions in an area over longer timescales (Miller and Cripps 2013; Paxton et al. 2017; Meekan et al. 2021). However, visibility is limited in most marine environments, and use of additional lighting to improve visibility, or use of bait to attract fish to the observable range, will likely influence natural behaviors. Direct visual observation by divers requires good visibility and favorable conditions for diving, so it is unlikely to be suitable in all environments. For safety reasons, dive surveys cannot be conducted during very loud activities like seismic surveys, so this method is limited to before-after-type surveys.

### Active Acoustic Methods

Active acoustics can be used to monitor fish abundance and swimming behavior in real time. Fishery sonars that can be tilted 180 degrees can be used to follow highly mobile fish schools over time to provide information about swimming depth, speed,

and direction and have been used to study responses of herring schools to seismic surveys (Peña et al. 2013). Fishery sonars are appropriate for highly mobile, pelagic, schooling fish.

Downward-looking echosounders are used in fishery acoustics and provide information about abundance and depth distribution of fish and, through additional sampling, information about species composition. This type of sampling has been used to study the before-during-after composition of fish in response to seismic surveys (Engås et al. 1996; Slotte et al. 2004; Løkkeborg et al. 2012). These methods generally require a vessel to follow the fish schools or to collect data along transects.

It is also possible to detect changes in fish schooling behavior without a vessel, for example, using moored upward looking echosounders (De Robertis and Wilson 2011; Kok et al. 2021); however a limitation with this approach is that species identification may not be possible (Kok et al. 2021). Echosounders are most appropriate for measuring abundance and biomass, as well as depth distribution, though tracking individual behaviors with echosounders can also be achieved (Handegard 2007).

## Catch Data Analysis

Analysis of catch data can allow detection of changes in abundance or catchability in relation to anthropogenic noise (Skalski et al. 1992; Engås et al. 1996; Løkkeborg et al. 2012; Bruce et al. 2018). Catch sampling studies can be relatively simple to conduct, as they do not require additional equipment (such as tags, video equipment, etc.) beyond what is already available on fishing or fishery research vessels, and samples can be taken opportunistically around real sound sources, such as seismic surveys.

Desktop studies using fishery data may also be able to detect changes in catch related to anthropogenic sound sources such as seismic surveys (e.g., Soldal and Løkkeborg 1993; Vold et al. 2012; Bruce et al. 2018). However, such methods need to consider possible confounding factors such as market forces, catchability, environmental variation, discarding, and variation in time of landings in relation to the disturbance (Vold et al. 2012; Bruce et al. 2018). It is also important to consider that observed changes in catch may be gear specific (e.g., Løkkeborg et al. 2012). Importantly, analysis of catch data can only provide indirect evidence of behavioral responses. It may be difficult in practice to directly link changes in catch rates to the physical, behavioral, or physiological responses that drive them (Bruce et al. 2018).

## Tagging Methods

Various methods of tagging are available and commonly used for studying fish behavior and population dynamics. Tagging methods include marking fish with unique identification tags (e.g., external T-bar tags (conventional tagging)) or

tagging fish with archival or data storage tags (DSTs) that record information about the fish's environment and/or behavior during its time at liberty.

Conventional tagging provides limited information on fish behavior, providing information only on release and recapture position, with no information on the fish's location and behavior during time at liberty. There is at least one example of an attempt to detect fish behavior in relation to a seismic survey using conventional tagging, with this method able to indicate that there was no obvious dispersal of bass from the study area in the year of the seismic survey compared to previous years (Pickett et al. 1994).

DSTs can provide detailed and high-resolution data on fish behavior and environmental variables. However, post hoc geo-positioning is required using recorded variables such as light levels or combinations of other factors such as depth and salinity. This can result in uncertainty in position estimates, which can make matching fish location to noisy human activities challenging, as evidenced by a study that tried to relate behavior of tagged tuna to seismic surveys (Evans et al. 2017).

## What Is Acoustic Telemetry?

Acoustic telemetry is an approach that can be used to monitor behavior of free-ranging fish in their natural environments, while avoiding many of the limitations of the methods described in section "Methods for Observing Behavioural Responses of Free-Ranging Fish." Acoustic telemetry studies involve the use of electronic tags that transmit a unique identification code through an ultrasonic signal. These tags are attached to individual fish, usually through surgical implantation into the body cavity. Study fish therefore need to be large enough to carry the tag. Tags may also be equipped with additional sensors, which transmit information about variables such as depth, acceleration, and temperature.

Tagging of fish is combined with deployment of receivers in the area of interest. These receivers can record time-stamped detection information from the fish tag when the animal is within range, thus collecting individual-specific information on fish presence in the study area. Receivers can be deployed in dense arrays that allow precise positioning of fish within the study area or can be spread out to cover a larger area to gain coarser-scale movement information.

Acoustic telemetry has been used to study fish movement and behavior since at least the 1990s (Abecasis et al. 2018). The use of acoustic telemetry in marine fish research has been greatly increasing in recent years, partly due to technological advancements leading to reduced tag size and increased battery life and memory size (Hussey et al. 2015). A recent review promotes acoustic telemetry as a suitable method to gather data on a range of topics relating to sustainable use of the oceans, including monitoring the impacts of various types of pollution (Alós et al. 2022). This highlights the potential for acoustic telemetry to become more widely used in research into the impacts of anthropogenic noise on fish.

## Using Acoustic Telemetry to Investigate Behavioral Responses to Noise

Acoustic telemetry is increasingly used to study behavioral responses of marine fish to anthropogenic noise, with an overview of published studies on this topic provided in Table 1. All but one of these studies are from the past 6 years, highlighting the very recent application of this technology to anthropogenic noise-related questions. Most of these studies have opportunistically investigated the effects of noisy human activities that were occurring in the environment, such as seismic surveys (Bruce et al. 2018; Meekan et al. 2021), pile driving (Iafrate et al. 2016; van der Knaap et al. 2022), and shipping (Ivanova et al. 2020; Rider et al. 2021). Others have used experimental sound treatments that nonetheless used realistic sound sources (Wardle et al. 2001; van der Knaap et al. 2021; McQueen et al. 2022, 2023).

The species of fish tagged in the studies in Table 1 have been limited to a handful of families, including gadoids (Wardle et al. 2001; Ivanova et al. 2020; van der Knaap et al. 2021, 2022; McQueen et al. 2022 2023), snappers (Iafrate et al. 2016; Meekan et al. 2021), porgies (Iafrate et al. 2016), and flatheads (Bruce et al. 2018), with two studies conducted on various shark species (Bruce et al. 2018; Rider et al. 2021) (full species names are provided in Table 1). Notably, these examples are mostly relatively hardy, large-, and round-bodied fish, which are therefore robust to the tagging procedure.

Acoustic telemetry studies have so far been used to infer a variety of large- and fine-scale behavioral responses to anthropogenic noise. The studies listed in Table 1 have been used to investigate changes in residency at an area of interest (Iafrate et al. 2016; Ivanova et al. 2020; Rider et al. 2021; McQueen et al. 2022; van der Knaap et al. 2022), home ranges (Ivanova et al. 2020; Meekan et al. 2021; McQueen et al. 2023), and various swimming behaviors such as step lengths, swimming depths, and acceleration (Wardle et al. 2001; Bruce et al. 2018; Ivanova et al. 2020; van der Knaap et al. 2021, 2022; McQueen et al. 2023) and behavioral states inferred from combinations of these behavioral metrics (Ivanova et al. 2020; van der Knaap et al. 2021). This level of detail about behaviors in situ, at an individual level and tracked for relatively long periods of time, would be difficult to achieve with the methods described in section "Methods for Observing Behavioural Responses of Free-Ranging Fish."

## Advantages and Limitations of Acoustic Telemetry

Acoustic telemetry is well suited to monitoring residency of fish at a specific site and has therefore proven particularly useful for assessing whether anthropogenic noise causes abandonment of habitats, feeding or spawning sites (Iafrate et al. 2016; Bruce et al. 2018; Ivanova et al. 2020; Meekan et al. 2021; Rider et al. 2021; van der Knaap et al. 2021, 2022; McQueen et al. 2022). Finer-scale changes in behavioral metrics

**Table 1** Overview of published studies investigating behavioral responses of marine fish to anthropogenic noise, using acoustic telemetry

| Species | Noise source | Behaviors studied | Responses observed | Combined with other method | References |
|---|---|---|---|---|---|
| Arctic cod (*Boreogadus saida*) | Ships | Home ranges, behavioral states (inferred from variables relating to step length, swimming speed, and bearing) | Home range displacement and change in swimming activity | No | Ivanova et al. (2020) |
| Bull shark (*Carcharhinus leucas*) | Boats | Residency | No change in residency related to boat density | No | Rider et al. (2021) |
| Great hammerhead shark (*Sphyrna mokarran*) | Boats | Residency | No change in residency related to boat density | No | Rider et al. (2021) |
| Nurse shark (*Ginglymostoma cirratum*) | Boats | Residency | No change in residency related to boat density. Hourly presence decreased with increasing boat traffic | No | Rider et al. (2021) |
| Atlantic cod (*Gadus morhua*) | Pile driving | Residency, spatial behavior within study area (step lengths, distance to scour bed, relative distance to piling event) | Moved closer to turbine bases, movement away from piling locations | No | van der Knaap et al. (2022) |
| Gray snapper (*Lutjanus griseus*) | Pile driving | Residency | Decrease in residency, potential displacement from area | No | Iafrate et al. (2016) |

(continued)

**Table 1** (continued)

| Species | Noise source | Behaviors studied | Responses observed | Combined with other method | References |
|---|---|---|---|---|---|
| Sheepshead (*Archosargus probatocephalus*) | Pile driving | Residency | No decrease in residency | No | Iafrate et al. (2016) |
| Atlantic cod (*Gadus morhua*) | Seismic survey | Residency, behavioral states inferred from step lengths and acceleration | Decrease in activity, change in daily patterns of behavioral states, displacement after survey | No | van der Knaap et al. (2021) |
| Gummy shark (*Mustelus antarcticus*) | Seismic survey | Step lengths, acceleration | Not enough data | Analysis of CPUE from commercial fisheries | Bruce et al. (2018) |
| Swell shark (*Cephaloscyllium laticeps*) | Seismic survey | Step lengths, acceleration | Not enough data | Analysis of CPUE from commercial fisheries | Bruce et al. (2018) |
| Tiger flathead (*Neoplatycephalus richardsoni*) | Seismic survey | Step lengths, acceleration | Some increase in movement, no change in residency | Analysis of CPUE from commercial fisheries | Bruce et al. (2018) |
| *Lutjanus sebae* | Seismic survey | Residency, home ranges | High residency, no long-term displacement | BRUV surveys | Meekan et al. (2021) |
| Pollack (*Pollachius pollachius*) | Air gun | Daily behavior patterns, movement | Interruption of daily behavior patterns, movement away from air gun | Video observations | Wardle et al. (2001) |
| Atlantic cod (*Gadus morhua*) | Air gun | Residency, home ranges, swimming depth, acceleration, step lengths | No change in residency, slight increase in swimming depth | No | McQueen et al., (2022, 2023) |

*CPUE*, catch per unit effort; *BRUV*, baited remote underwater video

detected on telemetry receivers have additionally been used to infer changes in movement behaviors and behavioral states within the study site (Ivanova et al. 2020; van der Knaap et al. 2021; McQueen et al. 2023).

The advantages of using acoustic telemetry over the methods described in section "Methods for Observing Behavioural Responses of Free-Ranging Fish" include the possibility to gather longitudinal behavioral data at an individual level, which would be difficult using active acoustic, visual observation, and catch sampling methods. Additionally, data can be collected over longer periods than would be possible using visual observation methods. The behaviors can be spatially resolved, which is a limitation of conventional or DST tags. Despite these advantages, there are also a number of limitations of this method that should be considered.

Acoustic telemetry is probably not an appropriate method for studying all fish species in all environments. The most successful of the studies reviewed in this chapter targeted species that showed high site fidelity to the study region. In studies that tagged more than one species, it was easier to draw conclusions about the responses of fish showing high site fidelity to the telemetered area, thus providing large sample sizes of data (Iafrate et al. 2016; Bruce et al. 2018; Rider et al. 2021). The situation reported in Bruce et al. (2018), where every tagged shark left the array before the seismic survey began, returning only occasionally and thus providing only a handful of data points, exemplifies the problems that can arise from tagging a species whose spatial movement ecology substantially exceeds the telemetry array.

Most studies listed in Table 1 were conducted in some kind of semi-enclosed environment such as bays (Ivanova et al. 2020; Rider et al. 2021; McQueen et al. 2022, 2023) or harbors (Iafrate et al. 2016) or around structures such as wind turbines (van der Knaap et al. 2021, 2022) which simplifies the designing of receiver arrays and may encourage retention of tagged fish within the system. Conducting studies offshore, with a lack of physical boundaries, could be riskier and increase the possibility that tagged fish quickly leave the area (as in Bruce et al. 2018). However, this risk is species specific, as shown by the high detection rates of tiger flathead and *L. sebae* in the offshore telemetry grids of Bruce et al. (2018) and Meekan et al. (2021), respectively.

In general, a key aim of studies that investigate the effects of anthropogenic noise on fish behavior is often to assess whether fitness-related behaviors such as feeding, reproduction, or migration are impacted by anthropogenic noise (Popper and Hawkins 2019). Dispersal from key foraging or spawning habitats is relatively simple to detect using acoustic telemetry (e.g., van der Knaap et al. 2021; McQueen et al. 2022). Detecting changes in actual spawning or feeding behavior is more challenging using acoustic telemetry, as behaviors cannot be directly observed. Some studies have attempted to translate behavioral metric data, such as step lengths and acceleration, to specific behaviors such as foraging (Ivanova et al. 2020; van der Knaap et al. 2021). However, it is difficult to assess the reliability of these categorizations using only the telemetry data. Ground-truthing behavior categorization using data from captive fish may be a means to improve the reliability of this approach (e.g., Clarke et al. 2021).

Telemetry studies generally produce large amounts of high-resolution data that often require statistically complex and computationally expensive analysis. It is important to consider structure in the data, for example, most telemetry data are serially autocorrelated and detections from the same fish are non-independent.

Spatial correlation, measurement error and false absences are also likely (Whoriskey et al. 2019). Fortunately, there are statistical tools available for dealing with such data, and new methods are being developed actively (Lennox et al. 2017; Whoriskey et al. 2019).

The most appropriate statistical method for analyzing telemetry data depends on the research question, but options include generalized linear mixed effects models, mark-recapture models, network analysis, and survival analysis (Whoriskey et al. 2019). Hidden Markov models are also becoming increasingly popular for analyzing aquatic telemetry data. These stochastic time-series models are well suited to telemetry datasets and can be used to infer hidden behavioral states from animal tracking data (Leos-Barajas et al. 2016; Pohle et al. 2017; Bacheler et al. 2019). There are also many studies that use telemetry data to estimate fish home ranges (e.g., Meager et al. 2010; McMahan et al. 2013; Dean et al. 2014), which can be relevant for asking questions about changes in space use (e.g., Ivanova et al. 2020; McQueen et al. 2023). In general, using complementary methods to analyze the same telemetry data can increase confidence in the conclusions drawn (Bacheler et al. 2019).

## Some General Field Study-Related Challenges

Some challenges associated with studying free-ranging fish were exemplified through this review, though are not unique to acoustic telemetry studies. For example, it may be challenging to disentangle behavioral responses to anthropogenic noise from the many other variables that influence behavior in natural settings. Movement of fish is the outcome of a complex interplay of different factors, including individual-specific factors and external factors (Cooke et al. 2022). Many variables must be taken into account when analyzing field data, including environmental variables (e.g., temperature, salinity, time of day, sun angle, tide, current speed (van der Knaap et al. 2022; McQueen et al. 2023) and presence of predators like marine mammals (Ivanova et al. 2020). Individual variation, variation between sexes, and variation between study years may also be important to consider (McQueen et al. 2023). Fish responses may also vary depending on time of year and motivational state (e.g., Fernö et al. 1998). Accounting for the numerous variables influencing fish movement behavior is another reason why analyses of telemetry data tend to be necessarily complex.

### Replicates and Control Sites in Field Studies

A key advantage of studies that have used acoustic telemetry is that the fish were exposed to realistic sound sources, such as full-scale seismic surveys (Bruce et al. 2018; Meekan et al. 2021; van der Knaap et al. 2021). In several studies, the sources were opportunistic, for example, pile driving (Iafrate et al. 2016; van der Knaap et al. 2022) or shipping and boat traffic (Ivanova et al. 2020; Rider et al. 2021). However,

this benefit may also make it difficult to conduct replicates or select appropriate control sites. This is an issue related to study design rather than acoustic telemetry and has also been highlighted in studies using other methods to monitor behavioral changes (Løkkeborg et al. 2012; Bruce et al. 2018). Given that fish behavior can vary according to a large variety of factors, as discussed previously, a lack of replicates and controls can limit the conclusions drawn from such studies.

In the studies included in this mini-review, experiments using large-scale exposures like seismic surveys did not include replicates (Bruce et al. 2018; van der Knaap et al. 2021). Conversely, studies focusing on shipping noise or pile driving were able to include replicates by analyzing results over several events (Ivanova et al. 2020; van der Knaap et al. 2022). Studies that used an experimental treatment, rather than an opportunistic exposure, were able to incorporate replicates into study design (McQueen et al. 2022, 2023).

Control sites were not included in some of the studies included in this review (Ivanova et al. 2020; van der Knaap et al. 2021, 2022). It can be extremely challenging to find suitable control sites in spatially and temporally dynamic marine environments, as control sites should be far enough away from the test site that they are not affected by the disturbance, yet as similar as possible in terms of habitat and species composition (Methratta 2020). In some studies, nearby, suitable control sites unaffected by the sound exposure were identified and exploited (Iafrate et al. 2016; Bruce et al. 2018; Meekan et al. 2021; McQueen et al. 2022). If it is not possible to establish suitable control sites, a gradient design may be an alternative, as has been recommended for studying effects of offshore wind farms on fish (Methratta 2021).

Overall, this indicates that acoustic telemetry does not avoid all limitations associated with conducting behavioral response studies in the field. In many situations it remains expensive and logistically challenging to incorporate replicates and control sites into large-scale field studies using realistic sound sources. Careful study design is required to ensure effects of anthropogenic noise can be disentangled from natural variation and confounding variables.

## Conclusions

Acoustic telemetry provides an opportunity to gather behavioral data at relatively long temporal scales and large spatial scales, with high temporal and spatial resolution. These data can be used to assess behavioral responses of wild fish to exposure to realistic sources of anthropogenic sound.

As with the other methods included in this mini-review, there are some limitations to the use of acoustic telemetry, for example, due to the inability to directly observe behaviors. Therefore, depending on the research question, it may be necessary to combine lab and field studies, as also promoted by Slabbekoorn (2016), or to combine observation methodologies, as in Wardle et al. (2001), Bruce et al. (2018), and Meekan et al. (2021).

Sample size can also be limited in acoustic telemetry studies, due to cost and practical limitations of tagging many fish, the possibility that tagged fish leave the

study area, and due to some methods of signal transmission resulting in signal interference if too many tags are transmitting within a limited area. Combining acoustic telemetry with other methods like catch sampling or visual observation can allow for increased sample sizes and sampling of additional species, providing at least some information on species and individuals that were not tagged (Wardle et al. 2001; Bruce et al. 2018; Meekan et al. 2021).

Not all species or life stages of fish are suitable for tagging, for example, due to size or fragility. However, as tag sizes continue to decrease through technological advancements, the variety of fish that can be tagged increases (Lennox et al. 2017). Extremely mobile species may also be unsuitable for behavioral response studies using acoustic telemetry, unless nationwide or international tracking networks are established (Abecasis et al. 2018; Lennox et al. 2022).

Overall, acoustic telemetry studies can provide data on wild fish behavior in unprecedented detail. Carefully designed acoustic telemetry studies can contribute to our understanding of the behavioral responses of free-ranging, marine fish to anthropogenic noise.

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# Effect of Broadband Boat Motor Sound on the Schooling Behavior of Invasive Bigheaded Carp

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Cassandra A. Kramer, Jackie Culotta, Marie L. Ervin, Allen F. Mensinger, and Brooke J. Vetter

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**Abstract**

Introduction of silver (*Hypophthalmichthys molitrix*) and bighead (*Hypophthalmichthys nobilis*) carp to the Mississippi River drainage has created ecological and economic distress. These species, collectively known as bigheaded carp, are closely related, and silver carp are well known for their jumping behavior in response to motorboats. Therefore, the impact of boat motor sound on bigheaded carp behavior has important implications for controlling range expansion. We examined the effect of broadband boat motor sound on the schooling behavior of bigheaded carp. In a two-choice shuttle tank, broadband sound (0.06–10 kHz; 140 dB re: 1 μPa) was played via an underwater speaker,

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and bigheaded carp responses were recorded using overhead cameras. The schooling behavior of silver and bighead carp during broadband sound playback was compared to a no-sound period. Broadband sound playback elicited negative phonotaxis and increased swim speed in all trials. However, the carp continued to school ($n = 5$ individuals swimming in the same direction and one body length apart) the majority (57.2% $\pm$ 11.2%) of time during the broadband sound playback periods. These results indicate that schooling could contribute to the consistent negative phonotaxis observed in previous studies reporting the efficacy of broadband sound in deterring bigheaded carp swimming.

**Keywords**

Negative phonotaxis · Acoustic deterrent · Broadband sound · Invasive carp · Behavior · Schooling

## Introduction

Silver (*Hypophthalmichthys molitrix*) and bighead (*H. nobilis*) carp, collectively known as bigheaded carp, pose ecological and economic threat to North American waterways. Since accidental introduction in the 1970s, both species have migrated north, establishing populations throughout much of the Mississippi River Basin. These invasive fish compete with native species for resources and physically damage watercraft and boaters (Kolar et al. 2007). Continued upstream migration could endanger the Laurentian Great Lakes, and the ecological stability of the largest freshwater system in North America could be compromised (Cudmore et al. 2012).

Bigheaded carp are otophysans and possess morphological specializations (Weberian ossicles) that connect the swim bladder to the inner ear and allow for detection of sound pressure. On the proposed continuum of hearing abilities in fishes, otophysans are well adapted for detection of sound pressure (Popper et al. 2021). Physiological studies (using auditory evoked potentials) report that bigheaded carp can detect frequencies up to 5 kHz (Vetter et al. 2018), while native non-otophysans are limited to lower frequencies (e.g., lake sturgeon *Acipenser fulvescens* and paddle fish *Polyodon spathula*; Lovell et al. 2005).

Acoustic barriers have been recommended to deter carp upstream migration (Putland and Mensinger 2019). When exposed to broadband acoustic stimuli (0.06–10 kHz), wild silver carp jump in large numbers (Vetter et al. 2017a), and captive bigheaded carp exhibit consistent negative phonotaxis (Murchy et al. 2017; Vetter et al. 2015, 2017b). However, schooling behavior in response to acoustic signals has not been directly evaluated in these species. This project examined the impact of broadband sound playback on the schooling behaviors of bigheaded carp in a controlled setting.

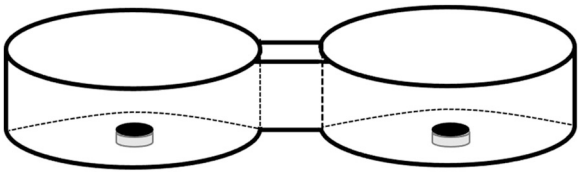


**Fig. 1 Two-choice shuttle tank.** Experiments were performed in a two-choice shuttle tank with a single speaker placed in the center of each side. Only one speaker was active at a time. An overhead camera mounted above the tank recorded fish behavior during pre-sound and broadband sound playback

## Materials and Methods

Experiments were conducted at the Minnesota Aquatic Invasive Species Research Center (MAISRC) at the University of Minnesota in St. Paul, Minnesota. All procedures were approved by the University of Minnesota Institutional Animal Care and Use Committee (Protocol: 2103-38930A).

### Animal Husbandry

Juvenile silver ($n = 15$ individuals; SL: 61.3 ± 1.3 mm SE) and bighead ($n = 15$ individuals; SL: 75.0 ± 1.5 mm) carp were maintained in 500 L circular holding tanks and fed a mixture containing equal parts *Spirulina* algae and dry feed (Starter Crumble, Skretting USA, Tooele, UT) once a day. Food was withheld from fish for 24 h prior to the testing period. Trials were conducted between 20 and 24 July 2021.

### Experimental Design

Experiments were conducted in a two-choice shuttle tank (Loligo Systems, Viborg, Denmark). The tank consisted of two circular chambers (50 cm diameter, 25 cm depth) connected by a central channel (10 cm length, 7 cm width, 25 cm depth), allowing fish to access both chambers (Fig. 1). Water depth was maintained at 20 cm. A single underwater Bluetooth speaker (JBL Clip 4, Harman International Industries, Inc., Stamford, CT) was placed in the center of each of the chambers. The speakers were situated in a PVC ring 5 cm from the bottom of the tank. An overhead camera (Model UI-1640SE-C-GL; IDS Imaging Development Systems, Inc.; Stoneham, MA) 2 m above the tank provided a full view of the tank to record fish position and behavior. A black curtain was placed between the experimental tank and the researcher to prevent interference with fish behavior.

## Sound Stimulus

The sound was broadcast through one of the two underwater speakers placed centrally within the tank chambers, selected at random. The inactive speaker remained in the other chamber as a control to keep the visual environment consistent. The acoustic stimulus consisted of a broadband sound (0.06–10 kHz; 140 dB re: 1 μPa) recorded underwater in the Illinois River from an outboard boat motor (100 Hp Honda four-stroke; Vetter et al. 2015). Sound was played from a laptop computer (VLC Media Player, Lenovo 80MK).

## Sound Mapping

Sound pressure levels (SPL) were measured throughout the two-choice shuttle tank using a hydrophone (HTI-96-Min, High Tech, Inc., Long Beach, MS) plugged into a PowerLab (Model 26T, AD Instruments, Sydney, Australia) data acquisition system, and the SPL was determined from the root mean square (RMS) voltage calculated in LabChart8 (AD Instruments). The hydrophone was placed at points distributed evenly throughout the tank. The tank was mapped under ambient conditions and during broadband sound playback. The entire mapping process was performed twice: with a single active speaker in each side of the tank. During each mapping process, the tank was mapped at two depths: 5 cm above the tank bottom and 5 cm below the water surface. A total of eight mapping locations were selected throughout the tank. The ambient sound level was 85–100 dB re: 1 μPa throughout the tank. During playback of the broadband sound stimulus, the maximum sound pressure level was 140 dB re: 1 μPa at the source (hydrophone placed 2 cm directly above each speaker). This SPL was chosen as it was at least 40 dB over the ambient sound (to act as an acoustic deterrent) but still allowed for a gradient of sound pressure levels in the tank such that the SPL was 20 dB lower in the opposite side (SPL was 120 dB re: 1 μPa directly above the inactive speaker in the chamber opposite to the active speaker).

## Behavioral Trials

Behavioral phonotaxis experiments were conducted for each school ($n = 3$ for both species). Before the beginning of the trial, either the right or left speaker was randomly selected for sound playback. A school ($n = 5$ individuals) of bighead or silver carp was placed in the experimental tank and allowed to acclimate for 30 min prior to the testing period. The last 5 min of acclimation was recorded via the overhead camera to serve as a control for fish movement and schooling. This period will be hereafter referred to as the "pre-sound period." After acclimation, testing began 5 s after the majority of the school (at least three fish) entered the chamber with the speaker selected for sound playback. The broadband sound was played for 5 min, and the school was allowed to swim freely throughout the shuttle during the

playback period. Fish behavior was recorded via the overhead camera (18 fps). After the 5-min playback period, the fish were removed and placed into holding tanks.

## Data Analysis

*Schooling Behavior:* Video data were analyzed for group size and behavior throughout the 5-min pre-sound and broadband sound playback periods. Schooling was characterized as all five individuals swimming with the same directionality while being within one body length from another individual. Smaller grouping behavior (e.g., majority groups of three or four fish swimming in the same direction within one body length) and no grouping behavior (e.g., fish swimming in pairs or individually) were also evaluated. Both pre-sound and broadband sound playback videos were evaluated for the total time the fish swam in schools of five fish or smaller groups of three or four fish (or a "majority" group) or demonstrated no group swimming. These values are reported as the percent of the total 5-min period spent in each condition. The percentage data were arcsine transformed for statistical analysis. To compare schooling behavior during the pre-sound swimming, a one-way ANOVA was performed on these residuals. A one-way ANOVA was also performed on the residuals for the broadband playback to compare schooling during broadband sound playback period.

*Behavioral Response to Broadband Sound:* During all broadband sound playback periods, the fish always swam in groups of three or more (or a "majority" group) and left the chamber during the first 10 s of sound playback. Therefore, the video data were further analyzed to assess the swimming velocity (m/s) of the majority groups (three or more) to the broadband sound stimulus using Noldus EthoVision software (EthoVision XT Version 11.5.1026; Noldus; Wageningen, the Netherlands). As a control, velocity also was evaluated during the pre-sound period. All statistical analyses were performed using SigmaPlot for Windows (version 12.5).

## Results

*Behavioral Response to Broadband Sound:* During the pre-sound period, the fish explored both sides of the shuttle tank while swimming at a consistent pace. During broadband sound playback however, a majority (a group of at least three individuals) exited the chamber with the active speaker and often remained in the other side (Fig. 2). There was no significant difference between the two species of carp for average swimming velocity during the first 10 s of the pre-sound or broadband sound playback periods (t-test, $p > 0.10$). Thus, these data were pooled to evaluate bigheaded carp response behavior ($n = 6$ schools). During broadband sound playback, a majority swam away from active speaker and increased swimming velocity (playback, 8.86 $\pm$ 0.46 cm/s; pre-sound, 6.86 $\pm$ 0.41 cm/s; mean $\pm$ SE; t-test, $p > 0.05$; Fig. 3).

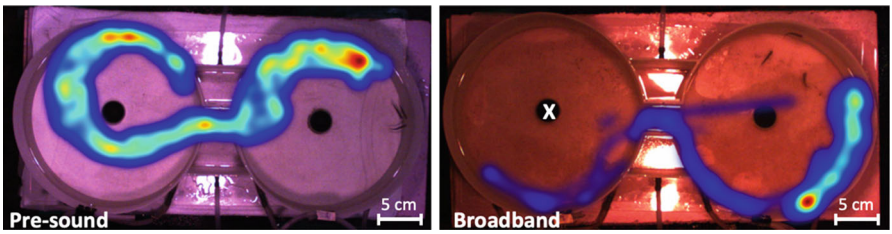



**Fig. 2 Negative phonotaxis in bigheaded carp.** Representative heat maps showing the pattern of swimming for a silver carp school during the 5-min pre-sound versus the 5-min broadband sound playback. The active speaker in the broadband sound playback period is denoted by an X. The colors represent the location of the school during the trial, with red indicating where the school spent the most time and blue representing minimal time in that location. During the pre-sound period, the school swam freely between the two chambers, while the school exited the chamber with the active speaker during the first 10 s of playback and spent the rest of the broadband period in the opposite chamber

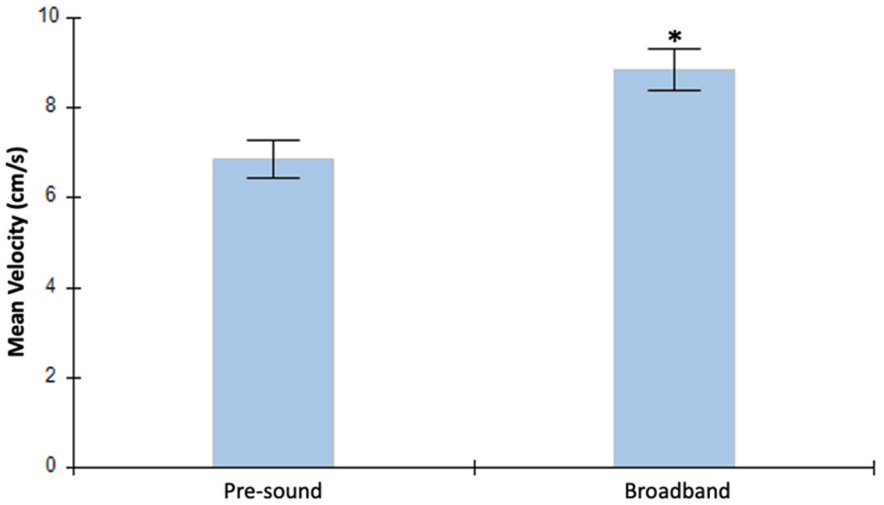



**Fig. 3 Mean swimming velocity during broadband sound playback vs. pre-sound period.** Mean velocity increased during the first 10 s of broadband sound playback compared with 10 s of pre-sound (control) swimming. Mean ± SE; asterisk indicates significant difference (t-test; $p < 0.05$)

*Schooling Behavior:* During both the pre-sound and the broadband sound periods, the carp continued to school the majority of the time (Fig. 4). Fish schooled in groups of five 48.0% ± 16.1% (mean ± SE) of the 5-min pre-sound and 57.2% ± 11.2% of the 5-min broadband sound playback periods. For the pre-sound period, smaller group swimming behavior was observed 23.2% ± 8.2% (group of four) and 18.2% ± 5.6% (group of three) of the time or not at all (10.6% ± 6.2% of the time). During the broadband sound playback, fish spent less

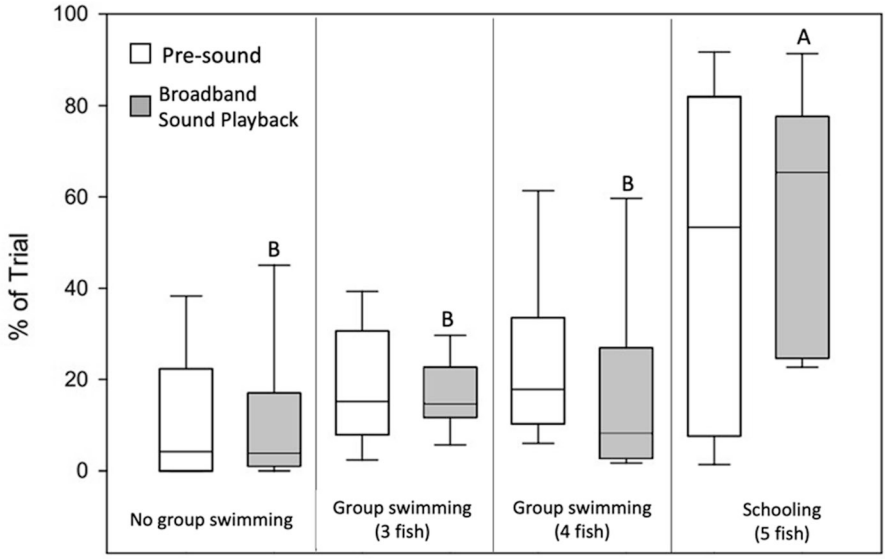



**Fig. 4 Group swimming behavior in bigheaded carp during the pre-sound and broadband sound playback periods.** During both the pre-sound and broadband sound playback periods, fish were most often in a school of five (though this relationship was not significant during the pre-sound period, ANOVA: $p > 0.05$). Letters above the gray bars (broadband sound playback) indicate significantly different groups (ANOVA: $p < 0.05$). For the statistical tests, the percentage data were arcsine transformed

time schooling in smaller groups (group of four, 16.1 ± 9.0%; group of three, 16.4 ± 3.3%) but spent a similar 10.3% ± 7.0% time not in a group.

The arcsine-transformed data for the pre-sound period were normally distributed (Shapiro-Wilk $p = 0.663$), but the variances were unequal ($p = 0.026$), so a one-way ANOVA on ranks was performed on the residuals and found no significant differences between the median values ($H_3 = 5.1$; $p = 0.162$; Fig. 4).

The arcsine-transformed data for the broadband sound playback period were normally distributed (Shapiro-Wilk $p = 0.139$) with equal variance ($p = 0.602$), and a one-way ANOVA found that fish were significantly more likely to school (group of five) during broadband sound playback than swim in the smaller groups (three or four) or exhibit no group swimming behavior ($F_{3,20} = 7.22$, $p = 0.002$; Fig. 4).

## Discussion and Conclusions

Similar to previous studies (Murchy et al. 2017; Vetter et al. 2015, 2017b), bigheaded carp avoided broadband sound. Furthermore, the present results show that carp schooling behavior is more likely to occur when fish are responding to a broadband sound stimulus. During sound playback, carp consistently remained in

schools of five. These results suggest that schooling could contribute to the consistent negative phonotaxis responses observed in previous studies. As individual fish cluster together, their negative phonotaxis responses result in strong schooling behavior. From a management standpoint, these results may indicate that deterrent strategies involving acoustic stimuli could be successful if they consider a school of fish as a single entity instead of considering the behaviors of individual fish; however, additional research is needed to evaluate schooling behavior under more naturalistic conditions (e.g., a larger tank or a field study). Further, the finding that fish prefer to school even under conditions of an active deterrent has interesting management implications. If a school is repelled after encountering a deterrent, it is possible that this could promote deterrence by other fish in the area. Finding deterrent methods that strengthen these negative group effects could help to maximize the number of fish affected by a single deterrent, ideally increasing the efficacy of the deterrent.

The sound mapping indicated that there was a gradient of sound pressure levels from ~40 dB above ambient at the active speaker to ~20 dB above ambient levels in the center of the opposite side of the tank, thus providing a "quieter" retreat for the bigheaded carp schools during playback periods. However, this study was done in a relatively small tank which poses a challenging acoustical environment, especially regarding the particle acceleration component of sound (Gray et al. 2016; Rogers et al. 2016). Particle acceleration was not evaluated and could have impacted the fish's behavioral responses. It is important that future research characterize particle acceleration, especially the acoustic impedance of the tank (Popper and Fay 2011).

Acoustic barriers have been evaluated to prevent the spread of invasive fish species (Putland and Mensinger 2019), and this study suggests that schooling could contribute to the negative phonotaxis behavior observed in species such as bigheaded carp. Future research includes evaluating schooling behavior in a more naturalistic setting and determining if a subset of individuals can influence the swimming behavior of an entire school.

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# Temporary Threshold Shift as a Measure of Anthropogenic Sound Effect on Fishes

57

Michael E. Smith and Arthur N. Popper

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**Abstract**

Vertebrates exposed to intense or long-duration loud sounds often show a temporary loss of hearing (temporary threshold shift, TTS). TTS has been examined in a number of fish species after exposure to white noise, pure tones, and impulsive sources, leading to the concern that TTS has the potential to impact fitness and survival of fishes exposed to anthropogenic sound. This chapter focuses on whether TTS is of real concern with regard to developing

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criteria for sound exposure for fishes. It starts with a review of the literature on TTS and ear damage as a result of exposure to various sounds and shows that the data are highly variable and that most of the significant TTS occurs in species that are defined as having specializations to enhance hearing sensitivity. The chapter then argues that TTS is not likely for free-swimming fishes since they can leave the areas of the most intense sounds and that most fish in a population exposed to such sounds will receive levels that are well below those that could produce TTS. At the same time, since data are limited, it is proposed that further studies using approaches that allow comparison between species and sounds be conducted.

**Keywords**

Noise · Hearing loss · Sensitivity · Impulsive sound · Continuous sound

## Introduction

In developing criteria for potential effects of anthropogenic sound on marine animals, a frequently cited issue is whether the sound has the potential to cause temporary loss of hearing, something referred to as temporary threshold shift (TTS). As discussed below, TTS is a change in hearing sensitivity as a result of temporary damage to the sensory hair cells of the inner ear and/or their innervating nerves as a result of exposure to intense or long-term sounds (e.g., Finneran 2016; Liberman 2016; Ryan et al. 2016). While the greater consideration has been for TTS in marine mammals (e.g., Finneran 2016; NMFS 2018; Thomsen et al. 2021), TTS has also been considered in developing interim criteria for effects of anthropogenic sound in fishes (Popper et al. 2014).

Popper et al. (2014) defined TTS as a change in hearing threshold of at least 6 dB and proposed that exposure to 186 dB re 1 $\mu Pa^2 \cdot s$ cumulative SEL (sound exposure level) would potentially result in TTS in all fishes for exposure to seismic air guns and pile-driving, while a level of 193 dB re 1 $\mu Pa^2 \cdot s$ cumulative SEL would be likely for low-frequency sonars. The likelihood of TTS for lower intensity continuous signals such as shipping was not thought to be great, and no criteria were proposed.

The purpose of this chapter is twofold. First it discusses what is known about TTS in different species exposed to a variety of sound sources. This includes consideration of the results from TTS studies, as well as the likely physiological mechanisms that result in TTS. Second, based on these data, the chapter considers the likelihood of TTS occurring in wild fishes and whether TTS is even an issue of any importance in developing criteria to protect fishes from anthropogenic sounds. Finally, since TTS data are very limited for fishes, the chapter

proposes additional areas of study that will help confirm if TTS is or is not an issue for most fishes.

## Anthropogenic Sound and Fish Hearing

### Potential Effects of Anthropogenic Sound

Within the last two decades, there has been a significant increase in research examining how sound from anthropogenic sources may potentially affect marine organisms. These sound sources include, among other things, powered vessels, pile-driving for marine construction, high-intensity low-frequency sonars, and geophysical research and energy exploration and production using seismic air guns. While the preponderance of earlier work focused on marine mammals, there is a substantial, and growing, interest on potential effects of anthropogenic sound on fishes and aquatic invertebrates (Scholik-Schlomer et al. 2023). Indeed, as fishes and invertebrates provide key components to aquatic ecosystems and both marine and human food chains, there is a need to fill the many research gaps in our understanding of how anthropogenic sound affects these organisms.

The potential negative effects of anthropogenic sound can be measured at various levels of impact on an animal's fitness. The most extreme effect is immediate death or physical injury ultimately leading to death (e.g., Halvorsen et al. 2012b; Dahl et al. 2020). Other potential effects include masking of important natural and biologically relevant sounds such as acoustic communication signals between individuals (Popper and Hawkins 2019) and behavioral changes such as displacement from the current impacted area which may be important feeding or spawning grounds (e.g., Engås et al. 1996; Carroll et al. 2017). Other potential effects include physiological impacts that produce hearing loss in fishes that could result in reduced fitness associated with an inability to hear biologically relevant sounds, even if masking does not occur.

### The Role of Hearing in the Lives of Fishes

Within vertebrates, hearing first evolved in the ancestors of fishes (e.g., Fay and Popper 2000; Clack et al. 2016). The selective pressures thought to have resulted in the evolution of hearing were the detection of distant sounds in the "auditory scene" such as those made by predators, prey, or sound-producing locations such as coral reefs (Fay and Popper 2000). Over 800 species of fishes have been reported to produce species-specific sounds (Kaatz 2002; Rice et al. 2022) for a variety of reasons including, but not limited to, agonistic signaling, stress response, alarm

calling, predator avoidance, and reproduction-associated displays (e.g., Hawkins and Myrberg 1983; Ladich 2019).

Since sound is so important to fishes, it can be expected that a reduction in their ability to detect such communication signals and the auditory scene in general will reduce the animal's fitness. Therefore, hearing loss is one of the adverse effects of anthropogenic sounds that has been focused on in fishes.

## Effects of Acoustic Overstimulation on Fish Hearing

The earliest studies documenting sound-induced hearing loss in fishes were done in the laboratory using tones or white noise as stimuli. The first report of hearing loss in fishes was done in goldfish, *Carassius auratus*, which were exposed to tones of 300, 500, 800, or 1000 Hz at approximately 149 dB re 1 μPa for 4 h, followed by determining behavioral hearing thresholds at 500 and 800 Hz (Popper and Clarke 1976). This exposure resulted in temporary threshold shifts (TTS) that depended upon stimulation and test frequency, but hearing returned to normal within 24 h.

Most subsequent experiments tested hearing loss by the recording of auditory evoked potentials (AEP), an electrophysiological method which allowed relatively rapid measurement of TTS in fishes compared to standard behavioral methods (e.g., Corwin et al. 1982; Kenyon et al. 1998). While using AEP recording only represent the responses of the ear or brainstem and do not measure hearing (Sisneros et al. 2016; Popper and Hawkins 2021a), they do provide a good indication of the function of the ear and are useful in determining if exposure to anthropogenic sound alters ear sensitivity.

Sound-induced hearing loss has been examined in very few of the more than 34,000 extant fish species. Table 1 lists these species, but does not consider the degree of hearing loss. The table also includes a number of species for which there was little or no hearing loss.

A general explanation for why some species are more prone to TTS than others is that there is considerable variation in hearing sensitivity between species. As can be seen in Table 1, most species that showed TTS are those that have anatomical specializations that enhance the frequency range of hearing and lower the sound level that can be detected (sensitivity threshold) (Popper et al. 2021). Most commonly, such specializations connect gas-filled chambers (e.g., swim bladders, suprabranchial chambers, and otic bullae) to the inner ear. For example, otophysan fishes (all species in the top section of Table 1) have modified vertebrae, the Weberian ossicles, that contact the swim bladder on one end and the inner ear on the other, allowing sound pressure-induced vibrations from the swim bladder to be transmitted directly to the inner ear (Popper and Hawkins 2021b). Such specialization allows for the detection of the pressure component of sound and results in lower hearing thresholds over a much broader frequency range compared to species without specializations (Popper and Hawkins 2021b).

Thus, it is important to note that the majority, but not all, of the species listed in Table 1 for which sound-induced hearing loss have been reported are otophysans,

**Table 1** Fish species for which TTS has been tested. Note that, the table does not include species where fish were exposed to intense sounds and ear damage, but no tests of hearing (e.g., McCauley et al. 2003; Smith et al. 2022)

| Common name | Species | Type of sound | Citations |
|---|---|---|---|
| **Species with specializations for hearing that showed TTS** | | | |
| Channel catfish | *Ictalurus punctatus* | Mid-frequency active sonar[a] | Halvorsen et al. (2012a) |
| | | Low-frequency active sonar | Halvorsen et al. (2013) |
| Goldfish | *Carassius auratus* | White noise | Smith et al. (2004a, b); |
| | | White noise | Amoser and Ladich (2003) |
| | | Pure tones | Popper and Clarke (1976) |
| Fathead minnows | *Pimephales promelas* | White noise | Scholik and Yan (2001, 2002b) |
| Koi | *Carpio carpio* | 150 Hz tone | Coffey (2014) |
| Lake chub | *Couesius plumbeus* | 5 or 20 air gun blasts | Popper et al. (2005) |
| Pictus catfish | *Pimelodus pictus* | White noise | Amoser and Ladich (2003) |
| Silver carp | *Hypophthalmichthys molitrix* | White noise | Nissen et al. (2019) |
| Zebrafish | *Danio rerio* | White noise | Breitzler et al. (2020) |
| **Species without specializations that showed some TTS** | | | |
| Oyster toadfish | *Opsanus tau* | Ship sounds, conspecific sounds | Rogers et al. (2020) |
| Meagre | *Argyrosomus regius* | Boat sounds | Vieira et al. (2021) |
| Northern pike | *Esox lucius* | 5 air gun blasts | Popper et al. (2005) |
| Rainbow trout | *Oncorhynchus mykiss* | Low-frequency active sonar[a] | Popper et al. (2007) |
| Sailfin molly | *Poecilia latipinna* | 150 Hz tone | Coffey (2014) |
| **Species for which no or minimal TTS** | | | |
| Bluegill sunfish | *Lepomis macrochirus* | White noise | Scholik and Yan (2002a) |
| Broad whitefish | *Coregonus nasus* | 5 air gun blasts | Popper et al. (2005) |
| Largemouth bass | *Micropterus salmoides* | Low-frequency active sonar | Halvorsen et al. (2013) |
| Nile tilapia | *Oreochromis niloticus* | White noise | Smith et al. (2004b) |
| Rainbow trout | *Oncorhynchus mykiss* | Ambient aquaculture sounds | Wysocki et al. (2007) |
| | | Mid-frequency active sonar | Halvorsen et al. (2012a) |
| Yellow perch | *Perca flavescens* | Low-frequency active sonar | Halvorsen et al. (2013) |

[a]Fish from some broods did not show hearing loss

most likely because their greater hearing sensitivity makes them more sensitive to sound exposure. In contrast, the species in Table 1 for which no or minimal hearing loss was reported have no specializations for hearing.

## Predicting Threshold Shift

One predictive model of fish hearing loss that considers the differences in frequency-specific hearing sensitivity between species is the linear threshold shift (LINTS) hypothesis (Smith et al. 2004b; Smith and Monroe 2016). This model predicts that the best indicator of potential TTS is the sound pressure level difference (SPD) between the intensity of the anthropogenic sound stimuli at a specific frequency and the baseline hearing threshold level of the species at the same frequency. Thus, the LINTS hypothesis predicts that fishes will have the greatest TTS at frequencies at which they are most sensitive (i.e., have the lowest hearing thresholds), but only if the anthropogenic sound is a certain level above that threshold. Or, put another way, an anthropogenic sound has to be a certain level above the hearing threshold to induce TTS.

This, in fact, is the case for goldfish and other otophysan fishes that have been tested (Smith et al. 2004a, 2006, 2011). In all cases, the anthropogenic sound was sufficiently above threshold to result in TTS. However, fishes without specializations and thus poorer hearing sensitivity may not show TTS to the same extent since the anthropogenic sound is not sufficiently above threshold. The only likely way to induce TTS in those species is to raise the level of the anthropogenic sound so it is the same level above threshold as in the hearing specialists.

While more work is needed to test and refine the LINTS model, it was able to predict TTS in three species of fishes that were exposed to a seismic air gun array (Popper et al. 2005). In that study, lake chub (*Couesius plumbeus*), an otophysan, exhibited the greatest TTS; northern pike (*Esox lucius*), a species without known specializations, were intermediate; and broad whitefish (*Coregonus nasus*), a species only likely to detect particle motion since it is a salmonid (Hawkins and Johnstone 1978), exhibited no hearing loss. As predicted by the LINTS model, this pattern was the inverse of the three species baseline hearing thresholds. That is, TTS was greatest in lake chub with the best hearing sensitivity and lowest in the broad whitefish with the poorest hearing thresholds. Whether this LINTS model is useful for species without hearing specializations is unclear since of those species tested so far (e.g., Nile tilapia, bluegill sunfish, rainbow trout, broad whitefish, sailfin mollies, northern pike), only northern pike and sailfin mollies showed TTS (Coffey 2014; Smith et al. 2004b; Wysocki et al. 2007; Popper et al. 2005).

An important issue, however, is that SPD has been reported in terms of sound pressure level (SPL) with units of dB re 1 μPa. The problem is that most fishes without hearing specializations only detect the particle motion component of sound; thus, pressure is not the most appropriate measure of hearing in those species (Nedelec et al. 2016; Popper and Hawkins 2018). Thus, the levels to which fish were exposed, in terms of particle motion, may be very different than the sound

pressure levels, particularly since the studies were done in tanks where the sound field is very complex and particle motion hard to measure (Radford et al. 2014; Rogers et al. 2016; Jézéquel et al. 2022).

## TTS and Sound Duration

While TTS increases linearly with SPL of the sound stimulus in goldfish and carp (Smith et al. 2004a; Nissen et al. 2019), TTS also increases with the duration of noise exposure (Popper et al. 2004; Smith et al. 2004b) or, in the case of a seismic air gun array, with an increase in the number of air gun shots (Popper et al. 2005). As anthropogenic sound sources like air gun shots and pile-driving produce transient impulsive signals with rapid increases and decreases in pressure that are very short in duration, measures other than the root-mean-square (RMS) of the SPL may be required to predict hearing loss in fishes. For example, cumulative sound energy and sound exposure level (SEL) are potential alternate measures that would account for both the amplitude and duration of a sound source (Popper and Hastings 2009).

## Recovery from TTS

While it is clear that anthropogenic sounds have the potential to produce hearing loss in fishes, the few studies that have performed hearing tests at multiple days post-sound exposure have shown that fish can recover to control hearing levels. For example, goldfish exposed to white noise (160–170 dB re 1 μPa) for 21 days exhibited significant threshold shifts in hearing up to approximately 28 dB. After 14 days, the fish fully recovered to control hearing levels (Smith et al. 2004b). In a subsequent study, goldfish were exposed to white noise (170 dB re 1 μPa RMS) for 48 h. By 7 days postexposure, goldfish hearing recovered significantly (mean TTS $< 4$ dB) (Smith et al. 2006).

Similar results have been found for zebrafish which were exposed for 24 h to white noise at various amplitudes (130, 140, and 150 dB re 1 μPa), and their hearing was tested at different times post-treatment (after 3, 7, and 14 days) (Breitzler et al. 2020). Recovery of hearing occurred within 3 d for fish exposed to 130- and 140-dB noise, while fish exposed to 150 dB took 14 days to recover to control levels. Thus, hearing recovery from TTS depends upon the intensity and duration of the sound stimulus. This is likely because the auditory sensory cells in the inner ear, referred to as hair cells, exhibit greater damage with increased intensity and duration of acoustic overstimulation (discussed below).

Recovery was also found after exposure to shorter and more intense signals. Rainbow trout (a non-specialist) exposed to low-frequency active sonar at a maximum received RMS sound pressure level of 193 dB re 1 $Pa^2$ for 324 or 648 s recovered from TTS within 24–48 h (Popper et al. 2007), and similar results were found for channel catfish (an otophysan) exposed to the same stimulus (Halvorsen et al. 2013) and to mid-frequency active sonar at 210 dB re 1 μPa. Similarly, both

lake chub (an otophysan) and young northern pike showed TTS to exposure to blasts from a seismic air gun at a sound exposure level of about 178 dB re 1 $\mu Pa^2$•s and recovered within 24 h (Popper et al. 2005).

## Effects of Acoustic Overstimulation on the Fish Ear

### The Inner Ear of Fishes

The inner ear of fishes consists of three semicircular canals and three otolithic end organs, the utricle, lagena, and saccule (reviewed in Popper and Hawkins 2021b). Within each end organ is a sensory epithelium (or macula) that contains large numbers of sensory hair cells surrounded by supporting cells and overlain by a dense calcareous otolith which is connected to the macula by a gelatinous matrix. The sensory hair cells are specialized mechanoreceptors with stereocilia and a single microtubule-based kinocilium in their apical surfaces. When sound passes through a fish, the differential density between the otolith and the underlying hair cells causes the stereociliary bundles to be deflected, opening up cation channels and producing receptor potentials that are transmitted to the brain (Flock 1971; Popper and Hawkins 2021b).

### Damage to Sensory Cells from Sound Exposure

One of the problems in studying hearing loss and TTS in many fish species is that it is difficult to measure hearing when animals have been exposed to sounds in the field or in some laboratory conditions (McCauley et al. 2003; Casper et al. 2013; Smith et al. 2022). To at least get some indication of potential hearing damage in these species, investigators have used microscopy to examine inner ear tissues. This approach, while not necessarily indicating hearing loss, parallels work in humans and other vertebrates that have shown close correlation between damage to sensory cells or the innervating nerves of the inner ear and hearing loss (e.g., Le Prell et al. 2012; Liberman 2016). Indeed, this approach has been validated in several studies in fishes that demonstrated close correlation between hair cell damage and hearing loss, followed by hair cell recovery and return of hearing (Smith et al. 2006, 2011; Uribe et al. 2013; Breitzler et al. 2020; Lara et al. 2022).

Sound-induced sensory hair cell loss in fishes was first reported for Atlantic cod (*Gadus morhua*) that were exposed to tones between 50 and 400 Hz that were 100–110 dB above the most sensitive hearing threshold for that species (Enger 1981). Examination of the saccular epithelia via scanning electron microscopy (SEM) revealed large patches of hair cells that were missing or almost completely lacking stereocilia. Lower-frequency tones damaged the caudal region of the saccule, while higher-frequency tones damaged the rostral region. Smith et al. (2011) found a similar tonotopic pattern of hair cell damage in goldfish exposed to tones

between 100 and 4000 Hz at 176 dB re 1 μPa RMS for 48 h, suggesting that this saccular organization may be common among fishes.

The first study to examine hair cell loss due to an anthropogenic sound source in the field was McCauley et al. (2003). They found that pink snapper (*Pagrus auratus*) exposed to air gun stimulation exhibited significant hair cell loss and damage 58 days postexposure (no fish were examined sooner so it could not be determined if hair cell loss showed up sooner postexposure). Pile-driving stimulation has also been shown to produce hair cell loss in fishes. Casper et al. (2013) found that 960 pile-driving strikes at 216 dB re 1 $\mu Pa^2 \cdot s$ cumulative SEL produced significant hair cell damage and loss in hybrid striped bass (*Morone chrysops x Morone saxatilis*). Similarly, exposure of caged Pacific mackerel (*Scomber japonicus*) up to 400 m from a single explosion of C4 explosives (4.5 kg) showed damage to sensory cells in the saccule (Smith et al. 2022).

While most work has focused on one end organ, the saccule, a study on the oscar (*Astronotus ocellatus*) found hair cell damage in small regions of the lagena and utricle, and not the saccule, following exposure to a continuous 300 Hz tone at 180 dB re 1 μPa (Hastings et al. 1996). Clearly, more work is needed on each of the inner ear end organs in order to understand potential hearing loss in fishes.

## Correlation Between Hair Cell Damage and TTS

There are only a few studies that have examined both sensory hair cell loss and TTS in fishes (Smith et al. 2006, 2011; Uribe et al. 2013; Breitzler et al. 2020; Lara et al. 2022), and initial analysis shows that there is a linear relationship between percent hair cell loss and the degree of TTS (Smith 2016). This pattern holds during the post-sound exposure period as well, with TTS decreasing, or recovering closer to control thresholds, as hair cell numbers return to control levels (Smith 2016).

At the same time, a possible exception to this relationship between TTS and hair cell loss was reported by Song et al. (2008) who found no hair cell damage using SEM in the same fishes for which air gun-induced TTS was reported (lake chub and northern pike; Popper et al. 2005). Similarly, while exposure to a US Navy Surveillance Towed Array Sensor System (SURTASS) Low Frequency Active sonar with a maximum received RMS SPL of 193 dB re 1 $\mu Pa^2 s$ produced a 20 dB threshold shift in rainbow trout (*Oncorhynchus mykiss*), no sensory hair cell damage was found in the ears of the same fishes (Popper et al. 2007).

The reason that there was TTS but no apparent hair cell damage is likely because these studies were limited in techniques to the examination of the surfaces of the sensory cells using various types of microscopy and not all regions of the ear were examined. In particular, TTS can occur in mammals when there is subtle damage to hair cells that are not easily detected without high-magnification scanning electron microscopy or other techniques. For example, acoustic over-stimulation of hair cells can result in broken tip links between the hair cell stereocilia which prevents mechanotransduction of sound signals or damages the

nerves innervating the hair cells (Lin et al. 2011; Indzhykulian et al. 2013; Liberman 2016).

## Is TTS Useful in Developing Sound Exposure Criteria for Fishes?

The data cited in this chapter demonstrate that anthropogenic sound sources can produce TTS in some fish species (Table 1) and/or damage to sensory hair cells. However, the number of species tested is few, and the experimental approaches used vary substantially, making it very difficult to compare results from different studies.

A fundamental question is whether it is even worth considering TTS when developing criteria to protect fishes from anthropogenic sounds. There are, of course, also questions as to whether a small level of TTS (e.g., 6 dB), if it even occurs, would have any short- or long-term impact on the fitness or survival of fishes (Popper et al. 2014).

While there are few data to directly address whether criteria should be developed to protect fishes from TTS, based on what is known about TTS and fishes, it is reasonable to suggest that TTS should not be an issue with regard to anthropogenic sound. This conclusion is based on several arguments.

First, and foremost, all reported effects on fish ears and hearing have been from exposure to very intense impulsive sources or long durations of moderately intense continuous sounds. While data are very few, it is likely that most species exposed to intense sounds, perhaps with the exception of animals that are "tied" to a site for breeding or other reasons, will likely move away from the source, and not stay in a region of intense sounds for long enough for TTS to occur, as demonstrated in the relatively few studies in the field (Fewtrell and McCauley 2012; Iafrate et al. 2016a, b; Krebs et al. 2016).

Second, considering that sound levels decrease as the distance from a source increases, it is reasonable to argue that most fishes in an area of anthropogenic sound will not be sufficiently near an intense source to be exposed to sounds intense enough to result in TTS. Thus, while some animals close to the source may be exposed to sufficiently intense sounds to result in TTS, these animals are likely to make up a very small part of the population of the species in an area, and even these animals are likely to move away from the source to areas of lower sound level (Iafrate et al. 2016a, b; Krebs et al. 2016; Watwood et al. 2016).

Third, as shown in Table 1, most species studied that have shown TTS to intense sounds were hearing specialists, which have substantially more sensitive hearing than non-specialists. Considering that most species of concern with respect to anthropogenic sound, including species considered threatened or endangered, and/or are of commercial value, are not hearing specialists, they are thus much less likely to develop TTS than specialists at the same level of sound exposure. Or, put another way, most marine species would need to be exposed to sounds far more intense than used in the studies in Table 1 before TTS would be expected.

Thus, the argument made here is that in considering potential criteria for effects of anthropogenic sound on fishes, TTS is not a particularly likely occurrence. At the same time, overall effects of anthropogenic sound on fish hearing, with particular

focus on masking and changes in behavior such as leaving feeding sites, are far more critical issues and need much more study.

Still, the argument made here regarding TTS is based on limited data and conjecture. In order to support the argument, more data on TTS and effects of sound on fish ears are needed. However, a much more structured approach needs to be taken in future studies using species that are likely to be exposed to anthropogenic sounds and using experimental approaches that allow more "natural" behavior of fishes exposed to the sounds (as discussed in Popper et al. 2019, 2022; Hawkins et al. 2020). Finally, far better control is needed for the sound field, and calibration of exposure levels needs to be in terms of particle motion since it is the stimulus that is most directly detected by the ears of all fish species.

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# Effects of Vessel Noise on Walleye Pollock (*Theragra chalcogramma*) Behavior

58

Mikhail Yu. Kuznetsov, Vladimir I. Polyanichko, and Igor A. Ubarchuk

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**Abstract**

The results of experiments on assessing the reaction of walleye pollock (*Theragra chalcogramma*) to the noise of passing research vessel are presented. A free-drifting buoy with built-in scientific echo sounder (38 kHz) was used. Pollock reactions were accompanied by horizontal and vertical movements from the noise source. The response distances to vessel noise ranged from 140 to 180 m, which is consistent with the model estimates. The total acoustic density curve has a tendency of a small initial rise and then a steady decrease with further approach of the vessel. A stronger decrease in the total density of fish in the upper horizons due to lateral avoidance is typical. The diving of fish when the vessel approaches causes a consistent increase in their density in a deeper layer. In the depth range of 0–100 m, the ratio of pollock density to the background at the closest point of approach was 0.56. At depths of 100–200 m, due to the deepening of fish, the decrease in total density was less significant ($K_{avoid} = 0.86$). The diving speed of pollock in the vessel's noise field is an order of magnitude higher than during daily vertical migrations. Such avoidance behavior can be a source of error in echo integration and trawl fish stock estimations.

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**Keywords**

Vessel noise · Walleye pollock · Avoidance reaction · Density · Aggregation · Response distance · Model · Behavior

## Introduction

Currently, it has been experimentally proven that vessel noise can have a significant impact on fish behavior and, as a result, on fish stock estimates and catch of fishing vessels (e.g., De Robertis and Wilson 2010; De Robertis and Handegard 2013; De Robertis et al. 2010; Kuznetsov 2011; Kuznetsov and Vologdin 2009; Mitson 1995; Ona et al. 2007; Vabø et al. 2002). The frequency ranges of maximum noise energy of the vessel and the best auditory sensitivity of most fish species coincide. Vessel noise levels far exceed fish auditory thresholds. This suggests that fish can perceive the noise of a vessel over long distances (Kuznetsov 2011; Mitson 1995). The main effect of the vessel noise influence is the avoidance reaction of fish (Fig. 1). The avoidance reaction is manifested by the desire of fish to leave the vessel noise field in the direction of its reduction (minimization of the stimulus) and is accompanied by horizontal and vertical movements of fish. These movements cause a change in the natural distribution and density of fish in the path of the vessel and create the possibility of error in the echo integration and trawl estimates of fish stocks.

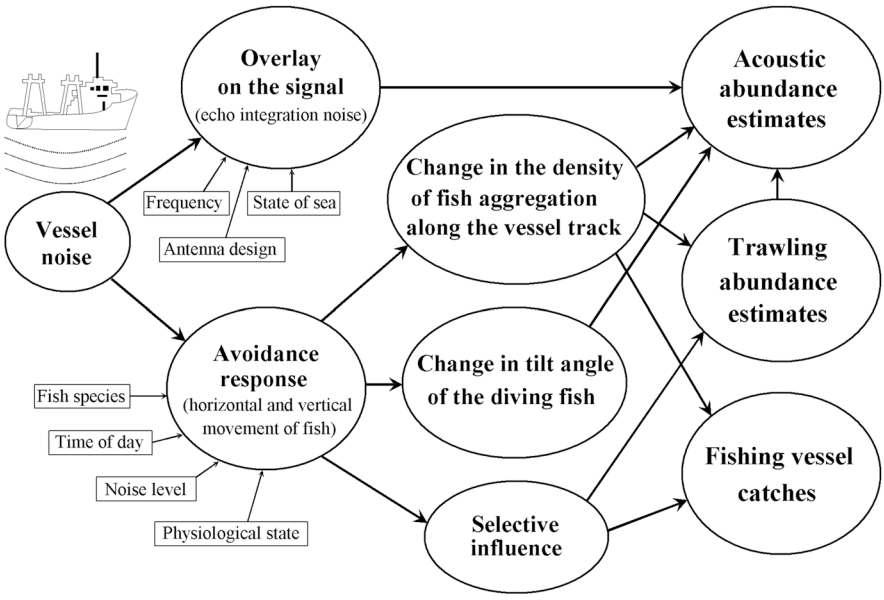



**Fig. 1** Effects of vessel noise influence on acoustic and trawl abundance estimates and fish catches. In the rectangles show the factors that determine the strength or degree of this influence

According to various researchers, the response distance of fish to the noise of the vessel varies significantly from 40 to 500 or more meters (more often from 100 to 200 m), depending on the species (hearing abilities of fish), the physiological state of the fish, the type of vessel (noise level), and environmental conditions (Kuznetsov and Vologdin 2009). The slope of the fish during diving causes a decrease in the values of acoustic reflectivity (backscattering strength), which leads to an underestimation of echo integration assessment of abundance (McQuinn 1999). The vessel noise can be an additional source of echo integration noise in scientific echo sounders (Korneliussen 2000).

The selective influence of the vessel noise field is important. This influence is especially pronounced in mixed aggregations of different species and age classes of fish, which have different energy capabilities (swimming speeds) and behavior (He and Wardle 1988). For example, larger (adult) fish respond more strongly to acoustic stimuli than their juveniles (Engås et al. 1996). The response to an approaching vessel to fish of different ethological groups is largely similar to the defensive response of these fish to the approach of a predator (e.g., Fréon et al. 1993; Skaret et al. 2005; Vabø et al. 2002). The nature of the reaction is determined by the species stereotype of the defensive behavior of fish. The defensive reaction of sedentary bottom fish is characterized to a greater extent by their hiding. Fast near-surface fish respond by more active horizontal avoidance of the vessel noise field. The reaction of vertically migrating fish is accompanied by both horizontal and vertical movements. The strength of the fish's response to the noise of the vessel varies significantly depending on the physiological state of the fish. Fish are usually easily excited in a state of increased locomotor activity (e.g., during migration) and are indifferent to acoustic stimuli during spawning.

This chapter reports on the locomotor reactions of walleye pollock (*Theragra chalcogramma*) to the noise of a passing vessel using an autonomous floating buoy with a built-in scientific sonar. The obtained results were used to simulate the motor behavior of fish in the vessel noise field, which will allow developing a strategy for its accounting and compensation when conducting trawl-acoustic surveys of fish stocks.

## Methods

Horizontal and vertical movements of fish from the noise source, as well as changes in the tilt angle of the diving fish, are accompanied by a decrease in the acoustic density and an increase in the depth of the fish location relative to the initial. This behavior of fish is interpreted as an avoidance response (Olsen et al. 1983).

Observations of the reaction of fish to the noise of the research vessel "Professor Kaganovsky" were carried out during the trawl-acoustic surveys of walleye pollock in the Sea of Okhotsk, Russia, in the spring seasons of 2014–2016. Autonomous hydroacoustic complex (AHC) was used for experiments. AHC is a floating buoy with built-in equipment (Fig. 2): scientific echo sounder Simrad EK60 with 38 kHz transducer, power supply, Wi-Fi module (router), and GPS receiver. The echo

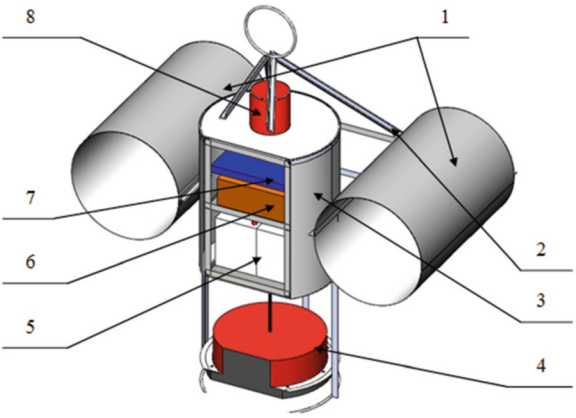



**Fig. 2** Arrangement of an autonomous hydroacoustic complex (AHC): (1) floating platform; (2) lifting truss; (3) instrument compartment; (4) echo sounder transducer (38 kHz); (5) rechargeable batteries; (6) echo sounder general-purpose transceiver (GPT); (7) laptop; (8) Wi-Fi module (router) and GPS receiver

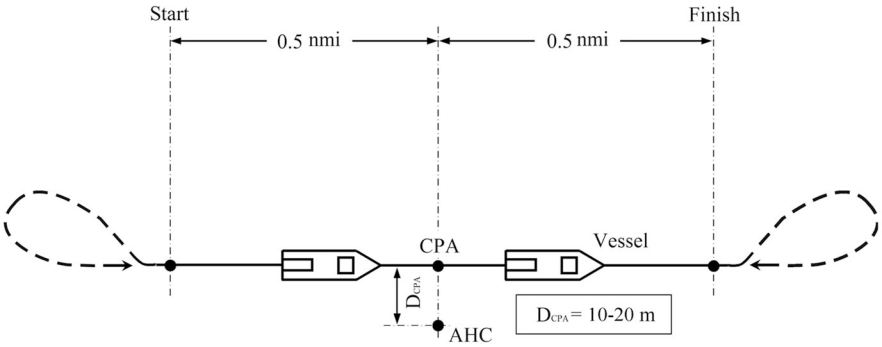



**Fig. 3** The pattern of vessel movement during measurements: *CPA* closest point of approach, *AHC* autonomous hydroacoustic complex, $D_{CPA}$ minimal distance between the passing vessel and the AHC

sounder is launched from the vessel via Wi-Fi and operates autonomously in the vertical location mode at a frequency of 38 kHz, used in the echo integration assessment of fish stocks. Acoustic and navigation data were recorded on the hard disk of a laptop computer in "raw" format using the Simrad ER60 program. All analyses of acoustic data were conducted using "SALTSE" software (Kuznetsov et al. 2021).

Experiments on the measurement of the avoidance reaction were carried out in the presence of uniformly distributed walleye pollock aggregations at a depth of not more than 200 m. Trawl samplings were carried out to identify echograms and determine the size and species composition of the studied aggregation before each series of experiments. The pattern of movement and maneuvering of the vessel during the experiments is shown in Fig. 3. The vessel departs from the buoy at a distance of 1 nautical mile (nmi), makes a circulation, and begins to move in the

direction of the AHC. The vessel passes in a straight line past the AHC, alternately to port and starboard, at a speed of 7–9 knots. The straight-line uniform motion of the vessel between the "Start" and "Finish" points is the transect. The echo sounder at AHC continuously measures the vertical distribution of fish during each transect. The minimal distance between the passing vessel and the AHC in the closest point of approach (CPA) varied from 10 to 20 m. The time interval between passes was at least 40 min in order to restore the background state of the fish aggregation.

The response of fish was assessed by the change in acoustic density (surface backscattering strength) and depth of the fish before and during the passage of the vessel past the AHC. In the mode of post-processing of echograms by the "SALTSE" software complex, nautical area scattering coefficient $s_A$ ($m^2$ $nmi^{-2}$) and weighted average depth of fish $H$ (m) were measured. First, their background values $s_{A(ref)}$ and $H_{(ref)}$ before the measuring transect when the fish were presumably undisturbed by the vessel and then during the transect $s_{A(pass)}$ and $H_{(pass)}$ – the observations at the time of vessel passage during 1- or 2-s echo integration intervals (~5 or 10 m horizontal distance) – were measured. The reference period was defined as 0.5 nmi before and after CPA, and observations during vessel passage were taken as CPA ±1 s (~5–10 m). The change in the density and depth of the fish during the passage of the vessel past the AHC was estimated in relation to the $s_A$ and H on transect to the background value as $K_{avoid}$ avoidance coefficient and $K_{depth}$ diving coefficient:

$$K_{avoid} = \frac{s_{A(pass)}}{s_{A(ref)}},$$

$$K_{depth} = \frac{H_{(pass)}}{H_{(ref)}}.$$

## Results and Discussion

Reducing the density of fish and increasing the weighted average depth (diving) of walleye pollock aggregations during the vessel's approach to the AHC were typical for most experiments (Fig. 4). In a series of experiments, where the proportion of immature fish was high, a stable effect of changing of density aggregations in the near-surface horizons was recorded at a distance from the vessel of 140–150 m. At distances from 30 to 50 m after the passage of the CPA, the avoidance coefficient reached a minimum value, and diving speed decreased to the background. Then the slow rise of pollock began with a wave-like increasing speed to 0.06–0.12 m/s. In aggregations with a predominance of mature pollock, the horizontal avoidance reaction began to appear somewhat earlier: at a distance of 160–180 m in front of the vessel. The decrease in the total density of fish often stopped ($K_{avoid}$ minimal) before the vessel approached to CPA at distances up to 30 m. Then the reverse process of restoring of walleye pollock density began. In aggregations of juvenile pollock, a decrease in the total density at $\leq$90% of the background was recorded at a distance of about 100 m with a tendency for a very slight decrease as the vessel

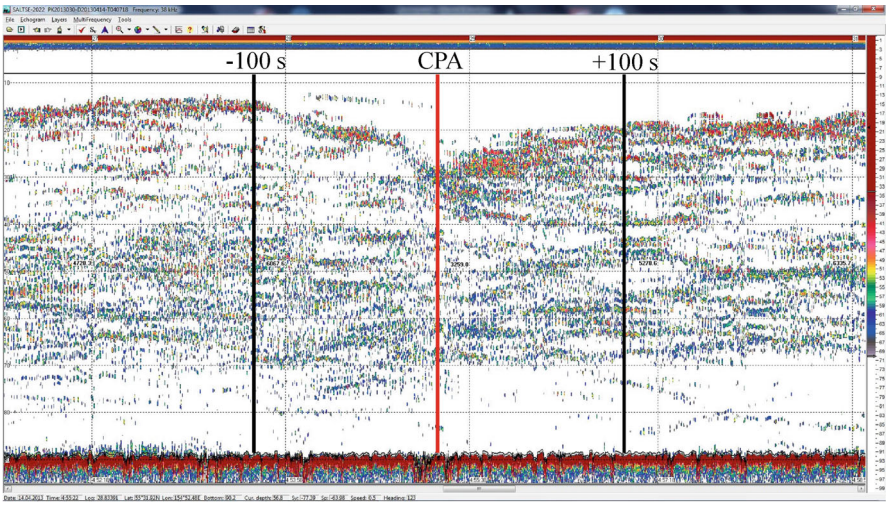



**Fig. 4** Echogram illustrating the walleye pollock's reaction when RV "Professor Kaganovsky" passes by the AHC. See text for explanation

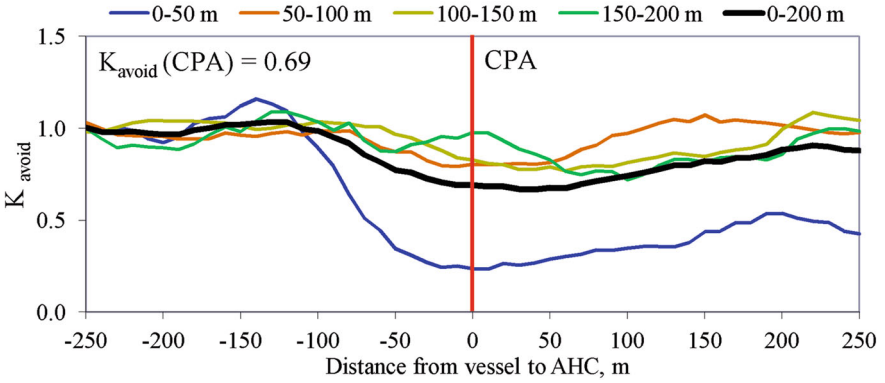



**Fig. 5** Change in density of walleye pollock aggregations in sub-ranges of depths depending on the distance between the vessel and AHC (Sea of Okhotsk, spring 2014–2016). See text for explanation

approached. The diving of fish became noticeable only near the vessel at a distance of about 50 m.

Generalized graphs of the density change (coefficient of avoidance) of walleye pollock aggregations in the vessel's noise field are shown in Fig. 5. At distances to the vessel of 180–140 m, the total density curve tends to have a slight initial rise ($K_{avoid} > 1$) and then a steady decrease with its further approach. Horizontal movements of the fish reacting forward in the course of the vessel and already there, apparently, cause the increase in density, due to which there is an effect of "crowding" of walleye pollock at response distances.

A steady decrease in the total density of walleye pollock aggregations $s_A$ was recorded at a horizontal distance to the vessel of about 120 m. At first, this decrease was mainly due to the intensive scattering of fish in the near-surface horizons of 0–50 m. Then, as the vessel approached, a decrease in density was observed in the deeper layers – 50–100 and 100–150 m (Fig. 5). Since the reaction of fish is accompanied not only by horizontal avoidance but also by their deepening, the density reduction curves in the subsurface layers of depth (50–150 m) summarize these effects and have a more gentle character.

In the depth range of 150–200 m (i.e., at the pollock response distances) near the CPA, the reverse process of increasing the density of aggregations was observed. This is due to the movement of fish from the upper horizons there, similar to the "crowding" of the pollock in front of the vessel.

Thus, part of the fish in the way of the vessel goes to the sides (dissipates). A stronger decrease in the density of fish in the upper horizons due to lateral avoidance is typical. It is also typical to redistribute the density of pollock by depth layers. The diving of fish when the vessel approaches causes a consistent increase in their density in a deeper layer. With a sufficiently wide vertical distribution of aggregations and if horizontal avoidance is small, at certain depths, the noise field of the vessel increases the density of fish relative to the initial (background). These results confirm the data of model calculations (Kuznetsov et al. 2015). In almost all experiments, the decrease in the total density during the vessel passage was due to a decrease in the density of fish in the upper horizons, and its growth – due to an increase in the density of pollock in the lower layers of depth.

The coefficient of avoidance in the CPA at depths less than 100 m according to the generalized data was 0.56 (Fig. 6a). This means that the density of pollock at the minimum distance from the vessel decreased by almost 1.8 times compared to the initial, measured with the natural (undisturbed) state of the fish in front of the vessel. In the depth range from 100 to 200 m (Fig. 6b), the trend of the initial rise and subsequent decrease in the total density of aggregations in CPA due to the deepening of fish was less significant ($K_{avoid} = 0.86$). The underestimation of the number of pollock in the depth range of 100–200 m did not exceed 15% under the vessel and 20% behind the vessel. The ratio of the total density of fish to the background in CPA in the entire depth range of 0–200 m was 0.69 (Fig. 5).

After the passage of the CPA, the density in the layer of 150–200 m decreased as the vessel departed, i.e., the fish left the densification zone, which was formed at response distances under the vessel. Then, at a horizontal distance from the vessel of more than 50 m, the process of reverse restoration of the density of fish in other layers, starting from a deeper one, became noticeable. Restoration of the total density of fish occurred more hollow than its decrease, at a distance of 220–240 m after the passage of the vessel, and only partially: up to 80% of the initial in the depth range 0–100 m and up to 90% of the initial in the depth range 0–200 m. In the depth range 100–200 m, the decrease and restoration of the total density behind the vessel went approximately at the same speed. The fish density was restored completely ($K_{avoid} = 1$) at a distance of 400–500 m from the vessel (after 1.5–2.0 min.). However, the weighted average

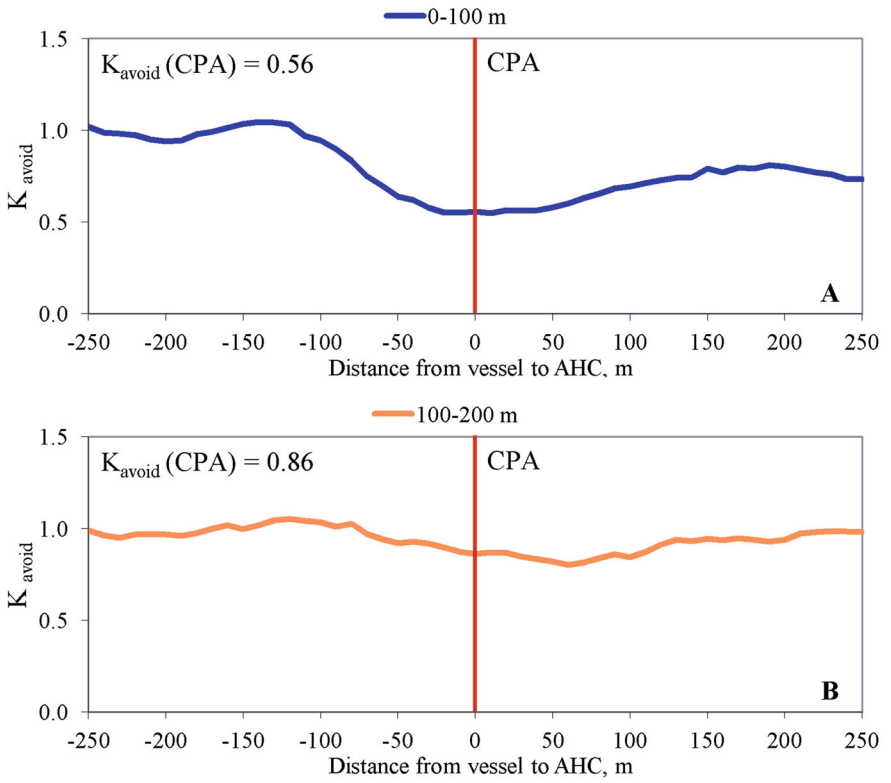



**Fig. 6** The density of walleye pollock aggregations varies depending on the distance between the vessel and AHC in the depth ranges: (**a**) 0–100 m; (**b**) 100–200 m

depth of the fish remained 7–10 m below the background. The initial vertical distribution of fish was restored for more than 20 min.

In a number of experiments, diving of fish was found at a distance of 350–500 m or more from the vessel. At the same time, the density of the aggregations did not change significantly (pollock did not dissipate). This state conforms to the first phase of the reaction, when fish hear sound (the noise level exceeds the auditory threshold), but cannot determine the direction of the arrival of the sound wave (Kuznetsov 2011). The reaction of fast near-surface fish (e.g., sardine, mackerel, salmon) is characterized by more active than background horizontal movements inside of the school (Kuznetsov and Kuznetsov 2007). Less mobile species (which include pollock), as long as the source of noise is not localized, respond mainly with vertical movements of fish (known as "squatting" of pollock). At distances to the vessel at which the increasing sound pressure of the noise exceeds the threshold of the avoidance reaction, the response of fish becomes directed. This reaction is accompanied by both vertical and horizontal movements of the fish from the noise source.

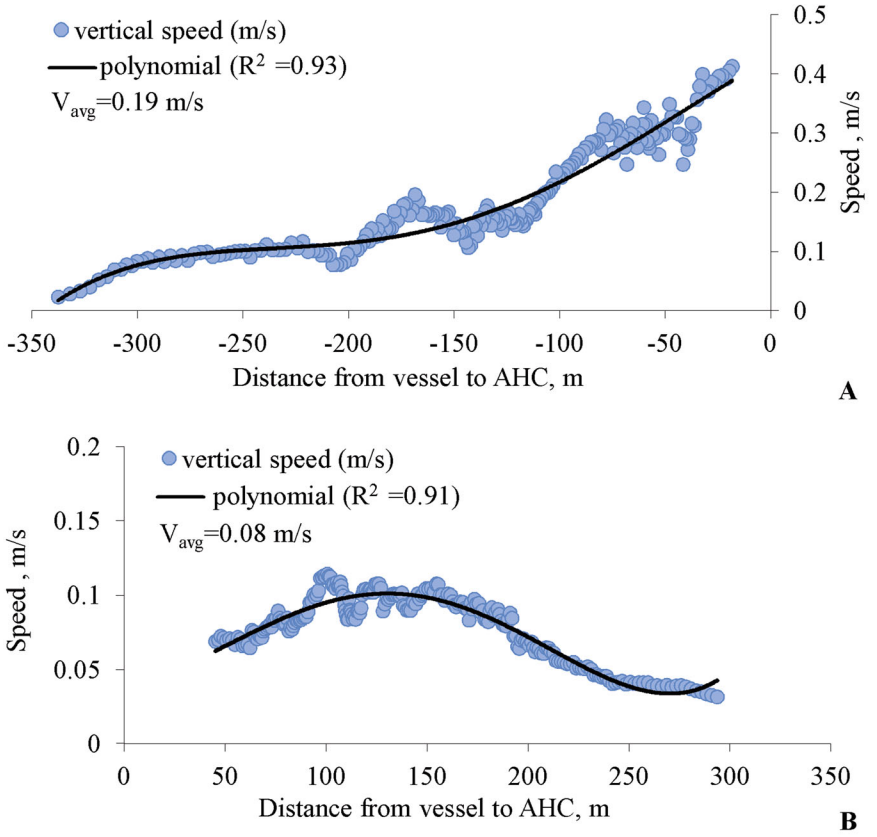



**Fig. 7** Dependences of the speed of diving (**a**) and lifting (**b**) of walleye pollock on the distance between the vessel and the AHC in the depth range up to 100 m. The dots represent a vertical speed (a running mean of 11 s). $V_{avg}$ average velocity

Generalized for all experiments, the dependence of the speed of diving and lifting of pollock on the distance between the vessel and the AHC is shown in Fig. 7. The speed values are well approximated by a fourth-order polynomial. The diving speed gradually increases from 0.04 to 0.15 m/s between 350 m and 150–180 m. At these distances, the behavior of the fish by noise level conforms to the first phase of the reaction (horizontal avoidance is absent). At distances less than 150 m (the noise source is localized), the speed of diving increases sharply and at the minimum distance from the vessel becomes the maximum – up to 0.42 m/s (at distances less than 50 m, the average speed is 0.35 m/s). The obtained results show that the dive rate of walleye pollock in the vessel's noise field is significantly (more than an order of magnitude) higher than those during their natural migrations.

The speed of lifting of pollock after the passage of the vessel is less than the speed of diving, but also significantly higher than during daily migrations. At a distance of

about 100 m from the vessel, the lifting speed becomes maximum – up to 0.12 m/s. The lifting speed of pollock slows down smoothly as the distance from the vessel increases (Fig. 7).

A simple model of the locomotor behavior of fish in the noise field of the vessel has been developed. The model uses experimental estimates of the pollock response distance, data on the change in the acoustic density $s_A$, and the depth and speed of diving and lifting of the pollock during the passage of the measuring transect. The initial distribution of fish in the model is assumed to be uniform in horizon and depth. By specifying the species and size of the fish and the type and speed of the vessel (acoustic field characteristics) in the model, it is possible to predict horizontal and vertical position of the fish relative to the vessel at any time.

Model estimates of the change in the density of walleye pollock aggregations are well consistent with experimental results. In the depth range of 0–100 m, the model curve, like the experimental results, tends to have a slight initial rise due to horizontal avoidance and densification of fish in front along the course of the vessel (Fig. 8). The ratio of the density of the fish aggregation to the background in the depth range

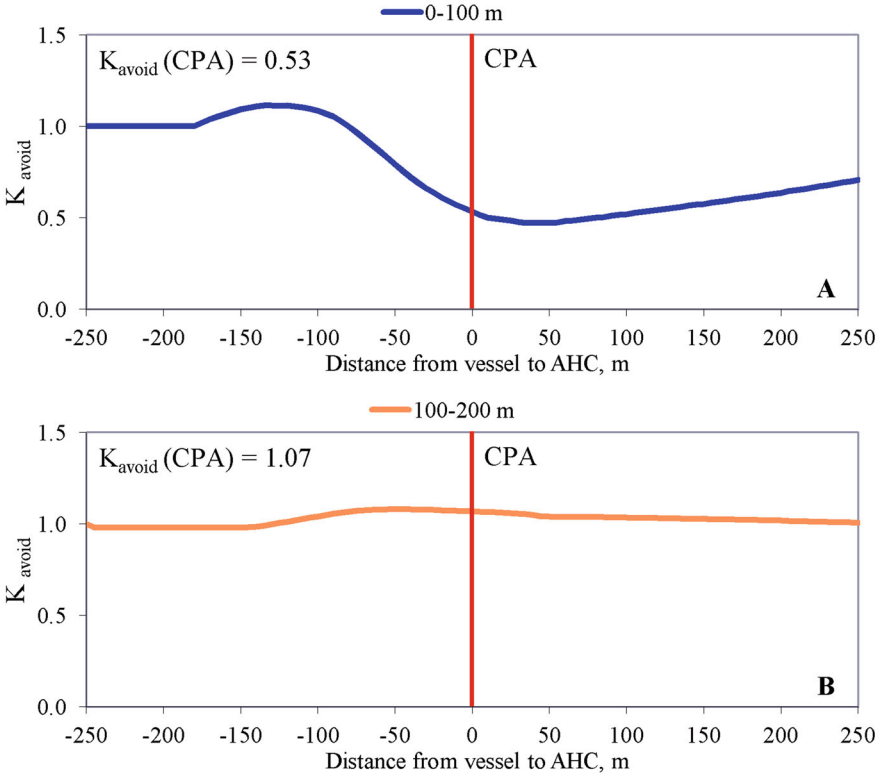



**Fig. 8** The change in the density of walleye pollock aggregations on the distance to the vessel in the depth ranges: (**a**) 0–100 m; (**b**) 100–200 m (model estimates)

of 0–100 m in the CPA according to the model was 0.53, which is close to the experimental estimates (0.56). In the layer of 100–200 m, where part of the fish moves from the upper horizons during the reaction, model and experimental estimates differ somewhat. The uniform initial distribution of fish adopted in the model is rarely observed in experiments. In the depth range of 100–200 m, the effects of horizontal avoidance (scattering) and an increase in density due to the movement of fish there from the upper horizons are simultaneously observed, which, with a uniform initial distribution in the model, mutually compensate for each other. As a result, a layer is formed where, according to model estimates, the pollock density may be higher than the initial one ($K_{avoid} > 1$).

Figure 9 illustrates the horizontal distribution of pollock in the noise field of the vessel depending on the size of fish and depth. The perturbed area, where the avoidance reaction and distortion of the spatial distribution of fish occur, is clearly visible. The square of the perturbed area (fish reaction zone) and the number of deviated fish significantly decrease with depth. Horizontal and vertical displacements of fish, minimizing the impact of noise, decrease as they move away from the vessel. As a result, at the edges of the disturbed area, a zone is formed, where the density of reacting fish becomes higher than the initial one. This zone is formed both on the sides and under the keel of the vessel and looks like a hemisphere, the radius

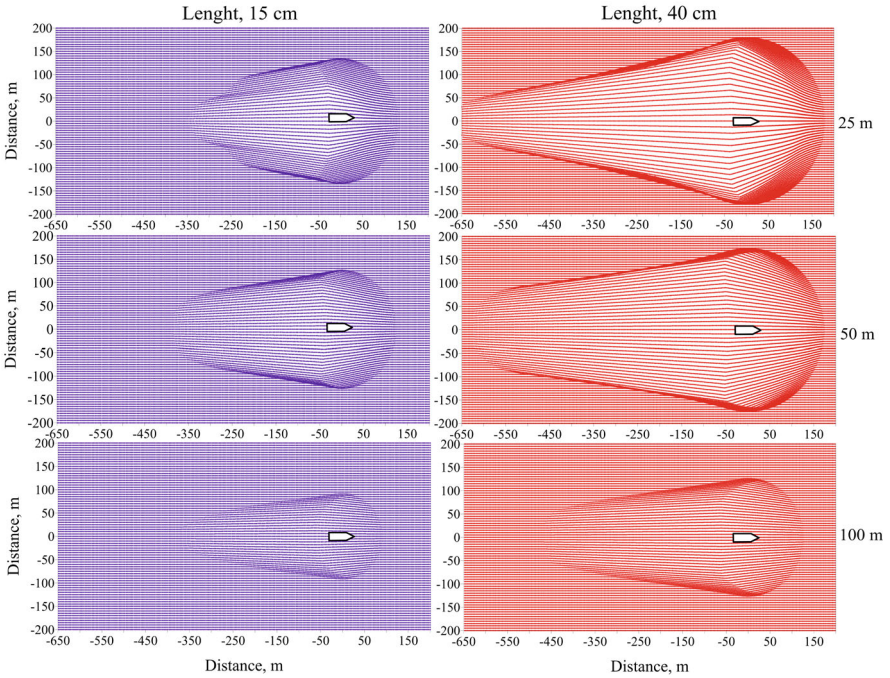



**Fig. 9** Model of walleye pollock behavior in the noise field RV "Professor Kaganovsky." Top view, section at depth 25, 50, and 100 m, speed 9 knots. See text for explanation

of which is the response distance of the fish. The restoration of the spatial distribution and density of fish is completed at a distance of 500–700 m after the passage of the vessel.

The large fish with higher energy and auditory abilities, in the process of reaction, move farther from the vessel and have a larger disturbance area than small objects, which demonstrates one of the manifestations of the selective effect of the noise field. The final location of the fish at various speeds of the vessel is different. The perturbed area where the avoidance reaction occurs, at a lower speed of the vessel significantly more fish manage to leave than at a higher vessel speed. In this case, the square of the perturbed area does not change.

In the transverse plane, a plurality of fish in the vessel's noise field forms a typical tunnel shape. Visually noticeable are the effects of reducing the density of fish aggregation relative to the initial one in the active reaction zone near the surface and densification (concentration) of pollock in a more comfortable zone along the edges of the tunnel.

## Conclusions

The results of experiments showed that the reaction of walleye pollock to the noise of the approaching vessel is manifested by their diving and dispersion. The response distances of pollock to the vessel noise ranged from 140 to 180 m from the vessel and are consistent with the calculated estimates. At such distances, the reaction of the fish becomes directed and is accompanied by their initial densification and subsequent decrease in density as the vessel approaches. The speeds of vertical movements of walleye pollock significantly exceed those of its daily migrations. The results confirm that the noise of the vessel is a strong irritant for fish, stimulating a pronounced defensive reaction. Experimental and model estimates showed a decrease in the acoustic density of pollock near the vessel of more than 40% at depths of up to 100 m. Such avoidant behavior can be a source of error in echo integration and trawl estimates of fish stocks.

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# Effect of Boat Noise on Chorusing Behavior of a Marine Fish (*Argyrosomus regius*, Sciaenidae)

59

Manuel Vieira, Paulo J. Fonseca, and Maria Clara P. Amorim

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**Abstract**

Aquatic noise has increased in the last decades, especially along the routes of freighter and in harbor areas. The increasing sounds are likely augmenting the pressure on aquatic animals' acoustic communication and causing enhanced stress on populations inhabiting such regions. One of such areas, the Tagus estuary (Portugal), holds important maritime harbors and a network of ferryboat routes. It is also used as breeding grounds for several fish species, namely the meagre (*Argyrosomus regius*), which might be impacted by anthropogenic noise since it produces loud breeding choruses. This study assessed the impact of boat noise on this species by inspecting changes in chorus sound levels in the field during boat passages. Results show that the sound level of meagre choruses is often reduced during a boat passage, a behavioral effect that might ultimately impact reproductive success of this species.

**Keywords**

Passive acoustic monitoring · Behavior · Anthropogenic noise · Acoustic communication · Reproduction · Machine learning

## Introduction

Anthropogenic noise is changing underwater soundscapes around the globe (Duarte et al. 2021). Some of the most common sources of underwater noise in shallow waters are boats and vessels. Such sounds are known to affect the behavior and physiology of various fish species (Bruintjes and Radford 2013; Graham and Cooke 2008; Picciulin et al. 2012; Amorim et al. 2022).

Sciaenidae are one of the best-known fish vocal families. They can produce conspicuous chorusing behavior associated with the breeding season (Luczkovich et al. 2008; Montie et al. 2017; Vieira et al. 2022). The choruses produced by these species are so extraordinary that they are described in the famous book "20,000 Leagues Under the Sea" by Jules Verne (1870) as follows: "Some authors more poet than naturalist – believe that these fish sing melodiously, and when they unite in a choir human voices scarcely equal them. I do not dispute this, but the sciaenidae did not serenade our passage, to my regret."

The meagre (*Argyrosomus regius*) is a large (up to 2 m) long-lived (can reach 40 years or more) sciaenid that, as other sciaenids, forms large aggregations during the breeding season (Haffray et al. 2012). An important spawning and nursery area is the busy Tagus estuary (Portugal) that holds important maritime harbors and a network of ferryboat routes that generate significant low frequency noise (Vieira et al. 2021a). Exposure to boat noise might interfere with chorusing behavior and ultimately with reproduction outcome. Vieira et al. (2021b) presented the non-stochastic decrease of the meagre calling activity during a boat passage. Here, this study go further to report changes in calling activity associated to boat passages, taking into account different chorus intensity, which should reflect different number of calling individuals and/or different individual calling rate.

## Methods

Continuous acoustic recordings were obtained in Tagus estuary (Air Force Base 6, Montijo, Portugal; 38°42′N, 8°58′W) at a site known to have meagre choruses and boats passing by regularly (Vieira et al. 2021a). Recordings were made using a High Tech 94 SSQ hydrophone (sensitivity – 165 dB re. 1 V/μPa, flat frequency response up to 6 kHz ± 1 dB) anchored near the bottom and connected to a stand-alone 16 channel datalogger (LGR-5325, Measurement Computing Corp, Norton, MA, USA; 4 kHz sampling rate, 16 bit). The data set consisted of ca. 27-day continuous recordings (from the 1st to the 30th of April 2018, with some interruptions).

To annotate the continuous recordings, two automatic sound recognition systems using hidden Markov model (HMM)-based supervised machine learning were applied to detect and identify both the boat passages (system-1, as described by Vieira et al., 2020) and the presence of meagre choruses (system-2, as described Vieira et al. 2021b). In short, multiple HMMs were trained using sounds of each defined category and then used to classify the recordings according to their maximum likelihood, i.e., the model with the best distribution to represent a given sound type (a vector representing a time series) was selected. Note that in an HMM-based sound recognition, it is assumed that the time series of an observed sound can be represented by a linear Markov model. For detailed information, see Vieira et al. (2021b).

For each boat passage annotated using the automatic recognition system, sound level was measured in third-octave bands (TOL) centered at 500 and 1584 Hz (computed according to Merchant et al. 2015; averages for every 10 s were used) to quantify the variation of the acoustic behavior during each detected boat passage. These bands were selected according to the description of Tagus soundscapes by Vieira et al. (2021a): TOL 500 Hz as proxy of the meagre chorus and TOL 1584 Hz as a proxy of the boat noise. Note, however, that to standardize the analysis to measure the changes on the sound level throughout a boat passage, it was considered 180 s before and 180 s after the TOL 1584 Hz peak at each boat passage. To evaluate changes in meagre calling activity, it was measured the minimum TOL 500 Hz before(−180 to −110 s), during(−10 to 80 s), and after (110–180 s) the TOL 1584 Hz peak. Because TOL 500 Hz levels also receive some contribution from boat noise, we used the minimum values registered in the different periods to evaluate changes in meagre calling activity, as these levels should have a minimum influence from boat noise. Note that, although the ferryboats follow approximately the same route between Lisbon and Montijo, the distance of ferryboats to the hydrophone can vary (usually between 100 and 300 m). The before, during, and after periods were selected based on the results reported by Vieira et al. (2021b), namely the during period was considered from −10 to 80 s to encapsulate the decrease of chorus sound level that usually occurred shortly after the peak of the boat noise sound level. These values were compared using a factorial two-way ANOVA with interactions using the following categories: (1) period relative to the boat passage (before, during, and after) and (2) the meagre chorusing level that was divided in three classes according to the minimum TOL 500 Hz before the boat passage (low, from 90 to 100 dB; medium, from 100 to 110 dB; and high intensity,

above 110 dB). This approach aimed at assessing if any changes in calling behavior due to boat noise could be influenced by chorus intensity, a proxy for meagre aggregation calling activity. Each boat passage was included as a random effect factor. ANOVA was followed by Tukey post hoc test. For all the statistical analyses, standardized values were computed for each meagre chorus class because they have different magnitudes. To respect the ANOVA assumptions, logarithmic transformations and removal of outliers were conducted. Assumptions were evaluated by visual inspection of residual plots (predicted vs. residuals values, normal probability plots).

## Results

On average, there was a decrease in the TOL 500 Hz, a proxy for the intensity of meagre choruses, during ferryboat passages (Fig. 1). This reduction usually had its maximum after the peak of the boat noise. A significant effect of boat passage period was found

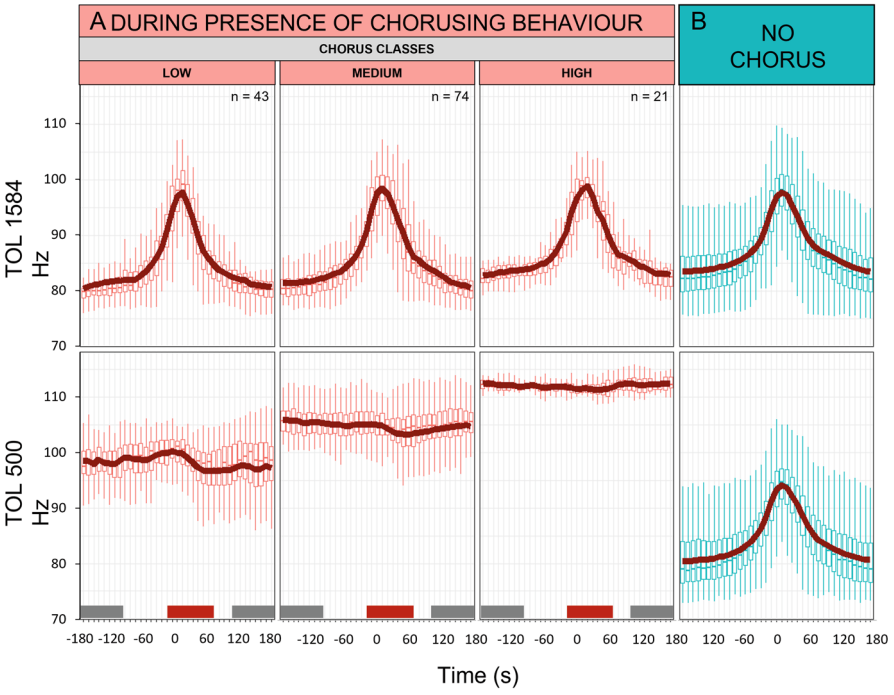



**Fig. 1** Effect of boat passages on the meagre chorus sound level. Red line represents the mean TOL around ferryboats passages in two bands (centered on 500 and on 1584 Hz). (**a**) represents ferryboat passages in the presence of meagre chorus at different classes of intensity, and (**b**) represents sections without meagre choruses. Zero seconds represent the energy peak of the ferryboat passages. Colored bars represent the before (gray), during (red), and after (gray) periods

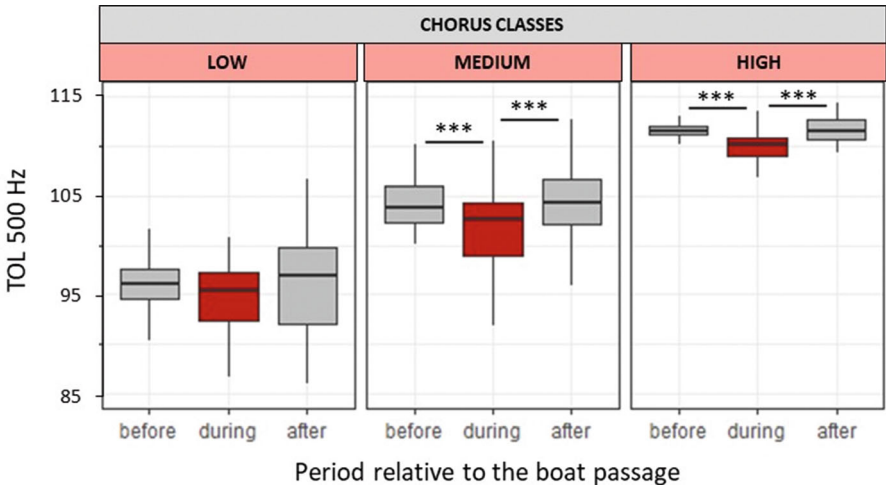



**Fig. 2** Change in TOL 500 Hz before, during, and after the boat passages. ***$p < 0.001$, Tukey test for pairwise comparisons within each chorus class. Note that statistical analysis was performed on standardized values

($F_{(2,254)} = 26.8$, $p < 0.001$) but no significant effect of the intensity of chorus ($F_{(2,135)} = 0.09$, $p = 0.9$). The interaction of these variables was significant ($F_{(4,254)} = 3.9$, $p = 0.004$). In choruses of medium (100–110 dB) and high intensity (above 110 dB), values before and after were significantly higher than during a boat passage (Tukey test, $p \leq 0.001$; Fig. 2), but no significant changes were observed in choruses of low intensity (before-during, $p = 0.26$; during-after, $p = 0.43$). No significant changes were observed between before and after periods ($p = 0.95$, $p = 0.82$ and $p = 0.98$ in chorus of low, medium and high intensity, respectively). A reduction in the chorus (TOL 500 Hz) from before to during a boat passage was observed on 67%, 85%, and 85% of boat passages with choruses of low, medium, and high intensity, respectively. Figure 3 shows two examples of chorus energy reduction associated with a boat passage.

## Discussion

We investigated how meagre choruses can change in the presence of a ferryboat. The sound level of meagre choruses decreased during boat passages, the levels resuming shortly after. Vieira et al. (2021b) showed that these changes are not random, meaning that on average, when boat passages occur, the reductions observed are more pronounced than what would be expected by chance. Furthermore, Vieira et al. (2021b) also presented cases where call activity was interrupted, suggesting that the sound level reduction was due to a reduction in the number of fish calling. This reduction in calling activity is consistent with the findings from other species, such as the reduction in chorusing of silver perch (another sciaenid, *Bairdiella chrysoura*) in response to whistles from predator bottlenose dolphins (Luczkovich et al. 2000) or

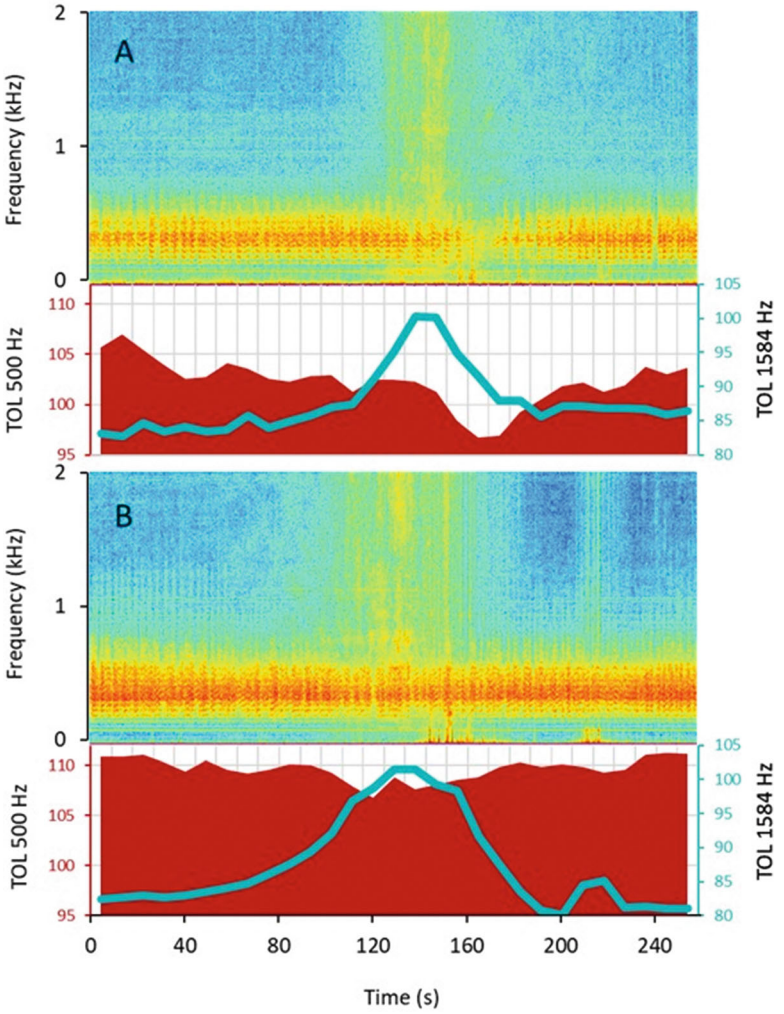



**Fig. 3** Example of the effect of a boat passage in the meagre chorus of (**a**) medium (100–110 dB) and (**b**) high intensity (>110 dB). Note that in (**b**), it is possible to observe a chorus dominated by long grunts with a high rate of overlap, forming a mostly continuous roar

the response of the Lusitanian toadfish that did not produce calls during boat noise playbacks (Alves et al. 2021).

Interestingly, other species have been shown to attempt to compensate for communication loss due to increased noise levels. For example, Picciulin et al. (2012) observed that the brown meagre *Sciaena umbra* increased pulse rate with repeated boat passages. Notice, however, that in cases where real boats were used (including the results presented here), the effect of boat noise could not be disentangled from those of visual and mechanical stimuli associated with the boat passage.

Here, it was further evaluated if the reaction in calling activity was affected by chorus intensity, and significance was found in the interactions between the main effect factors (period relative to the boat passage and chorus intensity classes). It was observed that when the chorus is of medium or high level, the reduction in energy of the chorus occurred more consistently (85% of the boat passages analyzed). In choruses of low intensity, a decrease in the chorus was observed in only 67% of boat passages. However, a higher variability in the response when the chorus had lower intensity was noticeable, i.e., the range in chorus sound level upon boat passages was larger (see boxplots in Fig. 1). Still, in most cases, a decrease in sound level was observed, and in some cases, disruption in the chorusing behavior was observed. Note that at the lowest chorus intensity, the increase in TOL 500 Hz due to the boat noise might be a confounding factor. Also, other confounding effects are associated with the fact that initial chorus intensity may result from the combination of the number of active calling individuals, individual calling activity, and the proximity to the hydrophone.

Whether the observed changes impact the reproductive behavior remains to be investigated. Recently, Amorim et al. (2022) reported a field experiment aiming to determine boat noise effects on Lusitanian toadfish reproduction. This study reported that boat noise elicited a stress response, reduced calling activity, and decreased the likelihood of a male receiving eggs. Although Lusitanian toadfish might be considered particularly vulnerable as they will lose their offspring if they abandon their nesting site, meagre may also be impacted. Sciaenids, as many other groups of species like groupers or cod (Zemeckis et al. 2019; Hughes et al. 2020), form spawning aggregations at predictable times and locations, which might make them particularly vulnerable (Erisman et al. 2017). Although these species are not attached to nesting sites, they are dependent on particular spawning grounds, and their fitness may be affected if they need to escape their spawning grounds to avoid noise or other stressors. Meagre use estuaries for spawning grounds where shipping and boating are often pervasive. If individuals continuously react to boat presence throughout the breeding season, this might have a significant impact on the mating rituals of this species. The permanence of aggregations in particular sites with chronic boat noise calls for further studies addressing possible fitness impacts (for further discussion, see de Jong et al. 2020).

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# Impacts of Anthropogenic Sounds on Reef Fish

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Aléxia A. Lessa, Viviane R. Barroso, Fábio C. Xavier, and Carlos E. L. Ferreira

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**Abstract**

Anthropogenic noise is now recognized as a global pollutant and a major threat to marine biodiversity. It affects the behavior, physiology, communication, and reproductive success of many marine organisms. However, there are several remaining gaps in its effects on certain taxa, such as reef fish. This chapter provides an overview of studies regarding the impact of noise on reef fish. Important steps to guide future research are also discussed.

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Anthropogenic noise · Reef fish · Behavior · Physiology

## Introduction

Biodiversity is a critical functional link to key ecological processes in reef systems and the consequent quality and provision of ecosystem services (Moberg and Folke 1999). Impacts such as overexploitation, eutrophication, and climate change are some of the major drivers acting synergistically and contributing to eroding reef resilience worldwide (Hughes et al. 2017). Noise pollution, despite having been exponentially growing in the last decades (Bittencourt et al. 2020; Duarte et al. 2021), is still neglected as a major threat to biodiversity with no proper management actions.

The underwater soundscape is formed by three main sound sources: biotic, abiotic, and anthropogenic, and it is an important proxy for understanding many ecological processes (Pijanowski et al. 2011). The understanding of how anthropogenic noise affects marine ecosystems has exponentially increased in the last few years (Duarte et al. 2021). Many studies have shown the increasing contribution of different anthropogenic noise sources in marine systems (Duarte et al. 2021). However, there are still many gaps in the effects of noise on species and ecological processes. Anthropogenic noise can affect behavior (Leduc et al. 2021; Nedelec et al. 2022; Velasquez-Jimenez et al. 2020), physiology (Fakan and McCormick 2019; Mills et al. 2020; Staaterman et al. 2020), and survival (McCormick et al. 2018) of different reef fish species, and doing so can exert direct and indirect ecological effects in the dynamics of reef systems.

Sound plays a crucial role in the dynamics of different species and consequently marine environments, including reefs. Due to the efficient propagation of sound waves, hearing is a primary sense in aquatic environments (Mooney et al. 2012; Putland et al. 2019). The acoustic signature of a reef, for example, is used by larvae for guidance during settlement and habitat use (Lillis et al. 2014). These acoustic cues can also be used in intra and interspecific interactions, including reproduction, territory and nest defense, foraging, predator-prey interactions, and many others (Popper and Hawkins 2019). Low-frequency noise (below 1000 Hz) from small vessels is the most frequent source of noise impact in shallow water areas (Bittencourt et al. 2014, 2020), and can overlap the same frequency range of hearing and sound production in many reef fish species (Lobel et al. 2010; Putland et al. 2019). A detailed understanding of the effects of different anthropogenic noise frequencies on reef species is critical to elaborate management rules to minimize impacts on species and critical functional processes.

Reef fishes are a diverse and abundant group in reef systems (Parravicini et al. 2013), being a classical ecological model for understanding the effects of different anthropogenic pressures (e.g., anthropogenic noise). Reef fish, as well as all vertebrates, can hear, but the aspects of their hearing capabilities are less known. There is

still a large gap in which species are vocal, and the types of vocalizations that are used (Ladich 2014). Many reef fish species use acoustic cues throughout their lifecycle, from settlement to habitat use (Gordon et al. 2018; Radford et al. 2011; Simpson et al. 2005). Damselfishes, for instance, are a great model for acoustic ecology studies, as territorial behavior makes them comparatively easy for experiments both in the lab and in the field. Also, vocalization in Damselfishes has been widely studied for territory and nest defense to reproductive interactions (Frederich and Parmentier 2016).

This chapter summarizes the main results found in the last few years regarding the effects of anthropogenic noise on reef fishes. The main topics for future research are also discussed.

## Effects of Anthropogenic Noise on Reef Fish

### Behavior

Anthropogenic noise affects different reef fish behavior across their life cycle, with behavioral changes usually being the first response of organisms to different environmental drivers (e.g., temperature, salinity). Looking at early life stages, Holles et al. (2013) showed primary evidence of how boat noise can negatively affect the selection of habitat by reef fish species. Later, a similar study indicated a disruptive effect of anthropogenic noise on the natural rate of settlement in reef fish larvae (Simpson et al. 2016a). The effects of noise on larvae settlement have been detected not only for tropical species but also for temperate ones (Jung and Swearer 2011). Those significant effects at early life-history stages could have detrimental effects on both populations and community levels. Survival rates are other important metrics used to evaluate the effects of anthropogenic noise on reef fishes. Recently, Nedelec et al. (2022) used both field and lab experiments to demonstrate how limited boat activity leads to greater survival rates of offspring. In "limiting-boat" conditions, the offspring of the spiny chromis *(Acanthochromis polyacanthus)* were more likely to survive, when compared to "busy-boating" conditions. The same pattern was seen for the ambon damselfish *(Pomacentrus amboinensis)* where less than one-third of recruits survived in boat noise treatments (Simpson et al. 2016b). Though, this is not always the pattern, as the survival rates of two other damselfishes did not alter after noise exposure in the laboratory (Fakan and McCormick 2019). Thus, these data suggest that more studies are needed to determine the differential effects of anthropogenic noise at earlier life stages of reef fishes.

Foraging is a crucial behavior not only for the performance of individuals but also for maintaining population dynamics since habitat use and food selection by consumers have been widely recognized to affect lower and higher trophic levels (Stephens et al. 2008). Some studies using damselfishes showed that individuals submitted to noisy conditions had lower foraging rates. For instance, a planktivorous damselfish, *Chromis chromis,* reduced its bite rates in the presence of heavy nautical traffic (Bracciali et al. 2012). The same pattern was observed for land-based

(Leduc et al. 2021) and underwater noise pollution (Lessa et al., *in prep)* on the *Stegastes fuscus* foraging behavior. Leduc et al. (2021) also experimented with flight-initiation responses to predators under elevated noise conditions, and found significantly reduced fleeing distance in these same conditions. Foraging in rays was also affected by noise pollution, since boat noise induced escape behavior (Berthe and Lecchini 2016; Mickle et al. 2022).

Increased refuge behavior is a common response of some species to a threat (Cooper and Blumstein 2015). For instance, territorial species are unable to move away in the presence of a stressor, due to their limited home range and swimming abilities. In a study with the red-mouth goby *(Gobius cruentatus)* noise playback significantly increased its refuge time for short-time exposures (Sebastianutto et al. 2011). A similar pattern was detected for the domino damselfish *(Dascyllus trimaculatus)* under noisy conditions (Nedelec et al. 2016).

Acute stress caused by noise may also increase aggression levels and, therefore agonistic interactions in reef fishes. Some studies found increasing aggressiveness in the presence of noise. Mills et al. (2020) highlighted that orange-fin anemonefish *(Amphiprion chrysopterus)* increased agonistic interactions toward heterospecifics when exposed to boat noise playback in two different exposure experiments. The same pattern was observed for the spiny chromis *(Acanthochromis polyacanthus)* males guarding nests exposed to boat noise playback (Nedelec et al. 2017). Increased territoriality in the presence of noise can affect other important aspects of reef fish life cycle. If they spend more time being territorial, they have less time to spend on other crucial daily activities, including parental care and mating, which may directly affect energy budget and individuals' fitness. However, Lessa et al. (*in prep*) found that noise does not necessarily increase agonistic interactions of the dusky damselfish *S. fuscus.* Therefore, the trade-off between territorialism and other crucial behaviors (e.g., parental care and mating) during noise exposure should be investigated.

## Physiology

Physiological responses, mostly associated with stress, are a central topic of discussion on the effects of noise on marine life. Noise can lead to primary (e.g., hormone release) and secondary responses (e.g., ventilation and heartbeat changes). For the first response, Armstrong-Smith (2016) observed elevated levels of corticosteroids in two reef fish species: starved green chromis *(Chromis viridis)* and spiny chromis *(Acanthochromis polyacanthus),* while Staaterman et al., (Staaterman et al. 2020) showed heightened whole-body cortisol levels of *Halichoeres bivittatus* to boat noise. Another physiological marker recognized as a stress response to boat noise was testosterone. Orange-fin anemonefish *(Amphiprion chrysopterus)* had elevated testosterone levels in short- and long-term experiments of noise exposure (Mills et al. 2020).

Noise can also have secondary physiological responses in reef fishes. An experimental study with embryos of the staghorn damselfish *(Amblyglyphidodon*

*curacao)* showed that heart rates significantly increased after exposure to 2-stroke and 4-stroke boat noises (Jain-Schlaepfer et al. 2018). Interestingly, the 4-stroke had a lesser effect on heart rates than the 2-stroke engines, where the latter had more than twice the effect. Possibly because the spectral content in terms of sound pressure levels, sound exposure levels and particle acceleration of the 2-stroke powered-boats was greater, when compared to the 4-stroke engines. Similarly, Fakan and McCormick et al. (Fakan and McCormick 2019) tested whether boat noise affected the early life history traits of the cinnamon clownfish *(Amphiprion melanopus)* and the spiny chromis *(Acanthochromis polyacanthus)*. The results showed that both species had elevated heart rates with the playback of boat noise compared to ambient noise. However, when looking at morphological traits the results diverged between species.

## What's Next?

### Starting from Scratch

Acoustic signals are used throughout a fish life cycle for orientation and settlement, and for different social interactions, including intra- and interspecific ones (Amorim 2006; Amorim et al. 2013; Gordon et al. 2018; Lobel et al. 2010; Radford et al. 2011). However, many species do not have their hearing range or the vocal repertoire fully described. This huge gap in acoustic ecology is keeping us distant from an integrative evaluation of the impacts of noise on wildlife. Therefore, future studies must focus on basic information about reef fish bioacoustics (e.g., hearing sensitivity, acoustic signatures), rather than just using applied science to evaluate the effects of noise.

### Integrative Approaches

Most studies focused on behavioral responses, nonetheless only a small number of studies explore the physiological responses of reef fishes to noise or even the interaction of both (see Mills et al. 2020). In addition to behavioral responses, physiological indicators of stress can also be used as proxies for the deleterious effects of noise. Physiology plays a vital role in understanding the mechanisms behind the effects of what we can see, such as changes in behavioral patterns. Thus, an important question may arise: how can physiological and behavioral responses be combined to address conservation efforts? Having longer experiments using combined response approaches (e.g., behavioral and physiological) is crucial to better understanding the effects of noise on wildlife. Also, it is important to evaluate the responses of key species, with different functional roles, to have a representative set of the ecological effects of noise, at both community and ecosystem levels.

## Multiple Contexts

From a conservation perspective, the effects of noise under multiple stressors (e.g., overexploitation, climate change, habitat loss) are still a topic we know little about. The responses of organisms to multiple stressors can be synergistic when the effect is greater, or antagonistic when the effect is minor (Côté et al. 2016). In natural systems, where the multiple-stressor-scenario is usual, isolating the effect of a single threat is a major challenge, and that is why most studies use a single-stressor approach in the field or prefer to run lab experiments, with controlled conditions. However, these results are not always realistic, as individuals and species may vary their responses depending on different contexts (Purser et al. 2016; Radford et al. 2016). Thus, experimental designs for multi-stressor studies should be explored and conducted to have a more naturalistic risk assessment of noise and, therefore, assist in conservation efforts.

## Across Scales

It is clear that there are still remaining gaps in reef fish responses to noise (Fig. 1) at different ecological and time scales. Such responses can vary with the type (e.g., low or high frequency sounds), duration, and time of exposure as well (chronic or acute, daily fluctuations) (Holles et al. 2013; McCormick et al. 2018). Although anthropogenic noise can be transient, there is an increasing number of noisy sources in the marine environment, such as ships, boats, wind farms, coastal constructions, seismic surveys, and many others (Duarte et al. 2021; Popper and Hawkins 2019). The continuous exposure to these sources may lead to habituation in some fish species, sometimes with recovering and even compensating for those stimuli (Nedelec et al. 2016; Picciulin et al. 2012). Therefore, extrapolation of short-term responses ranging from individual to population levels must be treated with caution, as some species may habituate and have different responses over time (*Pomacentrus amboinensis*: Holmes et al., 2017; *Dascyllus trimaculatus*: Nedelec et al. 2016; *Halichoeres bivittatus:* Staaterman et al. 2020).

Habituation itself is a controversial theme, as it may be associated with other unexplored topics, such as temporary and/or permanent noise-induced threshold




**Fig. 1** Conceptual diagram of the main topics for future research

shifts. Furthermore, defining response patterns and applying them from one species to another is a very simplistic approach, as some responses may be species-specific. For instance, Fakan and McCormick (2019) showed different responses in the morphological development of two damselfish species to the same noise. In addition, intrapopulation variation should also be investigated in more detail. Individuals of the same population cannot be treated as a homogeneous unit, as they can respond differently to anthropogenic stressors, due to intrinsic and extrinsic factors (Harding et al. 2018). Further research is needed to better understand: (1) how fish responses may change over time, and (2) how those responses change inside a population and through different taxa. Moreover, (3) how those individual responses have ecological significance and are indeed affecting population dynamics.

## Conclusions

Anthropogenic noise is a major concern for marine systems since noise has been recognized as a global pollutant. Despite recent efforts, the extent of knowledge on how it affects certain groups, like reef fish species, is limited. Some species can be affected by noise; however, a wide range of this impact on reef systems has not yet been determined and much work remains to be done. For instance, information about which species are more or less vulnerable to noise is crucial for management and decision-making in the future. There is also a lack of important basic bioacoustics information on reef fishes that would considerably enhance our understanding. Future research may explore the topics highlighted here for a better understanding of the range of effects of noise in reef systems.

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# Impact of Motorboat Noise on Vocalizations of Nesting Plainfin Midshipman Fish

61

Mackenzie B. Woods, William D. Halliday, Sigal Balshine, and Francis Juanes

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**Abstract**

As noise pollution in coastal environments continues to grow, it is imperative to understand how it may be negatively impacting marine life. The study in this chapter tested whether noise from a real motorboat would alter the vocalizations of nesting plainfin midshipman fish (*Porichthys notatus*), a soniferous toadfish

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that breeds in the intertidal zone and uses its vocalizations for courtship and defense. The results show evidence that wild plainfin midshipman guarder males may decrease their agonistic vocalizations in periods where they are exposed to large amounts of boat noise. There was an overall decrease in agonistic vocalizations on the nights when boat noise trials were run compared to control nights. While there was not a statistical difference in the number of agonistic vocalizations exhibited in sequential 10-minute periods with and without boat noise in the same night, there was a trend of fewer vocalizations during boat noise periods. This study helps elucidate how boat noise affects midshipman vocalizations in their natural habitat at the level of the individual, which has never been tested in plainfin midshipman fish and has scarcely been studied in other species.

**Keywords**

Sound · Behavior · Noise pollution · Toadfish · Soniferous fishes · Acoustic communication

## Introduction

Many marine animals rely on sound to communicate, perceive environmental cues, and detect prey and predators (Myrberg 1997). Anthropogenic noise in the ocean has risen drastically over recent decades, causing widespread changes to underwater soundscapes (Duarte et al. 2021). Contributing to these changes are activities such as construction, seismic exploration, and commercial shipping. In coastal environments, one of the primary contributors to noise pollution is recreational boating (Hermannsen et al. 2019). Accordingly, understanding the impacts of noise pollution on underwater soundscapes has become a major focus of current conservation research (Slabbekoorn et al. 2010; Duarte et al. 2021).

Many fishes use sound to communicate, and even nonvocalizing fish species rely on low-frequency sound for environmental perception, prey and predator detection, and navigation (Popper 2003; Slabbekoorn et al. 2010; Simpson et al. 2016). Noise pollution can have many negative effects on fish, including increased mortality by predation (Simpson et al. 2016), temporary hearing loss or altered auditory thresholds (Smith et al. 2004; Vasconcelos et al. 2007), physical damage to hearing structures (McCauley et al. 2003), distribution shifts away from noisy environments (Paxton et al. 2017), physiological stress responses (Wysocki et al. 2006; Nichols et al. 2015; Mills et al. 2020), numerous behavioral changes (Sarà et al. 2007; Sebastianutto et al. 2011; Nedelec et al. 2017; Mills et al. 2020), and masking of vocalizations (Vasconcelos et al. 2007). In addition to masking, noise can cause individuals to alter the amplitude, frequency, or duration of their vocalizations or to reduce how often they vocalize in the presence of noise (Holt and Johnston 2014; Brown et al. 2021).

In this study, a model soniferous toadfish species, the plainfin midshipman (*Porichthys notatus*), was used to investigate how boat noise affects fish vocalizations. Plainfin midshipman fish rely on acoustic communication for mate attraction

and defense (Zeddies et al. 2010). By rapidly contracting specialized sonic muscles attached to the gas-filled swim bladder, they produce three vocalizations: hums (tonal and lasting several minutes to over an hour), grunts (~0.5 s and either produced alone or in a series called a grunt train), and growls (frequency modulated and lasting up to several seconds; Brantley and Bass 1994; McIver et al. 2014). As the plainfin midshipman is nocturnal (Brantley and Bass 1994; McIver et al. 2014), they vocalize most often at night – hums in particular are often solely produced at night (McIver et al. 2014), with the chorus of hums peaking around midnight to 1 a.m. (Halliday et al. 2018; WDH and MBW, pers. obs.).

Every year in the late spring, plainfin midshipman fish migrate from deep waters to the shallow intertidal zone to breed. Nesting males endure a long (>60 days) and energetically costly parental care period in the intertidal zone (Bose et al. 2016), a habitat exposed to frequent noise from nearshore boat traffic (Halliday et al. 2018), during which they attract females using a mate advertisement hum, and care for their multiple clutches of eggs while continuing to attract new females (Brantley and Bass 1994). For spawning to occur, females must be able to hear and locate a male within his nest (Zeddies et al. 2010). Additionally, plainfin midshipman males use agonistic grunts and growls to defend themselves, their nests, and their young from rival males and predators (Brantley and Bass 1994; Woods et al. 2022). Consequently, masking of these vocalizations could have severe reproductive consequences and interfere with their ability to protect themselves and their territory.

The low-frequency noise generated by boats (e.g., recreational motorboats, commercial shipping vessels) can be particularly deleterious to species whose hearing is most sensitive to low-frequency sounds, and to those species that produce low-frequency vocalizations necessary for con- or heterospecific communication (e.g., defense) or for mate attraction. Like many fishes, the plainfin midshipman is most sensitive to low-frequency sound (Ladich 2014). The hum produced by nest-guarding, or "guarder" males, has a fundamental frequency of ~100 Hz, with dominant harmonics between 200 Hz and 400 Hz (Sisneros 2012; Halliday et al. 2018), while grunts and growls have a fundamental frequency of ~60–120 Hz (McIver et al. 2014). Therefore, plainfin midshipman vocalizations fully overlap the frequency range of boat noise (Halliday et al. 2018), resulting in the potential for masking of their crucially important mating hums, in addition to their agonistic low-frequency grunts and growls used for defense.

Animals sometimes alter the spectral (e.g., frequency, amplitude) or temporal characteristics of their vocalizations to overcome masking – a phenomenon known as the Lombard effect (Brumm and Zollinger 2011). Animals may also reduce their overall rate of vocalizations in the presence of noise in some cases (Miksis-Olds and Tyack 2009). This could be detrimental when the vocalizations are necessary to convey important signals, such as attracting a mate or deterring intruders. Recent work has shown that plainfin midshipman males can exhibit the Lombard effect when exposed to an artificial noise stimulus producing a tone of ~118 Hz along with several weaker harmonics (Brown et al. 2021). In that study, the males increased the amplitude of their hums, lowered the frequency of their vocalizations, and produced fewer of all three vocalizations (hums, grunts, and growls) in the presence of

artificial noise. However, that study used a relatively low-amplitude artificial noise stimulus that was both overpowered by and greatly resembled plainfin midshipman hums (Brown et al. 2021). How boat noise affects plainfin midshipman vocalizations in the wild has never been investigated.

This study expands the investigation of the Lombard effect by using a true anthropogenic noise stimulus (a motorboat driven near the nests) to explore whether plainfin midshipman alter their vocalizations as a strategy to overcome masking by anthropogenic noise pollution. It was predicted that boat noise would reduce the number of vocalizations produced, resulting in fewer vocalizations during boat noise periods compared to ambient control periods within the same night. In the future, the data will be further analyzed to quantify additional parameters of the vocalizations, including any changes to frequency and amplitude in response to boat noise.

## Methods

### Study Location and Nest Construction

Experiments were performed in June 2022 at a private beach in the Hood Canal region of Washington, USA. Four artificial nests were constructed from 929 $cm^2$ concrete tiles placed on small rocks to partially prop them up off the substrate on one side to encourage fish to enter and take them up as a nest. An area of the beach with the fewest natural nests was selected, and the artificial nests were placed at least 5 m away from each other and from all suspected natural nests (i.e., medium to large rocks that looked like they would make a good midshipman nest and/or rocks that had indications of being a nest, such as having a dug-out entrance or a shell midden outside a potential entrance).

The first three nests were placed in a horizontal row, parallel to the shoreline. As there was not room for a fourth nest in line with the first three due to the presence of many suspected natural nests, the fourth was placed 6 m behind (away from the shoreline) the third nest (Fig. 1).

The artificial nests were checked for occupancy every few days at low tide or by snorkelers when the nests were underwater. Once a male was discovered in one of the nests guarding a brood of eggs, the nest was covered with plastic mesh (1 $cm^2$ openings) weighed down by rocks along the edges to ensure that the same focal male remained for the duration of the experiment, as well as to exclude rival males, predators, or additional mates (Brown et al. 2021; Woods et al. 2022).

### Noise Stimulus

A 4-m aluminum hull boat with a four-stroke 9.9 horsepower engine was used as a noise stimulus. During the boat noise periods on each of the three trial nights, the boat was driven at full throttle continuously for 10-minute intervals, in loops with the closest point approximately 20 m from the nests (Fig. 1a). This distance was chosen

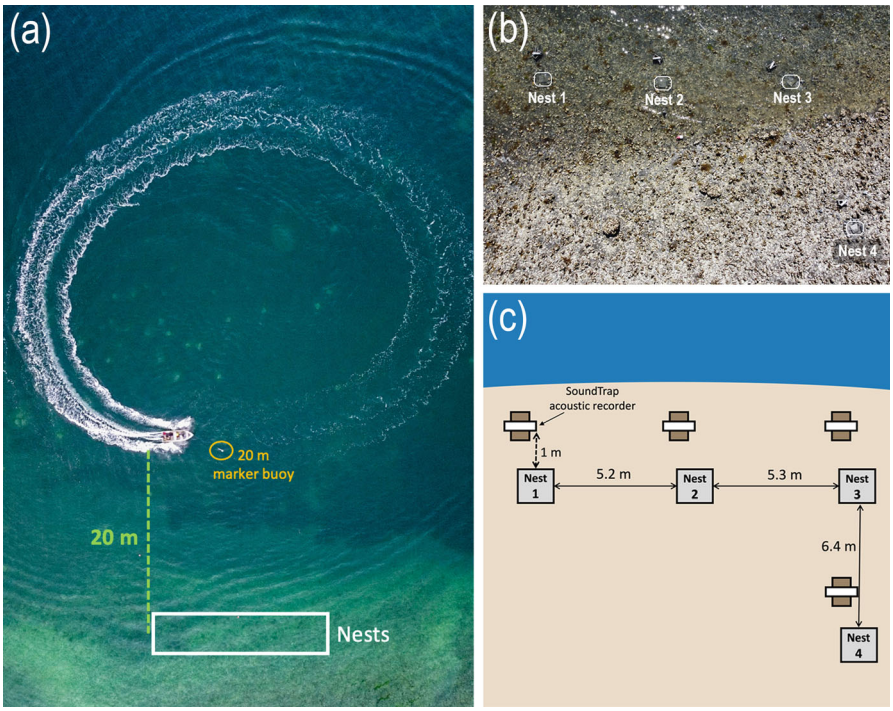



**Fig. 1** Nest layout and experimental setup: (**a**) An aerial photograph of the motorboat stimulus being driven in continuous loops during a boat noise period on a trial day. The four experimental nests are underwater in the area outlined in white. The closest point of the boat loop was 20 m away from the nests (shown in green) and was marked with a 20 m marker buoy (shown in yellow) to guide the boat loops; (**b**) aerial photograph of the artificial nests and acoustic recorders at low tide; and (**c**) schematic diagram of the layout of artificial nests and SoundTrap acoustic recorders on the beach. The distances between each of the four experimental nests and the acoustic recorders are noted. Recorders were placed 1 m in front of each nest. The waterline shown is at low tide; the nests were underwater during all experimental trials, as seen in (**a**)

because the nests were in shallow water, and therefore the boat noise they would naturally experience would rarely come from boats driven directly over the nests. The boat was always driven by the same individual (MBW) for consistency. SoundTrap acoustic recorders (Ocean Instruments New Zealand) placed 1 m away from each nest (Fig. 1c) were used to record the vocalizations of the focal fish as well as the received sound levels in both the boat noise and ambient control periods (Fig. 2).

## Experimental Trials

Because plainfin midshipman fish are nocturnal, this experiment was conducted at dusk. Trials were conducted between 20:10 and 21:30 for 5 consecutive days

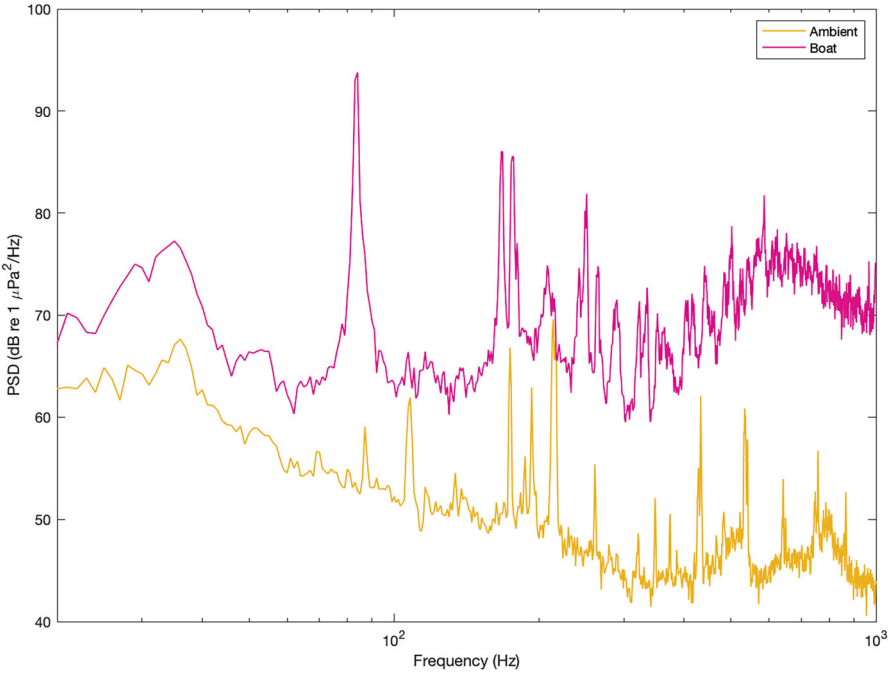



**Fig. 2** Power spectral density (PSD) of the sound levels in the ambient trial periods (gold), with no motorboat or introduced noise, and the boat noise trial periods (pink), during which a motorboat with a 9.9 hp engine was driven in continuous loops at full throttle

between June 23 and 27, 2022. The 5-day sequence consisted of two control nights (with no boat noise) and three trial nights (with boat noise) in the following order: Pre-Trial Control Night, Trial Night 1, Trial Night 2, Trial Night 3, and Post-Trial Control Night (Fig. 3). On the first day, acoustic recorders were deployed, and that night (the "Pre-Trial Night") the fish were left undisturbed, and their vocalizations were recorded. The second night (Trial Night 1), boat noise trials began. Immediately before trials, the boat was quietly rowed until it was in deep water far away (>150 m) from the nests, where the engine was then started and allowed to warm up before exposing the fish to the boat noise. Starting approximately 45 min before sunset, the first boat treatment began.

Each boat noise treatment was 10 min long and consisted of the boat being driven in loops continuously (Fig. 1a). After 10 min, the boat engine was turned off and the first 10-minute ambient control period began. This was repeated until there had been three boat noise periods. There was then a 10-minute follow-up period once the boat engine was shut off for the final time (Fig. 3). Before each boat noise period, the boat was started far away (>150 m) from the nests (after a full 10 minute of engine-off ambient), and the 10-minute noise period only commenced when the boat started the full-throttle loops. The interval between the boat engine being started and the beginning of each boat noise period were not analyzed for vocalizations.

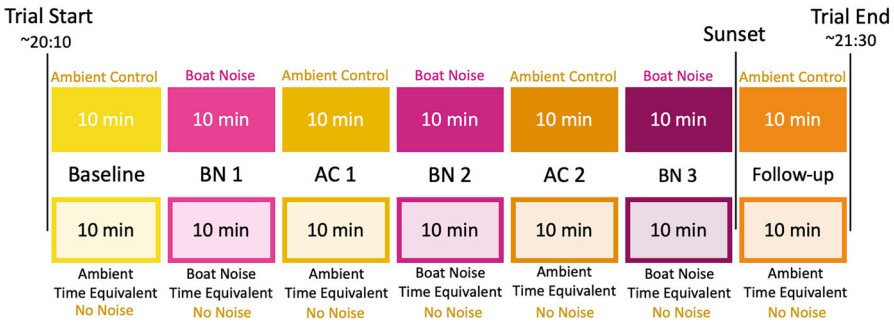



**Fig. 3** Nightly timeline showing the order of trial periods during which vocalizations were analyzed. Trial nights (top) had alternating ambient and boat noise periods, and control nights (bottom) had no boat present and no noise introduced. The colors correspond to the colors of each trial period in the results figures, with variations of yellow/gold/orange as ambient control periods and variations of pink/purple as boat noise periods. (Note: The darkening colors correspond only to time of night and not stimulus intensity)

This sequence was repeated on each of the next two nights (Trial Night 2 and Trial Night 3). On the fifth and final day (Post-Trial Night), the boat was not present and no noise was introduced, but the recordings were analyzed for vocalizations. The two control nights (Pre- and Post-Trial Nights) were used to control for any patterns or variation in vocalizations relating to time of night.

## Acoustic Analysis

All acoustic recordings were analyzed using Raven Pro acoustic software. Spectrograms were built in Raven Pro with 7000 samples, a Hanning window with 50% overlap, and time and frequency axes set to 10 s and 1000 Hz, respectively. The 10 minutes before the boat was initially turned on was scored as a baseline period, and each of the 10-minute boat noise and ambient control periods were scored. All vocalizations from the focal male of each nest were annotated in Raven Pro, where the peak frequency and duration of each vocalization were measured, and number of vocalizations per 10-minute period were counted. Vocalizations were assigned to the focal male based on the higher relative amplitude of the vocalizations suspected to be from the focal male compared to fainter vocalizations within the same file.

## Statistical Analysis

All statistical analyses were conducted in RStudio (v.2022.07.2; R Core Team 2021). Agonistic vocalizations were defined as the sum of grunts and growls for each fish within each trial period, over each of the 5 days.

A negative binomial generalized linear mixed effects model (GLMM) (*glmmTMB* package; Brooks et al. 2017) was fit to these counts of agonistic vocalizations over the 3 trial days (i.e., days that included the boat noise treatment) to examine whether there was a difference in the number of agonistic vocalizations during boat noise periods versus ambient control periods. Nest ID was included as a random intercept to account for the within-subjects study design, and acoustic treatment (ambient control or boat noise) was the fixed effect. Likelihood ratio tests were used to assess the statistical significance of the acoustic treatment effect.

A second negative binomial GLMM was then used to assess whether there was a difference in overall vocalization count during control nights (Pre-Trial and Post-Trial Nights; no boat noise) or experimental trial nights (Trial Nights 1–3; boat noise present). Again, the nest ID was included as a random effect and the fixed effect for this model was night type (trial or control). Likelihood ratio tests were used to assess the statistical significance of the night type effect.

Statistical analyses were not performed for mating hum vocalizations because there were too few hums present in the experimental recordings.

## Ethical Note

All procedures in this study complied with guidelines set by the ASAB/ABS (2012) and the Canadian Council on Animal Care (Olfert et al. 1993) and were approved by the University of Victoria Animal Care Committee (AUP: Juanes-2021-012). All procedures implemented were noninvasive. Following experimentation, the mesh was removed from the nests, all equipment was removed, and fish were allowed to continue guarding their brood in their artificial nests at the study site for the remainder of the breeding season.

## Results

There was no significant difference in the number of vocalizations during the ambient control periods (including the baseline period) and the boat noise periods on the three trial nights ($\chi^2 = 1.08$, $df = 1$, $p = 0.30$; Fig. 4). However, the figures show that there may have been slightly more vocalizations in the ambient periods compared to the boat noise periods, particularly on the first and second trial nights, during which no fish vocalized in any of the boat noise periods (Fig. 5); however, this pattern was not statistically significant. As expected, there was also no difference in the number of agonistic vocalizations during the ambient and boat noise time-equivalent periods on the control nights ($\chi^2 = 0.27$, $df = 1$, $p = 0.61$; Fig. 5). However, there were overall more vocalizations during the control nights compared to the trial nights ($\chi^2 = 6.95$, $df = 1$, $p = 0.0084$; Fig. 6).

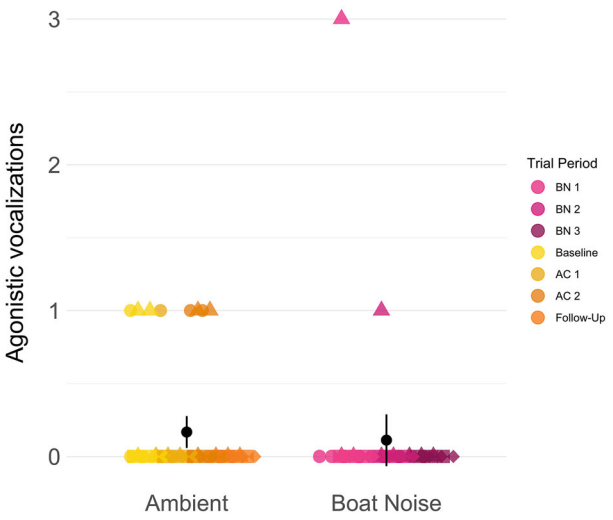



**Fig. 4** The number of agonistic vocalizations (grunts and growls), exhibited by each of four nesting plainfin midshipman guarder males during each 10-minute trial period, comprised of alternating ambient control periods and boat noise periods, pooled across the three trial nights. The shapes correspond to the four individual fish tested. The means and 95% confidence intervals are shown in black

## Discussion

There was evidence that nesting plainfin midshipman guarder males may decrease their agonistic vocalizations (grunts and growls) in periods when they are exposed to large amounts of boat noise. While there was not a statistical difference in the number of agonistic vocalizations exhibited in sequential 10-minute periods with and without boat noise, there was a pattern of slightly fewer vocalizations during boat noise periods. Furthermore, there was a significant overall decrease in agonistic vocalizations on nights when boat noise trials were conducted, compared to control nights.

Our findings are consistent with Brown et al. (2021) which found that plainfin midshipman guarder males elicited fewer vocalizations (grunts, growls, and hums) in the presence of introduced artificial noise. However, the present study is the first to test whether plainfin midshipman fish alter their vocalizations in response to boat noise. Brown et al. (2021) also found that the plainfin midshipman males increased the amplitude of their hums and lowered the frequency of their vocalizations (i.e., exhibited the Lombard effect) in response to the artificial noise stimulus, which had a fundamental frequency of ~118 Hz with several weaker harmonics, and which strongly resembled a midshipman mating hum. The results presented in this chapter are preliminary and represent only a small part of a larger study, which will also examine whether the males at the study site exhibited the Lombard effect in response to motorboat noise by altering the amplitude or frequency of their vocalizations. The

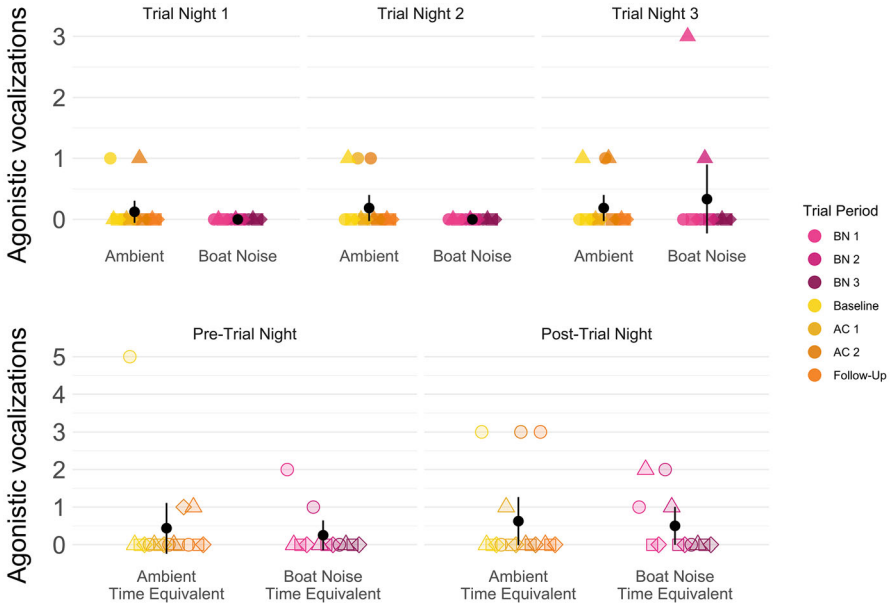



**Fig. 5** The number of agonistic vocalizations (grunts and growls), exhibited by each of four nesting plainfin midshipman guarder males during each 10-minute trial period, comprised of four ambient control periods and three boat noise periods, on each night in the five-night trial sequence. Top: the three trial nights; in these trials, the boat periods consisted of a motorboat being driven continuously at full throttle. Bottom: control nights before and after the three trial nights; there was no motorboat or introduced noise present during any part of the control trials, as the control nights were analyzed to compare the vocalizations in each corresponding period to the trial nights. The shapes correspond to the four individual fish tested. The means and 95% confidence intervals are shown in black

results of this study will help us understand whether the previous results reported in Brown et al. (2021) were due to the males trying to overcome masking by low-frequency anthropogenic noise or if they were trying to outcompete what resembled the hum of an additional rival guarder male.

Several studies have found decreases in fish calling rates in the presence of noise (Ladich 2019), consistent with the results of both the present study and those of Brown et al. (2021). For example, de Jong et al. (2018) tested two species of goby in the lab and found a reduction in acoustic courtship during noise playbacks in both species. This reduction in courtship vocalizations may have contributed to the lower spawning success of painted gobies in the noise treatment group. In the field, Mackiewicz et al. (2021) observed a 32% decrease in male oyster toadfish vocalizations post-exposure to an idling boat engine compared to pre-exposure calling rates. In contrast, increases in calling rate in the presence of noise are commonly observed in birds and mammals to increase signal redundancy – an additional tactic that can be used to overcome masking (Shannon et al. 2016). This increase in calling rate has also been found in fish in a few studies (e.g., Picciulin et al. 2012), but it

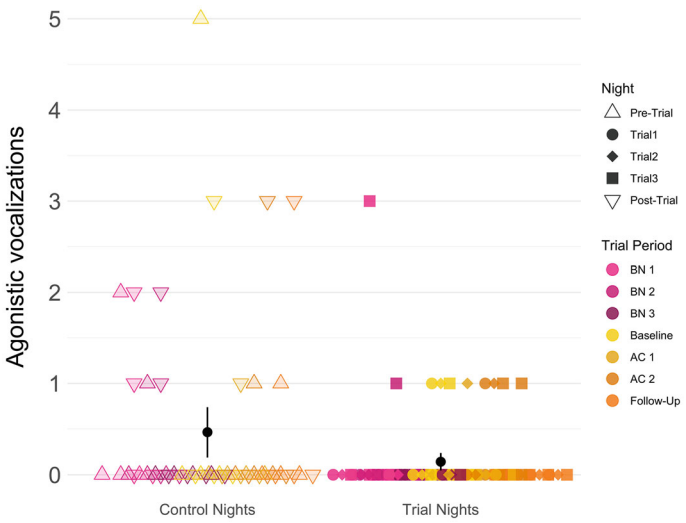



**Fig. 6** The number of agonistic vocalizations (grunts and growls), exhibited by each of four nesting plainfin midshipman guarder males during each 10-minute trial period, comprised of four ambient control periods and three boat noise periods, pooled across the two control nights and the three trial nights to compare the overall number of vocalizations present on nights when boat noise was (trial nights) and was not (control nights) present. In the control night "boat noise" periods (BN 1–3; pink data points), no boat or introduced noise was present, as the control nights were used to compare equivalent time periods of the evening to the trial nights when boat noise was present. During the boat noise periods (BN 1–3; pink data points) on trial nights, a motorboat was driven in continuous loops at full throttle. During ambient control periods (Baseline, AC 1–2, and Follow-Up; gold points), the boat engine was shut off and the boat was far from the nests. The means and 95% confidence intervals are shown in black

appears that it may be a less commonly used tactic among fishes given the number of studies reporting the opposite pattern of decreased calling rates during noisy periods.

The primary limiting factor of the current study was the small sample size. This may be why there was only a slight trend of fewer vocalizations during boat noise versus ambient trial periods but not a significant difference. However, the individual-level data of the four fish discussed in the current chapter are only one component of a larger overarching study; additional fish were tested at the group level. The group-level data contain far more vocalizations and will enable us to better assess the effect of the experimental boat noise on all three types of vocalizations – agonistic grunts and growls and the mating hum. The preliminary group-level data are pointing toward a decrease in the number of agonistic vocalizations during boat noise periods, as well as a potential decrease in humming. Further analysis of the data collected in both the group- and individual-level experiments will contribute to our growing understanding of the ways in which anthropogenic noise alters fish vocalizations.

## Conclusion

It was evidenced that plainfin midshipman males may reduce their calling rate in the presence of motorboat noise, but those results were not conclusive, and additional data would be required to draw firm conclusions. The overall decrease in vocalizations on trial nights compared to control nights indicates that boat noise did decrease the rate of agonistic vocalizations. However, when comparing sequential boat noise and ambient control periods on trial nights, the data visualization suggests that there might be fewer vocalizations during boat noise periods, but that did not turn out to be a significant pattern. The results were likely impacted by the small sample size; however, a complementary experiment was conducted on groups of nesting plainfin midshipman males and the addition of those data (once analyzed) will allow stronger conclusions to be drawn. Future analysis, both on the individual-level data presented here and the group-level data still being analyzed, will include assessing whether frequency or amplitude of midshipman agonistic vocalizations or mating hums are altered by boat noise.

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# Red Drum (*Sciaenops ocellatus*) Interactions with Vessel Noise in the Northern Gulf of Mexico

62

Bennett H. Price and Kelly S. Boyle

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**Abstract**

Red drum (*Sciaenops ocellatus*), a vocal fish species found in waters along North America's East Coast and northern Gulf of Mexico, forms large, vocal spawning aggregations in near-shore waters. These in-shore regions also experience frequent vessel traffic, both commercial and recreational, and resulting noise pollution is predicted to negatively impact sound-producing fishes. Noise pollution can increase hearing thresholds, damage sensory epithelia, and alter behavior. The aim of this research was to determine how often red drum calls are masked and how they respond via sound production. During 2021–2022, vessel noise and fish sounds were obtained via passive acoustic recorders in Saint Andrews Bay, Panama City Beach, Florida. Presented here are preliminary analyses of October 2021. Red drum sounds occurred frequently at this site during spawning season (June–November). Red Drum calling peaks immediately after sunset and significant overlap was found between chorusing and vessel noise during this calling period. The contribution of call pulse number, vessel noise, and sound pressure was tested. Preliminary results indicate red drum decreases call rate when intense boat noise occurs, and calls are masked. This suspected behavior modification and subsequent call masking could decrease spawning success and impact fitness.

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Ongoing analyses of 2021–2022 spawning seasons will address these questions further.

**Keywords**

Noise pollution · Sciaenid · Masking · Passive acoustics

## Introduction

In recent decades, anthropogenic noise has steadily increased in marine coastal habitats, such as bays and estuaries. Vessel noise associated with commercial and recreational boat traffic is a major component of this noise pollution, in addition to construction of renewable energy infrastructure and seismic exploration (Duarte et al. 2021; Hildebrand 2009). Coastal regions experience disproportionate anthropogenic impacts as human activity is coastally concentrated (Crossett et al. 2013). Coastal habitats are important to marine animals, biologically productive, and used by many species at different life stages for foraging, shelter, migration, spawning, and nursery habitat (Seitz et al. 2014).

In addition, sound travels farther in water than in air, making the region of impact of noise pollution much larger, particularly from sources of low-frequency sound (Rogers and Cox 1988). Vessel noise is often low frequency and overlaps with the call spectra and bandwidth of hearing of many fish species (Duarte et al. 2021; Ramcharitar et al. 2006). This makes it ever more critical to assess implications of vessel noise in coastal waters as its effects may be widespread and varied.

Noise pollution has been shown to cause organ and tissue damage, produce shifts in hearing sensitivity in fishes, and alter animal behavior (Duarte et al. 2021). Shifts in behavior can include conspecific interactions (de Jong et al. 2018), habitat use, and changes in acoustic behaviors (La Manna et al. 2016; Picciulin et al. 2012; Vieira et al. 2021). With so many impacts already documented, it is critical to continue to study effects of noise pollution on fish and their sound production. Thankfully, fish sound production can be readily studied via passive acoustic monitoring (PAM) in coastal waters.

## Study

The goal of this study was to observe red drum (*Sciaenops ocellatus*) spawning aggregation vocal behavior in the presence of anthropogenic noise. Red drum is a member of the family Sciaenidae, which includes several economically and ecologically important species (National Marine Fisheries Service 2022). This species ranges throughout the Gulf of Mexico and along the East Coast of North America as far north as Massachusetts. Individuals in the Gulf can reach ~110 cm total length (Bennetts et al. 2019; Hightower et al. 2022). In the summer and autumn, fish form

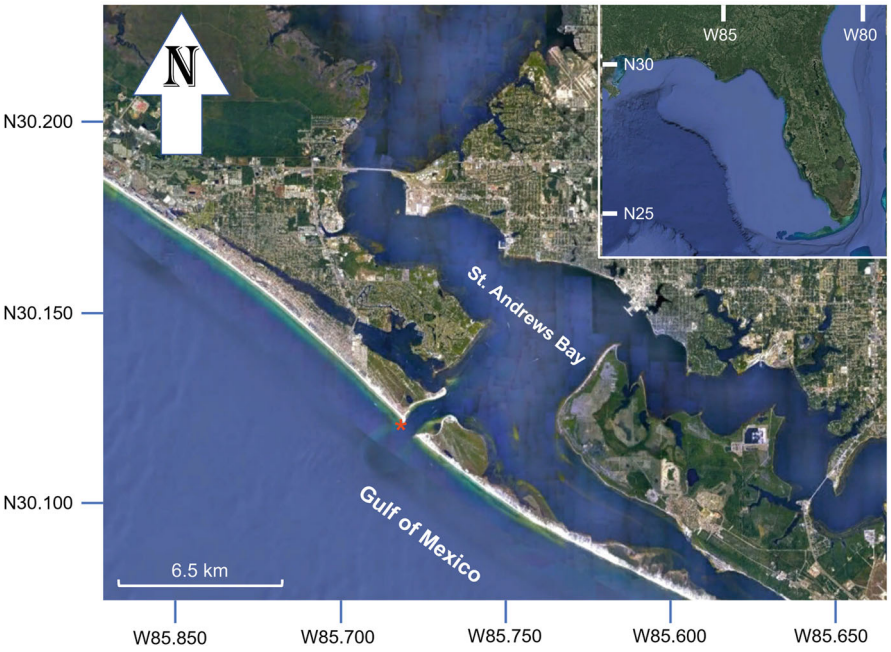



**Fig. 1** Site of recorder (*), St Andrews Bay, Florida, USA

large, high density spawning aggregations in coastal waters (Monczak et al. 2022). These aggregations are associated with intense choruses made by male fish that peak August through October. Peak chorus activity is also associated with seasonal abundance of young of year fish in the months after (Monczak et al. 2022). These sounds consist of multiple repeated pulses in rapid succession and are produced via activation of the sonic muscles near the swim bladder (Tower 1908).

Coastal waters in the northern Gulf of Mexico experience both recreational and commercial boat traffic that co-occur with fish sound production behavior. In 2021–2022, Saint Andrews Bay Pass, which runs through Saint Andrews State Park, Florida (Fig. 1), was monitored with a semiautonomous recorder (SNAP recorder, Loggerhead Instruments) during the red drum spawning season (June–November). The recorder was set to record 45-second files on a 15% duty cycle. For preliminary analysis, every file from a 2-h crepuscular period (30 min prior to sunset, 90 min post sunset) was screened for each day in October 2021. This time was chosen as it is when red drum spawning activity is most prevalent (Holt et al. 1985; Lowerre-Barbieri et al. 2008; Monczak et al. 2022). Files 12 h prior to this period were screened for vessel activity with the focus being on files with high-intensity SPL (>120 dB re 1 μPa) in the 0–1000 Hz band, which encompasses the dominant bandwidth of red drum calls (Fish and Mowbray 1970). Through this, fish and vessel interactions and how fish responded to anthropogenic sound were observed via PAM.

## Hypotheses and Predictions

Sound-producing fish may respond to vessel noise in several different ways. One hypothesis (disturbance hypothesis) is that fish may be disturbed by noise which would cause fish to reduce vocal activity. An alternative hypothesis is that fish may compensate (compensation hypothesis) to increase or alter calls to compete with higher background noise. Finally, fish may ignore vessel noise and not modify vocal behavior (nonresponse hypothesis).

A prediction of the disturbance hypothesis (prediction 1) is that fish might be disturbed and lessen or completely cease calling immediately when exposed to noise pollution which was termed masking avoidance. Vessel noise could also disrupt fish calling immediately after vessel noise (immediate disruption). Fish could also show a decrease in calling outside of vessel noise periods events as the level of exposure to vessel noise throughout the day is increased (daily disturbance) or as the amount of boating during the crepuscular period is increased (evening disturbance). If fish reduce calling at any point due to vessel noise, this will reduce the time spent calling to conspecifics and potentially reproductive success.

A prediction of the compensation hypothesis prediction (prediction 2) is that fish may compensate for background noise via an increase in their call rate during vessel noise (masking compensation) Fish could also increase pulse rate immediately after vessel noise exposure (immediate compensation), increase call rate when prior noise exposure during diurnal hours occurs at a higher rate of occurrence (daily compensation), or increase overall pulse rate throughout the entire crepuscular period, even during intervening quiet times, when vessel noise occurrence increases during these periods (evening compensation). Another means of compensation for increased background noise is the Lombard Effect. This is displayed when an animal modifies its call frequency or intensity to be heard while exposed to competing sounds (Lombard 1911; Zollinger and Brumm 2011). A compensation response during boating events would be wasted effort of the part of the calling fishes expending energy if conspecifics are unable to perceive the acoustic signal.

The nonresponse hypothesis (prediction 3) predicts that fish will not change their call rate, amplitude, or frequency during or immediately after vessel noise, or based on exposure to vessels over a given period (daily or evening exposure). This also represents wasted efforts for reproductive fishes if calls are being masked during these events to the degree that signals are not received, and fish might have greater reproductive successful if waiting to call outside of vessel noise.

## Analysis

A positive correlation between fish pulse rate and the SPL of the 0–600 Hz band was observed. This allowed for use of SPL in this band as proxy for fish calling outside of masking events. During vessel noise, fish calls are only discernible below an SPL where boats fully mask fish calling. This study is also limited in its ability to address whether single vessel noise events impact fish calling rate or amplitude immediately after a vessel passed (immediate disturbance) because the duty cycle and brief file

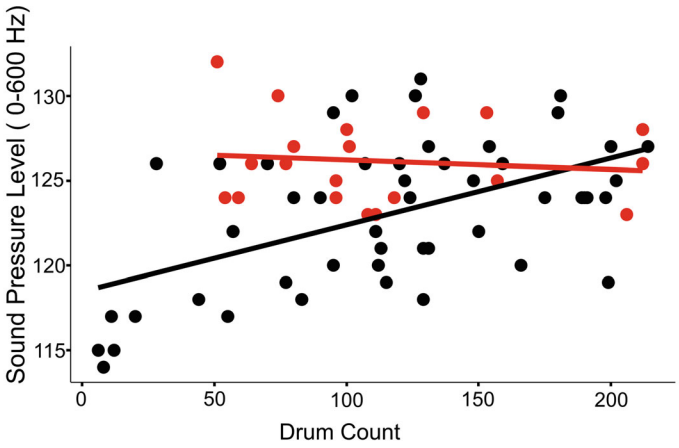



**Fig. 2** Relationship between band (0–600 Hz) sound pressure level (SPL) and red drum call pulse number in the presence and absence of vessel noise ($R^2 = 0.30$). SPL has a positive relationship with drumming in the absence of vessel noise. During vessel noise, however, lower pulse rates are associated with higher SPL, which is consistent with the prediction that high vessel noise (which increases overall SPL) reduces calling

duration limits the opportunity for vessel noise detection, vessel noise cessation, and potential vocal response by fish to occur within 45 s. Analysis of continuous recordings made at the study site in 2022 is in progress. However, preliminary data show that during vessel noise occurrence, the greater sound amplitudes were associated with lower pulse rates, which is consistent with predictions of masking avoidance**,** not masking compensation (Fig. 2). Continued analysis for both 2021 and 2022 full spawning seasons is ongoing.

This decrease in pulse rate could be because fish alter their acoustic behavior or vacate habitat when noise pollution is present. Previous studies have shown similar responses in other species. In another sciaenid (meagre, *Argyrosomus regius*), chorusing was diminished and at times fully disrupted by the passage of vessels in Portugal's Tagus estuary (Vieira et al. 2021). Also, two goby species (*Pomatoschistus pictus* and *Gobiusculus flavescens*) both showed decreases in sound production as part of their courtship when exposed to noise in the lab (de Jong et al. 2018).

This research also looked at how fish responded to vessel noise exposure outside of periods when vessel noise was present. To test for disturbance/compensation as a response to daily levels of exposure, call rate of periods without vessel noise during the crepuscular period were measured and compared to the rate of intense vessel noise (>120 dB, 0–1000 Hz band) 12 h prior to sunset for each day in October 2021.

There was no correlation between prior diurnal vessel noise and subsequent red drum pulse rate during crepuscular periods when vessel noise was absent (Fig. 3a) and thus no evidence for daily disturbance or daily compensation of red drum vocal behavior. Similarly, in October 2021 there was no correlation between vessel noise occurrence during the crepuscular period and red drum pulse rate during quiet intervening times during the same crepuscular period (Fig. 3b). This observation is

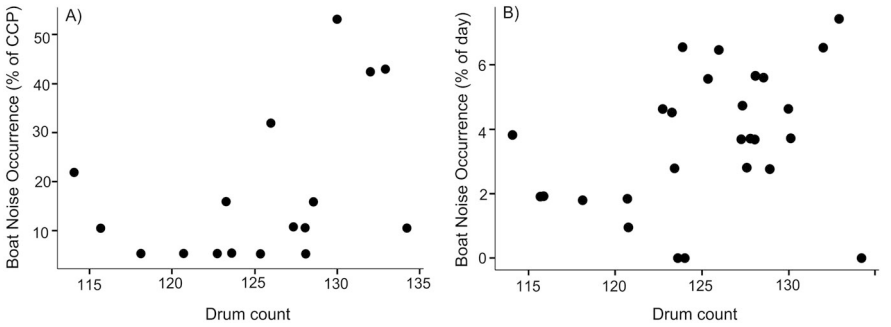



**Fig. 3** Percent of CCP (crepuscular calling period) calculated experiencing vessel noise (>120 dB, 0–1000 Hz) as compared to SPL of files without vessel noise (**a**). Percent of 12 h prior to evening calculated experiencing vessel noise (>120 dB, 0–1000 Hz) as compared to SPL of evening files without vessel noise (**b**). Neither show a significant relationship (**a**) $R^2 = 0.12$, (**b**) $R^2 = 0.18$

not consistent with predictions of evening disturbance or evening compensation. These preliminary findings indicate fish may be undisturbed by vessel noise immediately following vessel noise events. However, these data are still preliminary and limited to October 2021. Analysis of data over the two spawning seasons continues to better understand how vocalizing red drum respond to vessel noise.

These early findings indicate that red drum sound production may be disturbed during vessel noise, but it is not known if or how long sound production is disrupted following noise exposure. Sound production disruption during and following vessel noise exposure was observed in meagre, a related sciaenid species (Vieira et al. 2021). In contrast, in another sciaenid species (Brown meagre, *Sciaena umbra*), fish displayed increased vocalization rates after repeated boat exposure (Picciulin et al. 2012). Though the authors note, boat noise was too great to assess fish sounds during masking. However, in Sarasota, Florida, the presence of ferry boat traffic was found not to impact the vocalizations of Atlantic croaker (*Micropogonias undulatus*) (Luczkovich et al. 2012). Additionally, brown meagre in Venice, Italy did not show any avoidance of louder waterways in a study that used sound production as a measure for fish presence (Picciulin et al. 2022). It was hypothesized that other attributes, such as bathymetry, made noisy habitat desirable even over quieter waters (Picciulin et al. 2022).

Methods continue to be assessed and adjusted to better understand impacts on fish at the study site. Measuring vessel noise exposure 12 h prior to crepuscular calling period may not reveal the full extent of behavioral modification if it is occurring. The impacts of vessel noise in such a highly trafficked area may be compounded over days or even weeks of exposure. The current aim to address this question with further analysis over the two seasons of data collected and measuring vessel noise over greater periods of time.

It is possible that red drum exhibits the Lombard effect to adjust their call features to compensate for increased background noise when vessels are present. Both plainfin midshipman (*Porichthys notatus*) and oyster toadfish (*Opsanus tau*) were recorded with increased amplitude (Lombard effect) during noise exposure to compete with background sound (Brown et al. 2021; Luczkovich et al. 2017).

Interestingly, plainfin midshipman also lowered the frequency of their calls, in addition to increasing amplitude, when noise exposed (Brown et al. 2021). Red drum presents a challenge to measure the Lombard Effect as fish aggregations do not allow for responses in individual fish to be observed. With choruses, an increase in amplitude could indicate individuals in the group increased call amplitude or more fish in the aggregation vocalized. Methods continue to be explored which might better answer how choruses might change based on noise exposure and if the Lombard Effect is occurring with this species.

Another methodological challenge faced is vessel noise is highly variable, both in terms of vessel types producing sound and the source distances from the recorder. Noise source distance impacts the relative contribution of sound pressure and particle motion, which the HTI hydrophone does not measure. Particle motion is the primary way fish perceive sound and so should be considered in noise pollution studies (Higgs and Radford 2012). Additionally, with noncontinuous recordings, there exists a lack precision when addressing impacts of individual instances of boat noise during the time when fish are calling.

Call masking by background noise, either partially or wholly, presents a major issue in communication. Masking has been documented in terrestrial and aquatic environments (Halfwerk et al. 2011; Templeton et al. 2016) and is of major concern because it can obscure biologically important signals. In the May River, South Carolina, it was shown that spaces where fish were communicating via sound also contained vessel noise and this noise overlapped in the sound spectrum with fish calls. The researchers also saw overlap in the time when fish chorused and vessel traffic occurred. This interaction was observed with several sciaenid species during their spawning period (Smott et al. 2018). This study also confirmed multiple soniferous species co-occurring with vessel sound in Saint Andrews Bay Pass. However, the auditory system of red drum will need to be further investigated to fully understand what extent vessel noise is masking calls for this species.

Boat noise in Saint Andrews Bay Pass was highly prevalent and led to substantial levels of masking. Crepuscular calling periods on two nights were over 90% masked. This could lead to whole evenings during the spawning season representing wasted effort if fish are unable successfully communicate or reproduce in these conditions, potentially impacting population fitness. Further analysis of field recordings will be required to assess if this is a common trend throughout the spawning season and in other times of the year. However, the full extent of masking can be difficult to assess as background noise grows more intense. There is a finite window in which masking is observable via PAM (Figs. 4 and 5), and it is still important to know when fish sounds are complete masked. Future work in lab will better address if fish alter calling when exposed to such high-intensity noise.

## Impact

Passive acoustic monitoring Saint Andrews Bay showed that red drum is impacted by masking from vessel noise. During vessel noise, call rate was reduced as SPL of noise increased. This along with the extensive levels of vessel noise and masking

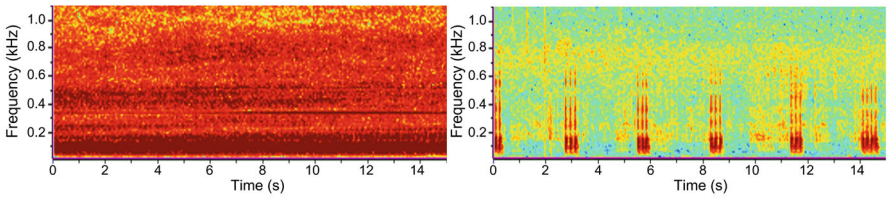



**Fig. 4** Spectrograms showing 15-second intervals of field recordings (0–1.1 kHz). Left – high-intensity vessel noise. Right – six instances of red drum calling with minimal background noise

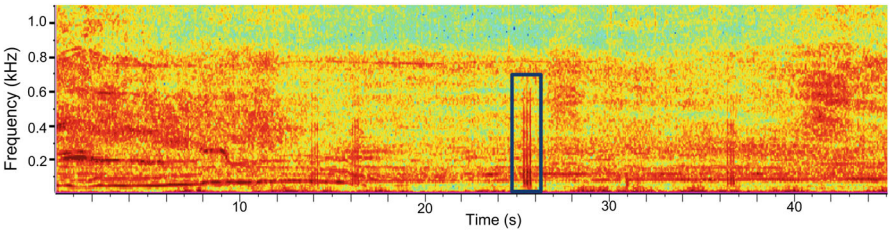



**Fig. 5** A 45-second spectrogram showing red drum calls (highlighted) coinciding with low-frequency (0–1.1 kHz) vessel noise resulting in observable masking

observed means that time spent with reduced calling is likely significant. This large amount of exposure to vessel noise could also be impacting animals in other ways not measurable through PAM, such as hearing threshold changes, tissue damage, or predator–prey interactions. Additionally, there were vocalizations from several other species observed on recordings that may also be impacted by this high rate of vessel noise.

Continuous PAM recording will also allow further insights to animal and vessel noise interactions that can be missed on duty cycle deployments. Immediate impacts post noise exposure may not always be seen on shorter recordings and the full scope of anthropogenic sound may not be fully represented. There is research currently working toward this goal currently with continuous recordings being made in the same location during the 2022 red drum spawning season. These recordings along with continued in lab experiments will allow for greater understanding of about red drum response to vessel noise in the northern Gulf of Mexico. Additionally, future analysis of both 2021 and the 2022 spawning seasons will allow for test of predictions (disturbance, compensation and nonresponse) of these alternative hypotheses of how vocalizing red drum respond to vessel noise.

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# Overview of the Effects of Boat Noise on Lusitanian Toadfish

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Manuel Vieira, Paulo J. Fonseca, and Maria Clara P. Amorim

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**Abstract**

Most marine soundscapes have changed due to the massive presence of anthropogenic noise. Lusitanian toadfish (*Halobatrachus didactylus*) is a vocal fish species that has been recurrently used as a model in both behavioral and physiological studies, making it an excellent species also to understand the effects of aquatic noise. This chapter aims to review what is known about the effects of boat noise on this species and its possible implications. Vocal behavior, hearing, reproduction, and early stages development of the Lusitanian toadfish are summarized, including several studies that observed effects of boat noise on this species in these different topics. Boat noise can disrupt and decrease calling activity, mask environmental and conspecific signals, reduce reproduction success, induce stress, affect parental care, and even affect larvae development. These results warn of the possible severe effects of noise pollution on fish and warrant the need of further studies addressing the consequences of noise at the population level.

**Keywords**

Fish · Behavior · Hearing · Boat noise pollution · Acoustic communication · Development · Reproduction

## Introduction

There are over 30,000 species of fish that have adapted to numerous aquatic environments showing a remarkable diversity in behavior, anatomy, physiology, and ecological adaptations (Helfman et al. 2009). However, human activities are affecting marine ecosystems at unprecedent rates (Villéger et al. 2017). Aquatic noise pollution is one important form of pollution. Indeed, it is so chronic and widespread that its effects have changed underwater soundscapes all around the globe (Duarte et al. 2021). The major sources of anthropogenic noise include commercial shipping, recreational boating, pile-driving, seismic surveys, and power stations, but in shallow waters one of the most prevalent sources of underwater noise are boats (Hildebrand 2009; Popper and Hawkins 2019; Vieira et al. 2021a).

Sounds can be a rapid, directional way to convey critical information through short and even long-distance to marine animals (Hawkins and Popper 2018). However, anthropogenic noise can mask sounds of interest, affect behavior, increase stress levels and trigger other physiological effects, or even cause physical injuries or death in fishes (Duarte et al. 2021; Pine et al. 2021). In this context, here is reviewed what is known from the highly vocal Lusitanian toadfish (*Halobatrachus didactylus*) as a model to understand the impact of this anthropogenic stressor on fishes.

## Vocal Behavior

The family Batrachoididae is known to include several highly vocal fish species that rely on acoustic communication for reproduction (Bass and McKibben 2003), such as the Lusitanian toadfish (Vasconcelos et al. 2012). This species has a remarkably large repertoire that includes calls like the boatwhistles (BW), croaks, double croaks, long grunt trains, and several combinations of these sound types (see Amorim et al. 2008, for details of the vocalizations). Males calling activity peaks from May to July, corresponding also to the peak of time nest occupation (Carriço et al. 2014). At that time of the year, they produce choruses that are some of the most dominant sounds on the Tagus estuary soundscape (Amorim et al. 2006; Vieira et al. 2021a). Although some other toadfish species are known to produce calls only at night, Lusitanian toadfish usually produce sounds throughout the day (Vieira et al. 2021a). Nevertheless, diel patterns of vocal activity can be affected by tide level, tide velocity, and time of day (Vieira et al. 2021b). It is still unknown if diel patterns of vocalizations change under anthropogenic noise, but a controlled experiment using boat noise playbacks has shown that Lusitanian toadfish exposed to this noise reduced the overall calling activity (Amorim et al. 2022). Control males produced on average 35 BW per hour while males exposed to boat noise reduced calling activity to only 13 BW per hour. It was also noted that the onset of vocal activity usually occurred earlier in nests exposed to control conditions. The detrimental effect found in the Lusitanian toadfish is consistent to what has been reported to other batrachoidids, namely, the oyster toadfish (Luczkovich et al. 2016), which showed depressed calling rates in a site with high boat traffic relative to a site with low boat traffic.

The BW, the most frequent sound of this species, is a highly stereotyped low-frequency signal that is mostly used as an advertisement call but can also be used in agonistic situations (Amorim and Vasconcelos 2008; Vasconcelos et al. 2010). This sound type presents significant differences between males that allow to distinguish different singing males based on the features of their sounds. If fish can use this information to recognize different neighbors, it could mediate interactions among territorial males (Amorim and Vasconcelos 2008; Vieira et al. 2015). Looking into vocal interactions of fish occupying nearby nests, it was observed that males can adjust the rate of BWs depending on whether they are calling alone or in a chorus (Amorim et al. 2011) and may also adapt calling activity to match a neighbor's calling rate (Jordão et al. 2012). Recently, Vieira et al. (2021b) showed that male-male interactions can be even more complex than previously thought. In a chorus, each male appears to be aware of its neighbors' calling activity, adjusting not only the calling rate but also the timing of each vocalization. Interactions like alternation or synchrony of the calls were observed, possibly dependent on advertisement or agonistic context, respectively. If anthropogenic noise disrupts acoustic communication, the complexity of the calling behavior in this species points to important implications for possible interference of boat noise on fish calling behavior. Indeed, the investigation of fine-scale temporal changes in the presence of boat noise playbacks revealed that fish avoided singing during each boat passage and vocal interactions between males were disrupted (Alves et al. 2021). The Lusitanian

toadfish's avoidance response to noise is consistent with a similar effect observed in the oyster toadfish (Krahforst et al. 2016) and the plainfin midshipman (Brown et al. 2021), i.e., these species also decrease the calling rate during the exposure to boat noise playbacks. However, although some fine-scale interactions are described in vocal patterns of other fishes (Mensinger 2014), further studies are needed to characterize in detail the fine-scale male–male vocal interactions and its possible disruption in the presence of noise.

## Hearing

Hearing sensitivity in the Lusitanian toadfish is best at low frequencies (<200 Hz), with best hearing sensitivity coinciding with the range of frequencies contained within conspecific calls, and decreases at higher frequencies (Vasconcelos et al. 2007). Nevertheless, recordings of auditory evoked potentials (AEP) revealed that Lusitanian toadfishes are capable of resolving temporal patterns, amplitude fluctuations, and frequency content of conspecific calls (Vasconcelos et al. 2011). This suggests that the auditory system of the Lusitanian toadfish is suited to discriminate conspecific vocalizations, which is likely important for communication and orientation during territorial defense and mate attraction. However, fish can also resolve the temporal patterns of boat noise, as this noise spectral components overlap with the Lusitanian toadfish hearing frequency range (Alves et al. 2021). The effects of boat noise on the hearing abilities of this species were evaluated in laboratory conditions. Vasconcelos et al. (2007) measured hearing sensitivities under quiet lab conditions and in the presence of ambient noise and ferryboat noise at levels encountered in the field. Using the AEP recording technique, these authors observed that, under ambient noise conditions, hearing was only slightly masked at lower frequencies. However, in the presence of boat noise the auditory thresholds increased 3–36 dB, with higher shifts occurring at the lower frequencies. More recently, Alves et al. (2021) tested how boat noise impacted the communication range of BWs. Using BWs recorded at different distances from the source in the intertidal zone, this study measured the signal detection threshold under ambient noise and boat noise conditions based on AEPs. As expected, boat noise impacted communication range. Small motorboat and ferryboat noise reduced the communication range from ca. 6 and 10 m to 2 m and 6 m, respectively. These experiments also showed that the masking effect depended on noise spectral content of both the BW and the boat noise. Both studies evidenced that hearing and acoustic communication are constrained during exposure to boat noise.

## Reproduction

During the breeding season in Portugal, Lusitanian toadfish can be found in estuarine or coastal shallow waters, where breeding males occupy nests under rocks or other hard substrates in the intertidal/subtidal areas (Amorim et al. 2010). As with other batrachoidids, males produce advertisement calls to attract reproductive females

(Amorim and Vasconcelos 2008). This vocal behavior is very important as females appear to choose the nest based on the acoustic information sent by breeding males. Indeed, calling rate and effort (percentage of time spent calling) reflect male quality (Amorim et al. 2010) and influence reproductive success (Vasconcelos et al. 2012). Amorim et al. (2016) confirmed the importance of the calling behavior in a field experiment that compared the reproduction success between muted and vocal males. These experiments demonstrated that if a male does not produce calls it will not be able to successfully mate. In addition, the number of eggs in the nests was positively associated with the male's maximum calling rate, supporting earlier studies by Vasconcelos et al. (2012). Note, however, that other factors may also influence mate choice, such as the easiness of directly sampling different males, i.e., accessing the male's nests (Amorim et al. 2016).

Since noise can interfere with vocal activity and communication range, one would expect that it should negatively affect reproduction. Recently, Amorim et al. (2022) reported another field experiment that examined the effects of boat noise on Lusitanian toadfish reproduction. This study used two sets of artificial nests deployed in the natural habitat, exposed to either ambient sound or boat noise playbacks. Nests were left undisturbed during a fortnight, and male toadfish spontaneously occupied both these sets of nests for breeding. Amorim et al. (2022) observed that boat noise did not affect nest occupation by Lusitanian toadfish males but decreased the likelihood of receiving eggs. This decrease is consistent with the study by Krahforst (2017) that showed oyster toadfish nesting in areas with boat traffic had lower reproductive success than in quiet areas. This, however, contrasts with the results reported by Brown et al. (2021) that did not find an effect of tonal noise exposure on plainfin midshipman reproductive success. In Amorim et al. (2022), concurrent with the decreased likelihood of receiving eggs, males exposed to boat noise also showed depressed vocal activity and slightly higher cortisol levels. In this study, oxidative stress and energy metabolism-related indicators revealed no increase in oxidative damage but pointed toward a decrease in the activity levels of energy metabolism-related biomarkers. These results suggest that boat noise depressed males metabolism and overall activity (including vocal behaviors) to cope with an acoustic stressor, thus decreasing the ability to attract mates. Although toadfishes might be especially vulnerable to the impact of boat noise due to the reliance on acoustic communication for mating and being attached to a nesting site, noise exposure effects can still be extrapolated to other fish species that may suffer similar or other fitness impacts (for further discussion, see de Jong et al. 2020). Detrimental noise effects should, however, be higher in fish species that reproduce in nests and provide parental care or in other species associated with restricted spawning or feeding grounds, i.e., that have higher costs to leave particular sites (de Jong et al. 2020).

## Eggs Survival and Development

Female Lusitanian toadfish lay large benthic adhesive eggs (>5 mm) in the nest and leave the eggs for male toadfish to provide parental care until the fry are free-swimming (Costa 2004). Note that Lusitanian toadfish larvae stay attached until

nearly all the yolk sac has been absorbed (Collette 2005). This means that larvae cannot escape from noise exposure for up to 2–3 months, depending on temperature (Felix et al. 2016; MCP Amorim, personal observation). Recently, Amorim et al. (2022) and Faria et al. (2022) have revealed some insights on the impact of boat noise in eggs' survival and early life stages development. These authors reported a field experiment where females laid the eggs on artificial nests during sound exposure to either ambient sound or boat noise playbacks. Eggs from each nest were divided in a split-brood design and exposed to both treatments in nests without a male to provide parental care. The first noticeable effect was an increase in the number of dead eggs in nests exposed to boat noise. Also, boat noise exposure affected larval development resulting in smaller larvae. Oxidative stress and energy metabolism-related indicators were also measured (Faria et al. 2022); consistent with the growth deficit, noise-exposed larvae showed larger overall stress responses, with increased levels of superoxide dismutase (SOD), DNA damage (oxidative stress-related responses), and electron transport system activity (ETS; energy-related metabolism). Note that this is a pivotal study providing evidence of effects of boat noise exposure in field conditions. It is thus urgent to conduct more studies to improve our understanding on the effects of anthropogenic noise on the early stages of fish development.

## Conclusion

Anthropogenic noise can affect the Lusitanian toadfish and other fish species at multiple levels and in different stages of development. However, we still lack insight into how these noise-induced effects translate into population-level consequences in this taxonomic group (Slabbekoorn 2019). Further studies addressing the consequences of noise at population and community levels are crucial to provide recommendations to reduce the impacts of anthropogenic noise. Nevertheless, the knowledge obtained until now demand for measures to limit boat noise at least in protected areas and sensitive coastal habitats such as major breeding areas.

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# Vessel Noise Pollution in Fish Habitats in the Northern Gulf of Mexico

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Kelly S. Boyle and Bennett H. Price

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**Abstract**

Vessel noise is expected to occur at sites with human activity, such as areas near boat channels, harbors and marinas, and petroleum exploration platforms. How noise varies among sites and how such variation may impact associated fish communities are not well-known. Passive acoustic monitoring (PAM) was used to examine spatial, diel, and longer temporal variation in vessel noise occurrence among five coastal sites in the northern Gulf of Mexico. Dramatically higher vessel noise occurrence was observed at some sites. Noise-related impacts on fishes are likely influenced by differences in species composition among sites, times of peak sound production among species, and bathymetry at sites that impacts propagation of lower-frequency vessel noise.

**Keywords**

Passive acoustic monitoring · Sciaenidae · Toadfish · Oil rig · Navigation channel

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## Introduction

Globally, aquatic vessel noise has increased over time and presents a pervasive global pollutant in marine waters (Duarte et al. 2021; Slabbekoorn et al. 2010). Vessel noise pollution, however, is expected to vary spatially because of established navigation channels, locations of marinas and ports, and proximity to coastal areas (Neenan et al. 2016). Further, human activity and industry around areas of petroleum exploration and extraction may be expected to have noise associated with construction and maintenance of drilling platforms and other infrastructure. Additionally, noise from some vessel activity may vary seasonally (Neenan et al. 2016). Noise associated with recreational fishing and pleasure boat activity may be predicted to increase in the summer, on weekends and holidays, and during calmer weather (Boyle et al. 2022a). On finer temporal scales, diel patterns of noise are influenced by patterns of human activity (Kaplan et al. 2016), such as ferry schedules and daylight patterns that are likely to influence when boating, fishing (Boyle et al. 2022a), and vessel traffic is more likely to occur. Together, these influences on vessel activity contribute to variation in vessel noise occurrence that complicates measurement of impacts on fishes and other marine life.

## Spatial and Temporal Variation in Vessel Noise Among Coastal Northern Gulf of Mexico Sites

Vessel noise is expected to vary among sites and over short term (diel) and longer temporal (seasonal scales). This variation has important ramifications for predicting impacts on fishes and other fauna. Passive acoustic monitoring was used to measure variation in noise among sites and across time over largely overlapping time periods at five sites along the northern Gulf of Mexico (Fig. 1). These sites include recordings made at two offshore oil rigs off of coastal Louisiana (Rig 1, Rig 2), ~18 m sea depth, with recorders placed at 12 m depth. These oil rigs lack permanent personnel but are serviced periodically by crew boats. The third site, Grand Isle, is along the southern edge of a vessel channel in Bayou Rigaud, Grand Isle, Louisiana, with a recorder placed at ~2 m depth. The fourth site, Big Lagoon, in western Florida, is along the Gulf Intracoastal Waterway, in a no-wake zone, where vessel speed is regulated and the recorder was at approximately 3 m depth. The last site, furthest east, was in St. Andrews Bay Inlet, in Panama City Beach, Florida. This site was a navigation channel into St. Andrews Bay, with the recorder placed at approximately 9 m depth. Intense vessel noise was defined as a sound file with a 1–3000 Hz band sound pressure level ($SPL_{1–3000Hz}$) $\geq$ 135 dB re: 1μPa. This 135 dB amplitude was chosen because it exceeds amplitudes shown to cause temporary threshold shifts in fish after 24 hours of exposure (Breitzler et al. 2020), and the number of sound files with amplitudes $\geq$135 dB provided a manageable dataset for confirmation of vessel noise as the source. This frequency band was examined because it encompasses the range of hearing of most fishes, including species with adaptations for detecting higher frequencies (Ladich and Schulz-Mirbach 2016; Popper et al. 2003). Average

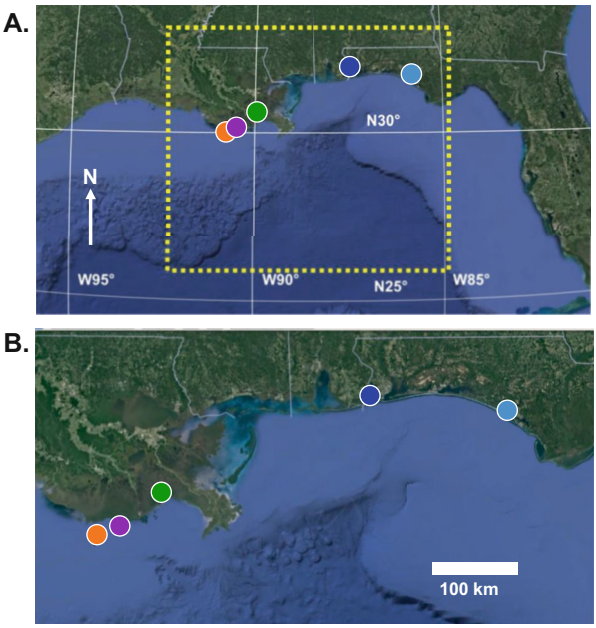



**Fig. 1** (**a**) Location of study sites in the northern Gulf of Mexico. Yellow-dashed line shows region of close-up view in (**b**). Sites from east to west: Rig 1 (orange), Rig 2 (purple); Grand Isle (green); Big Lagoon (dark blue); St. Andrews Inlet (light blue)

$SPL_{1–3000Hz}$ was calculated from PAM sound files collected at each site that ranged from 30- to 60-s duration depending on deployment and site (duty cycles ranged from 6.7 to 25%). For files in which the average was $\geq$135 dB, occurrences of vessel noise were verified aurally and visually from spectrograms.

## Fish Composition Among Sites That Vary in Noise Exposure

In the northern Gulf of Mexico, it is important to consider spatial variation in noise occurrence because fish species composition differs among habitats and location. Coastal marine fish species composition and abundance are influenced by habitat features such as proximity to freshwater input, bottom type (mud, sand, oyster reef, rocky, and artificial hard substrate), and depth (Gain et al. 2016; Garner et al. 2019; Nelson et al. 2021; O'Connell et al. 2019; Plumlee et al. 2020). Fish species that are more likely to occur in habitats modified by human activity may be more likely to experience high levels of noise exposure. Certain habitat modifications associated with human use, such as deepening of navigation channels by dredging, addition of hard bottom, vertical relief, and fouling community substrate in the development of wharfs, piers, jetties, and petroleum platforms, are likely to attract species that associate with these habitat features (Clynick et al. 2008; Putt et al. 1986; Wakefield et al. 2013). Thus, fish typically associated with artificial structures along ports and drilling platforms may be prone to exposure of vessel sound associated with these

developments. In addition, fishes that utilize deeper channels may be prone to vessel noise exposure in navigation channels.

Along the northern Gulf of Mexico, sites within bays and estuary passes include large aggregations of sciaenid fishes that produce sounds during spawning aggregations (Lowerre-Barbieri et al. 2008; Luczkovich et al. 2008). Spawning aggregations along bay entrances and navigation channels can result in exposure to vessel noise (Picciulin et al. 2022).

Fish composition varies among the five sites in the present study, and fish species present at these sites may be impacted by noise in different ways. The offshore Rig 1 and Rig 2 sites include many reef-associated species, such as red snapper *Lutjanus campechanus*, gray snapper, sheepshead *Archosargus probatocephalus*, and Atlantic spadefish *Chaetodipterus faber*. The inshore estuarine sites Grand Isle and Big Lagoon include several well-known soniferous species that were detected on recordings: gulf toadfish *Opsanus beta*, silver perch *Bairdiella chrysoura*, and spotted sea trout *Cynoscion nebulosus*. In addition, black drum *Pogonias cromis* vocalizations were detected at Big Lagoon. St. Andrews Bay Inlet includes vocal estuarine species like silver perch, red drum *Sciaenops ocellatus*, black drum, and gulf toadfish. In addition, St. Andrews Inlet has a rock jetty that provides an artificial reef habitat, and reef-associated species like gag *Mycteroperca microlepis* and cocoa damselfish *Stegastes variabilis* are present.

## Spatial and Temporal Variation of Vessel Noise Among Northern Gulf of Mexico Sites

Wide variation was observed in the frequency of occurrence of intense vessel noise among sites and months (Fig. 2). In September 2021, intense vessel noise was present for 15% of the month at Rig 1 (Fig. 2). This period occurred following Hurricane Ida, which passed very close to this site and may have resulted from service vessel presence at the platform. Noise levels at rig sites in other months were much lower. Among coastal sites, St. Andrews pass exhibited higher incidences of vessel noise occurrence in most months than other nearshore sites examined (Fig. 2). Broad variation among months indicates that long-term monitoring is necessary to assess noise occurrence among sites in the northern Gulf of Mexico. Further, because noise varies among months, the potential for negative impacts on fishes depends on seasonal variation of fish abundance at sites (e.g., within estuaries vs. further offshore) and seasonal activities of fishes (e.g., spawning aggregations and sonic activity).

## Impacts on Fish Vary by Taxa

Vessel noise impacts on fishes are expected to vary among species. In addition, noise impacts directly related to hearing may vary with auditory sensitivity and the behavior of affected species. Species that use hearing to detect sounds of predators

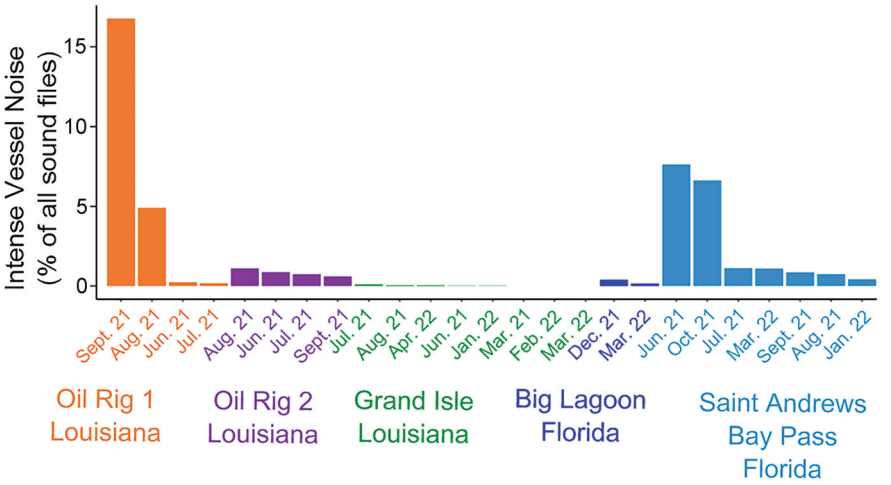



**Fig. 2** Spatial and monthly variation of intense vessel noise occurrence at five northern Gulf of Mexico sites in 2021 and 2022. Highest occurrence of vessel noise was observed at Rig 1 in September 2021; 15% of all files had $SPL_{0–3000} \geq 135$ dB re: 1μPa. Intense vessel noise also occurred more than 5% of the month during June and October 2021 at St. Andrews Inlet

and prey may be predicted to have less opportunity when such sounds are masked by vessel noise and would be expected to have reduced capacity to detect such cues when repeated noise exposure causes hearing loss. Soniferous fishes are predicted to be particularly susceptible to impacts from noise because exposure may not only damage hearing but can also mask calls from conspecifics (Codarin et al. 2009; Vasconcelos et al. 2007; Vieira et al. 2021). Co-occurrence of vessel noise and fish calls was observed at study sites in the Gulf of Mexico (Fig. 3). These events could potentially mask acoustic communication for these fishes.

Diel activity patterns of fishes, especially soniferous species, are likely to contribute to impact of vessel noise because boat traffic is often diurnally biased. Diurnally active soniferous species in noisy areas are likely to have more masking interactions with vessel noise. The timing of impacts on hearing and reception following vessel noise exposure is relevant because vessel noise is more prevalent during diurnal hours (Fig. 4). Impacts that quickly follow noise exposure, such as temporary threshold shifts in hearing (Scholik and Yan 2002), elevated stress (Celi et al. 2016), and altered behavior (Cox et al. 2018), are likely to disrupt responses by fish to calling conspecifics that may have roles in spawning behavior for species with mating calls or school and shoal cohesion for species with contact calls. At some of the study sites in the northern Gulf of Mexico, damselfish species (*Stegastes* spp.) produce diurnal calls associated with territory defense, mate attraction, and nest defense (Boyle et al. 2022b). When vessel noise is most prevalent in diurnal hours (Fig. 4), these fishes are more likely to have calls masked, and immediate effects of noise exposure on the auditory system may reduce efficacy of their calls.

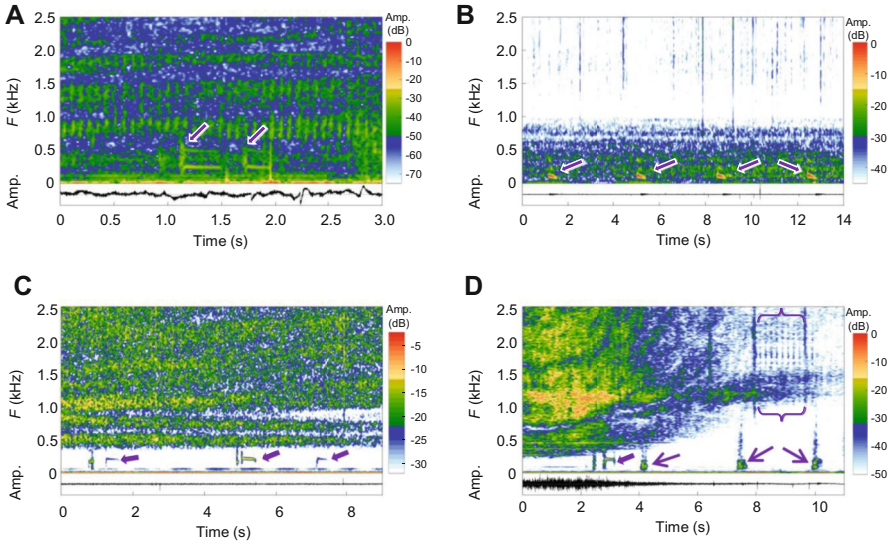



**Fig. 3** Vessel noise co-occurrence in the same frequency range as fish sounds in the northern Gulf of Mexico. (**a**, **b**) A shallow navigation channel in Bayou Rigaud, Grand Isle, Louisiana, results in a high-pass transmission of vessel noise because the shallow water prevents propagation of longer sound wavelengths. This low-frequency cutoff results in a quiet communication window below ~400 Hz for gulf toadfish boatwhistle (block arrows) and grunt sounds (open arrows) but overlaps higher-frequency sounds, like silver perch sounds (brackets in **b**), which are only audible when vessel noise is less intense. (**c**) Deeper channel along the Gulf Intracoastal Waterway in Big Lagoon, Florida, does not filter lower-frequency vessel noise. Thus, low-frequency fish calls, like gulf toadfish boatwhistles (block arrows), partially overlap frequencies even during weak vessel noise. (**d**) A channel >10 m deep at St. Andrews pass, Panama City Beach, FL, allows low-frequency vessel noise transmission that partially overlaps black drum calls (block arrows)

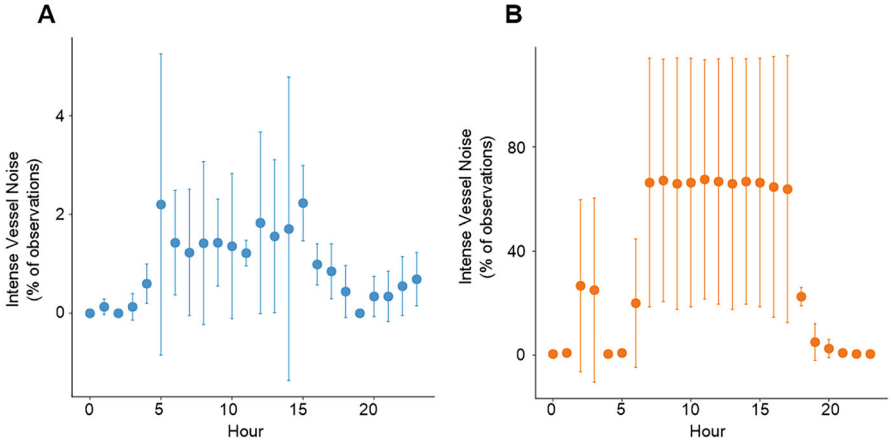



**Fig. 4** Diel patterns of vessel noise at (**a**) Rig 1 from September to October 2021 and (**b**) St. Andrews Inlet from December 2021 to March 2022. Mean + SD

Several sciaenid fish species present along the northern Gulf of Mexico spawn in the evening (Holt et al. 1985; Montie et al. 2016, 2017). At the study sites, vessel noise is less prominent in the evening than the day, though still occurs with some regularity. Thus, the amount of masking of calls associated with spawning is mitigated in part by the time of peak spawning for sciaenids. Another concern, however, is the potential impact of diurnal vessel noise exposure on calling and sound reception in the evening. If fish are present at a spawning ground prior to periods of calling, exposure to intense noise may disturb calling fish or increase stress before evenings when calling would normally occur. A sciaenid species in Portugal (meagre, *Argyrosomus regius*) was observed to reduce calling during and following ferryboat noise (Vieira et al. 2021). However, in another sciaenid species (brown meagre, *Sciaena umbra*), nocturnal sound emission rates were observed to increase following vessel noise exposure (Picciulin et al. 2012), while in another study conducted during diurnal hours, the same species did not change sound emission rates when boat noise was present, though fish increased hiding and flight reactions (La Manna et al. 2016). These differences in observed outcome may result from variation in behavioral responses of species, differences in the motivation associated with calls made during the day vs. calls made during evening spawning aggregations, or differences in the type of noise produced from vessels in each study. Daytime noise exposure may also temporarily decrease hearing sensitivity of fish, which would reduce the detection space available to fish listening to evening calls. Large schools of red drum were observed at the St. Andrews study site during the daytime during late summer and early fall when fishes produce spawning calls in the evenings.

Duration of exposure may vary depending on the degree of residency of fish at specific habitats. It is not known how long mobile species, like many sciaenids, remain in noisy locations within bays and estuaries. Spawning site location may involve many factors, and individual fish that selects noisy habitats may be predicted to remain at such locations multiple days if spawning continues to occur at the site (Picciulin et al. 2022). For intermittent vessel noise, it may not be possible for fish to avoid or easily discriminate between sites with vessel noise because noise builds gradually and travels over great distances. Studies have observed fish avoidance behaviors when vessels pass, which include diving to greater depth (De Robertis and Handegard 2013). At shallow sites, however, it is not possible for fish to flee to avoid intense noise while it is occurring. Over longer periods, fish may be predicted to leave a noisy habitat if they are able to assess longer-term noise patterns and if such information influences habitat preferences. Alternatively, fish that increases swimming activity in the presence of a noise stressor may be more likely to move and encounter quieter areas by chance.

Life history strategies of some fish species, such as species that nest or defend territories, also may influence the likelihood of a fish remaining in a noisy habitat. Gulf toadfish were observed aurally at Grand Isle, Big Lagoon, and St. Andrews (Fig. 3a, c), and at the latter site, a fish was visually observed next to the recorder during a PAM recorder recovery scuba dive. Male toadfish use calls to advertise for mates and defend nests (Gray and Winn 1961). Site attachment over the period of

nest preparation, advertisement, and defense extends the duration of potential noise exposure in sites with high vessel traffic. In the closely related oyster toadfish (*Opsanus tau*), males were observed to guard eggs and free-swimming young for a period between 23 and 49 days (Gray and Winn 1961). Vessel noise incidence was shown to cause a reduction in boatwhistle calls in oyster toadfish (Mackiewicz et al. 2021). At sites further offshore, like the oil rig sites monitored in this study, reef-associated species may be site attached in part because of distance to suitable habitat: artificial and natural reefs. Reef-associated species were observed on Rigs 1 and 2, such as Atlantic spadefish, gray snapper, and red snapper. Red snapper in the Gulf of Mexico exhibits site fidelity (Schroepfer and Szedlmayer 2006; Topping and Szedlmayer 2011) and thus may experience high levels of noise when vessel operations are busiest at rigs.

## Bathymetry of Sites and Noise Exposure

Sites with shallower vessel channels may mitigate potential impacts of vessel noise because of the low-frequency cutoff imposed by depth that prevents propagation of longer wavelength sounds. Shallower channels reduce the total amplitude of vessel noise by preventing lower-frequency components from extending as far from the source (Rogers and Cox 1988). Further, lower-frequency fish sounds will not be masked because of the quiet window below the cutoff frequency. A quiet window below 500 Hz was observed at Bayou Rigaud in Grand Isle, which is a shallow vessel channel with commercial (shrimp trawler and petroleum vessels) and recreational vessels. This quiet window prevents vessel noise from masking gulf toadfish boatwhistle sounds that are likely within a wavelength to the recorder (Fig. 3c, d). Fish species that produce higher-frequency calls like silver perch, however, are masked by higher-frequency vessel noise spectral components (Fig. 3e). At sites along deeper vessel channels, low-frequency fish sounds from gulf toadfish and black drum are masked by vessel noise.

## Particle Motion Versus Sound Pressure

Sound particle motion is the default pathway for fish hearing, though some fish lineages have evolved secondary mechanisms to additionally detect sound pressure (Ladich and Schulz-Mirbach 2016). Thus, for fish without accessory structures to detect sound pressure, sound particle motion is predicted to be the source of noise-induced damage to hair cells of the sensory epithelium (Popper and Hawkins 2018). Sound particle motion attenuation occurs more rapidly than sound pressure and is frequency dependent. Thus, the duration of intense particle motion noise exposure of a passing vessel will be lower than that for sound pressure. The assumption that particle motion is the main source of damage to most fish ears, however, needs more study. Further, it is challenging to address the contribution of these two aspects of noise exposure on fish ears because studies that expose fish to noise in the lab occur

in the acoustic nearfield where particle motion predominates (Popper and Hawkins 2018). Estimating particle motion noise levels in the field is more challenging than sound pressure measurement (Nedelec et al. 2016; Popper and Hawkins 2018). Passive acoustic monitoring of vessel noise in fish habitats is most often conducted using hydrophones that measure sound pressure. Further, sound particle motion estimates can require knowledge of source distance and water depth (Nedelec et al. 2016), which is complicated with a moving vessel. Lastly, more research is needed to determine the relative impacts of particle motion and sound pressure of vessel noise pollution on fishes to determine the limitations of PAM with sound pressure.

## Conclusions

Our limited observations from coastal northern Gulf of Mexico sites with moderate human activity indicate broad variation in vessel noise occurrence among sites. Notably, within sites, vessel noise differs over longer temporal scales (weeks-months), which underscores the importance of long-term monitoring for assessing vessel noise pollution in fish habitats. Future research needs to assess the residency time of individual fish at locations with high vessel noise to adequately determine potential impacts. This is especially important for mobile species that form spawning aggregations, which include many sciaenid fishes.

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# Effect of Aquaculture Sound on Fish Development, Physiology, and Behavior

65

Rikeem K. Sholes and Allison B. Coffin

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**Abstract**

This chapter reviews studies on the effect of aquaculture sounds from pumps, generators, and other equipment on the development, physiology, and behavior of fishes. The chapter also examines select studies using boat playback sounds, as these studies can still inform understanding of noise impacts on commercially important aquaculture species. Anthropogenic sound can have a direct effect on the target species being reared in aquaculture facilities, including negative impacts on embryonic development, sensory function, and stress. Hatchery sounds can also impact critical behaviors such as predator avoidance or parental

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care, which in turn can adversely affect subsequent generations. The goal of the chapter is to provide a scope of the wide range of environmental impacts unmitigated noise has on fish populations.

**Keywords**

Aquaculture · Hatchery · Anthropogenic noise · Fish · Stress · Development · Inner ear · Lateral line

## Introduction

A significant byproduct of industrialization is the increasing negative effect on the environment. Habitat fragmentation, overharvesting, and climate change have altered ecosystems around the globe (Brander 2010; Haddad et al. 2015). Aquatic environments have not been spared from this impact, and conservationists have been toiling to preserve species in a rapidly changing landscape. Marine and freshwater fish species are exposed to shifting water chemistry and temperatures, causing changes in their development and behavior (Domenici et al. 2019; Huang et al. 2021). Governments worldwide have initiated aquaculture systems to propagate aquatic species of economic, ecological, and cultural importance to mitigate these anthropogenic threats.

Many aquaculture facilities were established to provide protection for developing juveniles of fish populations of interest. These complexes offer a habitat that is largely isolated from predation and where food is in ample supply. Whether the objective is to propagate fish for recreational or commercial use or to preserve a threatened population, a common goal of aquaculture facilities is to reduce the high mortality rate in early developmental stages, thereby increasing the number of fish that survive to reproduction or market. While these facilities do accomplish the goal of eliminating natural predation, they have the unfortunate side effect of altering the development and behavior of captive-reared fish (Olla et al. 1998; Marchetti and Nevitt 2003). Aquaculture facilities are industrial complexes where juveniles are raised in manufactured enclosures that are a poor reproduction of the complex habitat these fishes would naturally inhabit.

Typically, fitness assessments in aquaculture operations have focused on the presence of parasites and diseases and fish weight upon release (Piper 1986; Barber 2007). These factors are undoubtedly important in the long-term survival of captive-reared animals but do not fully capture proper development or physiological function. The ability to interact with the surrounding environment is crucial for survival to adulthood, and metrics to measure this ability should be incorporated into aquaculture practices. A substantial body of research has shown that fish raised in industrial aquaculture environments are morphologically and behaviorally different from their wild conspecifics (reviewed in Anderson et al. 2020). Exposure to artificial lighting, processed diets, and excessive sound have marked effects on the development of these fish (*e.g.,* Ridha and Cruz 2000; Fakan and McCormick 2019).

This brief chapter explores the impact of anthropogenic sound on the development, physiology, and behavior of aquaculture fish populations (see Popper and Hawkins 2019 for further review on this topic). We highlight contemporary literature to give the reader a broad understanding of conservationists and aquaculturists' challenges in their attempt to rear viable populations.

## Effects of Aquaculture Noise on Development

Anthropogenic sound exposure is generally not acutely lethal to fish larvae (Wysocki et al. 2007; Bruintjes and Radford 2014; Lara and Vasconcelos 2021; Faria et al. 2022). However, sound exposure can alter embryonic and larval development and these changes may impact subsequent life stages (Jain-Schlaepfer et al. 2018; Fakan and McCormick 2019). The effects of this exposure are species-specific and also depend on the time and intensity of exposure. For example, a lab study of two species of reef fish, cinnamon clownfish (*Amphiprion melanopus)* and spiny chromis (*Acanthochromis polyacanthus*), exposed embryos to a continuous ~120 dB (re 1 μPa) recording of a two-stroke engine (Fakan and McCormick 2019). Sound-exposed embryos of either species had a faster heart rate, suggesting increased stress. Further, mean yolk volume decreased in *Ac. polycanthus* exposed to boat noise. Yolk sac reserves in early development help to nourish young fish before they can feed, so early absorption can lead to reduced survival. Interestingly, sound-exposed *Ac. polycanthus* had an increase in body length and eye diameter (Fakan and McCormick 2019). By contrast, a field experiment showed that boat noise playback reduced larval growth in Lusitanian toadfish (*Halobatrachus didactylus*) (Faria et al. 2022), demonstrating a complex relationship between anthropogenic sound and fish growth that likely depends on species, noise stimulus, and other parameters.

The type and regularity of anthropogenic sound also impacts fish development. While boat engine noise increases heart rate in some reef fishes, as described above, exposure to sound from a 2-stroke engine increased heart rate significantly more than similar exposure to a four-stroke engine (Jain-Schlaepfer et al. 2018). Along similar lines, a study in larval Atlantic cod (*Gadus morhua*) demonstrated that while exposure to either random or consistent noise reduced larval body size (compared to fish reared in ambient sound levels), larvae exposed to consistent noise depleted their yolk sac faster and had a lower body width-length ratio, suggesting lower energy reserves. Width-length ratio was positively correlated with predator avoidance behavior in a simulated predation experiment, indicating that constant sound exposure may attenuate predator avoidance behavior (Nedelec et al. 2015). Interestingly, a study in the daffodil cichlid *Neolamprologus pulcher* found no significant effect of intermittent boat noise exposure on larval growth (Bruintjes and Radford 2014). The authors concluded that moderate sound levels (127 dB re 1 μPa RMS) did not impact larvae (Bruintjes and Radford 2014). However, subsequent studies described above suggest that a continuous sound stimulus may have produced a different result.

### Effect of Noise on Gene Expression

Sound exposure can also alter gene expression in fishes, with potential impacts for both adult animals and their offspring. RNA-seq analysis of brain tissue from the African cichlid species *Astatotilapia burtoni* found that females exposed to 3 h periods of anthropogenic sound (~140 dB) expressed transcripts associated with feeding and brooding behavior differently than those not exposed to sound (Butler and Maruska 2021). Specifically, sound exposure increased expression of appetite-stimulating neuropeptides and downregulated leptin receptors, molecules that collectively regulate food intake. Sound exposure also reduced expression of prolactin and vasotocin, which are peptides linked to maternal care behavior (Butler and Maruska 2021). *A. burtoni* is a mouthbrooding species, and many noise-exposed females cannibalized or prematurely ejected their young, reducing brood success by 55%. Boat noise exposure also had a negative effect on parental care behavior in male spiny chromis, a nest-brooding damselfish, leading to a reduction in larval survival (Nedelec et al. 2017). These studies demonstrate that sound exposure may have a greater impact on larval mortality in species that require parental care versus those without such care, owing to noise-induced effects on parental behavior. Furthermore, offspring of noise-exposed *A. burtoni* females had upregulated isotocin levels in the brain (Butler and Maruska 2021). Isotocin is the fish homolog of oxytocin and helps regulate social behavior, including parental care in cichlids (Reddon et al. 2014). Therefore, maternal sound exposure could affect breeding success in subsequent generations.

While many of these studies used boat noise exposure in cichlids and damselfish, rather than aquaculture noise exposure in common hatchery species with greater commercial importance, they collectively highlight the detrimental impact of sound exposure on reproductive success and larval development.

## Effects of Hatchery Noise on Sensory Systems

Given that intense noise can cause hearing loss, it is unsurprising that several studies have examined the effect of anthropogenic noise – including sounds from aquaculture operations – on fish ear structure and function (Wysocki et al. 2007; Halvorsen et al. 2012; Casper et al. 2013; Popper and Hawkins 2019; Sapozhnikova et al. 2020). This section briefly discusses a few studies (of many) as part of our broader survey of the impacts of aquaculture noise on fishes.

The fish inner ear contains three otolithic end organs, the saccule, utricle, and lagena (Popper and Fay 1993, 2011). Each end organ is comprised of a sensory epithelium that contains a population of sensory hair cells associated with a calcareous otolith, where motion of the otolith relative to the epithelium initiates the signal transduction cascade in hair cells, ultimately resulting in perception (Popper and Fay 1993, 2011). In most fishes, the saccule is the primary auditory end organ, although the lagena and utricle may play roles in both hearing and balance (Mann et al. 2001; Meyer et al. 2012; Vetter et al. 2019; Rogers and Sisneros 2020). Broadband sound

exposure for 1–2 days causes saccular hair cell damage and hearing loss in model fish species such as goldfish (*Carassius auratus*) and zebrafish (*Danio rerio*), two otophysan species with relatively sensitive hearing thresholds over a broad range of frequencies (up to 4 kHz) (Smith et al. 2006; Breitzler et al. 2020).

However, sound-induced damage is much more variable in typical hatchery species and is likely correlated with both stimulus paradigm and relative hearing sensitivity of that species. For example, 18 days of exposure to a 300 Hz pure tone (corresponding to the peak frequency of a recirculating aquaculture system) at ~180 dB (re 1 μPa) caused damage to saccular hair cells and associated sagittal otoliths in peled (*Coregonus peled*), an important aquaculture species in parts of Russia (Sapozhnikova et al. 2020). By contrast, rainbow trout (*Oncorhynchus mykiss*) exposed to 150 dB (re 1 μPa) broadband noise showed no hearing loss, even after 8 months of continuous exposure (Wysocki et al. 2007). This lack of hearing loss in rainbow trout may reflect the lower intensity sound used in this study. However, hatchery rearing can reduce auditory sensitivity in Chinook salmon (*Oncorhynchus tshawytscha*), a fish with similar hearing sensitivity to rainbow trout, by altering the composition of the sagittal otolith (Oxman et al. 2007). It is unknown if changes in otolith composition are caused by hatchery sounds or other features of the artificial rearing environment.

## Effect of Noise on the Lateral Line

In addition to the inner ear, most fish also possess an external lateral line system used to detect nearfield water movement to mediate crucial behaviors such as schooling, predator avoidance, prey detection, and orientation to water flow (Coombs et al. 2014). Similar to the inner ear, each lateral line sensory organ (neuromast) contains a population of sensory hair cells. These hair cells are sensitive to low frequency stimuli (Weeg and Bass 2002; Higgs and Radford 2016), making them potentially susceptible to damage from vibration and low-frequency sound generated by hatchery equipment. A few hours of broadband sound exposure is sufficient to damage lateral line hair cells in larval zebrafish, with potential consequences for orientation behavior (Suli et al. 2012; Uribe et al. 2018). Hatchery rearing can also negatively impact lateral line morphology in steelhead (*O. mykiss)*, with fewer superficial neuromasts observed in fish reared in conventional concrete raceways than in wild-spawned conspecifics (Brown et al. 2013). Interestingly, fish reared in net pens in the center of an isolated freshwater lake (a relatively quiet environment) had an intermediate number of superficial neuromasts as compared to the other rearing conditions. However, it is unknown if lateral line development is specifically impacted by hatchery sound levels.

## Effect of Noise on Hatchery-Reared Chinook Salmon

To address this question, we exposed hatchery-reared Chinook salmon (unpublished observation) to one of three different noise treatments from yolk sac absorption to

release 4 months later: a quiet treatment in which sound attenuation techniques were implemented; an ambient treatment in which no changes were made to experimental tanks; and a white noise treatment where developing fish were treated with 150 dB (re 1 μPa RMS) broadband white noise. Juveniles exposed to white noise had a significantly lower number of trunk neuromasts than juveniles in the "quiet" treatment (unpublished observation), suggesting that hatchery noise is sufficient to affect lateral line development, with potential behavioral consequences. Future studies will examine the impact of hatchery noise on stress responses in Chinook salmon, as such responses can influence postrelease survival.

## Effects of Aquaculture Sound on Stress and Immune Responses

Excessive sound exposure can reduce disease resistance in fish and potentially impact survival (Wysocki et al. 2007; Quinn 2018; Masud et al. 2020). This threat is exacerbated by the high stocking densities observed in many aquaculture facilities, facilitating the spread of pathogens (Iguchi et al. 2003; Crosbie et al. 2010). In some fish, excessive sound can lead to changes in tissue and blood chemistry that diminish the immune system's ability to combat disease. For example, exposing largemouth bass (*Micropterus salmoides*) to 60 days of recirculating aquaculture system noise (115 dB SPL re 1 μPa) significantly increased catalase (CAT) activity and decreased malondialdehyde (MDA) activity in liver tissue (Hang et al. 2021). CAT scavenges free radicals and is produced in response to oxidative stress to protect cells, while MDA increases due to oxidative stress, suggesting that noise exposure increased oxidative load in this species. Similarly, exposure to ambient offshore aquaculture noise increased oxidative and cellular stress markers in juvenile gilthead sea bream (*Sparus aurata*), a marine aquaculture fish (Filiciotto et al. 2017). Oxidative stress is regulated by several intracellular processes, with mitochondria playing a key role. Unlike mammalian red blood cells, mitochondria are present in mature red blood cells in nonmammalian vertebrates, including fishes (Phillips et al. 2000; Pica et al. 2001). Noise exposure in peled (see above) decreased mitochondrial activity in red blood cells, accompanied by a decrease in red blood cell area (Sapozhnikova et al. 2020). Collectively, these results suggest that sound exposure in fish can increase oxidative stress via multiple mechanisms.

The stress hormone cortisol is a particularly important health metric, as elevated levels of cortisol decrease immune response, growth, and food conversion (Gamperl et al. 1994; Wendelaar Bonga 1997; Harris and Bird 2000). Broadband sound exposure caused a transient, intensity-dependent increase in plasma cortisol level in juvenile and adult Atlantic cod populations (Sierra-Flores et al. 2015). Atlantic cod broodstock exposed to this noise paradigm also had reduced egg fertilization rates, reduced egg viability, and increased egg cortisol levels, suggesting that maternal cortisol levels may impact egg cortisol to decrease embryo viability (Sierra-Flores et al. 2015). Sound exposure also increased cortisol levels in two freshwater aquaculture fishes, the common carp (*Cyprinus carpio*) and the European perch (*Perca fluviatilis*) (Wysocki et al. 2006). Interestingly, these cortisol increases

were only seen with ship playback noise but not continuous Gaussian noise (Wysocki et al. 2006). Radford et al. (2016) noted a similar finding in European seabass (*Dicentrarchus labrax*), where stress responses (ventilation rate, in this case) increased in response to impulse noise from pile driving or seismic survey, but not continuous boat noise. Long-term stress can lead to chronic conditions and make fishes more susceptible to diseases. Monitoring fish health is a crucial aspect of any aquaculture program, and excessive anthropogenic noise may increase the effort and cost to keep pathogens at bay. Further, sound exposure and the resulting chronic stress can negatively impact fish behavior.

## Effects of Aquaculture Noise on Behavior

Environmental conditions and stocking densities can alter social, exploratory, food-searching, and avoidance behaviors (Jørgensen et al. 1993; Papoutsoglou et al. 1998). Anthropogenic sounds have a particularly pronounced effect on what behaviors fish express and to what degree. Aquatic animals exposed to noise pollution often exhibit increased aggression, flight reaction, and hiding behavior, likely due to stress (Kunc et al. 2016). Hasan et al. (2018) found that exposure to boat noise playback attenuated the response to a conspecific chemical alarm cue in fathead minnows (*Pimephales promelas*), greatly reducing refuge-seeking behavior. This muted predator response puts affected minnows at a greater chance of predation. Brief anthropogenic sound exposure also altered behavior in European eels (*Anguilla anguilla*) in a lab experiment (Simpson et al. 2015). Juvenile eels were exposed to recorded ship noise (148 dB re 1 μPa) while experiencing a visual stimulus to simulate a predator attack. Eels exposed to ship noise were less likely to startle in response to the perceived presence of an ambush predator, and fish that did respond had slower startle responses. Boat noise exposure also reduced anti-predator behavior in a field experiment with juvenile ambon damselfish (*Pomacentrus amboinensis*), ultimately reducing survival (Simpson et al. 2016). Much more research is needed to analyze how sound-induced behavioral changes affect hatchery-reared fish that are subsequently released into the wild.

In addition to the direct individual antipredator response, aquaculture sound is likely to alter group swimming behavior. Schooling is a vital swimming behavior that serves social and defensive purposes, used to confuse predators or, in certain instances, to increase feeding efficiency (Partridge 1982). Largemouth bass exposed to sound from recirculating aquaculture systems showed a more irregular schooling pattern than fish exposed to ambient sound levels (Hang et al. 2021). Sound-induced erratic schooling behavior was also observed in an important marine aquaculture species, bluefin tuna (*Thunnus thynnus*). Bluefin in a very large sea pen exposed to boat noise swam in looser schools and with more vertical movement than fish exposed to ambient sound (Sarà et al. 2007). Just like many other responses to anthropogenic noise, changes in swimming behavior are species-specific, with some fish showing more pronounced reactions. Boat noise exposure elicited an obvious escape response in spottail shiner (*Notropis hudsonius*), a species with

specializations that increase hearing sensitivity, while bluegill sunfish (*Lepomis macrochirus*), a species considered a hearing generalist, showed no discernable escape behavior (Stasso et al. 2023). Both species in this study showed a decrease in overall swim behavior, and the kinds of behavior they exhibited tended to be more erratic than in a quiet environment. Given that food can be sparse in natural environments, energy conservation becomes key to survival. The erratic behavior observed in fish exposed to noise serves no function as an antipredator or food-seeking behavior. Superfluous movements deplete critical energy by increasing respiration, pulse rate, and utilizing muscles.

## Conclusion

This chapter demonstrates that aquaculture and related anthropogenic sounds can impact fish at many levels, from development to behavior to reproductive success. The better the consequences of noise are understood, the better is the ability to design solutions to counteract these effects. To form reasonable solutions, much more research is needed to determine if and to what degree long-term survival is affected in aquaculture species. In many cases, fishes raised in aquaculture facilities are either released into the wild or harvested for food, making obtaining reliable survivor measures difficult. The chapter summarized in this review adds to the pool of knowledge that conservationists can use to form a viable way forward.

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# Effects of Low-Frequency Continuous Noise on Fishes

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**Abstract**

Anthropogenic low-frequency continuous noise is on the rise in aquatic environments due to increased shipping, resource exploration, and infrastructure development. Tank-based studies using low-frequency continuous noise caused temporary hearing loss and behavioral alterations or masked relevant acoustic signals in some species. Various examples of acoustic communication in fishes reflect the importance of the reception of acoustic signals. Recent results have proven that noise also affects free-ranging fish, but regulating measures and mitigation approaches have not yet been taken. The European Union Marine Strategy Framework Directive is currently working on the assessment and monitoring of continuous noise, with focus on vessel noise. A necessary new energy resource such as wind farms, although expanding, is not specifically included. Prevention and mitigation measures, e.g. speed restrictions and technical improvement in vessels, have been discussed but have not been made mandatory to reduce underwater source levels. Current expertise demonstrates that source-level reduction is also reducing impact area and mitigation could be implemented regardless of knowledge gaps.

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Fish · Low-frequency noise · Continuous noise · Temporary threshold shift · Noise mitigation · Offshore wind farms

## Introduction

The increase of anthropogenic noise in the ocean, due to globalization, population growth, and expanding transportation sectors, is contributing to an ongoing rise of low-frequency noise of up to 3 dB per decade (Andrew et al. 2011; Chapman and Price 2011). Fishes are capable of detecting, discriminating, and localizing sounds. Sounds are used for orientation and habitat selection, predator and prey detection, and intra- and interspecific behaviors such as aggression, territory protection, defense, and reproduction (Popper and Hawkins 2019). This makes them vulnerable to a wide range of effects of acoustic pollution.

Assessing the effects of underwater noise is complex due to sound source and environment-dependent variability in things such as amplitude, spectral, and temporal properties. Further, noise can derive from single or from multiple sound sources simultaneously. These factors determine the acoustic characteristics of noise in a certain area and are mainly described in terms of intensity, frequency, and wavelength. The acoustic characteristics used in describing noise evidently determine the degree of behavioral and physical impact of noise on fishes (Popper and Hawkins 2019). Studies on the effects of high-intensity impulsive noise have found behavioral changes in fishes, such as higher state of vigilance (Spiga et al. 2017), affected shoaling behavior (Herbert-Read et al. 2017), and potential displacement (Iafrate et al. 2016). Besides behavioral effects, particularly high noise levels can also induce physical injuries. Impulsive noise, from pile driving or air guns used in seismic surveys, damages sensory hair cells (Casper et al. 2013) or swim bladders (Halvorsen et al. 2012a, b), both important elements in hearing. Temporary effects on hearing thresholds can also derive from the exposure to impulsive noise in fishes (Smith and Monroe 2016). Fish species with better hearing abilities were found to be the most vulnerable for effects on hearing (Popper et al. 2005).

Up to now, most efforts focused on mitigating the effects from high-intensity impulsive sounds on aquatic animals, since immediate physiological injury to tissues and behavioral reactions have been clearly demonstrated (Halvorsen et al. 2012b; Neo et al. 2014, 2015; Southall et al. 2019). However, continuous noise from shipping, construction work, rotating wind, and tidal turbines occurs temporally and all around the oceans and potentially affects fishes over wide geographic areas (Popper and Hawkins 2019). Growing transportation sectors are increasing the global vessel fleet each year and raise the anthropogenic low-frequency noise levels (Andrew et al. 2011; Chapman and Price 2011). Offshore wind farms are expanding as a renewable energy solution in recent years, especially in European waters. This rapid expansion of offshore wind energy will inevitably increase low-frequency noise levels, which should be considered for marine spatial planning and

environmental impact assessments (Tougaard et al. 2020). These anthropogenic sources produce long-term continuous sounds that increase the overall sound level in the aquatic environment, interfering with critical aspects of fish behavior and potentially hearing.

This chapter reviews current knowledge of the impacts of low-frequency continuous noise, its ecological consequences for wild fishes, current assessments, and noise mitigation measures.

## Behavioral Effects of Continuous Noise Exposure

Continuous noise can affect the behavior of fishes in various ways, but data is restricted to few species showing considerable variability. Most data is from tank experiments with playbacks often not resembling actual sound sources (Akamatsu et al. 2002), and behavioral responses to stimuli may differ from wild fish (Leplage et al. 2000; Calisi and Bentley 2009; Benhaïm et al. 2012).

Vessel noise exposure can reduce the foraging rate (Purser and Radford 2011; Magnhagen et al. 2017; Pieniazek et al. 2020) and lead to increased stress levels (Wysocki et al. 2006). Fish are less successful in repelling intruders (Sebastianutto et al. 2011) and are compromised in their antipredator behavior during playbacks of vessel noise compared to "silent" controls (Velasquez Jimenez et al. 2020). Increased aggregation in fish was found immediately upon white noise exposure with a return to control condition after 2 h (Mauro et al. 2020). Continuous noise can delay reproduction (Blom et al. 2019) and lead to decreasing exploratory activities and to alterations in home ranges (Ivanova et al. 2020; Kok et al. 2021).

In addition to behavioral effects, continuous underwater noise can also affect auditory detection of biologically relevant signals in fishes. Continuous noise has a high potential for masking if it overlaps in time and frequency with fitness-relevant sounds (Popper and Hawkins 2019). Detection thresholds are found to increase in the presence of background noise in goldfish and catfish (Wysocki and Ladich 2005). Continuous noise that is 34 dB re 1 μPa higher than ambient sound delayed female spawning decisions due to a reduced ability to assess male acoustic signals (Blom et al. 2019). Such reduced effective vocalization radii are a well-described problem for fishes (Radford et al. 2014; Stanley et al. 2017). Masking may be overcome by adjustments of amplitude or frequency of signals, also known as the Lombard effect (Codarin et al. 2009; Luczkovich et al. 2016; Brown et al. 2021). However, biological constraints exist in the ability to compensate for masking effects.

## Temporary Threshold Shifts Induced by Continuous Noise

Exposure to intense sounds or sounds of long duration can lead to sensory hair cell damage of the inner ear and is possibly correlated to temporary threshold shift (TTS) in hearing (Smith et al. 2006). Hearing may recover after exposure, which can be

enhanced by simultaneous repair or replacement of sensory cells in some species (Smith et al. 2011; Smith and Monroe 2016).

Continuous noise with much lower received sound levels than pile driving can also cause TTS in fishes. Likewise, hearing thresholds can be elevated by 5 dB after a 10-min exposure to white noise with 160–170 dB re 1 μPa and around 28 dB after 24 h (Smith 2004). The fish had similar threshold shifts after more than 24 h of exposure, indicating that an asymptotic threshold shift (ATS) was reached (Smith 2004). An ATS was also reached after 2 h of white noise exposure of 142 dB re 1 μPa (Scholik and Yan 2001). Even lower levels of 130 dB re 1 μPa could induce a TTS after 24 h exposure (Amoser and Ladich 2003; Breitzler et al. 2020).

Susceptibility to hearing loss in fishes differs between individuals and different species. The presence or absence of a swim bladder as well as other specializations that can contribute to enhanced hearing sensitivity in different species may alter the occurrence of hearing impairment (Popper and Hawkins 2019). Furthermore, impact variability can also be affected by hearing abilities of the exposed fish; a more intense sound will be required to produce TTS in fishes with low hearing abilities (Smith et al. 2004).

## Continuous Noise Exposures in Wild Fishes

It is debatable whether TTS-onset thresholds are regularly exceeded in real-life scenarios and if this is a problem occurring in harbors, wind farms, and shipping lanes. The source levels of large moving vessels can exceed 190 dB re 1 μPa (MacGillivray and de Jong 2021), while source levels from static vessels, when in dynamic positioning mode, can be 10 dB higher (Rutenko and Ushchipovskii 2015). If animals do not move away from static vessels or if a shipping lane is close to a structure with stationary animals, received levels may be exceeded, capable of inducing a TTS. In high traffic areas, cumulative sound exposure levels exceed reported TTS onsets (e.g., 24 h × 130 dB re 1 μPa = 180 dB re 1 $\mu Pa^2s$) (Merchant et al. 2014). However, so far, no hearing tests have been conducted after in situ vessel exposures with wild animals. In addition to vessel noise, dredging is also able to reach levels (SPLs up to 190 dB re 1μPa at 1 m) that are able to induce a TTS, but no direct effects have been investigated yet (Scholik and Yan 2002; Heinis et al. 2013; Liu et al. 2013; Wenger et al. 2017). Operational noise from wind turbines has sound levels reaching up to ~137 dB re 1 μPa at a distance of 40 m, depending on wind and rotational speed, pitch angle of the turbine's blades, the type of foundation, and possibly the age of the turbine (Tougaard et al. 2020). This combination of continuous noise emissions and artificial reef structures, which may attract fish, must not necessarily be a TTS threat, but cumulative contribution from multiple turbines should not be disregarded (Tougaard et al. 2020).

Noise exposure has proven to cause TTS in fishes, but there is no evidence for permanent hearing loss (PTS) (Smith et al. 2006; Smith and Monroe 2016). This raises the question about ecological relevance of temporary hearing effects. While having impaired hearing, animals may have lower survival and reproduction rates in

terms of poorer communication, inability to detect predators or prey, and difficulty in assessing their environment. To our best knowledge, no studies have been done on free-ranging fish to investigate whether a small change in hearing threshold has any real impact on fitness. Although hearing of fishes is more likely affected by impulsive noise, continuous noise still has the potential for inducing TTS as well, which must not be neglected.

In recent years more studies have been done in the wild with free-ranging fish that have the potential to escape from areas with disturbing factors. Field experiments showed decreased presence of wild fish and fewer foraging attempts made during boat noise exposure (160 dB re 1 μPa) (Pieniazek et al. 2020). However, variation in hearing abilities may determine the extent to which their behavior changes and the resulting degree of negative consequences (Pieniazek et al. 2020). A decreased magnitude in antipredator response was found in free-ranging sand gobies (*Pomatoschistus minutus*) during the exposure to boat noise with comparably low levels of 98–101 dB re 1 μPa (Kok et al. 2021). Prey species altered their antipredator behavior in ways that can potentially lead to higher mortality for individuals and ultimately have implications for the food web. Fish also increased hiding and their time spent inside the shelter during boat noise exposure (Picciulin et al. 2010; Nedelec et al. 2016). Tagged common roaches (*Rutilus rutilus)* showed increased swimming speed and increased their use of the central part of the lake (Jacobsen et al. 2014). Vessel noise and vessel presence can be perceived as a threat by Arctic cod (*Boreogadus saida*), which react by moving away and decreasing exploratory activities. These changes in behavior may represent an interruption in a critically needed exploitation of important and seasonally abundant food resources (Ivanova et al. 2020). Male plainfin midshipman fish (*Porichthys notatus*) exposed to noise produced fewer vocalizations, reduced the frequency of vocalizations, and increased the amplitude of their mating hum (Lombard effect). However, it did not significantly affect spawning success, suggesting that the Lombard effect allowed males to sustain signal efficiency (Brown et al. 2021). Likewise, the duration and intensity of chorusing fish were increased by noise from operating wind turbines (Siddagangaiah 2021). Nonetheless, possible limitations for further vocalization adjustments are unknown. Fine-scale telemetry data suggests that dredging does not create a barrier for adult Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) migrating to their spawning habitat (Balazik et al. 2020), while reductions in fish diversity were recorded during dredging most likely due to behavioral avoidance of the sediment plume (Pledger et al. 2021). However, the cause of avoidance cannot be clearly determined, as effects from dredging noise, increased turbidity, reduced light, contaminants, or gill damage cannot be distinguished in the wild (Wenger et al. 2017).

There have also been controversial results with instant effects in wild fish and a return to baseline behavior after an initial reaction during vessel sound exposure (Holmes et al. 2017; Staaterman et al. 2020). Additionally, Nedelec et al. (2016) found that fish increased hiding behavior and elevated ventilation rates during playback boat noise exposure but habituated within 1–2 weeks of exposure. There is evidence that population impacts due to noise disturbance exist, deriving from

subtle effects on fish behavior and physiology that, in turn, may reduce population growth rates (Soudijn et al. 2020). Future studies are needed to determine limits of adjustment possibilities, the potential of habituation, and fitness costs of altered behaviors. These information are critically required to estimate effects on spawning and reproductive success and therefore effects on population levels.

## Future Direction of Noise Mitigation

In 2008, aquatic noise pollution was included in the European Union Marine Strategy Framework Directive (MSFD; European Union 2008) working toward levels of underwater noise that do not negatively affect ecosystems and would ensure "Good Environmental Status" (GES). In recent years, research has proven severe impacts from noise on a broad spectrum of taxa in aquatic life (Duarte et al. 2021). Mitigation measures for high-intensity noise are already in use, mainly bubble curtains and acoustic deterrent devices during pile driving (Juretzek et al. 2021). Quieter alternatives to seismic air gun surveys have been developed and tested successfully but are not commercially used due to a lack of regulatory incentive (Merchant et al. 2022). While high-intensity impulsive noise had a lot of attention due to its capability of inflicting severe damage (Dekeling et al. 2020), progress on monitoring, assessing, and mitigating continuous noise pollution has been comparatively slow, yet regional maps have been produced for the Baltic, North Sea, and wider NE Atlantic (Merchant et al. 2022). Effects for impulsive noise are thought "of lesser relevance for continuous noise sources where received noise levels are insufficient to inflict damage for short exposures, and continuous exposures to high levels are less likely, due to the sources (e.g., ships) and receivers (animals) often move relative to each other, limiting the time with exposure to high levels of noise," states the assessment framework for continuous noise of the MSFD (Sigray et al. 2022).

Nevertheless, low-frequency continuous noise has also been recognized as a threat to the GES in the MSFD. Since shipping makes up the majority of underwater noise pollution, continuous noise is defined in annual trends in one-third octave frequency bands centered at 63 and 125 Hz (Sigray et al. 2022). Although the increasing occupation of wind farms of larger coastal areas contributes considerable cumulative energy to the soundscape, the monitoring programs are mainly considering vessel noise from AIS (automatic identification system) and VMS (vessel monitoring system) data in their continuous noise assessment and do not specifically address wind farms in their impact evaluation on aquatic life. The life cycle of offshore wind farms, includes a range of activities, generating underwater noise of varying intensity, duration, and spectra (Mooney et al. 2020). The effect of continuous noise has to be carefully considered to ensure that wind farms do not represent ecological traps for fishes (Reubens et al. 2014). For example, wind farms could be seen as good habitats with high food availability for fishes (van Hal et al. 2017),

where fishing is forbidden and building reef structures provide shelter and possibly hard substrate for spawning. However, they may compromise animals acoustically, masking necessary mating signals (Blom et al. 2019; Tougaard et al. 2020). Although wind turbines' new substrate habitats generally seem to attract fish, they potentially alter the fish composition with no knowledge about future habitat and population impacts (Bergström et al. 2013). Biodiversity loss and climate change are linked and should be approached simultaneously. Wind farms are essential as a solution as new energy source, but impacts on habitats should not be neglected.

The MSFD states that there is currently no sufficient data to include noise-induced hearing loss in continuous noise assessments. Since fewer fishes are possibly affected by noise-induced hearing loss from continuous sound than those affected by masking, the MFSD is addressing masking in determination of habitat status for continuous noise (Sigray et al. 2022).

The MSFD is ensuring that underwater noise "is at levels that do not adversely affect the marine environment," but the focus is on assessment and monitoring rather than prevention and mitigation (Merchant et al. 2022). To date, effects of continuous noise are often regarded as negligible, although evidence for an impact on hearing and behavior exists. A first step toward considering effects of continuous noise on hearing could be the monitoring of vessel noise on a temporal scale and a review of existing data about TTS in fishes that allows to estimate areas and times which exceeded TTS onsets. For mitigation of continuous noise, mainly shipping, only nonmandatory guidelines exist. Quieting technologies have long been applied to military and fishery vessels (Southall 2005) but are not used for larger commercial ships in general. Vessel noise mitigation can include technical adaptations to decrease underwater noise, while contributing to resource efficiency (Merchant et al. 2022). SATURN, a research project funded by the European Union under Horizon 2020, is currently working to identify the most promising options for measuring and reducing the negative effects of ship noise that can be applied to current and future ships. Operational measures, such as vessel speed restrictions, have proven to reduce noise levels from the majority of commercial vessels (MacGillivray et al. 2019). As the offshore wind industry expands fast globally, research priorities for sound and vibration effects on fishes and aquatic invertebrates have been suggested to fill in knowledge gaps and inform cumulative impact models, to understand effects on populations and ecosystems (Popper et al. 2022). It is often argued that additional in situ studies with actual sound sources and free-ranging fish are needed, since tank experiments do not resemble ideal soundscapes and captive fish do not show normal behavioral reactions. Reducing source levels of a sound reduces the affected habitat size and thus reduces impacts, both known and unknown, in many taxa, from individual to ecosystem level, and could be implemented with current knowledge (Risch et al. 2021). Even moderate-intensity continuous noise is able to elevate hearing thresholds and compromise important behaviors; therefore mitigation measures to decrease ambient noise levels are not unreasonable.

## Conclusion

The ability to perceive and react to acoustic signals is highly fitness relevant for many fish species. Rising low-intensity noise sources alter behaviors that evidently have high impact, such as foraging decrease and spawning reduction, which could affect population growth. Disturbance in perception, no matter if derived from stress, masking, or hearing impairment, must be avoided. Therefore, noise levels have to be monitored and effects on fishes in the wild have to be determined. This should be considered in marine spatial planning for wind farms and marine traffic. Wind farms are a needed energy resource, but biodiversity loss and climate change are interconnected issues that must be approached together. Current knowledge already makes it clear that source-level reduction is also reducing impact area. Therefore, regulatory and mitigation approaches could be implemented regardless of knowledge gaps.

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# Response Persistence to Repeated Sound Exposure During a Tank Study in Pelagic Fish

67

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**Abstract**

Marine fish inhabit an acoustic world and are often exposed to anthropogenic noise. Natural sounds play an important role in spawning aggregations, finding prey, escaping predators and guiding spatial behavior. Noise pollution at sea includes the prominent presence of sounds from vessels, but also pile driving, seismic surveys, and explosions, which can cause extreme overexposure. Such extreme events can cause physical damage and even mortality for nearby fish. Acoustic Deterrent Devices, using sounds at moderate levels, may be used to scare fish away from dangerous sites. Investigation of acoustic responsiveness to deterrent sound stimuli is challenging at sea due to low-visibility and limitations of tagging studies, while captive studies yield high resolution data that can be used to compare sound treatments. In this study, eight pulse train variants were tested on captive Atlantic herring (*Clupea harengus*) and methodological steps

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for measuring 3D behavior automatically for groups of four fish are reported. Potential decay in responsiveness was tested at two levels: within the repetitive exposure of ten pulse trains during a treatment, and across eight treatments over 2 days. Upon sound onset, herring showed an immediate increase in turning behavior and swam deeper in the basin. There was no decay or change in any behavioral response at either level. The discussion explores the importance of response persistence within the context of responsiveness tests in captivity.

**Keywords**

Anthropogenic noise · Acoustic Deterrent Device · ADD · Sound exposure · Habituation · Looming stimulus · Atlantic herring

## Introduction

Human activities at sea can be noisy and thereby affect marine fauna (Slabbekoorn et al. 2010; Kunc et al. 2016; Duarte et al. 2021). Such anthropogenic noise can lead to issues for marine animals through deterrence, disturbance, distraction, and/or masking of relevant sounds. High amplitude, impulsive sounds, such as from seismic surveys, pile driving, or explosions, may cause physical injury, which can even be fatal. The behavioral and physiological effects of anthropogenic noise may affect many individuals and species spread over a large area (Slabbekoorn et al. 2010; Cox et al. 2018). The physical and mortal effects are typically restricted to smaller areas, affecting fewer individuals and species (Ladich 2008; Popper and Hastings 2009; Halvorsen et al. 2012). The relatively short range at which impulsive sounds pose a direct danger opens up opportunities to exploit moderate acoustic stimuli to herd animals away from danger, for their own safety.

Moderate-level sounds can be used in Acoustic Deterrent Devices (ADDs) to protect aquatic animals from severe overexposure to loud sounds (Putland and Mensinger 2019). Minke whales (*Balaenoptera acutorostrata*), for example, moved away immediately after sound onset during an experimental test of an ADD, typically increasing horizontal speed, and directionality, away from the source, while extending dive duration (Boisseau et al. 2021). The so-called "Seal-scarer" and "FaunaGuard" were developed with the intention of guiding marine mammals away from dangerous acoustic events at construction sites, and there are reports of successful application for harbor porpoise (*Phocoena phocoena*) (Brandt et al. 2013; van der Meij et al. 2015; Geelhoed et al. 2017; Voss et al. 2023). ADDs are also often used to scare marine mammals away to prevent predation of the harvest at fish farms and marine mammal by-catch during fishing at sea, although the success of deterrence may be variable and fade over time (Carretta and Barlow 2011; Götz and Janik 2013).

Not only marine mammals, but also fish, may be spatially affected and redirected by sound, as is clear from several examples from studies in rivers. At short distance from the source, migrating individuals of European eel (*Anguilla Anguilla*) were

shown to avoid the river side from which infrasound was played, to guide them to a safe passage (Sand et al. 2000). Maes et al. (2004) reported large reductions in trapped Atlantic herring (*Clupea harengus*) and sprat (*Sprattus sprattus*) using an ADD at a cooling water inlet of a nuclear power plant. Combinations of sound, strobe lights, and bubble screens proved to be successful in guiding juvenile Chinook salmon (*Oncorhynchus tshawytscha*) away from a low-survival migration route (Perry et al. 2014), and in curbing the upstream migration of invasive silver carp (*Hypophthalmichthys molitrix*) and bighead carp (*H. nobilis*) (Ruebush et al. 2012). A recent review (Putland and Mensinger 2019) showed that most of the published deterrence studies showed positive results (>50% deterrence); however, these results varied among taxonomic groups (Popper and Carlson 1988) and publications may be biased to positive results. Examples of acoustic herding or redirecting of fish at sea are reported with respect to both deterrence and attraction in the context of fishing (Wahlberg 1999). However, detailed insights into general acoustic responsiveness patterns for marine fish at sea are lacking (Hawkins et al. 2014; Slabbekoorn et al. 2019), and very little is known about whether responsiveness persists or fades over time with repeated exposure (Wahlberg 1999; Bejder et al. 2009; Rankin et al. 2009).

Habituation is an important phenomenon, not only to understand the detrimental impact of anthropogenic noise, but also to understand the persistent efficiency of ADDs. Habituation concerns a decrease in behavioral response to repeated stimuli, when this decrease cannot be attributed to sensory adaptation or motor fatigue (Bejder et al. 2009; Rankin et al. 2009). Sensory adaptation refers to a decrease in response of the sensory system and motor fatigue to exhaustion of the muscles. Several marine animals, including fish (Johansson et al. 2016; Radford et al. 2016; Neo et al. 2018), cephalopods (Samson et al. 2014; Mooney et al. 2016), and bivalves (Hubert et al. 2022), have been shown to decrease their behavioral or physiological stress response over repeated sound exposures. Persistent potential to trigger a response at moderate sound levels, after repeated exposure, or long-term impact on swimming behavior and spatial avoidance, seem to be the most critical features for efficient stimuli for ADD applications.

However, data on pelagic fish are particularly limited, as they are difficult to gather not only due to the obvious exposure and observation challenges at sea, but also due to the challenges of testing them in captivity. Sound propagation in basins and tanks differ substantially from propagation at sea (Rogers et al. 2016; Campbell et al. 2019), and lack of escape possibilities are an inherent restriction, and thus determine the specific research questions that can be targeted. Nevertheless, studies in captivity are an important tool to study behavioral responses to acoustic stimuli in fish, complementary to observational and exposure studies in the field (Slabbekoorn 2013). Advantages of indoor studies include better visibility, known subject condition and exposure history, easier replication of comparable exposure events, and appropriate controls. Insights from indoor studies typically require outdoor follow-ups to confirm and test findings in real-world applications.

Variation in responsiveness in wild-caught Atlantic herring (*Clupea harengus*) to pulse train stimulus variants was tested with the intention of potentially improving

sounds for ADDs (Demuynck et al. unpublished). A pulse train was repeatedly played back in an indoor basin study: ten times within an exposure treatment of one sound stimulus, and eight different sound stimuli, in eight sequential treatments over 2 days. Persistence of immediate response patterns over repetition of ten of the same pulse trains of a treatment, and persistence of a response in a longer-term change, over the complete trial of eight different treatments per group of four fish, was explored. Changes over time, or lack thereof, at these two different time scales, may provide fundamental insights into auditory perception and anxiety-related responsiveness of pelagic fish, reveal validity of our experimental design, and suggest applied potential for ADDs.

## Methods

### Study Species and Housing

Atlantic herring are a common species of the North Sea, including the Belgian and Dutch parts, where they stay for the first one or two years of their life. Herring have a pelagic lifestyle and typically swim in large schools. In the early life stages, they live in relatively shallow coastal waters (up to 30 m depth in the southern part of the North Sea). They are among the better hearing fish species and hear frequencies up to 4 kHz or more (Enger 1967; Popper and Fay 2011). Young Atlantic herring were caught in the Westerschelde (sea inlet in Zeeland, the Netherlands), on July 16, 2021 (Demuynck et al. unpublished), with a large net by professional fishermen and then transported in two 500 L, well-aerated water tanks. The fish were kept in two stock basins (3 m diameter, 1 m height) until the exposure experiments at the marine fish facilities of Stichting Zeeschelp (Kamperland, the Netherlands). Fish were fed daily with crushed amphipods (*Gammarus spp*). The herring had an average length of 8.4 cm and average weight of 3.7 g when they were transferred for the experiment to a third basin, identical in size and shape to the stock basins. All basins had a continuous flow-through of filtered sea water from the adjacent Oosterschelde sea inlet (Zeeland, the Netherlands).

### Experimental Exposure Conditions

The herring were transferred in groups of four to the experimental basin to acclimatize for at least one night. In total, ten groups were exposed to eight different sound treatments over two subsequent days (see Demuynck et al. unpublished). Each sound treatment consisted of a different pulse train variant, which was repeated for a total of ten times. The pulse trains were 10 s long and made in Audacity® version 3.0.2 (Audacity Team 2021) with 35 white noise pulses of 0.02 s each, with a high-pass (150 Hz) and low-pass (4 kHz) filter applied. The eight different pulse trains varied in temporal pattern with respect to amplitude level and interval duration. The

amplitude was either constant (−1 dB relative to full scale (dB FS)), fading in (linearly from −31 dB FS to 1− dB FS), or fading out (linearly from −1 dB FS to 31− dB FS). The pulse rate was either regular (intervals of 0.275 s), accelerating (intervals linearly from 0.5 s to 0.05 s), or decelerating (intervals linearly from 0.05 s to 0.5 s). The sounds were played back with a Tascam DR-07 portable digital recorder, from a M033N, Kemo amplifier and a UW30, Lubell underwater speaker (frequency response 100 Hz – 10 kHz), hanging from a rod above with a rope, in the middle of the experimental basin.

## Behavioral Measurements via 3D Tracking

The behavior of the "schools" of four fish were recorded with five GoPro cameras (HERO3, HERO3+, HERO4, and HERO4+), which were evenly placed alongside the edge of the basin wall (see Demuynck et al. unpublished). Automatic tracking software (EthoVision XT, Noldus Information Technology) was used to extract the positions of each fish within the camera field of view, and subsequently to calculate the mean position of the group. Additionally, the mean distance between individuals was determined as a measure for group cohesion. By using the overlap of the fields of view of two adjacent cameras, the 3D position of the group center (x, y, z) could be calculated in R (Fig. 1). In this way, average swimming speed, swimming depth, and thigmotaxis (distance from the wall, or the complementary measure: distance from the speaker in the center of the circular tank) were automatically extracted.

Response variation was assessed at two-time scales (Fig. 2). Short-term responsiveness was measured as the difference between the 10 s bin just before the onset of the stimulus and a 10 s bin after the onset. Long-term responsiveness was measured as the difference between the 5 min bin before the start of a series of ten exposures of a treatment and a 5 min bin during the treatment period (Fig. 1). The silent periods between treatments were used as control (sham) before and during data, at both timescales. The change in behavior was determined (during – before) and compared during the exposures with the silent controls. To gain insight into the potentially changing response pattern over the ten consecutive exposures, the changes in behavior were tested with an ANOVA, in which the number of the pulse train was the explanatory variable, independent of the stimulus type and position in the series of eight exposure periods over the two days. In a similar way, the behavioral changes were tested for the number of exposure treatments in the sequence of eight, four on each day (here, treatment number was the explanatory variable in the ANOVA).

## Permits and Ethical Notes

This experiment was conducted in accordance with the Dutch Experiments on Animals Act and approved by the Dutch Central Commission Animal Experiments (CCD) under no. AVD1060020171085.

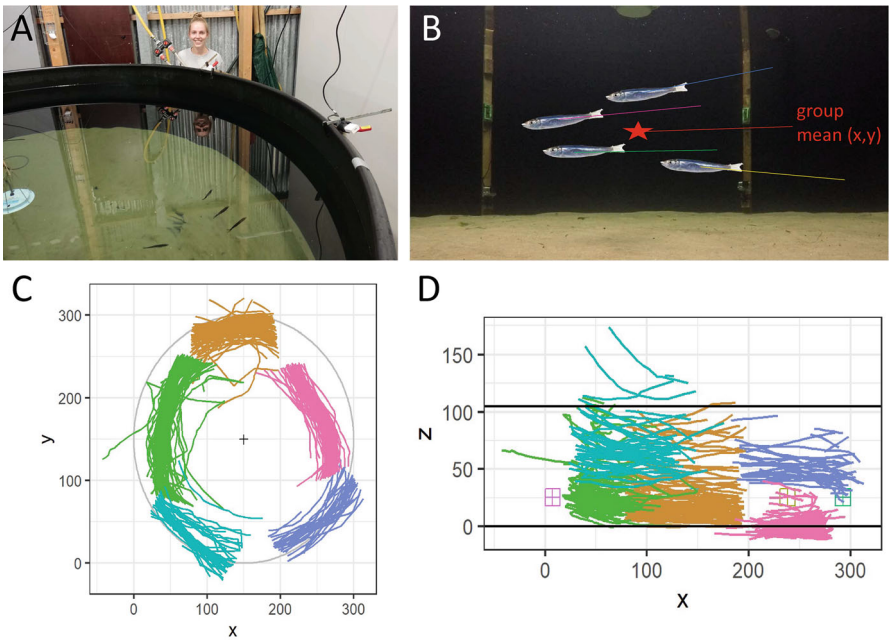



**Fig. 1** Overview of methodological steps for the behavioral assessment of responsiveness to sound stimuli by Atlantic herring: (**a**) Experimental basin with four juvenile herring, speaker in the center, and first author FD behind; (**b**) schematic representation of automatic tracking of the four fish to calculate group positions (indicated with red asterisk); (**c**) top view of 3D-tracking results of group averages for five different camera-pairs (each in different color) with x and y dimensions in cm; (**d**) side view of 3D-tracking results of group averages for five different camera-pairs (each in different color), with x and z dimensions in cm. Note that in (**c**) and (**d**) some tracks go outside of the basin (gray circle in **c** and horizontal black lines in (**d**), and that tracks of different camera-views are not perfectly connected. However, this was considered for the analyses, which looked at relative behavioral changes within certain time bins

## Results

The four fish of each group were typically swimming close together, relatively close to the tank wall (within 5–20 cm) (Demuynck et al. unpublished). The immediate responsiveness to the playback of sound stimuli typically included one, two, three, or all four fish showing a turning response, more than during the control periods (quasibinomial generalized linear model: intercept (control) = −2.41, slope = 3.07, $p < 0.001$). The immediate response to a sound exposure (first pulse train) at group level, not taking stimulus variant into account, was a significant decline in depth (ANOVA: $F1,44 = 5.60$, $p = 0.022$) and a nonsignificant trend for an increase in thigmotaxis (ANOVA: $F1,45 = 3.195$, $p = 0.081$). Group cohesion (ANOVA: $F1,44 = 1.628$, $p = 0.209$) and swimming speed did not change at the small timescale (ANOVA: $F1,44 = 0.836$, $p = 0.365$). Furthermore, none of the

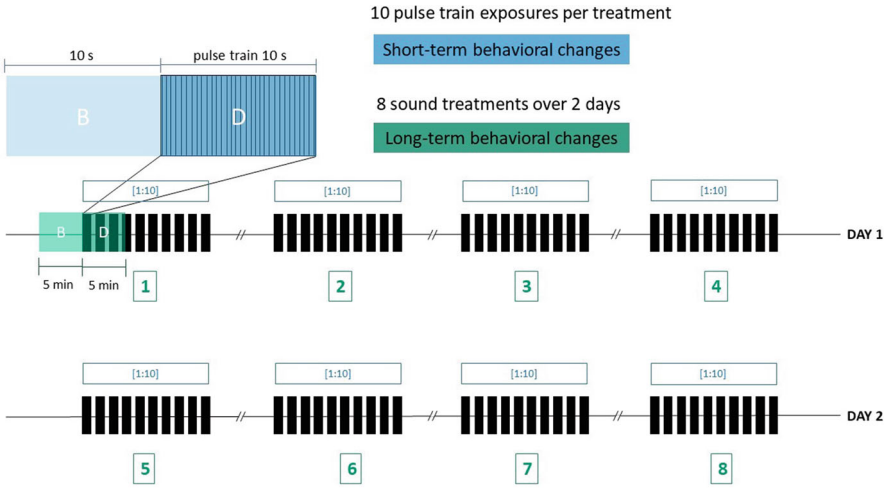



**Fig. 2** Overview of sound stimuli exposure design and two timescales of data processing. In the top left corner, the vertical lines in the black rectangular box represent the 35 pulses of a single pulse train variant (each pulse train was 10 s long). Each treatment period consisted of the playback of a single variant ten times. There were eight treatment periods in total, four on each of the two exposure days per group of four fish (n = 9, which were tested in sequence). The short-term response (in blue) was measured as an immediate change from a 10 s bin just before sound onset to a 10 s bin during the sound stimulus. The long-term response (in green) was measured as the change in behavior between a 5 min bin before the treatment periods and a 5 min bin during the treatment period (covering multiple 10 s stimulus playbacks in the overall 15 min treatment duration)

short-term behavioral measures revealed significant variation or change over time response in the ten subsequent exposures within a 15 min treatment period (ANOVAs: depth $F_{1,240} = 0.452$, $p = 0.502$; speed $F_{1,238} = 0.028$, $p = 0.868$; and group cohesion $F_{1,238} = 0.001$, $p = 0.976$, thigmotaxis $F_{1,241} = 2.07$, $p = 0.151$) (Fig. 3). The analyses of the longer-term response, not taking stimulus variant into account, revealed no significant changes in depth (ANOVA: $F_{1,91} = 0.216$, $p = 0.644$), speed (ANOVA: $F_{1,91} = 0.144$, $p = 0.705$), group cohesion (ANOVA: $F_{1,91} = 2.582$, $p = 0.112$), or thigmotaxis (ANOVA: $F_{1,91} = 0.836$, $p = 0.228$). None of the behavioral measures at this timescale showed a significant change or nonsignificant trend over time: There was no effect of treatment order, for the eight treatment periods over the two exposure days (ANOVAs: depth $F_{1,44} = 0.315$, $p = 0.577$; speed $F_{1,44} = 0.232$, $p = 0.632$; group cohesion $F_{1,44} = 0.0$ $p = 0.99$; and thigmotaxis $F_{1,44} = 1.078$, $p = 0.305$) (Fig. 3).

## Discussion

The groups of four herring significantly responded to sound exposures in the indoor conditions of our experimental setup. As an immediate response, the fish typically turned a few times, swam deeper, and showed a nonsignificant trend for swimming

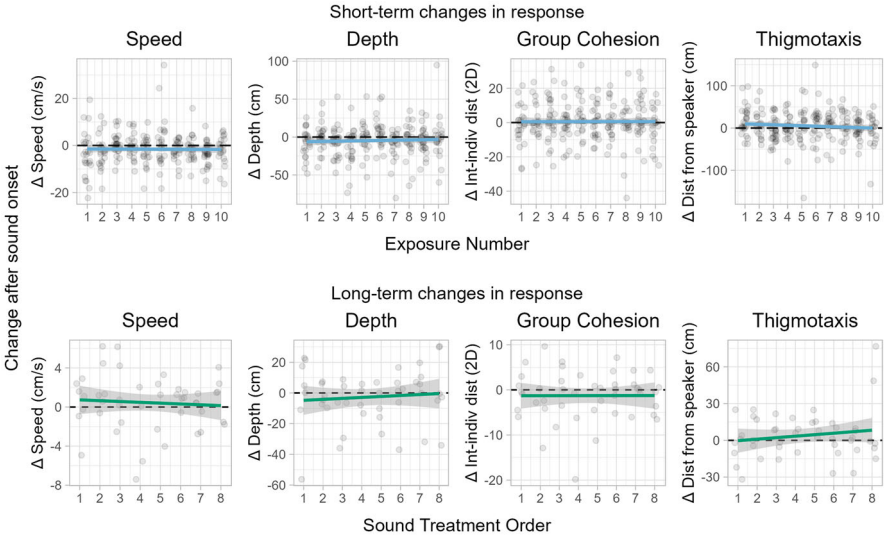



**Fig. 3** Results on persistence in behavioral responsiveness of Atlantic herring over repeated playback of sound stimuli (independent of specific pulse train variant) at two timescales. The short-term changes in response over ten sequential exposures during a treatment period are depicted in the top panel. None of the behaviors showed a significant change over the ten repeated exposures within the 15 min treatment periods. The long-term changes in response over eight sequential treatment periods, spread over two exposure days, are depicted in the lower panel. None of the other behaviors showed significant change over eight repeated tests

closer to the wall. None of the behavioral measures changed over subsequent exposures. The herring did not change their longer-term response, and their behavior also did not change significantly over the eight treatment periods over the two exposure days. These results indicated that after the initial, clear effect on turning rate and swimming depth, the four fish resumed swimming in the same baseline. Furthermore, the current analyses show that fading responsiveness, due to habituation, sensory adaptation, or motor fatigue, was not an issue for our measurements, at either timescale. This shows that the current experiment can be used to compare responsiveness among stimuli that varied in temporal patterns of change in amplitude and interval duration, despite the modest responsiveness in absolute sense (Demuynck et al. unpublished).

## Nature of Response Patterns

The immediate behavioral response patterns of our small artificial schools of four Atlantic herring included a significant response in turning and swimming depth, and a nonsignificant trend in thigmotaxis. Turning may be an important component for schooling behavior and has been described to be mainly related to the position of the nearest fish in a shoal (Katz et al. 2011). This could explain why turning behavior,

when increased during sound playback, was often displayed by all fish in the group. Increased turning during sound may be due to an increase in "alertness" or stress caused by the disturbance. A diving down response was expected as it is often found in other studies on captive fish, such as Atlantic herring (Doksæter et al. 2009, 2012a), European seabass (*Dicentrarchus labrax*) (Neo et al. 2014, 2018; Hubert et al. 2020), and in several marine fish in the wild (Sara et al. 2007; Handegard et al. 2010; Hawkins et al. 2014). The moderate, but significant nature of the vertical shift in the current study may indicate a moderate effect on anxiety or a spatial restriction of our relatively shallow basin conditions. Thigmotaxis, also called "wall-hugging," is a well-known measure of anxiety in laboratory studies in fish, especially zebrafish (*Dania rerio*) (see Schnörr et al. 2012; Kalueff et al. 2013). In our setup, closer to the wall also meant further away from the speaker. However, the fish were swimming relatively close to the wall most of the time as baseline behavior already, which seems to be a typical swimming pattern in captivity of pelagic fish in particular. Consequently, very little resolution remained for moving closer or away from the wall. This may be the reason for only a limited movement in the first exposure of the ten within an exposure period. While other studies have reported anxiety-related increases in group cohesion among schooling fish, forming tighter instead of less tight schools in conditions of apparent threat (Neo et al. 2014, 2015; but see Hubert et al. 2020), the results from this study lack any significant effect on group cohesion. However, even if the fish would have been swimming significantly closer together, this behavioral response does not appear to be advantageous for the efficiency of an ADD. On the contrary, like swimming downward, tighter schooling behavior may mean that they would stay at a particular location for longer. However, more studies on free-swimming fish in the wild are needed to confirm such an interpretation.

## Potential for Spatial Avoidance

As addressed above, the nonsignificant trend in thigmotaxis concerned an initial shift away from the speaker. Although this may be interpreted as a subtle spatial avoidance response, the effect was not significant, and it is not likely related to acoustic avoidance of the speaker in the center. Very little sound pressure level variation is expected over the scale of movement of the fish in response to the sound exposure, and they are relying on particle motion for directional hearing anyway (Popper and Fay 2011; Radford et al. 2012). Particle motion fluctuations in level may be similar sound pressure, but the directional patterns are highly complex in captive, shallow-water conditions, especially close to the basin wall and surface (Rogers et al. 2016; Hubert et al. 2016; Campbell et al. 2019), and are unlikely to provide useful information to the fish about speaker source location. Furthermore, despite the large size of the basin relative to the fish size and number, the fish may be restricted in revealing natural escape behavior in our captive conditions. Spatial avoidance in the wild can therefore not be excluded, but the current results should not be considered as evidence for a tendency of horizontal escape behavior. If anything, the thigmotaxis response may be related to a rise in anxiety. Whether a general rise in

anxiety yields any spatial avoidance in natural conditions at sea for herring or other pelagic fish remains to be investigated in situ.

## Conclusion

Atlantic herring in captive conditions responded to exposure of acoustic pulse trains as reflected by immediate turning behavior and by swimming deeper. Additionally, they showed an immediate response at small scale in swimming closer to the wall, although only to a moderate extent and this pattern was not significant. Consistent behavioral values across sequences of ten repetitions of the same stimulus were not surprising because of the overall low responsiveness. Nevertheless, it is argued that the nonsignificant trend in change in thigmotaxis may be an indication of a modest, initial rise in anxiety. At a larger scale, fish did not show any behavioral response to the sound treatment, which was persistent across eight treatment periods over the duration of two exposure days. The results indicate that a suitable behavioral assay was used to assess a variety of behavioral measures quantitatively and automatically, to evaluate relative variation in responsiveness to sound in juvenile Atlantic herring as a model species for pelagic fish. The methodology was sufficient to distinguish a response from no response at modest response levels. Responsiveness in captivity should not be translated directly to application potential of ADDs in pelagic fish at sea, but the increased turning behavior and trend in thigmotaxis may warrant further testing of ADD efficiency with fish in the field.

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# Chronic Boat Noise Impacts on the Early Life Stages of Cichlids (*Maylandia zebra*): Preliminary Findings

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**Abstract**

Anthropogenic noise is globally recognized as a stressor for animals. However, despite evidence of detrimental effects of noise exposure on fish, knowledge about chronic effects on critical early life stages is still scarce. Using a split-brood design to exclude the genetic effect, African cichlids (*Maylandia zebra*) from the same brood were exposed to boat noises (~120 dB) and to control condition (~100 dB) for 12 weeks, starting when mouth-brooding females released their young. Treatment fish were exposed to motorboat noises from 9 a.m. to 6 p.m. to mimic a typical daily boating activity. Larvae total length and foraging activity were measured in weeks 1 and 12. The study did not find a significant effect of chronic boat noise exposure on these variables. However, whether noise has any impact within the studied period or after week 12 is not clear. Future work will focus on analyzing the data weekly over the 12 weeks exposure period to further substantiate the current results. Addressing the effect of anthropogenic noise on early life stages may have implications for our understanding of the effect of this pervasive stressor on aquatic organisms.

**Keywords**

Anthropogenic noise · Long-term noise · Cichlidae · Early–life stage · Growth · Foraging

## Introduction

Anthropogenic underwater noise from commercial shipping, oil and gas exploration, naval operations, fishing, and construction work has been shown to have increased low-frequency ambient noise at an average rate of 2.5–3 dB/decade since the 1960s (McDonald et al. 2006; Miksis-Olds and Nichols 2016). The higher concentration of human activities in some areas increases the noise above the ambient level not only locally but also away from the source. As many anthropogenic activities generate low-frequency sound, it propagates over long range, making it a particularly pervasive pollutant (Miksis-Olds and Nichols 2016). Underwater sounds carry important information such as cues from the surrounding environment or from other animals. Aquatic animals use these acoustic information for communication, navigation,

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mating, foraging, agonistic displays, territorial defense, and reproduction (Kasumyan 2009). However, the hearing frequency range of most marine animals overlaps with anthropogenic noise, such as boat noise (Erbe et al. 2014; Merchant et al. 2020; Nikolopoulos et al. 2016). Underwater anthropogenic noise negatively affects aquatic animals, for example, by masking important acoustic information (Clark 2009; Clark et al. 2009; Codarin et al. 2009; McCauley et al. 2003) or impairing hearing by causing permanent and/or temporary hearing threshold shifts (Kastak et al. 2008; McCauley et al. 2003) or cause physiological changes including stress effects and changes in behavior (Popper and Hawkins 2019).

There are few studies focused on the effects of chronic noise during fish development (Faria et al. 2022). Early life experiences have been shown to have impacts on adult behavior, and the effects could be complex and long lasting (Carlson 2017). However, as increasing noise made by humans alters the underwater soundscape, it is expected that it will affect the early life stages of aquatic organisms, which are generally regarded as the most sensitive period in the whole life span. Fish during early life stages suffer high mortality even under normal environmental fluctuations and challenges (Catalán et al. 2020) and increased mortality could have profound consequences on populations (Slabbekoorn 2019). The limited studies on the impact of anthropogenic noise during the developmental period in aquatic animals point to species-specific effects. For example, noise was shown to decrease growth rate and increase overall stress responses in Lusitanian toadfish embryos (Faria et al. 2022); increase the heart rate of damselfish embryos (Fakan and McCormick 2019; Jain-Schlaepfer et al. 2018); increase heart rate, yolk sac use, and cortisol levels of the zebrafish larvae (Lara and Vasconcelos 2021); reduce feeding success in winter flounder larvae (*Pseudopleuronectes americanus*) (Gendron et al. 2020); and reduce the settlement of coral reef fish larvae (Simpson et al. 2016). Noise effects are also dependent on exposure duration. It is notable that most studies on anthropogenic noise effects are focused on the acute or relatively short-term exposure, likely due to long-term experiments being harder to carry out. There is evidence that long-term anthropogenic noise negatively affects animals (Barber et al. 2010; Crino et al. 2013; Kusku 2020), but knowledge about the chronic effects of noise on critical early life stages (larvae and/or juvenile period) is still scarce.

*Maylandia zebra* is an African cichlid living on the sediment-free rocky coast in Lake Malawi. This species is able to produce sounds during social interactions, such as territory defense (Bertucci et al. 2010; Chabrolles et al. 2017) or mating (Simões et al. 2008). This is an aggressive species with well-established dominance hierarchies. Maternal mouth-brooding is also an important feature of the species, lasting for 3 weeks before the female releases the larvae. Due to increased human activities and development of fishing and tourism industries in lake Malawi (Allison and Mvula 2002; Makochekanwa 2013), man-made noise has become a pollutant of concern. As *M. zebra* is an established study model in behavior and bioacoustics (Amorim et al. 2019; Bertucci et al. 2010; Chabrolles et al. 2017; Mellor et al. 2011), it provides an excellent opportunity to assess the effects of long-term anthropogenic noise on growth and behavior during the early life stages. This chapter investigates the impacts of chronic anthropogenic noise on growth and foraging during a critical developmental period and hypothesizes that long exposure to boat noise acts as a stressor in the early life stages and has the potential to affect growth and foraging.

## Materials and Methods

### Females and Juveniles Rearing Conditions

*Maylandia zebra* were purchased from Oxyfish (Verlinghem, France) and raised in stock tanks (l × w × h: 120 × 60 × 50 cm) at ENES laboratory (Saint-Etienne, France). Fish in each holding tank was fed daily with commercial cichlid food (JBL NovoRift sticks and Tetra flakes). All tanks were equipped with an external filter (Tetra EX filter, Tetra GmbH, Melle, Germany), aeration, and PVC tubes as shelters. The water temperature was maintained at 25 ± 1 °C with a pH of 8.0 and a 12:12 h light:dark cycle.

Stock tanks were checked daily for females at the beginning of the mouth-brooding period, which were then isolated individually. The mouth-brooding period lasted for 3 weeks after which the mouth of the female was gently opened by using round-head tweezers to release all larvae. These were transferred immediately into experimental aquaria (L × l × h: 30 × 20 × 20 cm). The aquaria were equipped with PVC tubes as shelters, sand as substrate (Aquasand Nature, Zolux déco), and aeration. Each aquarium was equipped with an underwater loudspeaker developed by Fonseca and Maia Alves (2012) and placed in the middle of the aquarium (see below). Experimental fish were fed daily with commercial food (JBL NovoMalawi sticks for Malawi cichlids, JBL NovoRift, and cichlid flakes, Tetra).

### Noise Playback

Motorboat sounds were originally recorded by V. Médoc in the Grangent lake (45°45′07.54″N, 4°25′56.47″E, Loire, France) at 1-m depth (hydrophone Aquarian Audio Products H2a-XLR, AFAB Enterprises, Anacortes, WA, USA; sensitivity: −180 dB re. 1 V $\mu Pa^{-1}$, frequency response within ±4 dB in the range 20 Hz–4.5 kHz, connected to a ZOOM H4 Handy recorder). Twenty-five original boat sounds were excerpted and linearly faded at both ends using Audacity 3.1.3 (http://audacity.sourceforge.net) to make them emerge from silence, before being used in the playbacks.

The aquaria background noise was around 100 dB re. 1 μPa and was used as the control condition in the experiment. To keep this background noise level, no underwater filters were installed in the experimental and control aquaria. Instead, one-third of the water was changed every 2 days.

Boat noise, played back through a DENON PMA-100M amplifier connected to a TASCAM US-366 preamplifier and controlled by a laptop running Audacity, was delivered through the abovementioned underwater loudspeaker. The loudspeakers were placed in the middle of each experimental aquarium, including the control aquaria but here without connecting them to the laptop. The excerpts of boat noise playbacks were adjusted to 120 dB re. 1 μPa before broadcasting to the fish. All modified boat noise was played back and then recorded by a pre-calibrated chain composed of a H2A-HLR hydrophone placed 3 cm away from the speaker and

connected to a ZOOM H4 recorder. Sound pressure levels of playbacks were calculated by integrating the average power spectra of the recorded sound files.

Audio tracks used in the playbacks contained different boat sounds randomly positioned along nine 1-h audio files to mimic the variability in the daily activity of small leisure boats. In the remaining time of the audio tracks, there was no noise. The number of boats varied from 6 to 15 boats/track. The boat noise was broadcast to the boat noise fish treatment from 9 a.m. to 6 p.m., every day during 12 weeks.

## Experimental Design

Five clutches from five different pregnant females were used for the experiment ($n$ = 12–21 larvae). Following a split-brood design, larvae from each clutch were divided randomly into two groups and transferred directly into the aquarium assigned to either boat noise or control treatment, where they stayed for 12 weeks.

Once a week during the experimental period, a 10-min video of daytime foraging activity was taken by using a video camera (HD 1080p C920; Logitech International SA, Lausanne, Switzerland) positioned in front of the aquarium. The frame rate of the camera was set at 120 fps (frames per second). Fish were fed with commercial food pellets every day. During filming of the foraging activity, fish were fed to satiation. All the videos were analyzed with Behavioral Observation Research Interactive Software (BORIS; Friard and Gamba 2016) to extract the foraging behavior events. Each time one fish opened its mouth to catch the food pellet was counted as a proxy for eating. The total number of foraging events was divided by the number of fish in the aquarium to get the average foraging activity per fish. Behaviors were counted at the group level (the sum of the numbers of the behaviors of all fish in one aquarium) with the software, because the larvae were too small to be labeled or identified. Every week, the total length (TL) of each individual was measured. A transparent box with 0.5 cm grid paper on the bottom was prepared for measuring the total length. To disturb the fish as little as possible, all fish in one aquarium were taken out simultaneously with a net that spanned the width of the aquarium, and fish were then placed together in the measuring box. The water in the box was changed between measurements from different aquaria. A photo from the top of the box was taken to later measure the fish length using ImageJ (Rueden et al. 2017). This was possible since the water depth was about the body height of the fish. After taking the photo, the fish were immediately placed back in the aquarium. From the total length of each fish, the average TL of each group under the two different conditions was calculated at weeks 1 and 12, i.e., the start and the end of the experiment.

## Statistical Analyses

Statistical analyses were conducted using R (version 4.2.1). According to the Shapiro–Wilk test, only the foraging per fish at week 12 was non-normal. Therefore,

a Wilcoxon signed-rank test was used to test treatment effect on foraging at week 12, and a t-test was used on total length (weeks 1 and 12) and foraging at week 1.

## Ethics Notes

The research carried out comply with relevant guidelines and regulations, including French national guidelines, permits, and regulations regarding animal care and experimental use (approval no. D42–218-0901, ENES lab agreement, Direction Départementale de la Protection des Populations, Préfecture du Rhône).

# Results

## Total Length

At the beginning of the experiment, the average total length (TL) of the larvae from the boat noise condition was 1.39 (±SD, range: ±0.07, 1.33–1.50 cm), and the average TL of the larvae for the control condition was 1.39 (±0.07, 1.32–1.52 cm). There was no difference in TL at the beginning of the experiment ($p$ value = 0.97) (Fig. 1a). After 12 weeks of exposure to boat noise, the average total length (TL) of

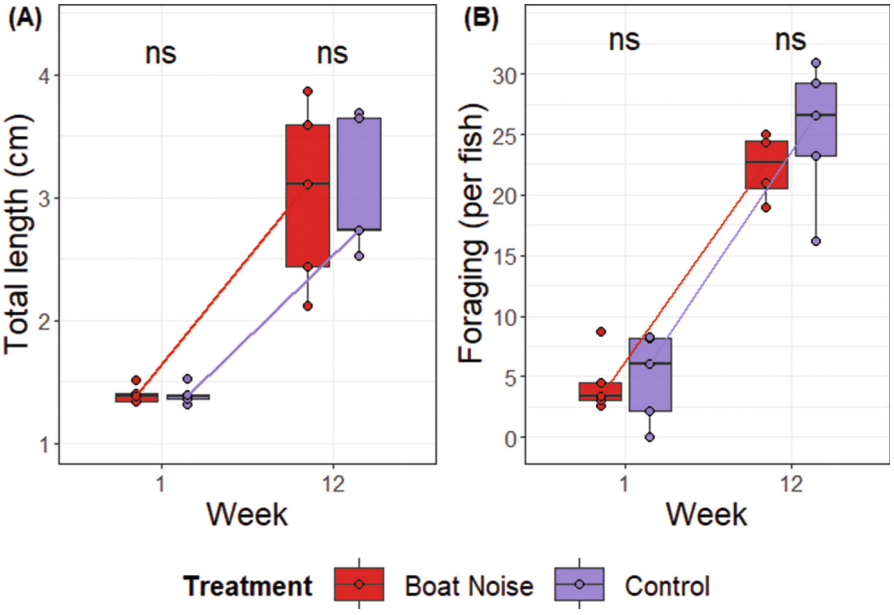



**Fig. 1** Effects of boat noise on total length (**a**) and foraging (per fish) (**b**) at weeks 1 and 12 of the experiment. Red represents fish exposed to boat noise condition, and blue represents fish exposed to control condition. Dots represent five different groups, and the boxes represent the 25th–75th percentiles

the fish under boat noise condition was 3.02 (±0.74, 2.11–3.86 cm), and the average TL of the control fish was 3.06 (±0.56, 2.52–3.69 cm). Total length of fish after 12 weeks of exposure to boat noise did not differ from the fish under control condition ($p$ value = 0.92, Fig. 1a).

### Foraging

Fish foraging activity was evaluated by the number of eating events per fish during the 10 min of filming. In the first week, the foraging activity per fish under the boat noise condition was 4.94 (±3.72, 0–8.33), while for the control condition, it was 4.44 (±2.47, 2.63–8.70). There was no significant difference in foraging per fish at the beginning of the experiment ($p$ value = 0.81, Fig. 1b). After 12 weeks of exposure, there were no foraging significant differences either ($p$ value = 0.84) (Fig. 1b) between fish under boat noise condition (25.21 ± 5.84, 16.17–30.86) and control condition (28.83 ± 14.74, 19–54.83).

## Discussion

### Growth

No significant effects were found after 12 weeks of boat noise exposure on both the total length and the foraging behavior of *M. zebra*. The lack of effects on growth is consistent with the findings for another cichlid (*Neolamprologus pulcher*) (Bruintjes and Radford 2014) in which 4 weeks of noise exposure did not affect hatching success or post-hatching larvae growth and survival. A study on rainbow trout (*Oncorhynchus mykiss*) juveniles reared under noise made by the intensive aquaculture systems for 5 months (Davidson et al. 2009) also suggested no impact in the growth and survival of the noise-exposed fish when compared to control. However, some other studies have shown that long-term noise exposure has detrimental effects on growth. For example, the total length of Lusitanian toadfish larvae (*Halobatrachus didactylus*) after 2 weeks of boat noise exposure was smaller than in the control condition (Faria et al. 2022). The larvae of freshwater sheepshead minnow (*Cyprinodon variegatus*) and killifish (*Fundulus similis*) exposed to high noise for 12 days were also smaller than those under quiet conditions (Banner and Hyatt 1973). Studies on larvae of Atlantic cod (*Gadus morhua*) showed that even larvae 2 days after hatching were smaller under noise than under control conditions, although there was growth compensation at later stages (Nedelec et al. 2015). Different effects on the total length under noise exposure might be species-specific, namely due to interspecific differences on fish hearing and the presence of a swim bladder or other resonant structures, which can cause different vulnerability to anthropogenic noise (Fay and Popper 2012; Fay et al. 2008). Differences in experimental procedures, including whether they are field- or lab-based (Popper and Hawkins 2019), the sound type (Bruintjes and Radford 2014; Kusku et al. 2020), boat type (Jain-Schlaepfer et al. 2018; McCormick et al. 2019), amplitude levels

(Lara and Vasconcelos 2021), and exposure duration (Popper and Hawkins 2019) could all have an impact on the fish's response to noise.

## Foraging

There was also no evidence that foraging activity was affected by chronic noise exposure during early life stages, when comparing that behavior between treatments on weeks 1 and 12. Although other studies have shown that foraging can be affected by additional man-made noise (Bracciali et al. 2012; Purser and Radford 2011; Voellmy et al. 2014), habituation, sensitization, and tolerance can occur in most animals after repeated and/or chronic noise exposure (Bejder et al. 2009). This suggests that young *M. zebra* fish may have habituated or gained tolerance to the added noise. However, as habituation should be the result of a longitudinal process, the comparison at only one time point (week 12) cannot provide strong evidence for distinguishing between habituation and tolerance (Bejder et al. 2009). Magnhagen et al. (2017) examined the effect of noise on foraging behavior in both roach and the Eurasian perch in the field. These authors found that perch can habituate to noise exposure as their feeding attempts first decreased and then increased gradually over the 5 days of the experiment (Magnhagen et al. 2017). However, the roach, which has higher hearing sensitivity, presented fewer feeding attempts than the perch and did not habituate to the noise during the experimental period (Magnhagen et al. 2017). The studies on perch and roach suggest that fish are able to habituate to anthropogenic noise or that the time needed for habituation has interspecific differences.

As hearing sensitivity in larvae and juveniles has not been studied in *M. zebra*, it is possible that the 120 dB boat noise level fell below hearing thresholds, thus not causing an effect. Noise is known to affect foraging behavior by directly reducing appetite (Charmandari et al. 2005) or indirectly through a reduction in activity and locomotion (Mendl 1999). It is also possible that it alters cognitive processes involved in food detection or distracts the attention of the fish (Chan and Blumstein 2011; Mendl 1999).

Only two time points were considered in this chapter, at the start and the end of the experiment, preventing from acknowledging possible effects of noise over the full 12 weeks period. Future analyses should focus on more time points within the long-term noise exposure. This will give a better insight regarding the effects of noise along the development of fish and on whether these effects are reversible or not.

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# Physical Effects from Underwater Explosions on Two Fish Species

69

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**Abstract**

There is broad concern about the potential effects of anthropogenic sound on aquatic animals including fishes. While there have been a number of studies focusing on intense impulsive sounds, most, including all recent studies, have focused on pile driving and seismic air guns. Recently, however, several studies have examined the effects of exposure to explosives, using a dose-response approach, with animals at different distances from the source. Studies were conducted offshore of San Diego, CA, USA, in 2018 with Pacific sardines (*Sardinops sagax*) and in 2019 with Pacific mackerel (*Scomber japonicus*). Primary effects were damage to the swim bladder and kidney. These two injuries displayed a monotonic dependence with acoustic dose and range from the explosion for Pacific mackerel. In contrast, Pacific sardine injuries showed a non-monotonic dependence, with increased rates of injury both near the detonation and at 100–150 m that may have been due to rapid decompression of the swim bladder caused by near-simultaneous arrival of bottom and surface reflected waves. Overall, differences in injury rates between these two species may have been related to anatomy and/or to differences in sound propagation conditions between years which could have influenced injuries within a certain range of distances from the detonation.

**Keywords**

Swim bladder · Kidney · Rupture

## Introduction and Background

Fishes can be injured or killed by underwater explosions (Yelverton et al. 1975; Dahl et al. 2020; Jenkins et al. 2022). Previous studies have related injury in fishes to the acoustic dosage from underwater explosions (Popper et al. 2014). Many of these studies, however, had few to no controls, restricted acoustic environments, inadequate sample sizes, and/or insufficient examination procedures (e.g., Hubbs and Rechnitzer 1952; Yelverton et al. 1975). Despite these shortcomings, this previous research, combined with more recent investigations (e.g., Settle et al. 2002; Govoni et al. 2003; Govoni et al. 2008), provides useful insight into the susceptibility of fishes to injury from underwater explosions. These studies demonstrated that swim bladder rupture occurs when it compresses and then rapidly over-expands, which can also result in damage to nearby organs and tissues. Moreover, swim bladder over-expansion can be intensified by the arrival of a negative pressure signal reflected from the water's surface creating a rapid pressure drop due to (Wiley et al. 1981) or convergence of surface and bottom reflected waves (Dahl et al. 2020).

The work discussed here examined nonauditory injuries on adult caged fishes located at different distances from an explosion using adequate sample sizes, detailed examination procedures, and ample controls. These studies were done in

open water, allowing relatively farther exposure geometries than possible in enclosed test locations.

The anatomical differences between different fish species could affect vulnerability to underwater explosions; these studies examined nonauditory injury to two species: Pacific sardine (*Sardinops sagax*) in 2018 (Dahl et al. 2020) and Pacific mackerel (*Scomber japonicus*) in 2019 (Jenkins et al. 2022). Additional exposures to Pacific mackerel were conducted in 2021. This chapter provides an overview of the experimental methods and findings for the first 2 years of this 3-year field effort. For more detailed results beyond what appears in this chapter, please refer to Dahl et al. (2020) and Jenkins et al. (2022). In addition, the ears of Pacific mackerel used in the study were also examined (Smith et al. 2022).

## Approach

This experiment was conducted on 4 trial days in October 2018 using Pacific sardines and 5 trial days in September 2019 using Pacific mackerel, with a single explosion each day. Pacific sardines had a mean standard length of 166 mm (SD 9 mm) and mean weight of 62 g (SD 11 g). Pacific mackerel had a mean standard length of 212 mm (SD 17 mm) and mean weight of 106.8 g (SD 27.9 g).

The explosive in 2018 consisted of 3.4 kg of C4 explosive, corresponding to 4.7 kg of trinitrotoluene (TNT), whereas 4.5 kg of C-4 (equivalent to 6.2 kg of TNT) was used in the 2019 trials. Detonations occurred at a depth of 10–11 m, near the middle of the water column. The detonations were conducted by Navy explosive ordnance disposal (EOD) personnel in accordance with the US Navy's Hawaii-SOCAL Environmental Impact Statement (www.hstteis.com) and associated permits.

All trials occurred approximately 5 km offshore from San Diego, California (USA), within the US Navy's Silver Strand Training Complex. All experimental cages were located along a transect with average water depth of 19.5 m (Fig. 1). Each

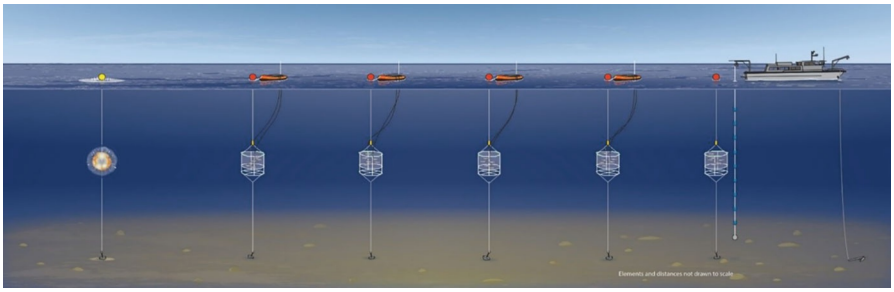


**Fig. 1** Basic layout (not to scale) for each explosive trial. Five cages are deployed midwater, with the explosion to the far left. Tourmaline blast sensors on the four cages closest to the explosion are cabled to instrument packs in the orange rafts. (Figure from Dahl et al. (2020))

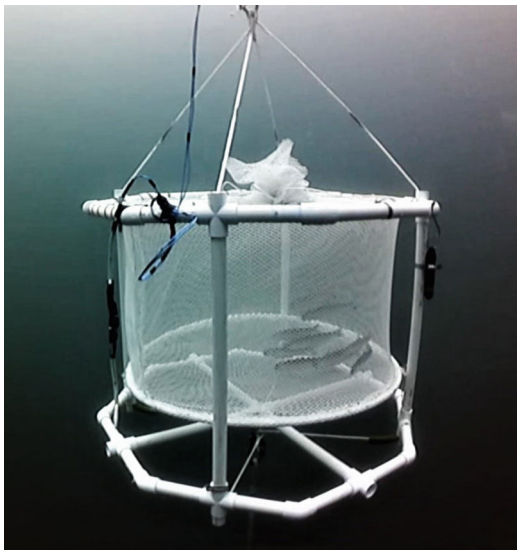


**Fig. 2** Photo of experimental cage captured by a remotely operated vehicle (ROV) just before the explosion. Details of the cage, instrumentation, and its suspension are discussed in Dahl et al. (2020) and Jenkins et al. (2022). On the right side of the cage is depth sensor with the pair of blast probes on the left. Pacific sardines are schooling in the cage. (Figure from Dahl et al. (2020))

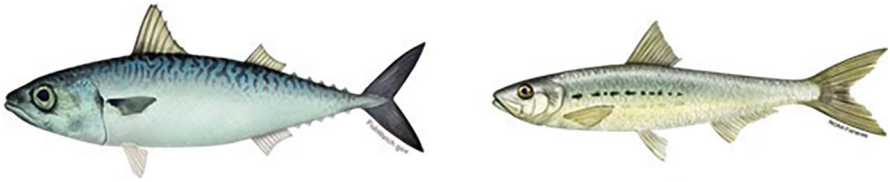


**Fig. 3** Illustrations of Pacific mackerel (left) and Pacific sardine (right) (https://www.fisheries.noaa.gov/species/pacific-mackerel and https://www.fisheries.noaa.gov/species/pacific-sardine)

cage (Fig. 2) was instrumented with either tourmaline blast sensors or autonomous hydrophone systems, depending on the expected received level.

Pacific sardine experimental cages ($n = 20$) were placed at distances ranging from 18 to 246 m, and Pacific mackerel cages ($n = 25$) were deployed at distances ranging from 21 to 807 m from the point of detonation. Control cages were placed at the same depth as experimental cages (mid-water) but retrieved prior to the detonation. Once all experimental fish cages were deployed, the EOD dive team detonated the explosives. Beginning approximately 15 min post-detonation, the test cages were recovered and all live fish were euthanized (Fig. 3).

Examination and dissection procedures were adapted from past studies (e.g., Casper et al. 2012; Halvorsen et al. 2012). This included measuring weight and length, external examination, and dissection of the abdominal and pericardial cavities to examine the organs and tissues. The fish's treatment was not known to the investigators performing examinations.

All procedures were approved by the Naval Information Warfare Center Pacific Institutional Animal Care and Use Committee (IACUC, protocol #131-2018), and

the Navy Bureau of Medicine and Surgery. The study followed all relevant US Department of Defense guidelines for the care and use of laboratory animals.

## Overall Findings

### External Effects

No differences in number or type of external injuries were seen between control and experimental animals. It is likely external injuries, such as fin tears or redness at the fin base, were the result of handling and transport prior to use. External injuries were not considered further.

### Internal Effects

Results showed five injuries that were significantly different between treatment and control Pacific sardines in 2018, including burst capillaries, fat hematoma, reproductive blood vessel rupture, swim bladder rupture, and kidney rupture (chi-square test $<0.05$ for pooled treatments versus pooled controls). For Pacific mackerel in 2019, kidney rupture, swim bladder rupture, blood pooling in swim bladder, swim bladder bruising, and blood pooling in abdominal cavity showed a significant difference between controls and treatments (Fisher's exact test of $p < 0.05$ for four or more treatment cages compared to pooled controls [see Jenkins et al. 2022]). In Pacific sardines, the reproductive blood vessel was an obvious blood vessel on the ventral surface of the swim bladder; similar reproductive blood vessel morphology was not observed in the Pacific mackerel. Eight of 209 experimental Pacific sardines were retrieved dead, and 24 of 253 Pacific mackerel were retrieved dead. As expected, most mortalities occurred closer to the detonation in cages with higher rates of injury.

Figure 4 provides the acoustic results from each test cage location for peak sound pressure level, sound exposure level (SEL), and pressure impulse. Received levels monotonically decreased with range during both experiments (2018 and 2019) as would be expected (Urick 1983). Although the size of the explosive varied between years, received levels as a function of range were similar.

Consistent with past findings (Gaspin 1975; Yelverton et al. 1975; Goertner et al. 1994), swim bladder and kidney rupture were the most prevalent injuries overall across the two study species and highly correlated with one another within years (Fig. 5). The kidney lies directly dorsal to the swim bladder in most teleost species. Thus, the proximity to the rapidly expanding and contracting swim bladder is the likely cause of kidney rupture. Similar conclusions were made for fish exposed to impulsive pile driving (Halvorsen et al. 2011, 2012).

Results show that both swim bladder and kidney rupture rates were high in Pacific sardines exposed within approximately 50 m of the source, as might be predicted; however, rates were much higher within this range for Pacific mackerel. Although a

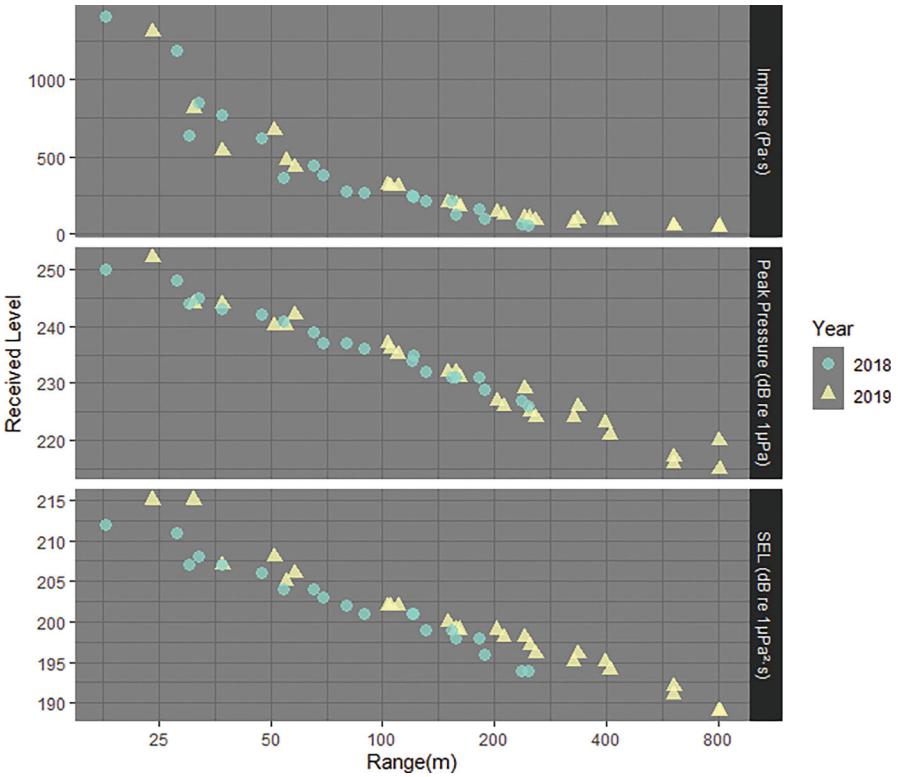



**Fig. 4** Received levels at each cage for 2018 (Pacific sardines) and 2019 (Pacific mackerel). Note the general monotonic transmission loss over the ranges sampled

larger explosive charge weight was used in 2019 than 2018, the resulting acoustic measurements at the cages were similar (Fig. 4). This suggests that the Pacific mackerel were more susceptible to injury than the Pacific sardines at the same acoustic dose.

Some of the observed differences might be attributed to differences in anatomy. Specifically, Pacific sardines are laterally flatter than the fusiform mackerel, while Pacific mackerel have a larger swim bladder (relative to body size) with a looser connection to the abdominal lining as compared to the sardines. The larger volume of air in the Pacific mackerel swim bladder may have contributed to more excitement of the air space by the impinging acoustic wave and the looser connection allowed for more motion causing more internal damage. Additionally, Pacific sardines used in 2018 had a large amount of abdominal fat not present in Pacific mackerel tested in 2019, which may have cushioned the internal organs and reduced incidents of injury in sardines.

Results for Pacific sardines also suggested a second range of severe injury to the swim bladder and kidney from about 100–150 m, while beyond that region, swim bladder and kidney rupture rates decreased to zero at the farthest ranges used. This effect was not observed in the same manner for the Pacific mackerel, where injuries

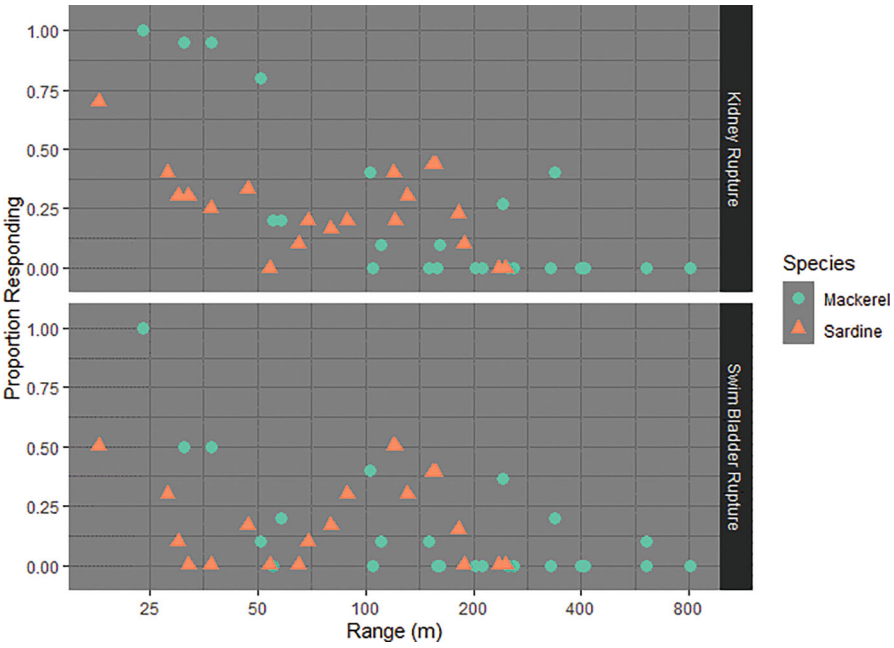



**Fig. 5** For Pacific sardines, rates of both kidney and swim bladder rupture initially fall off monotonically with increasing range; however, beyond about 50 m, the rate of injury again increases, attaining a maximum between 100 and 150 m. This was not the case with Pacific mackerel where injuries decreased across the entire range

generally decreased with increasing range, although there were instances of kidney rupture swim bladder rupture significantly higher than controls out to 333 m.

In previous research efforts on explosive and other impulsive effects on fishes, peak pressure, pressure impulse (Yelverton et al. 1991; Govoni et al. 2003, 2008), and SEL (e.g., Casper et al. 2012; Halvorsen et al. 2012) were dosage metrics used to predict injuries. It was anticipated that acoustic dosage would produce similar trends in this research. Pressure impulse, sound exposure level, and peak pressure were all generally good predictors of swim bladder and kidney rupture for both species, with injuries generally increasing with increasing acoustic dosage (Jenkins et al. 2022). However, Dahl et al. (2020) showed an increase in kidney and swim bladder ruptures at ranges between approximately 100 and 150 m for Pacific sardines which were correlated poorly with peak sound pressure level, SEL, and pressure impulse. Pacific mackerel did not experience a similar increase within these ranges.

Dahl et al. (2020) proposed a plausible explanation for the increased injuries in Pacific sardines within this range (100–150 m) observed in 2018. A rapid pressure drop caused by the near-simultaneous arrivals from the sea surface reflected and bottom bounce paths may have amplified the amount of swim bladder over-expansion per a process proposed by Goertner (1978) and Wiley et al. (1981). Sound speed profiles showed more mixing of near-surface water causing disturbance of the thermocline in October 2018 as opposed to September 2019 when

stratification of the water column was evident. This stratification may have disrupted the convergence of surface reflected and bottom bounce arrivals observed in 2018, thereby reducing the injurious effects to Pacific mackerel in 2019 from this phenomenon.

## Conclusions

This and past studies show that acoustic dosage metrics such as peak SPL, SEL, and pressure impulse are largely good predictors of injury in fishes exposed to underwater explosives (e.g., Yelverton et al. 1975; Wiley et al. 1981; Govoni et al. 2003). However, the 2018 study with Pacific sardines (Dahl et al. 2020) and in past studies (Yelverton et al. 1975; Wiley et al. 1981), potential waveguide effects were observed that increased injuries within certain ranges from the source that cannot be attributed to these acoustic doses alone. In the studies discussed here, exposures were conducted using similar methods to purposely limit factors, making results easier to understand; however, the timing of multipath arrivals at any given distance (i.e., fish cage) from the explosion was similar within a given year. Since the sound arrival waveform can vary with water depth, fish depth, explosive charge depth, and sound velocity profile, it is vital that future research varies these to better understand how these waveguide characteristics affect injury in fishes. Other significant variables to study in the future are different species/morphologies, exposure depth, explosion depth, bottom depth, and types of bottom. A follow-up study in 2021 using Pacific mackerel with a similar study design investigated the onset of recoverable injury, short-term survival, and the effect of fish depth on nonauditory injury. Analysis of the 2021 results is currently ongoing, and publications are in progress.

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# Section X

# Noise Effects: Marine Mammals

# Temporary Threshold Shift in Porpoise Hearing: Effect of Experimental Protocol

70

Jakob Tougaard, Kristian Beedholm, and Peter T. Madsen

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**Abstract**

Two recent studies of auditory temporary threshold shift (TTS) in a harbor porpoise at 25 and 28 kHz are compared to previously published results revealing a very large difference in sensitivity to TTS correlating with experimental paradigm. For fatiguing noise above 10 kHz, the TTS-onset thresholds measured with behavioral (psychophysical) methods following noise exposure to free-swimming porpoises are 40–60 dB higher than TTS-onset thresholds measured with auditory brain stem recordings on porpoises exposed while stationary. The uncertainty about sensitivity of porpoises to noise exposure above 10 kHz have very large implications for regulation of underwater noise and call for immediate further experiments. Such experiments should disentangle and quantify the confounding effects of exposure (stationary vs. free-swimming) and threshold measurements (behavioral vs. brain stem recordings), and the results are required

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to establish the VHF weighting function above 10 kHz, applicable to porpoises in assessment of risk of impact from high-frequency noise sources.

**Keywords**

TTS · Impact · Noise · Pile driving · Frequency weighting · Hearing

## Introduction

A recent review (Tougaard et al. 2022) showed good correspondence between experimental thresholds for onset of temporary threshold shift (TTS) in porpoises (Phocoenidae) and predictions from frequency-weighted sound exposure levels (SEL; from Southall et al. 2019) for fatiguing sounds below 10 kHz. For frequencies above 10 kHz, however, there were significant differences between predictions and experimental data. Since the publication of the review, two additional studies of TTS on harbor porpoises (*Phocoena phocoena*) induced by fatiguing sounds at 25 kHz and 28 kHz, respectively, have become available. These new results make it clear that TTS results for these frequencies fall into two categories: some with very low TTS-onset thresholds and others with very high TTS-onset thresholds. As noise exposure criteria related to injury typically are based on TTS-onset thresholds (used directly or extrapolated to thresholds for onset of permanent threshold shifts, PTS), it has significant implications for regulation of underwater noise, how these discrepancies between different experimental results are handled. The results are reviewed below, and possible explanations for the differences are offered and discussed.

## 25 and 28 kHz Experiments

Both experiments were conducted on the same animal (female harbor porpoise *Freja*) at the Fjord&Belt facility, Denmark, and included measurements of hearing thresholds by auditory brain stem responses on the porpoise exposed to brief fatiguing sounds of varying levels while stationary under water. One experiment used a single pure tone pulse at 25 kHz, approximately 600 ms in duration, as the fatiguing sound (Ruser et al. 2015; Andersson and Nöjd 2016), whereas the other study used a single 17-s sequence of 1024 tone pips, where each pip consisted of five cycles of a 28 kHz sinusoid, shaped with a raised cosine envelope (Schaffeld et al. 2022). TTS-onset thresholds, defined as the SEL required to elicit 6 dB of TTS, were estimated by fitting curves to the individual data points obtained at different SEL, with the method described by Finneran (2015). The TTS-onset threshold for the porpoise was 140.8 dB re 1 $\mu Pa^2s$ for both experiments, which translates into 140.3 dB re 1 $\mu Pa^2s$ and 140.5 dB re 1 $\mu Pa^2s$ VHF weighted (sensu Southall et al. 2019) for the 25 kHz and the 28 kHz fatiguing sounds, respectively. These two data points are shown in Fig. 1, on top of the data for very high-frequency (VHF) cetaceans plotted in the original Fig. 3 of Tougaard et al. (2022). The inclusion of the two additional TTS-onset thresholds at 25 and 28 kHz highlights a clustering of

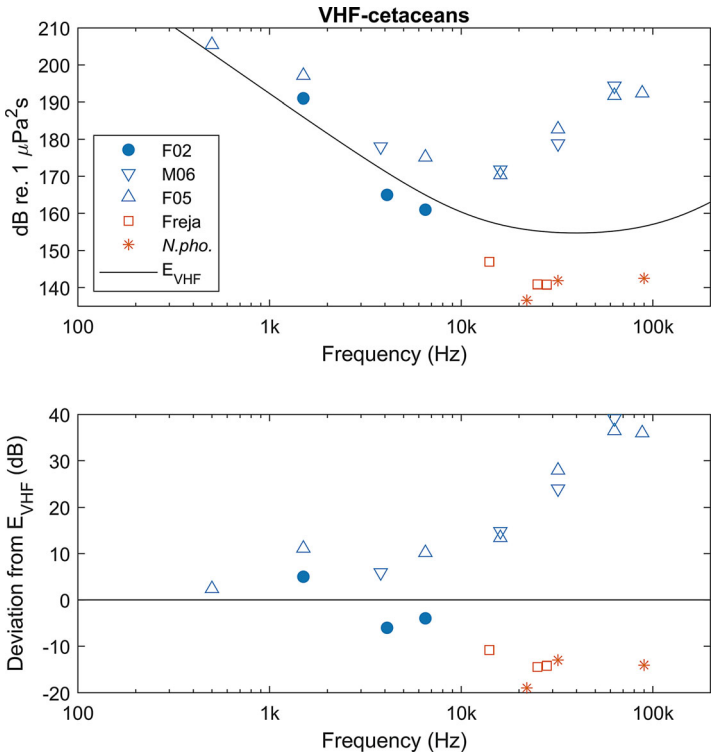



**Fig. 1** TTS-onset thresholds for porpoises exposed to non-impulsive sounds of different frequency. Color indicates experimental paradigm. Blue, exposure with fatiguing noise to free-swimming animals, thresholds measured by behavioral audiometry; orange, exposure to stationary animals, thresholds measured by auditory brain stem responses. Open symbols represent data not included in the most recent guidance regarding auditory weighting functions (Southall et al. 2019). Solid line is the VHF auditory exposure function of Southall et al. (2019)

the TTS-onset thresholds above 10 kHz in two groups. The first group consists of the data points on harbor porpoise *Freja* with fatiguing noise (pure tones up to 0.5 s duration) at 14, 25, and 28 kHz (Ruser et al. 2015; Schaffeld et al. 2019, 2022) together with the data points with fatiguing noise (pure tones up to 30 min long) at 22, 32, and 90 kHz on finless porpoises (*Neophocaena phocaenoides*, Popov et al. 2011). The second group is formed by the data points from harbor porpoises *M06* and *F05* with continuous fatiguing noise (1/6 octave band noise) at 16, 32, 63, and 88 kHz for 6 h (Kastelein et al. 2019a, b; 2020a, c).

## Discussion

Three data points were included by Finneran (2015) in his derivation of the VHF auditory weighting function, indicated by filled circles in Fig. 1. Although the measurements on *Neophocaena* (Popov et al. 2011) were available, they were not

included because TTS-onset thresholds could not be derived with the methodology otherwise used (interpolation based on data points on both sides of the level required to elicit 6 dB of TTS, the criterion used for TTS onset). Furthermore, the *Neophocaena* measurements were obtained by auditory brain stem recordings (ABRs), which are known to result in lower TTS-onset thresholds than behavioral thresholds (Finneran et al. 2007; Finneran 2015). However, three out of four data sets in total in Popov et al. (2011) provided good fits to the equation used for interpolation by Finneran (2015), and TTS-onset thresholds could be extrapolated (see details in Tougaard et al. 2022). These data points are included in Fig. 1 as stars.

When all currently available data on TTS-onset thresholds are included, as in Fig. 1, two main conclusions become evident. First, new data below 10 kHz (Kastelein et al. 2017; Kastelein et al. 2020b; Kastelein et al. 2021) are in good correspondence with the exposure function derived by Finneran (2015). Second, new data above 10 kHz cluster in two groups: one group with very high TTS-onset thresholds, 15 to nearly 40 dB above the exposure function; and a second group, 10–20 dB below the exposure function. The difference of 40–60 dB between the two groups is not trivial, as it amounts to a 4–6 orders of magnitude difference in the acoustic energy at TTS-onset threshold. A difference of 40 dB in sound exposure level (SEL) amounts to the difference in SEL of an exposure of 1-s pulse and 3 h, both with the same sound pressure level.

For several reasons it is important to understand what underlies the dichotomy of the thresholds above 10 kHz. The difference between the two groups of data points above 10 kHz is unlikely to be explained by differences between experimental animals. The lower-threshold group was obtained from three individuals and the higher-threshold group from two individuals at two different facilities many years apart, with both groups showing very little intragroup variation. This leaves two central differences between the groups as candidates for explaining the difference between groups: differences in the threshold measurement technology and differences in experimental protocol for exposure to the fatiguing noise.

## Threshold Measurements

The lower group of TTS-onset thresholds (Popov et al. 2011; Ruser et al. 2015; Schaffeld et al. 2019; Schaffeld et al. 2022) was measured by auditory brain stem responses, whereas the higher group was measured with behavioral audiometry (Kastelein et al. 2019a, b, 2020a, c). The measured amount of TTS for the same fatiguing noise depends on the way thresholds are measured, with auditory brain stem responses resulting in higher amounts of TTS measured than when measured by psychophysical methods (Finneran et al. 2007). The TTS also manifests itself at lower exposures in ABR thresholds than in behavioral thresholds (Finneran et al. 2007), resulting in lower TTS-onset thresholds. This difference likely lies in the way ABR thresholds are measured. Determination of thresholds from ABR data depends on a very persistent brain stem potential, allowing noise reduction by averaging across a large number of ABRs. A high-amplitude and stable ABR requires stable

recruitment of many nerve fibers but also that the nerves fire consistently, i.e., with stable latency. Recruitment of nerve fibers depends on the threshold of individual hair cells and synapses to the auditory nerve fibers, and changes are therefore likely to parallel changes seen in behavioral thresholds. Latency and especially variance in latency are unlikely to affect behavioral thresholds to any significant degree, as the animal's auditory system is not reliant on averaging across many neurons in the same way as an ABR measurement is. The lower TTS-onset thresholds seen in ABR measurements are therefore likely to be related to the fatiguing noise affecting timing of neural responses at levels where recruitment is not significantly affected. Near-simultaneous measurements of TTS by ABR and behavioral methods by Finneran et al. (2007) showed 20–25 dB more TTS, when measured by ABR for the same fatiguing stimulus (see Fig. 1 in Finneran 2015), which also was manifested as a 20–25 dB difference in TTS-onset thresholds (see Fig. 14 in Finneran et al. 2007). The difference in methodology for measuring TTS can therefore likely explain at least part of the difference between the two groups of results, but even if this can explain 25 dB of the difference, there is still 10–20 dB left unexplained.

## Delivery of Fatiguing Stimulus

The other main difference between the two sets of experiments, confounded with threshold measurement technology, is the way the fatiguing noise is delivered to induce TTS. In the high TTS-onset threshold group, the fatiguing noise was delivered to free-swimming animals over an extended exposure time (6 h), whereas much shorter exposures (up to 17 s for *Phocoena* but up to 20 min for the *Neophocaena* measurements) were made to stationary animals in the low TTS-onset threshold group. Could it be that the noise field of the free-swimming porpoise offered places in the tank with much lower exposure levels than assumed? This would mean that the cumulated sound exposure levels that free-swimming porpoises were exposed to are overestimated, leading to similar overestimation of TTS-onset thresholds.

## Resolving the Confounding Factors

No matter what caused the pronounced difference in TTS-onset thresholds, it must be a priority to resolve and understand this substantial discrepancy, as these data have important implications for management of underwater noise. The VHF auditory weighting function (inverse of the exposure function in Fig. 1) forms the foundation of assessment and regulation of noise exposure to VHF cetaceans in several countries (e.g., National Marine Fisheries Service 2018; Danish Energy Agency 2022). Depending on which data sets are included in establishing exposure limits, the current exposure limit (determined based on the VHF weighting function) could potentially be off with as much as up to 40 dB overestimation (including only the behaviorally determined thresholds) or down to 20 dB underestimation (including only the ABR thresholds).

Several experiments can be considered to test hypotheses about the cause of the observed differences in TTS onset and disentangle the confounding factors. An easy first would be to investigate changes to the spectrum of brain stem responses with TTS onset. Jitter in the timing of neural responses should have a more pronounced effect on the faster (mostly earlier) components than on the slower (mostly later) response peaks. The hypothesis would then be that there should be a low-pass effect on the auditory responses to the test signals, close to or at the level of the fatiguing sounds, where TTS sets in. Second, a partial switch of methods between experimental facilities seems a good start, i.e., combining noise exposures to stationary animals with behaviorally measured TTS and likewise conducting noise exposures with free-swimming animals followed by auditory brain stem recordings. This could be coupled with measurements in the same setup, but with different types of fatiguing noise: short exposures to loud pulses of noise versus long exposures to weaker, continuous noise, both with the same SEL. Furthermore, as no TTS-onset thresholds have been measured by auditory brain stem recordings in porpoises exposed to fatiguing noise below 14 kHz, it must be of importance to do this, to determine whether the difference between experimental protocols extends to lower frequencies as well.

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# Kurtosis Analysis of Down-the-Hole Pile Installation Sounds Based on Marine Mammal Auditory Capabilities

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Shane Guan and Tiffini Brookens

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**Abstract**

Down-the-hole (DTH) pile installation is a relatively new technology that uses percussive drilling to install piles, either by striking both the drill bit that cuts into the bedrock and the metal pile to advance the pile (DTH pile driving) or by striking only the drill bit, with no contact with the pile (DTH pile drilling). Recent studies have shown that DTH pile installation generates complex sounds containing both impulsive and non-impulsive features. In this chapter, kurtosis was used to quantify the impulsiveness of sounds from DTH pile installation for six marine mammal functional hearing groups. Results show that, during DTH pile driving, kurtosis values decreased as the distance to source increased. However, no clear trend was detected for kurtosis versus range during DTH pile drilling. Additionally, kurtosis values based on marine mammal auditory weighting functions generally were greater for species with enhanced low-frequency sensitivity (i.e., low-frequency cetaceans, pinnipeds, and sirenians) than unweighted values, whereas kurtosis values for mid- and high-frequency cetaceans were similar to, or in some cases less than, unweighted

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values. Given that sound exposure criteria are currently lacking for marine mammals exposed to complex sound, research in this area is critically needed.

**Keywords**

Down-the-hole pile installation · Complex sound · Kurtosis · Impulsiveness · Auditory weighting functions · Marine mammal · Functional hearing groups

## Introduction

In-water pile driving generates intense sounds that could adversely affect marine mammals (Branstetter et al. 2018; Kastelein et al. 2018). Conventional pile driving mechanisms include impact pile driving, where an impact hammer strikes the pile into the sediment, and vibratory pile driving, where a vibratory hammer pushes the pile into the sediment through oscillating forces. For both impact and vibratory pile driving, the hammers act directly on the pile heads, and acoustic characteristics and sound-generating mechanisms from these activities are well studied (Reinhall and Dahl 2011; Lippert et al. 2016). Specifically, impact pile driving produces short, impulsive sounds, with hammer strike rates of approximately 1 Hz; vibratory pile driving produces intermittent, non-impulsive sounds, with the vibratory hammer oscillating at a rate of approximately 20 Hz.

More recently, down-the-hole (DTH) pile installation has been increasingly used to install piles for marine and coastal construction projects. Instead of applying force on the pile head, a DTH hammer strikes a drill bit at the base of the pile and cuts into the substrate to advance the pile using a combination of percussive and drilling mechanisms. At the same time, drill cuttings and debris at the seabed are removed by an air-lift exhaust inside the pile (Bruce et al. 2013). Studies on the underlying acoustic characteristics have shown that two different techniques can be used during DTH pile installation: (1) DTH pile driving, in which the hammer strikes the drill bit, as well as the metal pile shoe at the base of the pile, and (2) DTH pile drilling, in which the hammer strikes only the drill bit with no physical contact with the pile or any pile shoe (Guan and Miner 2020; Guan et al. 2022a).

Given that sounds generated from DTH pile installation contain both impulsive components from hammer strikes and non-impulsive components from drilling and debris removal, they would be defined as complex sounds based on human psychoacoustics studies (Ahroon et al. 1993). However, current regulatory guidelines for marine mammal impact assessments involving underwater sound use a binary system to categorize sound types: impulsive and non-impulsive sounds. Separate thresholds are used to estimate noise-induced threshold shifts (NITSs) for marine mammals exposed to these two mutually-exclusive sound types (National Marine Fisheries Service 2018). There is no agreed-upon method for assessing marine mammal NITSs from exposure to complex sound, such as those generated during DTH pile installation.

Conversely, NITSs from exposure to complex sound have been relatively well studied in human psychoacoustic research over the past few decades (e.g., Lei et al. 1994). Instead of applying a binary approach to classify sound source types, researchers use a statistical measure of kurtosis to quantify the impulsiveness of the sounds and to evaluate any auditory effect in humans (Zhao et al. 2010). Under this approach, random noise with a Gaussian distribution has a kurtosis value ($\beta$) of 3. As impulsiveness increases, $\beta$ also increases due to the high variability of the acoustic pressures. Based on NITS research in chinchillas, $\beta > 40$ is considered fully impulsive for airborne sound (Hamernik et al. 2007).

In the past couple of years, researchers have begun to use kurtosis measurements of acoustic pressure to quantify the impulsiveness of anthropogenic underwater sounds (Martin et al. 2020), including unweighted sound from DTH pile installation (Guan et al. 2022b). This chapter further expands the use of kurtosis to quantify the impulsiveness of sounds from two DTH pile installation activities based on the six marine mammal functional hearing groups used by the US regulatory community, including low-frequency (LF) cetaceans, mid-frequency (MF) cetaceans, high-frequency (HF) cetaceans, sirenians (SI), otariids (OW), and phocids (PW) (Finneran 2016).

## Sound Levels and Impulsiveness from DTH Pile Installation

Underwater acoustic recordings were collected during a two-day DTH pile driving event for a dock replacement project off Biorka Island in August 2018 (Guan and Miner 2020) and during a two-day DTH pile drilling event for a cruise ship terminal construction project in Ward Cove near Ketchikan in June 2020 (Guan et al. 2022a). Both locations are in Southeast Alaska. Detailed descriptions of study areas, data collection methods (including details regarding data collection and recording equipment), and data analysis procedures are provided in Guan and Miner (2020) and Guan et al. (2022a). Tables 1 and 2 show the median sound levels from both studies, including single-strike sound exposure level (SEL) or $L_{E,ss}$, root-mean-square sound pressure level (SPL) or $L_{p,rms}$, peak SPL or $L_{p,pk}$, and cumulative SEL or $L_{E,cum}$.

Based on studies by Guan and Miner (2020) and Guan et al. (2022a), sound levels (SPLs and SELs) were generally higher during DTH pile driving of the two 18-inch piles than during DTH pile drilling of the two 33-inch shafts at the same distances (Tables 1 and 2, respectively). A simple explanation is that, during DTH pile driving,

**Table 1** Median sound levels and cumulative SELs for DTH pile driving of two 18-inch piles off Biorka Island, Alaska, from Guan and Miner (2020)

| | Pile F14 | | Pile E14 |
|---|---|---|---|
| Acoustic parameters | 10 m | 200 m | 10 m |
| Median $L_{E,ss}$ (dB re 1 μPa$^2$s) | 147 | 127 | 145 |
| $L_{E,cum}$ (dB re 1 μPa$^2$s) | 192 | 171 | 191 |
| Median $L_{p,rms}$ (dB re 1 μPa) | 162 | 140 | 161 |
| Median $L_{p,pk}$ (dB re 1 μPa) | 173 | 148 | 170 |

**Table 2** Median sound levels and cumulative SELs for DTH pile drilling of two 33-inch shafts in Ward Cove, near Ketchikan, Alaska, from Guan et al. (2022a)

| | Pile 137E | | | Pile 131 W | | |
|---|---|---|---|---|---|---|
| Acoustic parameters | 10 m | 200 m | 1300 m | 10 m | 90 m | 150 m |
| Median $L_{E,ss}$ (dB re 1 μPa$^2$s) | 138 | 122 | 109 | 142 | 127 | 124 |
| $L_{E,cum}$ (dB re 1 μPa$^2$s) | 185 | 170 | 150 | 193 | 180 | 177 |
| Median $L_{p,rms}$ (dB re 1 μPa) | 150 | 134 | 122 | 155 | 140 | 137 |
| Median $L_{p,pk}$ (dB re 1 μPa) | 158 | 143 | 130 | 163 | 149 | 147 |

the hammer struck the metal pile shoe, but during DTH pile drilling, it did not (Guan et al. 2022a).

In determining whether a specific sound type is impulsive or non-impulsive, a method proposed by Harris (1998) and adopted by Southall et al. (2007) compares the difference of the $L_{p,rms}$ of a 1-s sound clip to that of a 35-ms sound clip. If the difference is 3 dB or larger, then that sound is considered impulsive. Otherwise, it is considered non-impulsive. Using this method to examine the two DTH pile installation datasets from Alaska, the average sound-level difference was 3.40 dB for DTH pile driving and 2.05 dB for DTH pile drilling (Guan et al. 2022b). Sounds from DTH pile driving would have been considered impulsive, and those from DTH pile drilling would have been considered non-impulsive.

However, this method is difficult to implement and at times can be subjective, as time-gating of the signals for comparison is a tedious and sensitive process, especially when the signal-to-noise ratio is low. Additionally, sounds from DTH pile installation contain both impulsive (from hammer strikes) and non-impulsive (from drilling and debris pumping) components, and using kurtosis values that represent a continuum would be more appropriate than the current impulsive vs. non-impulsive binary approach to assess the impulsiveness of such sounds.

## Kurtosis Values for Marine Mammal Functional Hearing Groups

To compute kurtosis values, all acoustic recordings obtained during the measurements are divided into 60-s segments. A 60-s window duration was chosen for consistency with other studies that calculated kurtosis values for assessing the impulsiveness of various underwater sounds (e.g., Martin et al. 2020; Guan et al. 2022b). More importantly, Tian et al. (2021) showed that 60 s was an optimal window duration that allows for both efficient computation and high accuracy for noise-induced hearing loss in humans. If one or more hammer strike signals were present within the 60-s segment, the sound clip was classified as "hammer on"; otherwise, it was classified as "hammer off."

In calculating $\beta$ for each of the six marine mammal functional hearing groups, custom MATLAB (MathWorks Inc., Natick, MA, version R2021a) codes were developed using the following equation:

$$\beta = T{\cdot}f_s{\cdot}\sum_{i=1}^{N}[p(i)-\overline{p}]^4 / \left\{\sum_{i=1}^{N}[p(i)-\overline{p}]^2\right\}^2 \quad (1)$$

where $f_s$ is the sampling frequency, $T$ is the window duration, $p(i)$ is the frequency-weighted acoustic pressure for specific marine mammal functional hearing groups, and $\overline{p}$ is the arithmetic mean acoustic pressure for $p(i)$ within $T$.

The acoustic pressure for each marine mammal functional hearing group was determined based on the auditory weighting function developed by Finneran (2016)

$$W(f) = C + 10\,log_{10}\left\{\frac{(f/f_1)^{2a}}{\left[1+(f/f_1)^2\right]^a\left[1+(f/f_2)^2\right]^b}\right\} \quad (2)$$

where $C, f_1, f_2, a$, and $b$ are constants that define the shape of the function for each functional hearing group. To ensure that the original time series information was preserved, computations of frequency-weighted acoustic pressures were conducted using Fourier and inverse Fourier transforms in linear space.

Table 3 presents median $\beta$ for each marine mammal functional hearing group during DTH pile driving when the hammer was on and off, and Table 4 includes similar results during DTH pile drilling. Unweighted $\beta$ also are included for DTH pile driving and DTH pile drilling. For unweighted sounds previously reported by Guan et al. (2022b), median $\beta$ ranged from 10.4 to 30.0 during DTH pile driving and from 4.2 to 5.8 during DTH pile drilling (Tables 3 and 4, respectively).

The results show that for DTH pile driving, $\beta$ values were overall much greater than 3 but less than 40 (Table 3), indicating sounds from DTH pile driving were somewhere between non-impulsive and fully impulsive. Additionally, $\beta$ for DTH pile driving showed clear differences between hammer on and hammer off, with much greater values when the hammer was active (Table 3). However, although $\beta$ for DTH pile drilling were generally greater than 3, they were overall less than those from DTH pile driving, which indicate sounds emitted during DTH pile drilling were less impulsive than those during DTH pile driving. In addition, there was no clear trend in $\beta$ between hammer on and off during DTH pile drilling (Table 4). Furthermore, for DTH pile driving, decreases in $\beta$ were observed as the distance from the pile increased, which indicate that the sound was becoming less impulsive as it propagated away from the source (Table 3). This trend was not observed for DTH pile drilling (Table 4).

When comparing kurtosis values among different marine mammal functional hearing groups, weighted $\beta$ for LF cetacean, SI, OW, and PW were greater than unweighted values at various distances during DTH pile driving (Table 3). This trend likely is due to the fact that most acoustic energy from DTH pile installation was in the low-frequency (<2 kHz) range, which corresponds to the more sensitive hearing range of these functional hearing groups (Guan and Miner 2020). Similar patterns also were observed for LF cetacean, OW, and PW during DTH pile drilling at all distances except 1300 m. These findings could have implications regarding auditory perception and the potential for resulting auditory damage.

**Table 3** Median kurtosis values for each marine mammal functional hearing group during DTH pile driving of two 18-inch piles off Biorka Island, Alaska

| Distance from pile and measurement date | Hammer on/off | LF | MF | HF | SI | OW | PW | UW | *n* |
|---|---|---|---|---|---|---|---|---|---|
| 10 m, 15 August 2018 | On | 30.38 | 28.80 | 29.32 | 30.82 | 31.97 | 29.86 | 30.00 | 86 |
| | Off | 5.09 | 4.48 | 4.57 | 4.28 | 4.33 | 4.78 | 4.28 | 24 |
| 10 m, 13 August 2018 | On | 25.96 | 22.89 | 22.73 | 33.48 | 37.41 | 31.27 | 21.43 | 50 |
| | Off | 4.52 | 4.28 | 4.48 | 3.78 | 3.79 | 3.77 | 4.90 | 19 |
| 200 m, 13 August 2018 | On | 16.76 | 10.31 | 10.33 | 9.77 | 14.24 | 13.91 | 10.40 | 50 |
| | Off | 7.70 | 4.47 | 4.52 | 3.98 | 3.95 | 4.13 | 4.66 | 19 |

Note: *LF* low-frequency cetacean; *MF* mid-frequency cetacean; *HF* high-frequency cetacean; *SI* sirenian; *OW* otariid; *PW* phocid; *UW* unweighted

**Table 4** Median kurtosis values for each marine mammal functional hearing group during DTH pile drilling of two 33-inch shafts in Ward Cove, near Ketchikan, Alaska

| Distance from pile and measurement date | Hammer on/off | LF | MF | HF | SI | OW | PW | UW | *n* |
|---|---|---|---|---|---|---|---|---|---|
| 10 m, 20 June 2020 | On | 5.61 | 4.72 | 4.72 | 4.79 | 4.90 | 4.92 | 4.68 | 186 |
| | Off | 5.83 | 4.46 | 4.46 | 4.53 | 4.87 | 4.77 | 4.46 | 19 |
| 10 m, 19 June 2020 | On | 5.09 | 4.18 | 4.18 | 4.22 | 4.40 | 4.45 | 4.20 | 209 |
| | Off | 6.51 | 5.03 | 4.99 | 5.22 | 5.34 | 5.66 | 5.00 | 43 |
| 90 m, 20 June 2020 | On | 6.43 | 5.81 | 5.82 | 5.80 | 6.05 | 5.96 | 5.83 | 186 |
| | Off | 5.72 | 4.81 | 4.83 | 4.75 | 4.56 | 4.65 | 4.84 | 19 |
| 150 m, 20 June 2020 | On | 8.18 | 5.61 | 5.60 | 5.62 | 6.43 | 6.47 | 5.60 | 186 |
| | Off | 9.02 | 5.99 | 5.86 | 6.40 | 6.34 | 6.73 | 5.81 | 19 |
| 200 m, 19 June 2020 | On | 6.15 | 5.21 | 5.21 | 5.07 | 5.51 | 5.51 | 5.23 | 209 |
| | Off | 6.72 | 5.24 | 5.34 | 4.42 | 4.42 | 4.83 | 5.60 | 43 |
| 1300 m, 19 June 2020 | On | 5.39 | 5.41 | 4.46 | 3.93 | 4.11 | 4.45 | 5.09 | 42 |
| | Off | 3.09 | 3.09 | 3.10 | 3.12 | 3.16 | 3.15 | 3.15 | 13 |

Note: *LF* low-frequency cetacean; *MF* mid-frequency cetacean; *HF* high-frequency cetacean; *SI* sirenian; *OW* otariid; *PW* phocid; *UW* unweighted

## Kurtosis as a Parameter for Marine Mammal Threshold Shifts from Sound Exposure

For marine mammal impact assessments involving sound exposure, the binary scheme that categorizes sounds into two mutually-exclusive types continues to be used: impulsive and non-impulsive sounds. Although research on marine mammal NITS has only focused on animals being exposed to one of these two sound types (Guan and Brookens 2021), animals are rarely exposed to purely impulsive or purely non-impulsive sounds in the wild. In addition to the scenario presented with sounds emitted during DTH pile installation, coastal construction activities often involve simultaneous impact and vibratory pile driving or drilling (e.g., Department of the Navy 2022a), simultaneous rock hammering and drilling (e.g., Department of the Navy 2022b), or impact pile driving from a barge that operates dynamic positioning.

As impulsive sounds propagate through the ocean environment, multipath and scattering render the sound less impulsive. Marine mammals also are unlikely to remain within the immediate vicinity of impulsive sources (i.e., an impact hammer or seismic airgun arrays), in most circumstances. As such, the sounds they are exposed to likely would be complex sounds.

It has been shown that, given the same $L_E$, exposure to complex sounds produces more severe hair cell damage than continuous Gaussian sound in human and terrestrial mammals (Lei et al. 1994; Hamernik et al. 2007). However, whether marine mammals exposed to complex sounds would experience similar effects as terrestrial mammals is unknown, as temporary threshold shift studies conducted thus far have exposed animals to either purely impulsive or purely non-impulsive sounds.

Therefore, there is a critical need to obtain NITS data on marine mammals that are exposed to complex sound at different degrees of impulsiveness. Studies also are needed to review the current binary scheme used to classify sound source types and to establish appropriate standards for classifying sound types based on metrics of impulsiveness from different sound sources.

## Conclusion

Kurtosis values were used to quantify the impulsiveness of sound generated during DTH pile driving and DTH pile drilling at two locations in Southeast Alaska. Using $\beta$ to evaluate impulsiveness, which ranges from 3 for non-impulsive Gaussian sound to greater than 40 for fully impulsive sound, sounds generated during DTH pile installation were in the continuum between the two extremes. $\beta$ values during DTH pile driving were all greater than those during DTH pile drilling. Thus, sounds from DTH pile driving were more impulsive than those from DTH pile drilling. This finding likely is due to the DTH hammer directly striking the metal pile shoe during DTH pile driving, resulting in more impulsiveness to the sounds emitted. Additionally, for DTH pile driving, a decrease in $\beta$ with increasing distance was observed, which is consistent with the common understanding that multipath and scattering reduce the impulsiveness of an impulsive signal (Urick 1982).

Among the six marine mammal functional hearing groups, $\beta$ values tend to be greater for groups that are more sensitive to low-frequency sounds – such as LF cetaceans, SI, and both pinniped groups – than those of unweighted sounds. This trend also is consistent with previous studies that determined the majority of acoustic energies from DTH pile driving and DTH pile drilling were within the low-frequency range (Guan and Miner 2020; Guan et al. 2022a).

Currently, assessments regarding the impacts of sound on marine mammals are based on a binary scheme that categorizes sound source types as either impulsive or non-impulsive. There is no mechanism to evaluate marine mammal auditory impacts from exposure to complex sound, such as exposure from DTH pile installation or from a real-world construction activity that includes a combination of impulsive and non-impulsive sources. Studies on human and terrestrial mammal NITS have shown that under the same amount of $L_{E,cum}$, increased impulsiveness can cause more hair cell damage than exposure to non-impulsive Gaussian sound. Therefore, research is needed to investigate whether marine mammal exposure to complex sound follows a similar pattern to that of human and terrestrial mammals and to establish impact assessment guidelines that reflect the real-world sound exposure environment.

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# Adaptive Call Design to Escape Masking While Preserving Complex Social Functions of Calls in Killer Whales

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**Abstract**

Since first described as moans or squeals, pulsed calls of killer whales (*Orcinus orca*) have been extensively studied. Most research focused on their social functions, such as maintaining contact by members of family groups or hunting parties, recognizing members of different communities of the same ecological type (killer whale ecotypes are differentiated by food preferences [Ford, Killer whale: *Orcinus orca*. In: Encyclopedia of marine mammals. Academic, pp 650–657, 2009]), or members of populations from different ecotypes. Social learning and intergenerational social transmission of call structure are seen as the main drivers for stability of group- or population-specific repertoires of stereotyped calls that appear to change very slowly over generations. Other ecological drivers potentially influencing call stability have been given less attention, including the influence of propagation loss and ambient noise on structural conformity. The stereotypic structure of calls (ordered occurrence of distinct components), however, allows identification of groups and populations at considerable distances. In order for calls to be recognizable at greater distances, their structure needs to allow reliable propagation in differing ambient noise conditions. Model simulations of variation in automated call detection range suggested that different spectral call frequencies are responsible for optimal detection in environments with different ambient noise levels. In a number of experimental field trials, the influence of noise on propagation of calls from two different populations of fish-eating killer whales was examined. The goal of these experiments was to determine if call structure allows some degree of compensation from noise masking. Preliminary results of the field experiments presented here showed that the propagation of dominant frequencies of calls, which varied between call components of call types, showed variations in propagation distances. The propagation was correlated with the ambient noise levels present at different receiver locations, such as upper or lower parts of the water column. Furthermore, the use of certain buzz-type call components, which are common initial components of many killer whale calls, differed in propagation, especially when they were used as standalone calls by members from a subpopulation spending more time in noisy environments. Spectral energy differences in calls, however, are currently not considered when anthropogenic noise impact, an identified threat for this endangered population, is assessed.

**Keywords**

Adaptive call design · Ambient sound · Buzzes · Discrete call types · High-frequency component (HFC) · Killer whale ecotype · Low-frequency component (LFC) · Masking · Propagation and propagation loss · Pulsed sounds

## Introduction

It is generally assumed that animal communication signals and the physiological receiver capacities have coevolved to create reliable information transmission through air, water, and substrate, in order for the sender of a signal to manipulate the receiver's behavioral response and for the receiver to gain information about the sender's intent (Endler 1993). The critical information for an understanding of the impact of anthropogenic noise on animal communication therefore needs to consider the adaptive structure of the signal that evolved to minimize propagation loss of relevant information between sender and receiver, taking the hearing and perception abilities of the receiver to recognize the signal into account to better understand both quality and magnitude of impact that the noise signature has on signal transmission loss, hearing, and perception. In this ongoing research project, the goal is to better understand the effects of noise interference on call propagation and the role of adaptive call design in masking escape in killer whales.

## Potential Evolutionary Influences of Ambient Noise on Stereotypic Killer Whale Call Structure

Killer whales produce a variety of communication signals including whistles (Ford 1989; Thomsen et al. 2001, 2002), which can have a stereotypic design (Riesch et al. 2006) and may be ultrasonic for humans (Samarra et al. 2010), as well as amplitude-modulated pulsed calls, which have been early on described as "squawks" (Schevill and Watkins 1966) or "screams" (Steiner et al. 1979) and which often occur as "discrete" call types due to their stereotypic design (Moore et al. 1988; Ford 1991; Strager 1995; Miller and Bain 2000; Yurk et al. 2002; Deecke et al. 2005; Filatova et al. 2007; Schall and Van Opzeeland 2017; Wellard et al. 2020). In the literature, both terms "calls" and "whistles" are sometimes used interchangeably to describe pulsated sounds. The contours within the spectrogram of a whistle depict primarily the frequency modulation of the sound types, while the contours in a spectrogram of a call or moan can be the result of both frequency and amplitude modulations. The latter is due to a signal processing artifact that produces spectrographic sidebands as a result of the pulse repetition rate (Watkins 1966). This confusion generally does not negatively influence the recognition of the discrete structure of a call but needs to be considered when the goal is to determine the negative effects of noise on call propagation because the fundamental frequency of a pulsated call is not automatically the contour with the lowest frequency in the spectrogram but can be one of the multiples of that frequency. If one assumes that the fundamental frequency is carrying information relevant to caller and receiver, one needs to look at the impact of noise on each of the multiple contours of a call because higher frequencies generally attenuate more than lower frequencies over distance and can be affected

by masking noise in higher-frequency bands. Higher frequencies of killer whale calls also tend to transmit more directionally than lower frequencies (Schevill and Watkins 1966; Miller 2002, 2006). This feature allows killer whale calls to reduce the impact of masking noise when the noise is not between the caller and the intended receiver of the call, while masking is pronounced when the noise source is opposite to the receiver from the caller's location. Directionality plays an important role in acoustic disturbance in areas where sound exposure to anthropogenic sounds is not omnidirectional, which is mostly the case in nearshore environments where foraging often takes place for some species, such as killer whales.

## Killer Whale Call Structure

Killer whale calls contain one or more distinct frequency components (Fig. 1), which are responsible for the discreteness of recognizable call types (Ford 1984). The number of distinct components varies and can exceed more than five in some populations, for which the term "compound" call was developed (Strager 1995).

Different call components tend to experience different propagation loss in part due to differences in directionality of components (Miller 2002) but also because of differences in spectral sound energy distributions (Fig. 2).

The depicted source-level (SL) distributions (Fig. 2), obtained from recordings made with a suspended hydrophone from a small vessel when whales were in close proximity and from acoustic recordings and movement tags attached to whales via suction cups (Dtags), may underestimate the relative contribution of HFCs because of

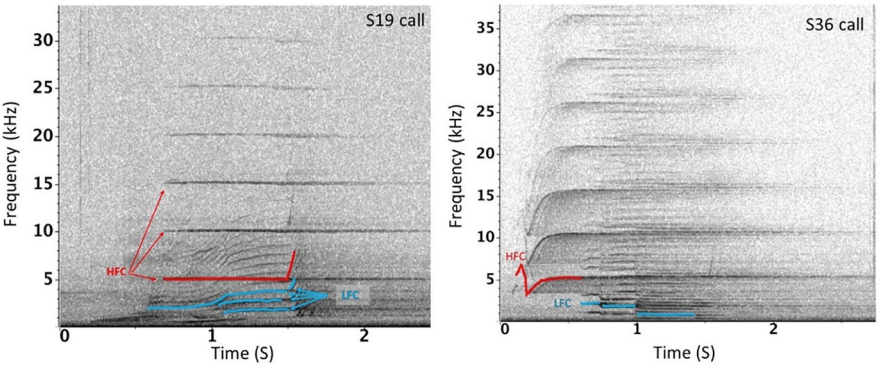



**Fig. 1** Spectrograms of two discrete calls of a population of fish-eating killer whales called Southern Residents. HFC (in red) indicates the presence of a high-frequency component, while LFC (in blue) indicates the presence of a low-frequency component. HFCs can overlap LFCs (two-voiced calls) or in sequence with LFCs. Calls can have more than one LFC but generally only one HFC. All spectrograms were produced using RAVEN Pro acoustic analysis software (Version 1.6, Cornell Lab of Ornithology)

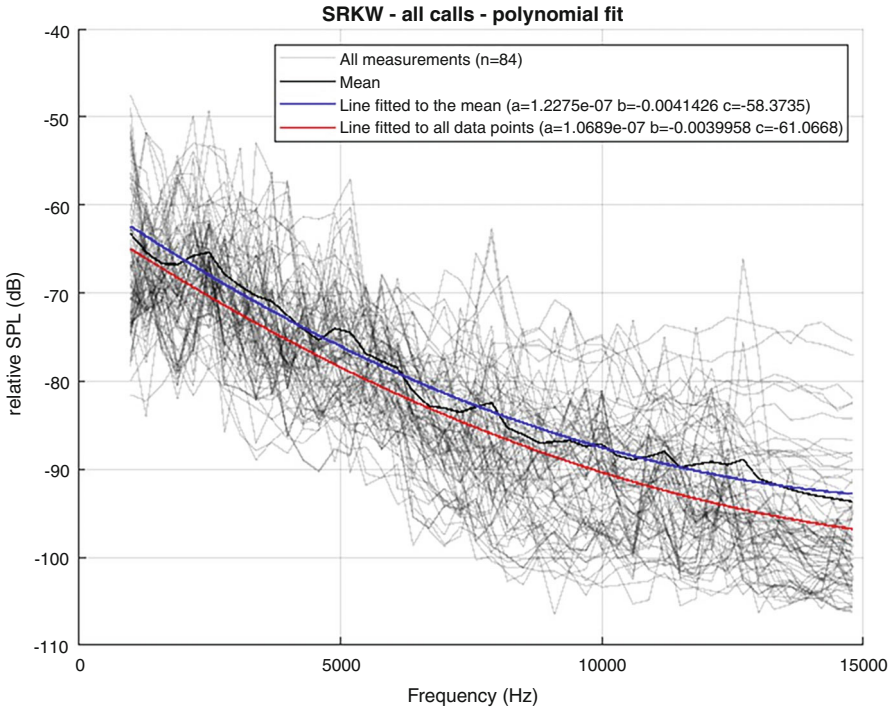



**Fig. 2** Relative spectral distribution of killer whale call source levels averaged over 300 Hz bands across a 1–15 kHz range and based on high signal-to-noise ratio (SNR) calls recorded from small vessels and from retrieved acoustic data collected during Dtag deployments

the directionality of HFCs. Both acoustic data acquisition methods may not have captured the received levels of directional signals appropriately. The recordings made from a vessel were done using an omnidirectional sensor, while the Dtag was placed behind the blowhole below which sound is generated in odontocetes (Cranford et al. 1996). Nevertheless, the distribution shown in Fig. 2 indicates a greater variation in SLs for higher frequencies. The importance of signal transmission quality is generally not realized when applying broadband source and received levels in acoustic impact assessments because these assessments focus on total exposure dosage. Neither will assessments that weigh impact across the frequency range of hearing sensitivities because they assume the impact is the same for all frequencies.

Aside from HFCs and LFCs, killer whale calls also contain burst-pulse components with lower pulse rates that aurally appear as buzzing sounds. These buzzes can either occur at the beginning of a discrete call (Ford 1991; Yurk 2005) or are produced as separate calls (Fig. 3).

Buzzes may play a much more important role in killer whale communication than previously thought and may be especially important when ambient sound levels are high because the HF parts of the transmitted burst-pulse are loud and directional.

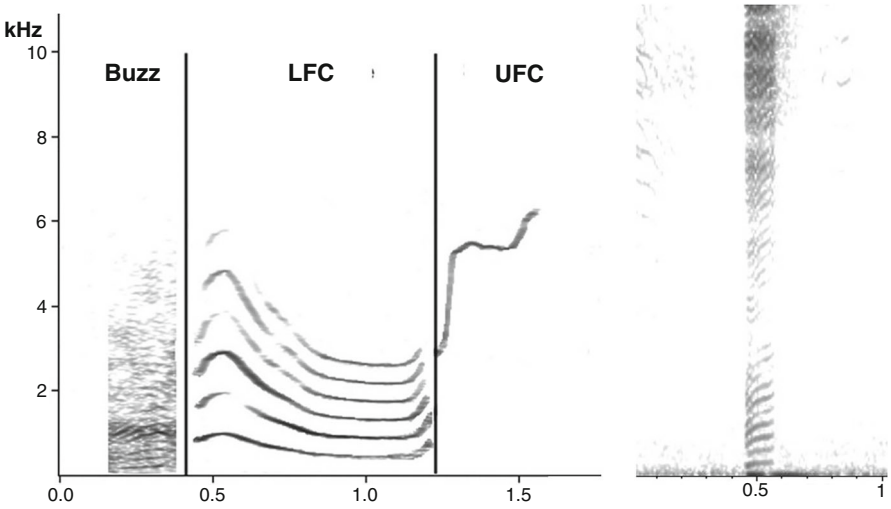



**Fig. 3** Discrete calls with initial buzz component (left) or consisting of a single buzz component (right)

## Experimental Study of Call Component Propagation Under Varying Noise Conditions

\Samples of prerecorded discrete killer whale calls were emitted via an underwater sound projector (M 18C 6.0, Geospectrum Technologies Inc.) mounted on a triangular-shaped metal frame together with either an analogue hydrophone (M36, Geospectrum Technologies Inc) or a digital hydrophone (OceanSonics) tethered to a vessel via a cable to determine the source level of the emitted signals (Fig. 4). The frame was lowered from the vessel, and the calls were recorded at various distances by a drifting acoustic buoy (CAB, SMRU Consulting) with two RESON TC 4014 suspended hydrophones at depths of 12 and 17 meters and a sea-bottom-mounted passive acoustic (PAM) mooring containing a digital recorder (AMAR, JASCO Applied Sciences) (Fig. 4). The CAB recorded WAV files with a sampling rate of 125 kilocycles per second (kcps), which provided an effective frequency range of recorded sounds from 10 Hz to 60 kHz. The bottom-mounted PAM mooring deployed at a location in Juan de Fuca Strait, British Columbia (Fig. 5), at a depth of 120 meters recorded WAV files at 256 kcps via M36 hydrophone, which provided an effective recording range of 10 Hz to 128 kHz. The projected sound files were WAV files with a sampling rate of 48 kcps allowing transmission of sounds up to a 24 kHz. The M18 projector had an effective transmission frequency range of 1–40 kHz, with a peak resonance frequency at 12.4 kHz and maximum source level of 218 dBV re. 1 μPa. Transmission did not exceed 175 dB re. 1 μPa peak pressure.

The source levels of the emitted calls were adjusted via a 600 W sound amplifier to assure that the signal emission stays within the ranges of broadband source levels reported in the literature (Miller 2006; Holt et al. 2011). Because the vessel and the acoustic buoy were both drifting during the sound emissions, the distance between

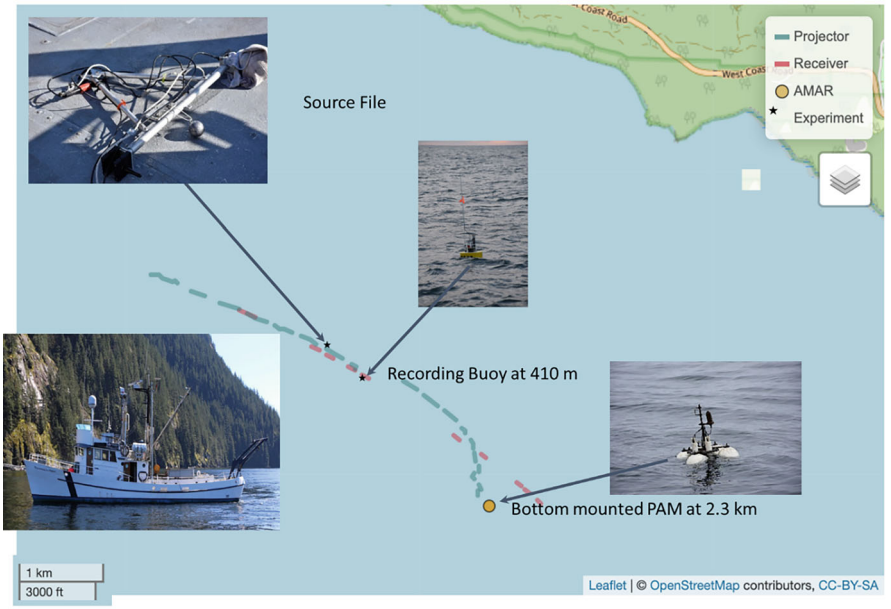



**Fig. 4** Image at top left: frame with mounted projector (covered by towel) and hydrophone at 1 m distance from projector; image at center: drift buoy with PAM and transmitter; image at bottom right: bottom-mounted PAM mooring (surfacing during retrieval); image at bottom left: vessel from which the projector was deployed. Depicted are the paths of the projector frame versus the drifting buoy and the respective locations of drifting PAM and bottom-mounted PAM during one sound emission experiment

sound projector and recording stations varied with time and was measured via GPS transmitters on both platforms and verified via laser range finder readings when possible. The exact position of the acoustic mooring on the sea bottom was determined via triangulation using acoustic pinging. The emitted sounds consisted of calls with LFCs and HFCs and buzzes. The recorded files were aurally and spectrographically inspected for the presence of calls or call components. The received sound pressure levels (RL) of contours of calls or components were determined and compared to the corresponding source sound pressure levels to allow for signal propagation loss (PL) calculations. Parallel to the signal PL assessment, ambient noise sound pressure levels (NL) at the receiver location were calculated.

The emitted sounds contained killer whale calls from three populations inhabiting the waters off British Columbia, Canada, and Washington State, USA. Two of the populations were of a killer whale ecotype that specializes on foraging fishes, particularly larger salmonid species (resident killer whales), while the third set of calls was from an ecotype that specializes on marine mammal prey (Biggs killer whales). Resident killer whales appear to use discrete calls to maintain contact and group cohesion over distances of several kilometers (Ford 1989; Miller 2002). The estimates of PL of calls over ranges of hundreds of meters to several kilometers and the influence of ambient sound on call RL are the target of this study.

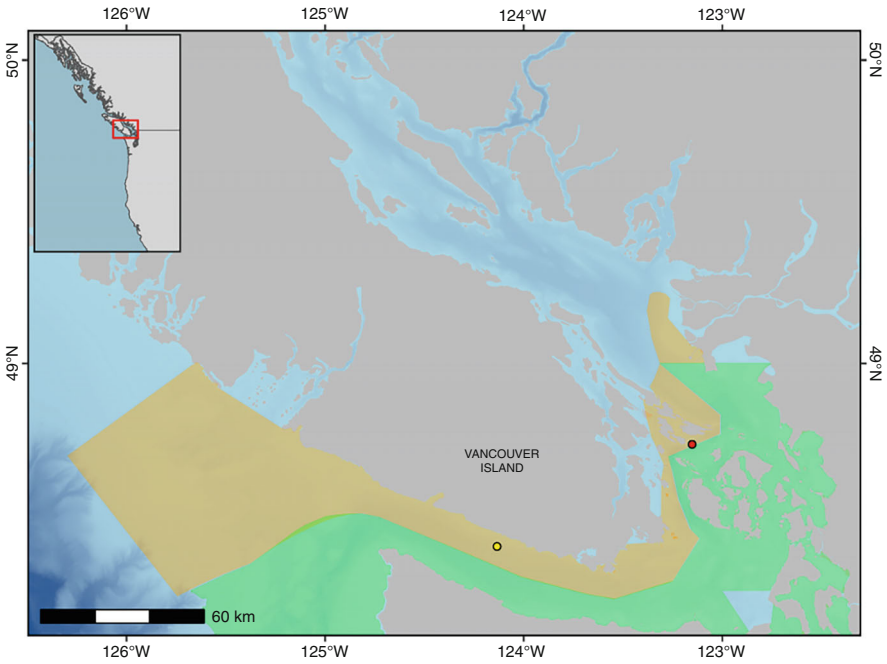



**Fig. 5** Map of designated critical habitat areas for SRKW in transboundary Canadian (beige) and US (green) waters. The yellow and red dots mark the locations of call propagation loss experiments for which spectrographic examples are shown in Figs. 6 and 7

The call propagation loss experiments took place within the designated critical habitat of one of the resident killer whale populations called Southern Resident killer whales (SRKW) that are considered endangered due to their low numbers (73 individuals (https://www.whaleresearch.com/orca-population)). Identified threats to SRKW include disturbance from anthropogenic noise sources (Fisheries and Oceans Canada 2018).

The area designated as critical habitat for SRKW (Fig. 5) is used regularly by SRKW year-round and is characterized by high commercial and noncommercial vessel traffic and other human activities and has higher ambient noise levels compared to more remote locations of the coast.

SRKW can be considered familiar with high ambient noise levels resulting from both natural sources such as wind, rain, and other precipitation and anthropogenic sources (Burnham et al. 2023). In the following sections, some interesting examples of how different call components propagate under varying noise conditions are presented.

## Call Propagation Loss Examples

Below are two examples of call propagation losses from two sequences of calls including buzzes from the SRKW population. These examples are from an experiment in Juan de Fuca Strait in March of 2019. Juan de Fuca Strait is the waterway at

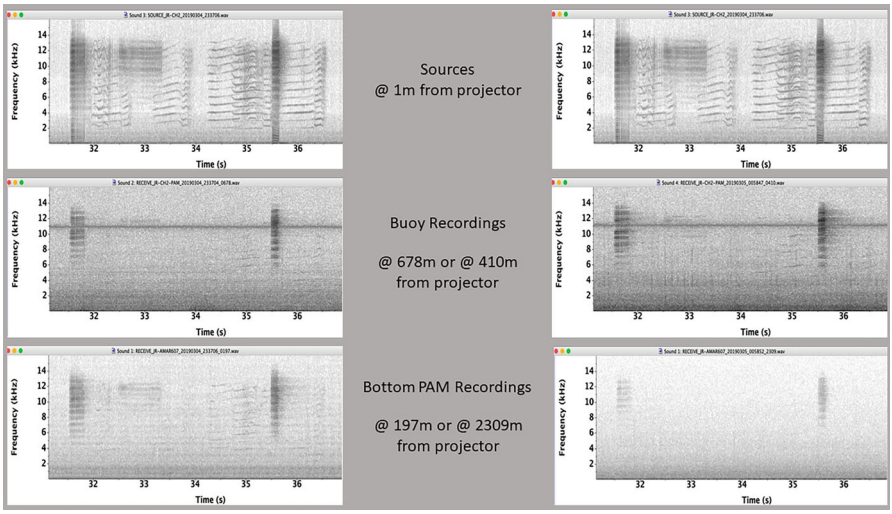



**Fig. 6** Call propagation loss between projector and two different receivers during two experiments using the same SRKW call sequences. Spectrograms at the top are the source sound levels of the call sequences. The spectrograms in the middle are received sound levels at two hydrophones suspended from a buoy (<20 m water depth). The two spectrograms at the bottom show received levels at a single hydrophone mounted on an acoustic mooring at an approximate water depth of 120 m

the bottom of Fig. 5. The experiment took place about 5 km from the shore of Vancouver Island in the North (Yellow dot in Fig. 5). The closest shipping lane for commercial vessels transiting was about 4 km away. Typical duration of sound transmissions was 6 min and transmissions were repeated up to ten times during each trial while drifting.

In the example shown in Fig. 6, the received ambient noise levels appear to be higher in the upper water layers (<20 m) than at the sea bottom at this location (>100 m). Lower frequencies are more readily masked by ambient noise, which at close distances may be dominated by the noise from the idling research vessel, but also appear to propagate less effectively between surface and bottom (recording at 197 m lower left spectrogram). Buzzes with distinct energy peaks in higher frequencies (>10 kHz) propagated better in the upper water layers and were the only calls/call components visible in the spectrogram of the associated recording at the sea floor at distances greater than 2 km.

Figure 7 shows the results from a second call propagation loss experiment also taking place in March 2019 near the confluence of three waterways called Swanson Channel, Boundary Pass, and Haro Strait (Fig. 5 – red dot). The area is used by SRKW and by commercial vessels traveling to and from the Port of Vancouver using Boundary Pass, Haro Strait, and Juan de Fuca Strait. Some commercial vessels also travel up and down Swanson Channel but less often. Commercial vessels passed the recoding buoy at ranges of <2 km. Significant small vessel traffic was a prominent feature in this area and occurred during the experiment.

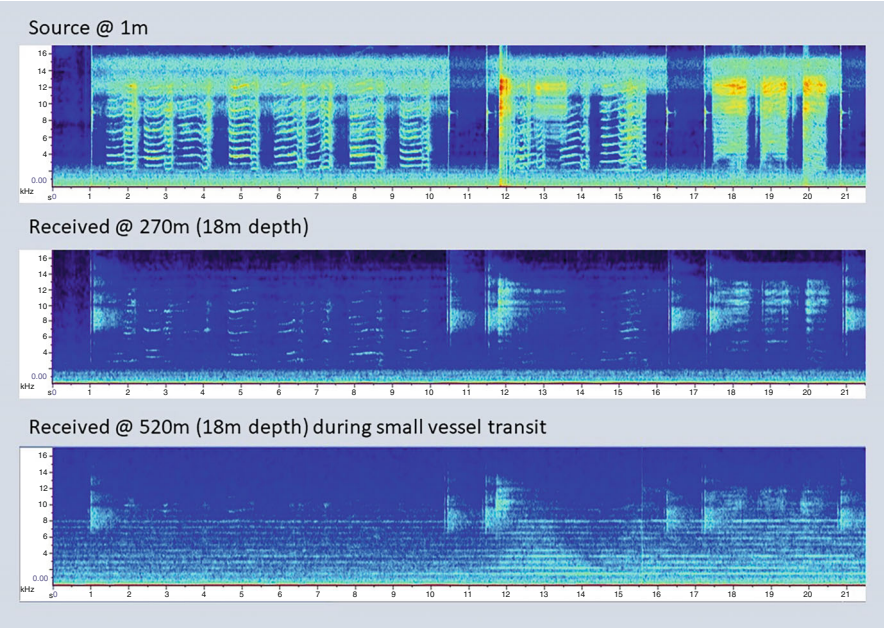



**Fig. 7** Propagation loss of SRKW call sequences in Swanson Channel/Boundary Pass with small vessel passing closer than 1 km of the recorder (bottom spectrogram). Received call sequences are recorded in the upper water column (<20 m) via a buoy with integrated PAM. Broadband signals at the beginning of call sequences are artifacts

The sound source in the bottom spectrogram in Fig. 7 is from a small vessel passing by the acoustic buoy closer than 1 km. When ambient noise levels at a receiver location were low (e.g., Fig. 7 – center spectrogram), most call components propagated according to the modeled propagation loss relative to their spectral source levels over distance (e.g., several hundreds of meters). LFCs, HFCs, and buzzes were detectable at distances of several kilometers in recordings made by hydrophones located at depths less than 20 m. At all distances, detectability decreased proportionally with the level of ambient noise present in frequency bands containing peak energy levels of LFCs, HFCs, and buzzes. However, when ambient noise was high because sources (i.e., vessels) were closer (<1 km) to the recorder, primarily buzzes were detectable because of their higher source levels in higher frequencies.

## Relevant Considerations for Anthropogenic Noise Impact Assessments on Killer Whales and Potentially Other Odontocetes

The frequency structure, i.e., spectral design of the calls, appears to play an important role in the propagation loss and detectability of calls, especially in environments with higher ambient noise levels. This matches results of a modeling study conducted by Mouy et al. (2020). The modeling study suggested that often higher

sound frequencies than the frequency with highest amplitude (around 1 kHz, Fig. 2) had the greatest influence on positive detection via an automated detector classifier (DC) and therefore play an important role for call detection (Fig. 8).

Call source levels appear to decrease linearly with increasing frequencies below 5 kHz (Fig. 2). However, the frequencies responsible for furthest detection range are not always the lower frequencies at every geographical location or location within the water column (Figs. 6 and 8). The call source level distribution may be influenced by the shape of the hearing sensitivity curve of killer whales (Fig. 9), which may have adapted to natural ambient noise variation, especially in the upper water column. Resident killer whales spend more time in the upper 20 m of the water column than at depth (Wright et al. 2017) and are therefore more likely to vocalize in these upper water layers. Future analysis of the call propagation loss experiments, using a more extended dataset, will show if these preliminary findings are more generally valid.

When ambient noise is low, LFCs of calls, as well as energy peaks of Buzzes at frequencies below 5 kHz, may travel the furthest. When ambient noise levels are higher, call components such as HFCs and Buzzes with energy peaks in higher frequencies may contribute more toward call detectability over greater distances due to typical low-frequency content of ambient noise from natural and anthropogenic sources.

Resident killer whales live in stable matrilineal groups and maintaining contact while traveling and foraging appears an important factor to maintain social cohesion (Bigg et al. 1990; Ford 1989). Because resident killer whales tend to search and capture prey individually but share prey afterward (Ford and Ellis 2006, 2014), the distance over which these killer whales can maintain contact may determine the area over which they can effectively search for food. Under high ambient noise levels, call components such as buzzes and directional HFCs may allow whales to maintain acoustic contact. HFCs and buzzes may also determine the distance over which the

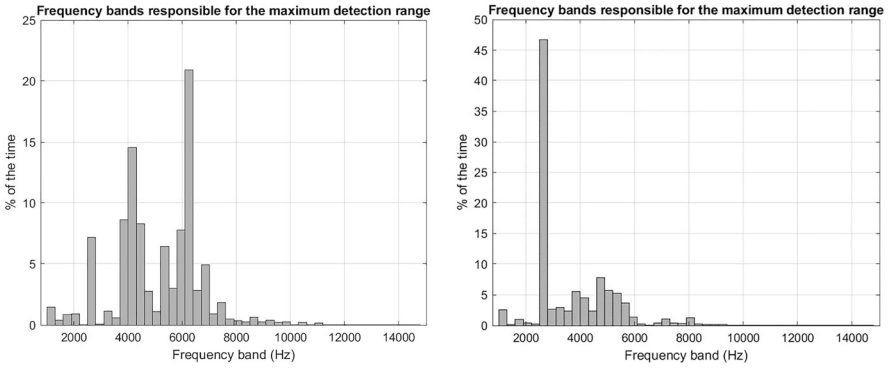



**Fig. 8** Frequency bands that were responsible for maximum detection ranges of DC system based on a stochastic model of detection range (Mouy et al. 2020). Left: at a location in Juan de Fuca Strait; Right: at a location in Boundary Pass. Results are for detection range simulations during the summer

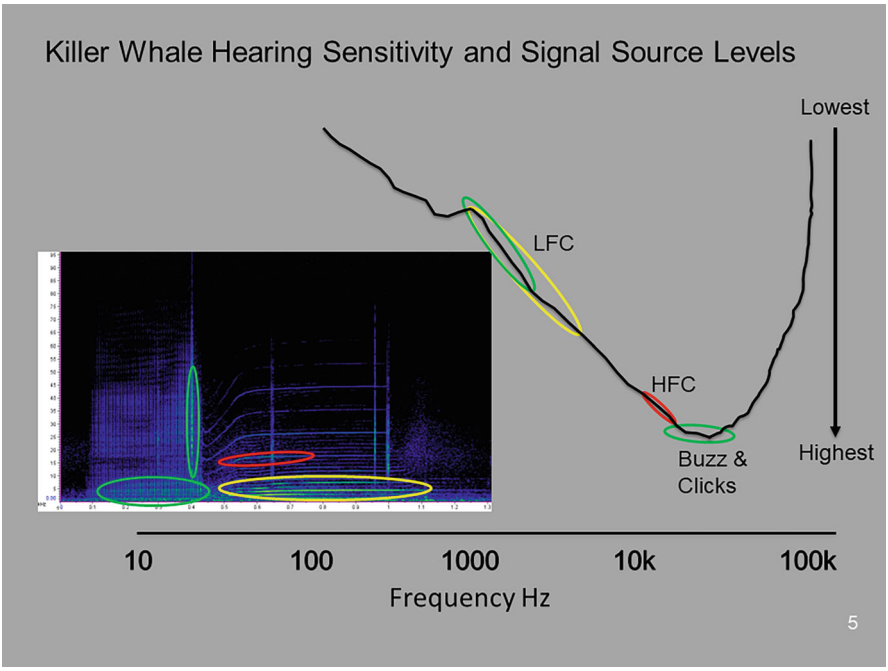



**Fig. 9** Killer whale call spectrogram and hearing sensitivity curve with projected areas containing peak signal source levels of call components including buzzes. The sensitivity curve is an adaptation based on reported hearing sensitivities of killer whales in captivity summarized by Branstetter et al. 2017

whales can spread out when traveling (Miller 2002). The call structure appears to allow killer whales to maintain contact under different propagation conditions and when ambient sound varies, hence call structure has adapted to the acoustic environment in which the whales forage, travel, and socialize. This adaptive call design allows whales to compensate for increased ambient sound levels not only by speaking louder (Holt et al. 2009), but perhaps also by paying attention to different call components.

While many anthropogenic noise impact assessments for killer whales and other cetaceans pay attention to the broadband received "noise dose" (e.g., Williams et al. 2014), the call structure and the respective source levels of call components rather than the broadband source level may need to be taken into account to accurately assess impact of noise in frequencies that are important to killer whales and to determine when noise impact becomes impossible to mitigate by the whales. Numerical broadband vessel noise measurements are skewed toward the frequencies in which the noise signature is the highest, which is typically below 250 Hz for vessel noise (Southall et al. 2017). Furthermore, most anthropogenic noise assessments rely on sound-level measurements taken at greater depths (e.g., the sea bottom in waterways used by large commercial vessels) at which the measurement of noise

generated by these vessels can be assessed accurately (MacGillivray et al. 2019). However, these measurements may not always reflect noise exposure levels relevant to killer whales in upper water layers and the impact on signals traveling between upper and deeper water layers correctly. Vessel noise appears to propagate well in the water layers in which they are generated, which is typically in the upper 20 m of the water column. Furthermore, vessel noise also propagates well at higher frequencies especially in shallow waters (Hermannsen et al. 2014) and in particular noise generated by smaller vessels may dominate the ambient noise in higher frequencies over short distances (Hermannsen et al. 2019). Small vessels may be around the whales more often than large commercial vessels and often closer to the whales. The impact of noise from these smaller vessels may need to be assessed based on their contribution in those higher frequencies. The limits to noise compensation due to the adaptive call design needs to play a bigger role in acoustic impact studies in order to correctly determine the extent of negative acoustic impact on whales and dolphins.

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# Response of Harbor Porpoises to Pingers and Acoustic Harassment Devices

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Julika Voß, Armin Rose, Vladislav Kosarev, Raúl Vílela, and Ansgar Diederichs

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**Abstract**

Pingers and acoustic harassment devices (AHDs) are used by industries (e.g., fisheries, aquaculture, and offshore wind development) all over the world to mitigate harmful impacts on marine mammals such as the harbor porpoise (*Phocoena phocoena*). This review compares different devices and their effects specifically on harbor porpoises and gives recommendations with regard to which device to use for the desired outcome. Pingers mainly emit acoustic signals at sound pressure levels between 115 and 155 dB re 1 µPa (rms) @ 1 m at frequency ranges between 2.7 and 160 kHz, whereas AHDs mainly emit signals at sound pressure levels between 172 and 194 dB re 1 µPa (rms) @ 1 m at frequency ranges between 1 and 150 kHz. Therefore, harbor porpoises generally show a smaller avoidance radius around pingers than around AHDs. Selecting an appropriate device should not only depend on the cost and user-friendly installation, but primarily on the desired range of the acoustic signals, keeping the size of the area to be exposed to sound and the duration as small as possible (e.g., pingers for a minimum deterrence distance of 100–300 m and APDs for a minimum deterrence distance of 500–750 m). Factors including habituation and effects on species other than the target species need to be considered in choosing mitigation devices

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and monitored on a regular basis. When selecting an appropriate device in a targeted manner, pingers and AHDs are a good tool to deter harbor porpoises from an area where they may suffer harm, such as hearing damage or entanglement in fishing nets.

**Keywords**

*Phocoena phocoena* · Temporary hearing threshold shift · Deterrence · Pinger · Acoustic harassment device · Bycatch · Impulsive noise · Offshore wind farms

## Introduction

Harbor porpoises (*Phocoena phocoena* L., 1758) are the most common cetacean in the North Sea (Hammond et al. 2013, 2017) with approximately 345,000 individuals inhabiting the area (Hammond et al. 2017). This species has a distinct hearing range from below 1 kHz to about 180 kHz with the lowest hearing threshold of about 33 dB re 1 µPa between 100 and 140 kHz (Kastelein et al. 2002). Below 16 kHz and above 140 kHz, sensitivity decreases by about 10 dB and 260 dB per octave, respectively, resulting in a sharp increase in the hearing threshold (Kastelein et al. 2002). Within the range of their best hearing sensitivity, i.e., around 130 kHz, are their click frequencies of echolocation (Møhl and Andersen 1973; Teilmann et al. 2002).

Therefore, threats to this species do not only include chemical contamination (Berggren et al. 1999; Weijs et al. 2010; Mahfouz et al. 2014), predation by grey seals (Van Bleijswijk et al. 2014; Leopold et al. 2015; Podt and IJsseldijk 2017), prey depletion due to overfishing and habitat destruction (MacLeod et al. 2007), and gillnet fisheries leading to bycatch (Berggren 1994; Vinther and Larsen 2004; Bjørge et al. 2013), but also noise pollution (Pirotta et al. 2014; Culloch et al. 2016; Wisniewska et al. 2018). Specifically, sound emissions can affect their behavior and lead to a temporary hearing threshold shift (TTS) or permanent hearing threshold shift (PTS) (Kastelein et al. 2011; Tougaard et al. 2015, 2022). For very-high-frequency cetaceans such as harbor porpoises, a TTS-onset threshold of 153 dB re 1 µPa$^2$s and a PTS-onset threshold of 173 dB re 1 µPa$^2$s are assumed as sound exposure levels (weighted) for nonimpulsive noise such as signals of most pingers and AHDs (Southall et al. 2019).

For deterrence of marine mammals from the danger area and thus mitigation of these threats, pingers and acoustic harassment devices (AHDs) are used all over the world around aquaculture facilities and other fishery activities, during offshore wind farm construction and before detonations. Namely, these devices aim to deter all animals from a certain area by emitting acoustic signals. Pingers emit acoustic signals at similar frequency ranges, but higher sound pressure levels than AHDs. Pingers can thus significantly reduce the risk of bycatch (Palka et al. 2008; Dawson et al. 2013; Larsen and Eigaard 2014). In contrast, AHDs can prevent hearing damage from impulsive noise such as pile driving from offshore wind farms or

detonations by deterring the animals further away than pingers (Brandt et al. 2013a; Popper and Hawkins 2016).

However, some studies suggest that also AHD signals can lead to a TTS and that these devices can thus have a much stronger deterrent effect than intended (Schaffeld et al. 2019; Findlay et al. 2021; Todd et al. 2021). Therefore, this review compares different devices and their specific effects on harbor porpoises and gives recommendations in regard to which device should be used for specific outcomes.

## Considering Studies with Different Methods

Studies of interest to this review were identified based on the criteria of (a) using pingers and/or AHDs as source of sound; (b) analyzing their effects on harbor porpoises; and (c) specifying avoidance distances. In total, this review includes 29 studies which partly include different types of pingers and AHDs. However, most of these studies only report individual details on avoidance distances, with varying degrees of avoidance reactions (e.g., closest observed approach as opposed to maximum zone of deterrence). The avoidance distances summarized in this review are therefore based on individual recordings. Besides, even with pingers and AHDs from the same production series, some variation in output is likely (Kastelein et al. 2010). Therefore, the results of this review should not be considered as total range of possible avoidance distances.

The studies included in this review are based on different methodologies; with their own strengths of weaknesses, these need to be taken into account when interpreting "apparent responses." Thus, for the purpose of this review, the methods were categorized as follows:

1. Noise propagation modeling: By combining the hearing range of harbor porpoises and the sound characteristics of a device, the radius around the pinger or AHD where the animals may avoid and/or hear the signals can be acoustically modeled (e.g., Northridge et al. 2010; Todd et al. 2019).
2. Passive acoustic monitoring (PAM): Detection rates using devices such as Cetacean Porpoise Detectors (C-PODs) provide an indication of harbor porpoise presence since the animals use their echolocation system almost continuously – also during sound exposure (Akamatsu et al. 2007; Brandt et al. 2013b; Haelters et al. 2015; Wisniewska et al. 2016).
3. Visual observations: Visual observations such as theodolite tracking can be an effective tool to monitor the behavioral response of harbor porpoises to pinger and AHD activation within a limited radius (e.g., Koschinski et al. 2003; Kyhn et al. 2012; Brandt et al. 2013a).

As all methods have their advantages and disadvantages; this review considers the results of studies using different methods to summarize current knowledge on the effects of pingers and AHDs on harbor porpoises. In total, 8 studies are based on noise propagation modeling (1 with pingers, 5 with AHDs, and 2 with both types),

10 studies are based on passive acoustic monitoring (5 with pingers and 5 with AHDs), and 11 studies are based on visual observations (3 with pingers and 8 with AHDs).

## Deterrence Effect of Different Kinds of Pingers and AHDs

All the pingers and AHDs considered in this review emit acoustic signals in a frequency range between 2.7 and 160 kHz (Table 1) and 1 and 150 kHz (Table 2), respectively. The frequency ranges greatly vary among the individual devices of a certain type, but less so between pingers and AHDs on average.

The sound pressure level, however, greatly differs between pingers and AHDs. All reviewed pingers emit acoustic signals between 115 and 155 dB re 1 µPa (rms) @ 1 m (Table 1). In contrast, all analyzed AHDs emit acoustic signals between 172 and 194 dB re 1 µPa (rms) @ 1 m (Table 2). Accordingly, AHDs produce higher sound pressure levels.

Due to the different frequencies and sound pressure levels, harbor porpoises are deterred to different extents depending on the type of pinger or AHD. Even though the avoidance distances summarized in this review are based on individual recordings and should not be considered as total range of possible avoidance distances, it is clearly visible that harbor porpoises generally show a smaller avoidance radius around pingers than around AHDs (Fig. 1). Namely, the closest observed approach during pinger operation is between 0 and 0.294 km, porpoises then showing strong avoidance behavior and severe disturbance and discomfort (Table 1), whereas the closest observed distance during AHD operation is between 0.017 and 1.9 km (Table 2). Furthermore, during pinger operation most harbor porpoises are deterred within 0–0.6 km, showing behavioral disturbance (Table 1). In contrast, during AHD operation most harbor porpoises are deterred within 0.01–4 km (Table 2). So in overall, AHDs deter porpoises farther away than the pingers considered here. But also between individual devices of pingers and AHDs there are large differences in the radius of disturbance depending on the frequency and sound pressure level (Fig. 1).

## Impact of Using Pingers and AHDs

Both individual fitness and ecological community structure can be affected by underwater noise (Southall et al. 2007, 2019). The source, the path and as well as the receiver affect how marine mammals react to underwater noise in each case (Erbe et al. 2016). For the source, such as the pinger or the AHD in this case, key factors are the source level, frequency, and temporal characteristics like the duration and number of operations within a short period. The path and associated environmental factors such as bathymetry, seabed sediment characteristics, and water column properties will determine whether sound is rather absorbed or reflected (Farcas et al. 2016). For the receiver, i.e., the animal's behavioral response, important factors

**Table 1** Deterrence ranges of harbor porpoises for different types of pingers and according to the intensity of disturbance

| Company | Device | Sound pressure level in dB re 1 µPa at 1 m (RMS) | Frequency in kHz | Duration in ms | Interval in ms | Method | Study area | Closest observed approach | Deterrence effect for most animals | Small behavioral response | Maximum zone of deterrence | Maximum zone of audibility | Reference |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Lien | Unknown | 115 | 2.7 | NA | 785 | Visual observations | Fortune Channel (Canada) | | 0.414 km distance | | | | Culik et al. (2001) |
| Unknown | Unknown | 115 | 2.9 | NA | 785 | Visual observations | Clayoquot Sound (Canada) | 0.133 km | | | | | Koschinski and Culik (1997) |
| Aquatec group | AquaMark 300 | 132 | 10 | 300 | 4000 | Passive acoustic monitoring | Jammerland Bay (Denmark), Gilleleje flak (Denmark) | | Reduction by 29–95% at 0 km | Reduction by 0–17% at 0.3 km | | | Kindt-Larsen et al. (2019) |
| Dukane | NetMark 1000 | 132 | 10 | NA | NA | Visual observations | Grand Manan Island (Canada) | | Reduction by 84% within 0.5 km | | | | Cox et al. (2001) |
| Dukane | Netmark 1000 | 130 | 10 | 300 | 4000 | Visual observations | Fortune Channel (Canada) | | 0.414 km distance | | | | Culik et al. (2001) |
| Dukane | NetMark 1000 | 133–145 | 10–12 | 300 | 4000 | Passive acoustic monitoring | Bloody Bay (UK), Lagabay (UK) | 0.294 km | Reduction by 50–100% within 0.5 km | | | | Carlström et al. (2009) |

(continued)

**Table 1** (continued)

| Company | Device | Sound pressure level in dB re 1 µPa at 1 m (RMS) | Frequency in kHz | Duration in ms | Interval in ms | Method | Study area | Closest observed approach | Deterrence effect for most animals | Small behavioral response | Maximum zone of deterrence | Maximum zone of audibility | Reference |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Dukane | NetMark 1000 | 139–145 | 10.5–12.6 | 300 | 3800–4000 | Noise propagation modeling | | | | | | 0.1 (sea state 2) – 0.6 (sea state 0) | Trippel et al. (1999) |
| Dukane | NetMark 100 (second generation) | 130 | 11.3 | 300 | 4300 | Noise propagation modeling | | 0 km | | | | 0.38 km (high ambient noise levels) – 2.79 km (sea state 1) | Taylor et al. (1997) |
| Airmar | Unknown | 132–155 | 10–160 | 300 | 4000 | Passive acoustic monitoring | Jammerland Bay (Denmark) | | Reduction by 40% at about 0.6 km | | Either <2.5 km or >5 km (unable to examine within study) | | Kyhn et al. (2015) |
| Aquatec group | AquaMark 100 | 145 | 20–160 | 200–300 | NA | Passive acoustic monitoring | Jammerland Bay (Denmark), St. Andrews Bay (UK) | | Reduction by 75–80% at 0 km, by 85–67% at 0.2 km and by 50–67% at 0.4 km | | 0.8 km | | Kindt-Larsen et al. (2019) |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| SaveWave | Black saver | 155 | 30–160 | 200–900 | 4000–16,000 | Passive acoustic monitoring | Jammerland Bay (Denmark) | | Reduction by 65% at about 0.6 km | | Either <2.5 km or >5 km (unable to examine within study) | | Kyhn et al. (2015) |
| Aquatec group | AquaMark 100 | 125 | 40–50 | 360 | 5000–30,000 | Noise propagation modeling | | | < 0.02 km distance | < 0.02 km distance | 0.3–1 km | 0.2 km (sea state 4) – 2 km (sea state 0) | Kastelein et (2010) |
| Fishtek | Banana pinger | 139.7 | 59–130 | 300 | NA | Passive acoustic monitoring | Kullaberg (Sweden) | | Reduction by 76% at 0 km and by 68% at 0.1 km | Reduction by 30–36% at 0.4 km | < 2.3 km | | Königson et al. (2022) |
| Fishtek | Banana pinger | 145 | 59–120 | 300 | 4000–12,000 | Passive acoustic monitoring | Mousehole (UK) | | | Reduction by 37% at 0 km and by 9% at 0.1 km | | | Omeyer et al. (2020) |
| PICE | Unknown | 20–160 | 145 | 300 | 5000–30,000 | Visual observations | Fortune Channel (Canada) | 0.13 km | 0.414 km distance | | | | Culik et al. (2001) |

**Table 2** Deterrence ranges of harbor porpoises for different types of AHDs and according to the intensity of disturbance

| Company | Device | Sound pressure level in dB re 1 µPa at 1 m (RMS) | Frequency in kHz | Duration in ms | Interval in ms | Method | Study area | Closest observed approach | Deterrence effect for most animals | Small behavioral response | Maximum zone of deterrence | Maximum zone of audibility | Reference |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| GenusWave | Unknown | 180 | 1 | 200 | 2000–40,000 | Visual observations | Sound of Mull (UK) | | | | < 0.25 km | | Götz and Janik (2015) |
| GenusWave | Unknown | 174–180 | 1 | 200 | 2000–40,000 | Visual observations | Argyll (UK), Orkney (UK) | | | | < 0.2 km | | Götz and Janik (2016) |
| Terecos | DSMS-4 | 178.5 | 4.9 | 8–16 | NA | Noise propagation modeling | | | | | | 37 km (sea state 6) – 99.1 km (sea state 0) | Todd et al. (2019) |
| Unknown | Unknown | 180 | 5.5–20.5 | 200 | 9400–59,400 | Visual observations | Luing (UK), Arsa (UK), Scarba (UK) | 1 km | 1.78 km distance | | | | Hiley et al. (2021) |
| Terecos | DSMS-4 | 177–179 | 7.5 | 1000–2000 | NA | Passive acoustic monitoring | Loch Leven (UK), loch Carron (UK) | | | Reduction by 9.9% at 0.3 km, by 7.4% at 0.438 km, by 4.3% at 0.855 km, and by 1.4% at 1.169 km | 1.2 km | | Northridge et al. (2013) |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Terecos | Unknown | NA | NA | NA | NA | Noise propagation modeling | | | | | | 14.7 km | Northridge et al. (2010) |
| Ace Aquatec | Silent Scrammer | 191 | 9.7 | 1100 | 2100 | Noise propagation modeling | | | 1.3 km distance | | | 33.5 km (sea state 6) – 68 km (sea state 0) | Todd et al. (2019) |
| Airmar | DB II plus | 181 | 10 | 2.5 | 17,000 | Visual observations | Bishop (Canada) | 0.991 ± 0.302 km | | | > 1.5 km | | Johnston (2002) |
| Airmar | Unknown | NA | 10 | 1.8 | 40 | Noise propagation modeling | | 0.017 km | | | | 2.77 km (high ambient noise levels) – 12.19 km (sea state 1) | Taylor et al. (1997) |
| Airmar | Unknown | NA | 10 | 1.8 | 40 | Visual observations | Retreat passage (Canada) | Reduction by 100% at <0.2 km, by 99.8% at 0.2–0.399 km and by 98.6% at 0.4–0.599 km | Reduction by 91.9% at 2.5–3.5 km | | > 3.5 km | | Olesiuk et al. (2002) |
| Airmar | Unknown | 192 | 10 | 2 | 46 | Passive acoustic monitoring/ noise propagation modeling | Sound of Mull (UK), loch Sunart (UK) | 0.2 km (feeding click trains) | | | | 14.7 km | Northridge et al. (2010) |
| Airmar | DB plus II | 192.5 | 10.3 | 1.4 | 40 | Noise propagation modeling | | | 1.3 km distance | | | 32 km (sea state 6) – 63.5 km (sea state 0) | Todd et al. (2019) |

(continued)

**Table 2** (continued)

| Company | Device | Sound pressure level in dB re 1 µPa at 1 m (RMS) | Frequency in kHz | Duration in ms | Interval in ms | Method | Study area | Closest observed approach | Deterrence effect for most animals | Small behavioral response | Maximum zone of deterrence | Maximum zone of audibility | Reference |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Ace Aquatec | Seal Scrammer | 186 | 10–20 | 5000 | 50,000–600,000 | Noise propagation modeling | | | 0.2 km (sea state 4) – 1.2 km (sea state 0) | 2–10 km distance | 17–57 km | 14 km (sea state 4) – 91 km (sea state 0) | Kastelein et al. (2010) |
| Ace Aquatec | Seal Scrammer | 186 | 10–20 | 5000 | 50,000–600,000 | Noise propagation modeling | | 0.38–0.59 km | 4 km distance | | | | Hermannsen et al. (2015); Kastelein et al. (2015) |
| Lofitech | Unknown | 189 | 12 | 500 | 600–90,000 | Visual observations | Helgenæs (Denmark) | Reduction by 100% within 0.19 km | 0.525 km distance | | | | Mikkelsen et al. (2017) |
| Lofitech | Unknown | 187.2 | 12.84 | NA | NA | Passive acoustic monitoring | OWF BOWL (UK) | | | Reduction by 50% within 21.7 km | | | Thompson et al. (2020) |
| Lofitech | Seal Scrammer | 189 | 14 | 500 | NA | Passive acoustic monitoring | OWF Dan Tysk (Germany) | | Reduction by about 90% at 1.5–3 km | | 12 km (possibly 18 km) | | Dähne et al. (2017) |
| Lofitech | Seal Scarer | 189 | 14 | 500 | 600–90,000 | Passive acoustic monitoring | North Sea (Germany) | | Reduction by 52–95% within 0.75 km | | 7.5 km | | Brandt et al. (2013b) |
| Lofitech | Seal Scarer | NA | 14.5 | 550 | 1000–90,000 | Visual observations | Fyns Hoved (Denmark) | Reduction by 100% within 1.9 km | Reduction by 50% at 2.1–2.4 km | | > 2.6 km | | Brandt et al. (2013a) |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Lofitech | Seal Scarer | 193 | 15 | 500 | 1000–30,000 | Noise propagation modeling | | | 0.1 km (sea state 4) – 0.6 km (sea state 0) | 10–24 km distance | 2–9 km | 18 km (sea state 4) – 91 km (sea state 0) | Kastelein et al. (2010) |
| Lofitech | Seal Scarer | 193 | 15 | 500 | 1000–30,000 | Noise propagation modeling | | 0.04–0.15 km | 2 km distance | | | | Hermannsen et al. (2015); Kastelein et al. (2015) |
| Ferranti-Thomson | Seal Scrammer | 194 | 25 | 20,000 | NA | Noise propagation modeling | | 1.03 km | | | | 6.88 km (high ambient noise levels) – 10.14 km (sea state 1) | Taylor et al. (1997) |
| Van Oord, SEAMARCO | FaunaGuard porpoise module | 172 | 60–150 | NA | NA | Passive acoustic monitoring | North Sea (Germany) | | Reduction by 30–100% at 0.75 km | Reduction by 25–60% at 1.5 km | 2.5 km | | Voß et al. (2023) |
| Van Oord, SEAMARCO | FaunaGuard porpoise module | 172 | 60–150 | NA | NA | Noise propagation modeling | | | 1.2–1.3 km distance | | | | de Jong and Binnerts (2014) |
| Van Oord, SEAMARCO | FaunaGuard porpoise module | 172 | 60–150 | 9000–18,000 | 3000–10,000 | Noise propagation modeling | | > 1 km | | | | | Geelhoed et al. (2017) |

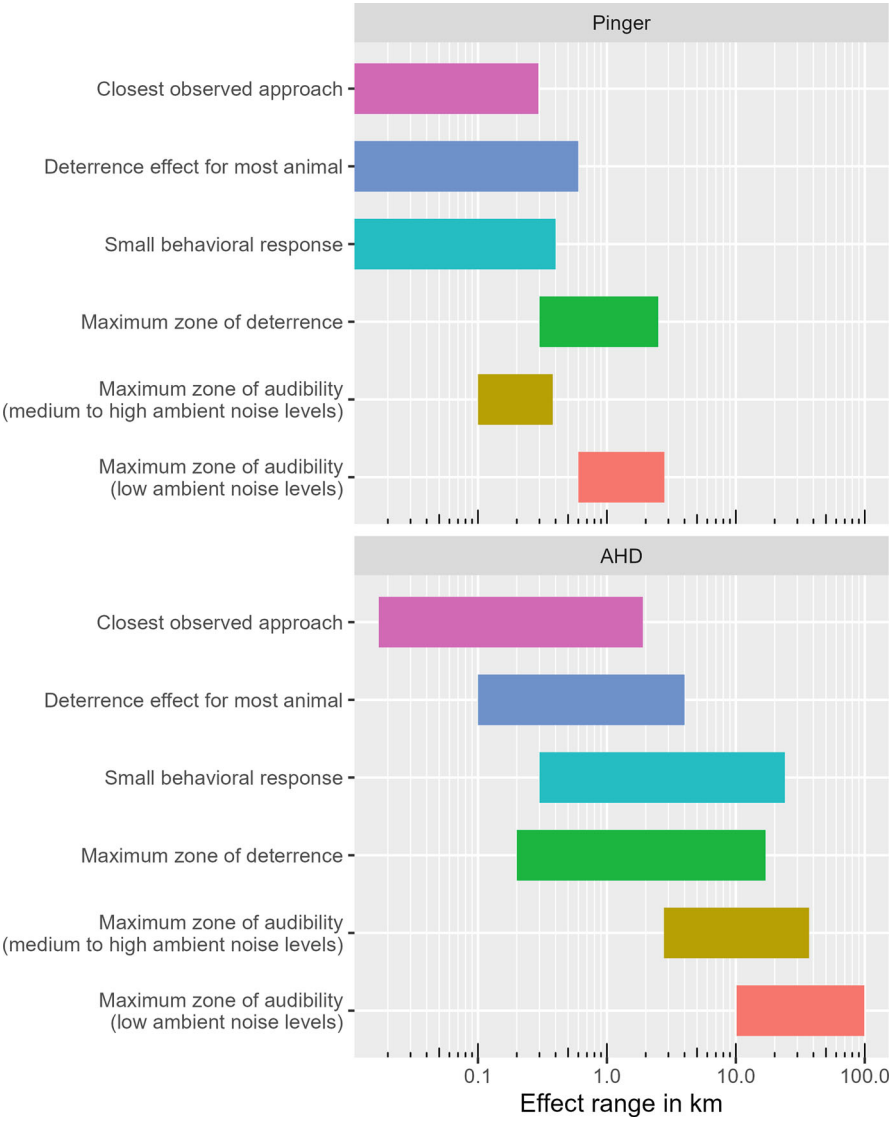



**Fig. 1** Deterrence ranges of harbor porpoises for pingers and AHDs listed in Tables 1 and 2 and according to the intensity of disturbance. Since most of the studies included in this review only report individual details on avoidance distances, with varying degrees of avoidance reactions (e.g., closest observed approach as opposed to maximum zone of deterrence), and since even with pingers and AHDs from the same production series, some variation in output is likely (Kastelein et al. 2010); the results of this figure should not be considered as total range of possible avoidance distances

include hearing ability, the behavioral context, distance to the source, previous exposures, demographics, and food availability (Erbe et al. 2016). Harbor porpoises, for example, are more likely to be displaced if suitable foraging opportunities are available in other areas without noise (Carlström et al. 2002). Therefore, harbor porpoises react to acoustic signals from pingers and AHDs in their individual contexts.

However, pingers generally deter harbor porpoises up to several hundred meters as they mainly emit sound up to pressure levels of 155 dB re 1 µPa (rms) @ 1 m (Table 1). AHDs, in contrast, lead to farther-reaching disturbance; they deter most animals up to a few kilometers away from the device (Table 2). Small behavioral responses during AHD operation, such as small changes in the surfacing rate, swimming pattern, and distance toward the AHD, even occur up to 21.7 km, and their acoustic signals are still audible to harbor porpoises up to 37 km when considering medium to high background noise levels (Todd et al. 2019; Thompson et al. 2020). This far-reaching disturbance is caused by the sound characteristics of AHDs: AHDs were primarily developed to reduce economic losses at fish farms due to seal predation (Fjälling et al. 2006; Graham et al. 2009; Harris et al. 2014) and were thus adapted to the hearing ability of seals during development. Grey seals, for example, usually communicate in a frequency range between 0.1 and 0.4 kHz, some of them with upsweeps up to 4.7 kHz (Au and Hastings 2008), while harbor seals have their best hearing ability between 0.5 and 40 kHz (Kastelein et al. 2009). Therefore, seal scarers mostly emit acoustic signals between 1 and 25 kHz at sound pressure levels up to 194 dB re 1 µPa (rms) @ 1 m (Table 2), resulting in signals being transmitted over long distances (Erbe et al. 2022).

Due to the sound characteristics of most AHDs, these devices appear to have the potential to induce a TTS (Schaffeld et al. 2019). For example, Hiley et al. (2021) calculated that harbor porpoises suffer a TTS in up to 704 m distance around an Airmar dB Plus II AHD when operating for 30 min, whereas Todd et al. (2021) calculated a TTS in up to nearly 31 km distance when the same kind of device operated for 24 h. These calculated threshold shifts might differ in reality, as environmental factors such as bathymetry, seabed sediment characteristics, and water column properties affect sound propagation (Farcas et al. 2016). However, due to the TTS potential of AHDs, it is discussed whether some devices are appropriate for the specific outcome.

Particularly in Scotland, the number of AHDs around aquaculture sites has rapidly increased in recent years (Cabinet Secretary for Rural Affairs and Islands 2022) with a suspected overuse (Todd et al. 2021). Namely, cumulative effects of AHDs can generate noise over large areas. Assuming a single device being installed at each site, Findlay et al. (2021) calculated that 23% of a certain Special Area of Conservation for harbor porpoises is exposed to AHD noise levels sufficient to cause a TTS. In the worst case scenario, 0.9% of the special area of conservation is exposed to AHD noise levels sufficient to induce a PTS.

However, alteration of sound pressure levels, frequency, and operational mode can reduce AHD impact (Todd et al. 2021). Accordingly, acoustic porpoise

deterrents (APDs) such as the FaunaGuard Porpoise Module were specifically designed for the deterrence of harbor porpoises (van der Meij et al. 2015). These devices emit acoustic signals in frequency ranges between 60 and 150 kHz at sound pressure levels up to 172 dB re 1 µPa (rms) @ 1 m (Table 2). APDs consequently show significantly larger propagation losses compared to other types of AHDs (Rosemeyer et al. 2021). Most harbor porpoises are thus still deterred up to about 0.75–1 km, but the maximum zone of deterrence is considered to be 2.5 km (Table 2). APDs are therefore not considered to cause far-reaching disturbance such as observed with other types of AHDs (Voß et al. 2023).

## Recommendations for Using Pingers and AHDs

This review shows that there are a variety of pingers and AHDs with different effect ranges, so device selection has to be carefully considered depending on the objective of use and overuse should be avoided. Selecting an appropriate device should not only depend on the cost and user-friendly installation, but primarily on the desired range of the acoustic signals, keeping the size of the area to be exposed to sound and the duration as small as possible.

Therefore, pingers are recommended for small-scale deterrence where about 100–300 m is sufficient, such as continuous pinger operation at fishing nets for avoiding bycatch. However, the response of harbor porpoises to pingers and AHDs needs to be monitored permanently to early notice if harbor porpoises show signs of habituation, as has been suggested in other studies (Cox et al. 2001; Carlström et al. 2009; Kyhn et al. 2015). Moreover, other species such as seals may perceive pingers as "dinner bells" and thus prefer to swim to nets with active pingers, leading to increased bycatch rates for these species (Carretta and Barlow 2011; Wargo Rub and Sandford 2020) and depredation events. Therefore, pingers should be chosen which are emitting noise outside the best hearing range of other resident marine mammal species such as seals and dolphins (Königson et al. 2022). In this way, these other species can only hear the acoustic signals at very short distances and are not attracted to them, so that no increase in bycatch rates for these species is to be expected.

For several years, additional research has been conducted on devices that synthetically simulate signals of harbor porpoises with which the animals usually warn each other. These so-called Porpoise Alarm (PAL) devices are also designed to keep porpoises away from nets and thus reduce bycatch. In a field study of three types of signals, two types were observed to increase the distance to the device by 23 and 32 m, respectively, and thus to deter the animals, whereas one type was observed to attract harbor porpoises with a decrease in distance of 29 m (Culik et al. 2015). The type that produced the largest increase in harbor porpoise deterrence (i.e., by 32 m) was tested in Western Baltic gillnet fisheries and resulted in a 64.9% decrease in the expected bycatch rate, and even 79.7% at a gear spacing of less than 210 m (Chladek et al. 2020). In contrast, in a field study in Iceland, no significant difference was observed between the PAL and control sets, and almost all of the harbor porpoises captured in the PAL sets (11 out of 12) were large adult males, while the sex ratio in

the control sets was 7 males and 4 females, suggesting that adult males may be attracted to the PAL sets (ICES 2018). Therefore, whether PAL devices can be used as pinger replacements on nets in any region to keep harbor porpoises away while avoiding the "dinner bell" effect on seals, and whether harbor porpoises habituate to the sounds or even might be attracted needs further investigation.

APDs are recommended for a minimum deterrence distance of 500–750 m. During the construction of offshore wind farms in Germany, for example, the sound exposure level is legally required to be below 160 dB re 1 $\mu Pa^2$ s at a distance of 750 m from piling locations (BMU 2013). Due to the continuous development of noise abatement systems, many construction projects now comply with this threshold or even fall below it. Beyond this distance, it is assumed that harbor porpoises no longer suffer hearing injuries, so deterrence of the animals up to 750 m and thus deploying an APD for this purpose is sufficient. Voß et al. (2023) indicated that it is sufficient for the APD to be switched on for about 20–25 min, as after this time effectively no detections were registered within this radius. Thus, it can be assumed that the APD deterred nearly all harbor porpoises from the danger area. To minimize potential habituation of harbor porpoise to the APD signals, it is recommended that the device should be active for around 25 min and no more than 30 min (Voß et al. 2023). For the same purpose, e.g., the FaunaGuard Porpoise Module uses different complex signal sequences (van der Meij et al. 2015). Furthermore, as these devices emit sounds outside the best hearing range of seals (NOAA National Marine Fisheries Service 2018), no effect of APD operation on these animals is assumed.

In case of other impulsive noise sources such as detonations, significantly higher sound exposure levels than with pile driving may occur at 750 m. During a high-order detonation of unexploded ordnance (UXO) with a charge mass of 140 kg TNT equivalent in the Danish North Sea, Salomons et al. (2021) measured a sound exposure level of 170.2 dB re 1 µPa (rms) at a distance of 12 km. The authors calculated that harbor porpoises might have suffered PTS up to about 4 km distance. During another high-order UXO detonation with a charge mass of 201.5 kg TNT equivalent in the Scottish North Sea, a sound exposure level of 160 dB re 1 µPa (rms) at a distance of 55 km was calculated (Robinson et al. 2022). Keeping the noise level as low as possible should be prioritized. For example, noise abatement systems and low-order detonations, using a smaller charge to disrupt and/or consume the UXO than high-order detonations, are recommended. In some cases, however, harbor porpoises still need to be deterred over a wider area than possible with APDs. Therefore, an APD should be used first, and in addition after about 25 min another kind of AHD which causes a farther-reaching deterrence. Deploying an APD for 25 min before deploying another type of AHD will prevent harbor porpoises from staying within 750 m distance to the other kind of AHD (Voß et al. 2023), thus avoiding TTS caused by AHDs such as seal scarers. According to Findlay et al. (2021), for example, animals would suffer a TTS after 85 s of operation of an Airmar AHD, assuming an animal is 100 m away from the Airmar AHD when it is switched on, and that it swims away from the AHD at a constant speed of 4.3 m/s. In this calculation, even an increased swimming speed was already assumed, since harbor porpoises usually swim at an average speed of 0.5–1.2 m/s without noise exposure

(Culik et al. 2001; Otani et al. 2001; Kastelein et al. 2018), but were, e.g., observed to swim up to 10 m/s during pinger operation (Brennecke et al. 2022). Since longer APD operation times only led to a moderate extension of the effect range, with a maximum of 2.5 km effect range for the longest available operation times (35–40 min; Voß et al. 2023), another kind of AHD is recommended thereafter, which causes a farther-reaching deterrence. An Airmar AHD, for example, is assumed to be able to deter most animals up to a distance of 3.5 km (Olesiuk et al. 2002). Assuming a swimming speed of 0.9 m/s with constant direction away from the AHD location, this device is recommended to operate for about 60 min to ensure that all harbor porpoises have moved away from a radius of 3.5 km around the detonation location and at the same time do not become used to the sounds and return. If a larger deterrence radius than 3.5 km is necessary, the animals must be gradually deterred, e.g., firstly, by 25 min of APD operation at the location of the detonation, then by 60 min of other kinds of AHD operation at the location of the detonation, and finally, by continuation of other kinds of AHD operation at the location of the detonation, but also by other kinds of AHD operation 2.5 km north, east, south, and west to the location of the detonation.

In general, pingers and AHDs are good tools to deter harbor porpoises from an area where they may suffer harm, such as hearing damage or entanglement in fishing nets. However, the appropriate device must be carefully selected for the specific purpose, i.e., there must be neither too little deterrence nor excessive usage. Furthermore, factors such as habituation and effects on species other than the target species need to be considered and monitored on a regular basis.

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# Beaked Whale Behavioral Responses to Navy Mid-Frequency Active Sonar

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E. Elizabeth Henderson

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**Abstract**

Although beaked whales remain a cryptic and elusive family, much has been learned about their dive and foraging behavior in the last few decades through the use of passive acoustic monitoring (PAM) and a variety of tagging methods. Species-specific echolocation pulses can be used to acoustically track individuals and groups, and biologging tags provide information on dive and movement behavior. These technologies have also provided a window into how beaked whales behaviorally respond to Navy mid-frequency active sonar, which in turn may elucidate maladaptive behaviors that could lead to stranding. This chapter reviews existing knowledge about beaked whale behavioral responses to anthropogenic sound sources in general, and to Navy sonars in particular. By understanding the breadth of ways beaked whales behaviorally respond, the scientific community can continue to work toward recognizing and mitigating responses that could lead to stranding, as well as determining the population-level implications of impacts on individuals.

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Cetaceans · Navy sonar · Impacts · Stranding · Dive behavior · Avoidance behavior · Review

## Introduction

Beaked whales (family: Ziphiidae) are one of the most ubiquitous but elusive families of cetaceans, distributed globally from Antarctic to Arctic waters (Macleod et al. 2006) but are rarely visually observed due to their cryptic surface behavior. Less than 20 years ago, very little was known about beaked whale behavior, distribution, or abundance, with many species described based on a few records of stranded animals (Dalebout et al. 2002; Macleod and Mitchell 2006), while only a few abundance estimates based on visual surveys existed (Barlow et al. 2005). In fact, as recently as a decade ago, there were concerns that some populations were declining, such as Cuvier's beaked whales (*Ziphius cavirostris*) in Southern California (Moore and Barlow 2013), which was hypothesized to be due in part to the use of naval sonar in the region. However, the discovery that beaked whales produce echolocation pulses unique to each species (Baumann-Pickering et al. 2013a; Johnson et al. 2006; Zimmer et al. 2005) has led to an explosion of Passive Acoustic Monitoring (PAM) and biologging efforts to identify the distribution and abundance of beaked whales globally. The main driver of this increased focused was a series of mass strandings of beaked whales in the 1990s and 2000s, where naval Low- and Mid-Frequency Active Sonar (LFAS and MFAS) were a plausible, if not likely contributing factor (e.g., Cox et al. 2006; D'Amico et al. 2009; U.S. Department of Commerce, U.S. Department of the Navy 2001; Filadelfo et al. 2009; Frantzis et al. 2002; Simmonds and Lopez-Jurado 1991). Therefore, in addition to learning more about basic questions, such as foraging and diving behavior, occurrence patterns, and abundance, researchers have been striving to assess the behavioral responses of beaked whales to MFAS to help global navies better understand and mitigate their impacts. This chapter is a review of the current knowledge and future direction of beaked whale behavioral responses to MFAS.

In the last two decades, researchers have definitively identified the echolocation pulses for 12 different species of beaked whales, including Cuvier's beaked whales (Frantzis et al. 2002; Zimmer et al. 2005), Blainville's beaked whales (*Mesoplodon densirostris*; Johnson et al. 2006; Madsen et al. 2013; Rankin and Barlow 2007), Arnoux's beaked whales (*Berardius arnuxii;* Barlow et al. 2021a; Rogers and Brown 1999), northern bottlenose whales (*Hyperoodon ampullatus;* Hooker and Whitehead 2002), Gervais' beaked whale (*Mesoplodon europaeus*; Gillespie et al. 2009), Longman's beaked whale (*Indopacetus pacificus*; Rankin et al. 2011), Baird's beaked whale (*Berardius bairdii*; Baumann-Pickering et al. 2013b; Dawson et al. 1998) Sowerby's beaked whale (*Mesoplodon bidens*; Cholewiak et al. 2013), True's beaked whale (*Mesoplodon mirus;* DeAngelis et al. 2018), and Shepherd's beaked whale (*Tasmacetus shepherdi*; Leunissen et al. 2018). Most recently, the BW37V

pulse was described by Griffiths et al. (2019) and was confirmed to be Hubb's beaked whale (*Mesoplodon carlbubbsi*) in 2021 (Ballance et al. 2023). Several echolocation pulses that have been recorded globally have yet to be linked to a source species (Barlow et al. 2021a; Baumann-Pickering et al. 2013a, 2014; Hildebrand et al. 2015; Giorli et al. 2018; LeBien and Ioup 2018; Manzano-Roth et al. 2022; McCullough et al. 2023; McDonald et al. 2009; Stanistreet et al. 2017; Trickey et al. 2015), although based on known distributions of beaked whale species, some linkages have been inferred. These PAM efforts have found several species to be far more broadly distributed than previously known (Barlow et al. 2021a; Griffiths and Barlow 2016; Stanistreet et al. 2017), new beaked whale habitats have been discovered (Yack et al. 2013), and it has been recognized that some population abundances may not be decreasing after all (e.g., Moore and Barlow 2017), are higher than previously estimated based on visual surveys alone (Barlow et al. 2021b), or may even be increasing (DiMarzio et al. 2020). Additionally, in the last decade, at least four new species or potential new species have been identified, including Deraniyagala's beaked whale (*Mesoplodon hotaula*; Dalebout et al. 2014), Sato's beaked whale (*Berardius minimus*; Brownell and Kasuya 2021; Yamada et al. 2019), and Ramari's beaked whale (*Mesoplodon eueu*; Carroll et al. 2021), along with an as-yet unidentified beaked whale off Baja California, Mexico (Barlow et al. 2022).

In addition to the increase in PAM efforts for beaked whales, the advent of biologging tags, including acoustic recording tags, satellite tags, and time-depth recording (TDR) tags, has led to our improved knowledge about foraging dive behavior. Early tagging studies revealed extreme deep diving depths in Blainville's and Cuvier's beaked whales (Baird et al. 2006; Johnson et al. 2004; Tyack et al. 2006; Zimmer et al. 2005). These pioneering studies determined that Blainville's and Cuvier's beaked whales dove to depths of 600–1900 m for durations of up to an hour, and that these deep foraging dives were interspersed with shallower dives of 100–400 m for a period of 60–90 min. Studies with sound and movement tags (i.e., DTAGs, Johnson and Tyack 2003) found that beaked whales produced echolocation pulses only on their deep foraging dives and only once the animals were deeper than 200–500 m, and produced buzzes associated with prey capture attempts (Johnson et al. 2004; Tyack et al. 2006). As more tag types were developed, additional studies built upon these early findings. Dives of previously unimaginable depth and duration were recorded (Schorr et al. 2014; Quick et al. 2020), and our understanding of the full dive cycle structure was further refined. The full dive cycle duration is defined as the time from the onset of one deep foraging dive to the next. Within that period occurs the deep foraging dive and an Inter-Deep-Dive Interval (IDDI) comprised of a series of intermediate "bounce" dives without foraging clicks interspersed with shallow surfacing bouts (Falcone et al. 2017; Minamikawa et al. 2007; Shearer et al. 2019; Tyack et al. 2006). The descent rate of the deep dive tends to be faster than the ascent rate with a higher fluke rate, and the vertical angle is steeper during the descent than the ascent (Baird et al. 2008; Tyack et al. 2006). The presence of bounce dives and the longer, slower ascent from the deep foraging dives are theorized to be mechanisms by which beaked whales avoid predators (Aguilar de

Soto et al. 2018; Bernaldo de Quirós et al. 2019; Hooker et al. 2009). Furthermore, beaked whales in a group dive with a high level of synchrony both in movement and vocal behavior until they reach their foraging depths, and then forage separately while remaining within a few hundred meters of each other (Aguilar de Soto et al. 2018, 2020; Cioffi et al. 2021; Alcázar-Treviño et al. 2021). As additional species have been tracked via tags and PAM, most dive characteristics hold true, although variability has been discovered among species. For example, the Cross Seamount beaked whale is the only species to forage solely at night (or at least to produce that specific echolocation pulse at night), and also appears to only dive to a few hundred meters depth to forage (Manzano-Roth et al. 2022; McCullough et al. 2023). In some regions, Baird's beaked whales may also forage at shallower depths of 400–700 m (DeAngelis et al. 2023), although in other regions they have been found to dive to depths comparable to Blainville's and Cuvier's beaked whales (Minamikawa et al. 2007; Stimpert et al. 2014). Sowerby's beaked whales dive to similar depths as other beaked whales, but their deep dives are shorter, their clicking and movement is more rapid, and their faster stroke rate leads to a higher ascent speed than found for sympatric Blainville's beaked whales (Visser et al. 2022). In addition, Sowerby's beaked whales perform fewer intermediate dives, instead spending more time near the surface (Visser et al. 2022). Additional information, including dive cycle periodicity, temporal and seasonal patterns in diving behavior, and the foraging behavior of individuals in groups has continued to be elucidated through tagging and PAM studies (Arranz et al. 2011; Barlow et al. 2020; Barlow and McCullough 2022; Cioffi et al. 2021; Henderson et al. 2016).

Beaked whale dive behavior has evolved as a behavioral and physiological adaptation to the preferred prey of this family of odontocetes. Beaked whales forage primarily, and in some cases exclusively, on meso- and bathy-pelagic squid and mid-water or benthopelagic fishes that do not undergo diel vertical migrations (MacLeod et al. 2003; MacLeod and D'Amico 2006; Smith et al. 2021; Visser et al. 2021). Differences and specializations among species may indicate niche separation in sympatric species (MacLeod et al. 2003; Visser et al. 2021, 2022) or geographic divergence in species with prey overlap (MacLeod et al. 2003). Sympatric species may further partition shared habitat by diving to different depths (McSweeney et al. 2007; Naito and Mitani 2007). Beaked whales occur in all waters deeper than about 1 km but appear to be particularly common in habitats with steep slopes and complex bathymetry, including submarine canyons and seamounts (Arranz et al. 2011; Hazen et al. 2011; Johnston et al. 2008; Lanfredi et al. 2016; MacLeod and Zuur 2005). These habitats in turn have been found to aggregate prey via interactions between deep sea currents and seafloor topography (D'Amico et al. 2001; MacLeod and D'Amico 2006; MacLeod and Zuur 2005), and areas of frequent beaked whale foraging activity have been found to have more dense concentrations of preferred prey than similar adjacent habitats (Benoit-Bird et al. 2020; Southall et al. 2019). Dynamic oceanographic features have also been found to correlate with beaked whale occurrence, such as the deep maximum oxygen layer (Lanfredi et al. 2016), or salinity and temperature in specific layers that may structure prey distribution (Hazen et al. 2011), although these relationships are less well studied. While

future shifts in distributions of beaked whale populations related to climate change are likely as prey ranges change (e.g., Hooker et al. 2019), little work has been done to investigate these possible shifts. Macleod (2009) included beaked whale species in their assessment of potential global changes to marine mammal distributions relative to climate change. They hypothesized that out of the 21 species considered, 6 would experience a range expansion, 8 a range shift, and 5 a range contraction, depending on if they were cold- or warm-water limited, and that 12 of these shifts would not be favorable to the species (Macleod 2009).

All this baseline information on distribution, habitat use, and foraging behavior is crucial when assessing and understanding behavioral responses to anthropogenic stressors, particularly in the case of naval sonar. This specific noise type, along with seismic airguns to a lesser extent, has been linked to several global cetacean mass stranding events that predominantly included beaked whales, particularly Cuvier's beaked whales. In the late 1990s and early 2000s, there was a series of mass strandings that were associated with multi-national navy training events that utilized LFAS and MFAS (Cox et al. 2006; D'Amico et al. 2009; U.S. Department of Commerce, U.S. Department of the Navy 2001; Fernandez et al. 2005; Frantzis 1998; Jepson et al. 2003; Martín et al. 2004; Simmonds and Lopez-Jurado 1991). Typically, operational MFAS uses a series of tonal pulses in the frequency range of 1–10 kHz (e.g., Henderson et al. 2022b), while LFAS uses frequencies less 1 kHz, although this varies by country. Ultimately, there appeared to be several major factors that, when occurring together, greatly increased the likelihood of a mass stranding of cetaceans. These included the use of multiple ships with sonar in a bathymetrically complex environment with limited egress for the whales; the presence of a strong surface duct or acoustic waveguide; the sonar source occurring within the depth of the surface duct; the activity occurring in an area where the shoreline is within 10s of km of the 1000 m isobath; and vessels moving faster than 5 knots (D'Amico et al. 2009; D'Spain et al. 2006; U.S. Department of Commerce, U.S. Department of the Navy 2001). Many international navies have adjusted their mitigation policies and changed areas of military readiness activities in response to these findings, and some have entered into regional multi-national agreements to further help reduce potential impacts on marine mammals, with an emphasis on beaked whales (e.g., Dolman et al. 2009, 2011; NATO 2018). In the Canary Islands, where MFAS use has been banned altogether, no additional atypical beaked whale stranding events have occurred (Fernández Rodríguez et al. 2012), while in the Mediterranean Sea and other regions, beaked whale strandings continue to occur (Podestà et al. 2016; Simonis et al. 2020). While Faerber and Baird (2010) pointed out that a lack of strandings in Hawai'i may not necessarily mean animals weren't impacted, just that a stranding response has not been detected due to differences in geography of the islands, a series of aerial surveys were flown in Hawai'i after seven US Navy training events, and no stranded animals were observed (Deakos et al. 2017).

The current best explanation for beaked whale strandings associated with anthropogenic events is the onset of decompression sickness (DCS), which seems to be a result of a combination of behavioral and physiological factors (Bernaldo de Quirós

et al. 2019; Fahlman et al. 2014; Hooker et al. 2012). Deep diving cetaceans such as beaked whales and sperm whales (*Physeter macrocephalus*) may already live at the edge of nitrogen supersaturation due to the long durations of their deep dives coupled with a long time spent at intermediate depths (Hooker et al. 2012). As discussed in detail below, behavioral response studies of beaked whales have found that these whales most often respond to MFAS by ceasing to forage, but remaining at depth while moving away from the sound source when the exposure starts during a foraging dive. Alternately, in areas with an extended shallow shelf, the directional movement into shallow depths could result in animals' inability to behaviorally regulate their dissolved gasses by diving. The avoidance behavior may also include an increase in aerobic effort with increased fluke rates (e.g., DeRuiter et al. 2013), which may increase $CO_2$ levels (Hooker et al. 2012), further coupled with a stress or flight-or-fight response. Increased $CO_2$ levels could in turn initiate nitrogen bubble formation in already supersaturated tissues, leading to DCS (Fahlman et al. 2014; Hooker et al. 2012). Coupling this physiological response with the behavioral response of longer deep dives and more time at intermediate depths as animals avoid the MFAS source, particularly in individual animals that may already have risk factors or be otherwise compromised, could be the perfect storm of factors that lead to stranding (Bernaldo de Quirós et al. 2019). Therefore, while long deep dives are part of the normal beaked whale behavioral repertoire (Schorr et al. 2014; Quick et al. 2020), when the existing nitrogen supersaturation is exacerbated by additional factors, stranding may occur in some individuals (Hooker et al. 2012), especially in the circumstances of close proximity to shore and limited egress (D'Spain et al. 2006). In necropsied animals that may have stranded in relation to MFAS use, there were seven consistent factors that seemed to be reliable diagnostic features indicative of DCS (Bernaldo de Quirós et al. 2019). These included one or more animals stranding in good body condition and with recent food in their stomachs; the presence of many gas bubbles made up of nitrogen (and some $CO_2$); fat hemorrhages; multi-organ gas and fat emboli; "disintegration" of interstitial tissue; and multi-organ hemorrhages in lipid-rich tissue (see Bernaldo de Quirós et al. 2019 for more details). Stranded animals presenting one or two of these features could often be linked to other causes of death, and bubble formation has also been found in other species unrelated to MFAS exposure (Bernaldo de Quirós et al. 2019; Fernandez et al. 2017; Moore et al. 2009); therefore, rapid assessment of stranded animals is critical in determining the cause of the strandings. There is ongoing research about the potential for occurrence of DCS in beaked whales and the link to naval MFAS activity, which is outside the scope of this review but is crucial in the understanding, mitigation, and prevention of future occurrences (Bernaldo de Quirós et al. 2019; Díaz-Delgado et al. 2018; Fahlman et al. 2021; Podestà et al. 2016).

## Behavioral Response Research

As discussed in Harris and Thomas (2015), research on beaked whale behavioral responses to sonar has utilized a few different approaches. There have been focused behavioral response studies (BRS) and controlled exposure experiments (CEE) on

individual tagged whales exposed to both simulated/scaled sources and real naval sonars. There have been opportunistic studies with tags, PAM, and combinations of both to identify behavioral responses of beaked whales to real world testing and training activity on active US Navy ranges. Finally, analyses of aggregated data from both data types have been conducted, where responses are modeled using data from a number of animals or risk functions created using the received levels of the sonar at the animals. While other reviews have been conducted on the impacts of anthropogenic noise or specifically sonar on marine mammals in general (Cox et al. 2006; Hildebrand 2005; Nowacek et al. 2007; Southall et al. 2007, 2016), this review is focused specifically on beaked whales that have been exposed to both real and simulated MFAS signals that are representative of real naval sonars. The sonar waveform used by different countries varies but is typically made up of one or more frequency-modulated (FM) or continuous wave (CW) pulses of varying bandwidths, durations, and interval periods (Henderson et al. 2022b). For example, the simulated waveform used in US-based BRS studies was typically characterized by an FM upsweep followed by two CW tones, with an overall bandwidth of 550 Hz from 3.2 to 3.75 kHz, 1.4 sec duration, and 25 sec interval period (Tyack et al. 2011). In contrast, the European-based 3S studies used an FM upsweep of 1 kHz from 6 to 7 kHz for the MFAS signal, and an FM upsweep or downsweep of 1 kHz bandwidth from 1 to 2 kHz (or 2–1 kHz), all of 1 sec duration and 20 sec interval period (Miller et al. 2011). In addition, exposures to other signal types, such as playbacks of killer whale (*Orcinus orca*) vocalizations, or echosounders, are also discussed in order to put the responses to MFAS in a broader context. Killer whale whistles are made up of a CW waveform and can occur from 1.5 to 18 kHz, while pulsed calls range from 250 Hz to over 30 kHz, with most energy in the 1–6 kHz band (Ford 1989; Deecke et al. 2005). Killer whales also produce a high-frequency modulated downsweep signals that occur between 20 and 27 kHz (e.g., Simonis et al. 2012). Echosounder pulses are typically short duration FM pulses at one or more specific frequencies, but often with broadband energy directed in a narrow beam (Risch et al. 2017). This review seeks to summarize and synthesize the current state of knowledge of beaked whale behavioral responses to MFAS, as well as the direction this field is moving with some ongoing studies. Note that all reported received levels are given in dB re 1 uPa root-mean-square (RMS) unless otherwise specified.

## Behavioral Response Studies/Controlled Exposure Experiments

Starting in 2006, researchers began conducting controlled experiments on tagged focal whales, where variables such as distance to the animal, presence of other conspecifics, source level of the signal, and received level at the animal could be controlled or known, and received levels and behavioral responses could be measured as accurately as possible. Several new types of descriptive and statistical analyses were developed or applied during these studies, including the severity response scale, the Mahalanobis statistic, behavioral change-point analyses, and Hidden Markov Models (HMMs) (Antunes et al. 2014; DeRuiter et al. 2013; Southall et al. 2007; Wensveen et al. 2019).

One of the first true controlled exposure experiment on beaked whales was a joint tagging and PAM effort on the AUTEC range in the Bahamas in 2007 and 2008 (Tyack et al. 2011). A Blainville's beaked whale was tagged with a DTAG and exposed to playbacks of simulated 0.55 kHz bandwidth MFAS from 3.2 to 3.75 kHz, ramped up to a maximum received level of 147 dB re 1 μPa. When the received level reached 138 dB re 1 μPa, the whale stopped echolocating and began swimming away from the source and ascending slowly. In fact, a similar response has been observed for most tagged beaked whales exposed to MFAS in subsequent BRS and opportunistic studies (e.g., DeRuiter et al. 2013; Falcone et al. 2017; Miller et al. 2015; Stimpert et al. 2014). A similar change in behavior and long, slow ascent was also observed for the same whale when exposed to playbacks of mammal-eating killer whale vocalizations in the 2–5 kHz band (Allen et al. 2014; Tyack et al. 2011), and to a second whale also tagged with a DTAG and exposed to a playback of a pseudo-random noise (PRN) signal, also in the 2–5 kHz band. The whale responded to the killer whale vocalizations at just above the detectability level of 98 dB re 1 μPa, while the response to the PRN occurred at 142 dB re 1 μPa. The PRN signal was used in this and other BRS studies as a control signal to determine if the animals are responding to received level alone or to acoustic characteristics of the sonar signal that may be perceived as similar to killer whale vocalizations (Miller et al. 2022; Tyack et al. 2011), thus indicating their response to sonar is similar to an anti-predator response.

Another set of CEEs called the Sea mammals, Sonar, and Safety (3S) project was conducted by an international team of scientists in European waters off Norway starting in 2006 and was initially focused on killer whales and herring (*Clupea harengus*) (Doksaeter et al. 2009; Kvadsheim et al. 2007; Miller et al. 2011) due to an apparent decrease in the presence of both species after an incidental exposure to Norwegian sonar. Sperm whales and long-finned pilot whales (*Globicephala melas*) were added as focal species in the first years of the study. During a follow-on effort, called $3S^2$, humpback whales (*Megaptera novaeangliae*), minke whales (*Balaenoptera acutorastrata*), and northern bottlenose whales (*Hyperoodon ampullatus*) were also added as focal species. One northern bottlenose whale was tagged with a DTAG during the main $3S^2$ effort (Kvadsheim et al. 2015; Miller et al. 2015; Sivle et al. 2015). In 2015 and 2016, an additional effort called 3S-ORBS (Off-Range Beaked Whale Studies) was focused on northern bottlenose whales in an acoustically pristine environment that had not likely been previously exposed to sonar (Miller 2015; von Benda-Beckmann et al. 2019; Wensveen et al. 2019). In the $3S^2$ experiment, the northern bottlenose whale was exposed to 1 kHz bandwidth FM upsweeps from 1 to 2 kHz for 35 min to a maximum cSEL of 162 dB 1 $\mu Pa^2s$. The exposure started while the whale was on a shallow 100 m dive and was about 5.4 km from the source. The whale initially surfaced for several breaths and then moved toward the source, with the received level at about 98 dB re 1 μPa. The whale dove while still approaching the source, and then at 130 dB re 1 μPa it turned away from the source and began diving deeper. The transmission stopped when the whale was 2074 m deep, but the whale continued diving to 2339 m where it finally began its

ascent. No foraging clicks were produced during this very long and deep dive. Following this dive, the animal dove in a series of shallow dives, as it continued very directional movement away from the source vessel for another 7 h and 35 km before the tag detached. A similar response was observed for another northern bottlenose whale tagged with a DTAG and exposed to a 1–2 kHz upsweep that was within 1 km of the animal (Wensveen et al. 2019). This whale stopped producing foraging clicks and began diving at a high speed while moving away from the source during and after the exposure for a total of 6.5 h, although this animal did resume foraging before the tag came off. A third northern bottlenose whale was exposed to a signal from 3.4 to 3.9 kHz at a distance of 17 km. This whale also dove deeper and began moving away from the source; it continued moving away without clicking for 7.5 h and almost 37 km. A fourth northern bottlenose whale was also tagged with a DTAG and exposed to the 1–2 kHz upsweep at close range; however, there was a transducer failure during this exposure and the received level never rose above 100 dB re 1 μPa. In fact, this whale approached the vessel and circled it throughout the exposure, and then began foraging within 24 min after the exposure ended (Wensveen et al. 2019). In addition to these whales tagged with DTAGs, nine other whales in the area and associated with the "focal" tagged whales were tagged with Wildlife Computers satellite tags. Received levels for these whales were modeled using propagation modeling (see von Benda-Beckmann et al. 2019 for details). These tags allowed the researchers to model behavioral responses to the sonar exposures at a variety of distances out to 346 km; most tagged animals within 28 km appeared to avoid the sound source at received levels of 117–126 dB re 1 μPa with at least one of the satellite tagged animals in apparent association with an animal with a DTAG and demonstrating very similar dive and avoidance behavior. Two of the satellite tagged whales at 38 and 346 km did not respond, but the received levels may have been at or well below detection thresholds (Wensveen et al. 2019). The 3S study continued with 3S3, focused on sperm and pilot whales and included a Norwegian Navy vessel, and will continue further in 2023 with 3S4.

The next series of controlled exposures of both simulated and real sonar signals to beaked whales occurred in Southern California (SOCAL Behavioral Response Study) and was conducted from 2010 through 2016 (Southall et al. 2014, 2015, 2016). While this study was largely focused on blue (*Balaenoptera musculus*) and fin (*B. physalus*) whales, beaked whales and delphinids were also included as focal species. This was the first CEE study to utilize real US Navy ships to conduct exposures at longer distances, along with the simulated source that had a lower source level and therefore had to be deployed within a few kilometers of the whales. This study also used the PRN signal to compare responses against the 3–4 kHz MFAS signal. Two Cuvier's beaked whales and one Baird's beaked whale were tagged with DTAGs and exposed to the simulated source. One Cuvier's beaked whale and the Baird's beaked whale were both conducting foraging dives with echolocation pulses when the CEE started, while the other Cuvier's beaked whale had started to dive on what was likely a deep foraging dive. Both foraging whales stopped clicking, and all three whales increased their fluke rates and increased their

swim speeds as they moved away from the source and conducted deeper, longer dives with slow ascents (DeRuiter et al. 2013; Stimpert et al. 2014). Received levels at the time of the responses were 98 dB re 1 μPa and 127 dB re 1 μPa for the two Cuvier's beaked whales and 127 dB re 1 μPa for the Baird's beaked whale. In all three exposures, the sound source was stationary and within 3 km of the animals. One of the Cuvier's beaked whales also had an incidental MFAS exposure from a US Navy ship conducting an exercise about 118 km away, and RLs from this exposure were in the range of 78–106 dB re 1 μPa. Interestingly, the whale did not appear to respond to this MFAS, which provided some initial insight into the role that source distance may play as a context cue for these species (DeRuiter et al. 2013; Southall et al. 2016). When a real US Navy ship was utilized in the same study a few years later, it was positioned 70 km away from a tagged Cuvier's beaked whale (a distance that was chosen based on modeled propagation through the region) and received levels were 100–120 dB re 1 μPa. No behavioral response was observed for the tagged animal from this distant source (Southall et al. 2014, 2016).

The most recent large-scale BRS study was conducted in the Atlantic Ocean off North Carolina from 2017 to 2022, following many of the protocols and lessons learned from the SOCAL BRS project (Southall et al. 2018). Both DTAG and satellite tags were deployed on Cuvier's beaked whales and pilot whales in order to maximize the numbers of animals exposed and the distances between the source and the animals. Real US Navy ships were utilized again, following the same paradigm as in Southern California, where the ship was positioned relative to the known locations of the tagged animals using propagation modeling to achieve the desired range of received levels. Satellite tags lack the motion sensors that allow the fine-scale movement analyses conducted using DTAG data, and also provide down-sampled dive data relative to DTAGs, making their recorded data more coarse. However, satellite tags can be deployed for weeks or even months, which provides a longer duration baseline and follow-on period for comparison with the exposure period. Furthermore, Bouchet et al. (2021) determined that the increased uncertainty in the data obtained through satellite tagging can be reduced through a larger sample size, and this study reached much larger sample sizes than earlier studies that relied on DTAGs alone. Analyses are ongoing and so final publications on behavioral response results are still forthcoming. However, new statistical and modeling methods have been developed to successfully analyze the satellite tag dive data in a manner comparable to the DTAG data. For example, Schick et al. (2019) explored improved methods of estimating positional uncertainty in satellite tag data in order to improve received level estimates. Hewitt et al. (2022) developed methods to detect a behavioral response in satellite tag dive data using a kernel density analysis when comparing the data to a large aggregate of baseline data. Using this method, changes in dive behavior similar to those observed with DTAGS have been detected, including extended deep dives after an exposure and reduced follow-on deep dives, which is presumed to indicate an extended cessation of foraging (Hewitt et al. 2022). Finally, a great deal of baseline behavioral and social data have been analyzed, providing a framework against which behavioral responses can be compared (e.g., Cioffi et al. 2021; Shearer et al. 2019; Quick et al. 2020).

## PAM Data on US Navy Ranges Coupled with Tags

Another approach to investigating behavioral responses to MFAS is to conduct tagging on an active US Navy range, thereby conducting an opportunistic behavioral response study to real world training activity. This type of effort has been conducted at three different US Navy ranges: the Atlantic Undersea Testing and Evaluation Center (AUTEC) in the Bahamas, the Southern California Offshore Acoustic Range (SOAR), and the Pacific Missile Range Facility (PMRF) in Hawai'i. As these efforts can often be repeated in the same area over long periods, they have the added benefit of the potential to provide additional information about the populations that are present in the area rather than just focusing on individuals. Photo-id catalogs and long-term abundance trends can be developed, which allow for inference on repeated exposures and population level impacts (e.g., Baird et al. 2011, 2012, 2018; Claridge 2013; DiMarzio et al. 2020; Falcone et al. 2009). Satellite tags have been used more often in these studies than DTAGs or other sound and movement recording tags, and therefore received levels have to be modeled rather than measured. However, new tag technologies are being developed to get longer-term deployments of days to weeks in tags with sound and movement recording capabilities that will provide valuable new insights into beaked whale behavior (Falcone et al. 2022; Sweeney et al. 2022).

At the AUTEC range in the Bahamas, along with the two Blainville's beaked whales that were tagged with DTAGs described above, a third animal was tagged with a Wildlife Computers satellite tag prior to a US Navy training exercise that was conducted on the range (Tyack et al. 2011). For the 3 days prior to the exercise, the whale's track remained within a mean of 37 km to the center of the range, whereas during the 72 h of the exercise that mean distance extended to 54 km from the center of the range. In the first 3 days after the exercise, the distance dropped to a mean of 29 km, and then again to a mean of 13 km in the next 3 days. The maximum estimated received level by the tagged whales during the training exercise was 146 dB re 1 μPa (Tyack et al. 2011). This was repeated on an additional six Blainville's beaked whales tagged with satellite tags through 2015, who were exposed to a total of four multi-ship training events (Joyce et al. 2019). Four of these whales also moved away from their initial locations by 28–68 km, although only three animals were on or near the range at the onset of MFAS. The other four animals were 10–73 km away from the range, and while two of them did move further away, the two more distant whales did not avoid the area. Propagation modeled received levels at depths of 5 and 100 m were estimated for the tag tracks that were smoothed with a continuous time random walk model (*crawl* in R, Johnson and London 2018), with maximum estimated exposures of 145–172 dB re 1 μPa. Estimated received levels were reduced as the animals moved away from the range. The three animals that had been on or near the range at the onset of the exposure period returned to the range within 1.7–3.9 days after the training ended. Finally, two (out of the seven total) satellite tags were equipped with depth sensors in order to examine dive behavior. While both animals continued to conduct deep dives during the exposure period, one animal did not conduct the very deep dives that were recorded before and after the exposure, and also conducted more mid-depth

(400–800 m) dives than were recorded before or after the exposure (Joyce et al. 2019). These tag data were also combined to develop a discrete-space continuous-time model of exposure in order to estimate transition rates as animals avoided the MFAS by moving outside the range as well as their return to the range over time (Jones-Todd et al. 2022).

A long-term study of habitat use and dive behavior has been conducted on Cuvier's beaked whales that inhabit the SOAR range in Southern California (Falcone et al. 2009). Dive behavior of individual tagged animals has been observed (e.g., Schorr et al. 2014), with detailed information on surfacing bout durations, shallow and deep dive depths and durations, and overall IDDI rates and inter-individual variability. Some of the longest and deepest dives on record for Cuvier's beaked whales were identified in this dataset, and the researchers acknowledged that while these could be within the normal dive repertoire of this species, these dives took place on an active US Navy range where MFAS and other anthropogenic noise regularly occurs (Schorr et al. 2014). In fact, the longevity of this study and the frequent occurrence of MFAS led to the development of models using the aggregated dive behavior of 16 satellite tagged animals to test for changes in foraging dive behavior relative to two types of MFAS (Falcone et al. 2017). This was one of the first efforts to look at helicopter-dipping MFAS in addition to the typical ship hull-mounted MFAS that has been the focus of most BRS efforts. This study used Generalized Additive Mixed-effects Models (GAMMs) to look for changes in dive behavior relative to both sources. For the helicopter-dipping MFAS, both deep and shallow dive durations increased as the proximity to the source decreased, although four unusually short and shallow deep dives were recorded as well. Although dive duration increased with source proximity, dive durations for both types of dives decreased as the amount of temporal overlap increased. The shallow dive duration increased more for helicopter-dipping MFAS than for hull-mounted MFAS and increased the most when both sources were active. The shallow dives also tended to be deeper when they were longer overall as would be expected, but were less deep during exposures of both types of MFAS, while surface intervals were longer and more variable during both types of exposures (Falcone et al. 2017). These results indicate that Cuvier's beaked whales may react more strongly to lower-powered sources (such as helicopter-dipping MFAS) at closer distances, possibly due to the more unpredictable nature of their deployment. Finally, overall IDDIs became longer with both source types at closer distances. The baseline IDDIs in this population range from 33 to 680 min, with a mean duration of 175 min, equating to approximately 7–8 foraging dives per day (Falcone et al. 2017; Schorr et al. 2014). In the overall dataset, there were 37 IDDIs longer than 300 min; 70% of these included the presence of one or both types of MFAS, indicating possible loss of foraging opportunities. While this study is the first to look at multiple types of MFAS, as well as the impacts of MFAS on different facets of dive behavior, it must be noted that all of the GAMM models also included other variables, including time of day, other dive variables, and region, which were also significant in the models and therefore affected dive behavior. Further, a follow-on study that modeled dive behavior relative to additional environmental variables, such as sun angle and

seafloor depth, found that although MFAS was present during at least 10.4% of the time the tagged whales were on SOAR, the effect of MFAS on most dive variables was less than 4% except for IDDIs, which continued to be longer in the presence of MFAS (Barlow et al. 2020).

Another long-term tagging and observation study has been conducted on the PMRF range off Kaua'i, Hawai'i, as well as throughout the Hawaiian Islands (Baird et al. 2008, 2009, 2018, 2019; Henderson et al. 2021; Schorr et al. 2009). This study has examined the behavior and residency patterns of a variety of odontocete species that occur in Hawaiian waters, including Cuvier's and Blainville's beaked whales. Resident populations of both species have been identified in the waters off the Big Island of Hawai'i (Baird et al. 2009; Schorr et al. 2009), in addition to a potential offshore population of Blainville's beaked whales (Baird et al. 2011), and both species have been detected acoustically on the PMRF range (Henderson et al. 2016; Martin et al. 2022, 2023). In 2021 and 2022, three Blainville's beaked whales were tagged with Wildlife Computers satellite tags prior to multi-ship training events on the range (Baird et al. 2021). These data are currently being analyzed, but dives from these whales appear to match acoustically detected dives on the range before the training events (Henderson and Baird, unpublished data), and the movements of the animals during the training events will provide insight into the behavior of these animals when they move off the range. These movements appear to be similar to those found at AUTEC, with animals moving off the range during the period of MFAS use and then returning to the range within a few days, although avoidance distances are only on the order of 10–12 km (Baird et al. 2021).

## Aggregated PAM Data and Risk Function Modeling

Finally, the vocal behavior of beaked whales relative to MFAS activity can be recorded using PAM to provide another type of observational study without tagging. While this type of study lacks the ability to identify individual responses, or to observe animals that cease vocalizing, it has the benefit of being able to assess behavioral responses over much larger spatial and temporal scales than tagging studies of individual whales. For example, multiple High Frequency Acoustic Recording Packages (HARPs) have been deployed throughout the Southern California Bight since 2006 (Baumann-Pickering et al. 2018). Among many other signals, these instruments have recorded MFAS, as well as Cuvier's beaked whale clicks, and potential impacts by MFAS on Cuvier's beaked whale echolocation can be modeled using these long-term datasets (Baumann-Pickering et al. 2018). Similarly, bottom-mounted Autonomous Multichannel Acoustic Recorders (AMARs) were also deployed off the Scotian Shelf in Eastern Canada from 2015 through 2017 and opportunistically recorded a large, multi-national naval training event (Stanistreet et al. 2022). Echolocation clicks from both sperm whales and four species of beaked whales (Cuvier's, Sowerby's, northern bottlenose, and an unidentified beaked whale) were recorded before and after the training event and at the same time the previous year. Clicks during the event were significantly

reduced for sperm whales and were reduced for both Cuvier's and the unidentified beaked whale although not significantly due to a low sample size (Stanistreet et al. 2022). PAM conducted on US Navy ranges can also assess both vocal behavior and changes in spatial use relative to the training activity over short and long timeframes. These data can also be utilized to develop received level-based risk functions of behavioral response to MFAS (Harris et al. 2015, 2018; Jacobson et al. 2022; Moretti et al. 2014; Tyack and Thomas 2019).

At the AUTEC range in the Bahamas, Blainville's beaked whale group vocal periods (GVPs) were recorded before, during, and after two multi-ship training events (McCarthy et al. 2011; Tyack et al. 2011). Significantly more GVPs were recorded before the MFAS activity than during, while GVPs remained low for 35–65 h after the MFAS activity, beyond which the number of GVPs returned to similar or higher levels than before (McCarthy et al. 2011). There was also a spatial change to the pattern of GVPs recorded on the range, such that before and after the training activity GVPs were distributed across the range, although predominantly occurred in the southwestern corner of the range. However, during MFAS activity, the reduced number of GVPs that occurred on the range were pushed to the edges of the range, with little to no acoustic activity at the center of the range. Estimated received levels for the groups that continued to forage ranged from 101 to 157 dB re 1 μPa, with a mean of 128 dB re 1 μPa (McCarthy et al. 2011; Tyack et al. 2011). Data from a third multi-ship training event at AUTEC was used to develop a behavioral response function based on a generalized additive model (GAM) that estimated the probability of a group of Blainville's beaked whales initiating a foraging dive in the presence of sonar (Moretti et al. 2014). The resulting risk function estimated a 20% probability of disturbance at 130 dB re 1 μPa and a 95% probability of disturbance at 180 dB re 1 μPa, with a 50% probability of disturbance at 150 dB re 1 μPa.

A similar reduction in the detection of GVPs during periods of MFAS activity and change in spatial distribution to the edges of the range has also been observed for another population of Blainville's beaked whales found on the PMRF range in Hawai'i (Henderson et al. 2019; Jacobson et al. 2022; Manzano-Roth et al. 2016). The risk function modeling conducted by Moretti et al. (2014) at AUTEC was repeated using data recorded on PMRF from six multi-ship training events (Jacobson et al. 2022). This was done to compare the responsiveness of another range-associated population of Blainville's beaked whales in a separate ocean basin with a different environment that did not have the deep ocean basin and limited egress found at AUTEC. This risk function differed in that it modeled three periods: the period before training activity, an initial 2–3 day period of training activity during which hull-mounted MFAS was not used but other noise sources were present, and then the 2–4 day period of training that included MFAS use. While both phases of the training activity occurred at AUTEC as well, that initial phase was not explicitly modeled and was included in the "before" period. In addition, received levels were modeled for the beaked whales in Hawai'i at an assumed dive depth of 1000 m, compared to the 100 m depths used in the Moretti et al. (2014) risk function model. Finally, in the PMRF model, the probability of a GVP occurring was only modeled

between 100 and 165 dB re 1 μPa, rather than the 80 to 200 dB re 1 μPa scale used in the AUTEC model. The resulting 50% probability of a disturbance occurred at 132 dB re 1 μPa; however, this was relative to the 44% decrease in GVPs that had already occurred in the first period of training that did not include MFAS. In fact, in a separate behavioral change-point analysis using GVP data aggregated from 16 multi-ship training events conducted at PMRF, the most significant decrease in GVPs occurred at the very start of that initial phase of training that does not include MFAS (Henderson et al. 2019; Martin et al. 2020). A follow-on analysis was conducted to look at the specific sources that may be leading to that initial decrease, including the closest point of approach (CPA) and approach angle of submarines and surface vessels, and the CPA, approach angle, and estimated received level of active training torpedoes. As this analysis was conducted outside of a classified environment, it was necessarily coarse, with CPAs of the three sources only estimated per half hour and per general hydrophone location, and with approach angles estimated relative to the hydrophone in 90-degree quadrants (approach, away, or parallel). No GAMs were significant for any of those variables relative to a decrease in GVPs unless the intercept was excluded from the model (Henderson et al. 2022a). Further analyses are planned that will use finer resolution spatial and temporal scales as well as single-source training events in order to determine what causes that initial drop in GVPs.

Other beaked whale species are also acoustically detected at PMRF, including Cuvier's beaked whales, Longman's beaked whales, and the unidentified species that produces the Cross-Seamount echolocation pulse (Martin et al. 2022; McDonald et al. 2009). A long-term analysis of Cross Seamount beaked whale GVPs found a similar spatial and behavioral response to Blainville's beaked whales, in that GVP detections were reduced during the biannual multi-ship training events and were compressed to a narrower, shallower region of the range, but returned to baseline levels within 3 days after the training ended (Manzano-Roth et al. 2022).

## Other Acoustic Sources

Behavioral responses to other acoustic sources have also been found for beaked whales via opportunistic and observational studies. A Cuvier's beaked whale that had been tagged with a DTAG was passed by a vessel during one of its foraging dives with a maximum broadband received level of 136 dB re 1 μPa and a CPA of 700 m (Aguilar de Soto et al. 2006). The broadband noise from the vessel had the potential to mask the beaked whale's echolocation pulses, while the energy in the 2–8 kHz band was 24 dB higher than noise in that band during other foraging dives, which was hypothesized could impact beaked whales if they were particularly sensitive to sound in that bandwidth (Aguilar de Soto et al. 2006). However, there were no changes in dive behavior, including depth, duration, or descent and ascent speed during this dive relative to the other dives recorded for this animal, so all impacts suggested by this analysis were hypothetical. In contrast, a true CEE of vessel noise was conducted with Blainville's beaked whales on the AUTEC range (Pirotta et al. 2012). A vessel was directed to approach hydrophones where GVPs

were being detected, and at the onset of echolocation pulses turned on their engines to full power for 2 min, up to six time per GVP. GVP durations did not change relative to the ship noise, but the number of hydrophone changes (indicating movement of the group) was reduced for groups close to the ship. This suggested that while the groups continued foraging, they significantly reduced their horizontal movement, or changed to a more directed movement, during the dive at distances up to 5.2 km from the ship (Pirotta et al. 2012).

During visual and acoustic cetacean surveys of the North Atlantic Ocean, many fewer acoustic detections of beaked whales occurred on days when a Simrad EK60 echosounder (operating simultaneously at frequencies of 18, 38, 70, 120, and 200 kHz) was active (Cholewiak et al. 2017). Fewer groups were also observed visually on days when the echosounder was active, but generalized linear models (GLMs) of sighting data found sea state and region to be the only significant variables for visual detections. In contrast, the GLMs of acoustic data found echosounder activity and region to be the significant predictors of beaked whale detections, with only seven acoustic events, or 4% of all detections, occurring when the echosounder was active (Cholewiak et al. 2017). In contrast, two 12 kHz multibeam echosounder surveys were conducted on the SOAR range in Southern California, and the number of Cuvier's beaked whale GVPs increased during and after the surveys relative to the number of GVPs detected before the surveys (Kates Varghese et al. 2020). There was also not a spatial change in where the GVPs were detected on the range during the surveys in 2017, and while the main area of foraging activity shifted during the 2019 survey, there was no reduction in the number of GVPs (Kates Varghese et al. 2021). A similar echosounder study was conducted at AUTEC in 2008, and observations of GVPs were opportunistically observed during another survey in 2015; both efforts found a similar lack of impact to Blainville's beaked whale foraging dives (Benoit-Bird et al. 2020; Hazen et al. 2011; Vires 2011). It may be that beaked whales inhabiting naval ranges are exposed to anthropogenic sound sources so often there is some degree of habituation (although see the discussion on site fidelity [e.g., Forney et al. 2017] below), leading to a lack of response to a highly directional echosounder, whereas in the North Atlantic beaked whales are less frequently exposed to these types of sounds and are therefore more reactive. Alternately, all of the studies on naval ranges used single frequency echosounders, while the vessel in the Atlantic Ocean survey used a multibeam echosounder with five frequencies; therefore, the increased responsiveness could have been due to the number and higher frequencies of the sources.

In addition to echosounders, evidence from gillnet mortality studies found that 10–12 kHz acoustic pingers affected beaked whale behavior positively (Carretta and Barlow 2011). In that study, the number of beaked whales entangled in gillnets was reduced to zero when pinger use became mandatory in a US west coast drift gillnet fishery. The effect of pingers in reducing bycatch was much greater for beaked whales than for any other cetacean species, indicating that they may be more sensitive to novel sound sources than other species.

Finally, new sources of acoustic noise are still being introduced that may be of concern. Acoustic ultrasonic antifouling devices are starting to be utilized by cruise

ship lines and other commercial vessels, and a preliminary investigation found an apparent response by Cuvier's beaked whales at Guadalupe Island off Baja, Mexico (Trickey et al. 2022). Both visual sightings and acoustic detections were reduced when a vessel was present at the island and using the antifouling device in the summer of 2019; no similar reduction in detections occurred in the following summer of 2020, when commercial vessels were not operating due to the COVID-19 pandemic. Nor was there a similar decrease in activity in the presence of other vessel noise or echosounders alone, so the reduction could not be linked to any other anthropogenic noise than the antifouling devices (Trickey et al. 2022). As commercial vessels in general, and cruise ships in particular, are found globally, this may become a prevalent source of noise impact as more companies adopt this technology.

## Other Factors that May Facilitate or Mediate Behavioral Responses

As mentioned, one hypothesis as to why beaked whales respond so strongly to naval MFAS is the sounds' possible similarities with killer whale vocalizations (Zimmer and Tyack 2007), as some killer whale ecotypes are known to prey on marine mammals (Jefferson et al. 1991) and vocalizations occur in a similar bandwidth as MFAS (Deecke et al. 2005; Ford 1989). This was first tested during the CEE conducted at AUTEC; one of the tagged Blainville's beaked whale was exposed to a playback of mammal-eating killer whale vocalizations during the foraging dive that directly followed the exposure to simulated MFAS (Allen et al. 2014; Tyack et al. 2011). The killer whale vocalizations were bandwidth limited to 2–5 kHz in order to match the bandwidth of the simulated MFAS. As soon as the received level of the playback was detectable above background noise levels, at 98 dB re 1 μPa, the animal stopped clicking and made the longest and slowest ascent that was recorded out of 32 other dives recorded on the range. The beaked whale continued to swim away from the playback location for an additional 10 h, which was a longer duration response than for any of the MFAS exposures described above in this and all other CEEs (e.g., DeRuiter et al. 2013; Miller et al. 2015; Stimpert et al. 2014). Similarly, killer whale vocalization playbacks were conducted with northern bottlenose whales, as well as long-finned pilot whales, humpback whales, and sperm whales, during the 3S CEEs (Miller et al. 2022). Northern bottlenose whales responded similarly to the killer whale vocalizations as they did to the MFAS exposures with a cessation of foraging, and thus the response to MFAS was hypothesized to be an anti-predator response. In fact, Aguilar de Soto et al. (2012, 2018, 2020) further hypothesized that beyond the strong response to MFAS, much of the typical baseline beaked whale dive behavior is an adapted anti-predator response, including not producing echolocation clicks in the first and last several hundred meters of a deep foraging dive's descent or ascent, the high level of vocal synchrony among group members once they do start echolocating, and the horizontal movement during the ascent phase of the dive. However, the long-slow ascent phase of a deep foraging dive has also been hypothesized to serve as a means to facilitate gas exchange (Hooker et al. 2012). Similarly, the increase in shallow dives and reduced time at the surface observed for

Cuvier's beaked whales during the day may serve as a visual anti-predator response (Baird et al. 2008; Barlow et al. 2020), while also functioning as a mechanism for gas exchange, lactic acid processing, or digestion (Baird et al. 2008; Hooker et al. 2009). However, extended time at intermediate depths could also lead to DCS (Bernaldo de Quirós et al. 2019; Zimmer and Tyack 2007) indicating a trade-off between anti-predator strategies and prevention of DCS.

In contrast, it has been theorized that animals with strong site fidelity, or animals with poor body condition, may not avoid an area with an acoustic stressor, such as MFAS, if their foraging habitat is limited to the exposed area or if the trade-off of additional foraging is worth the increased risk (Forney et al. 2017; Frid and Dill 2002; Siegal et al. 2022). However, most of the northern bottlenose whales tagged during the 3S studies demonstrated behavior assumed to indicate an anti-predator response, regardless of body condition (Miller et al. 2015; Siegal et al. 2022; Wensveen et al. 2019). In the Bahamas, it has been suggested that the Blainville's beaked whale population at AUTEC acts as a sink for the greater region, as more females without calves were observed compared to a less noise-impacted area at the nearby island of Abaco (Claridge 2013; Moretti 2019). Prey availability and richness seem comparable across these two areas (Benoit-Bird et al. 2020), which seems to indicate that differences in prey abundance alone are not a driving factor in the continued use of the range. However, in multiple surveys prey availability did match the areas of highest beaked whale foraging activity at AUTEC (Benoit-Bird et al. 2020; Hazen et al. 2011), indicating there is a relationship between beaked whale presence and prey abundance. Similarly, the area of preferred foraging habitat on the SOAR range in Southern California also seems to have a higher concentration of nutrient rich prey (Southall et al. 2019) than in adjacent areas. In fact, Southall et al. (2019) estimated that Cuvier's beaked whales foraging in their preferred habitat on the SOAR range would only need to conduct a mean of 6 dives per day to reach their energetic requirements, while Schorr et al. (2014) estimated that population conducts at least 7–8 dives per day, therefore possibly exceed their daily requirements. However, Cuvier's beaked whales that have been tagged near the SOAR range have demonstrated they are capable of and frequently conduct long-distance movements away from the range as part of their normal behavioral repertoire; therefore, they are also not restricted to that high impact foraging area (Schorr et al. 2014). Furthermore, they often return to the area after these long-distance excursions, and many individuals have been photographically recaptured over several years (Curtis et al. 2021; Falcone et al. 2009). Recent abundance estimates of this population, based on 11 years of mark-recapture data, put the number at 121 animals (71–219), with this being likely a portion of the overall population of the California Current Ecosystem, with a survival rate of 95% and calf percentage of 4.7–6% (Curtis et al. 2021). Although the annual rate of change was estimated as −0.8%, the survival estimate of this population is higher than that for a population in the Canary Islands where MFAS use was banned (Reyes Suárez 2018) and is comparable to the estimate at Abaco in the Bahamas (Curtis et al. 2021; Claridge 2013). However, the percentage of calves in the SOAR population is low (as is the case in Abaco), thus resulting in the negative overall growth rate.

Therefore, in all of these disparate beaked whale populations, while an immediate change in foraging dive behavior may indicate an anti-predator response that may continue for several hours, the animals continue to return to these habitats to forage. This may be due to high concentrations of preferred prey, or other as-yet undefined habitat characteristics. While these data demonstrate individual responses, and in some cases repeated exposures of individuals that could result in multiple lost foraging opportunities, population-level impacts have not yet been conclusively identified. In areas where prey concentrations are rich and estimations of the number of foraging dives needed in a day to reach energetic requirements are lower than the typical number of dives actually conducted per day (e.g., Benoit-Bird et al. 2020; Southall et al. 2019), it is difficult to say what might be the population level impacts of missed foraging opportunities. For example, for Blainville's beaked whale range-associated populations (e.g., Henderson et al. 2019, 2022a; Joyce et al. 2019; Manzano-Roth et al. 2016; Martin et al. 2020), it could be that the typical avoidance behavior observed on naval ranges during training activity are learned anticipatory responses rather than a subsequent response after an exposure. This may be why those responses are initiated at the start of activity, even before the onset of MFAS, and routinely last for up to 3 days past the activity. These generalized avoidance responses could indicate moves to comparable (or better, or worse) prey patches but not be hallmarks of lost foraging opportunities. Alternately, as suggested for AUTEC, high prey concentrations could lead to a continued use of the range, even though there are population-level impacts (Claridge 2013; Benoit-Bird et al. 2020; Moretti 2019), creating a local population sink that is continuously refilled by the larger regional population. The ongoing long-term efforts at these locations will hopefully provide answers to these questions as more photo-id and dive data are collected. Furthermore, there is ongoing effort to use these data in bioenergetic models that can then be utilized in Population Consequences of Disturbance (PCOD) models (e.g., Czapanskiy et al. 2021; New et al. 2013). PCOD models integrate what is known about life history demographics, behavior, and resources, along with impacts to individuals resulting from behavioral responses, such as missed foraging opportunities or poor body condition, to estimate the impacts of disturbance at the population level. Several PCOD studies are ongoing for beaked whales and other cetaceans to evaluate population level effects of noise exposures (e.g., Booth et al. 2020, 2022; Pirotta et al. 2018; Pirotta 2022).

## Summary and Conclusions

Advances in biologging and PAM capabilities, coupled with concerns over beaked whale stranding events, have led to a focus on beaked whale occurrence, distribution, habitat use, and behavior. Researchers have sought to understand how and why beaked whales respond to MFAS, and how these responses could lead to a stranding. When exposures occur during deep foraging dives, the primary response by beaked whales is to cease foraging but remain at depth while moving away, sometimes vigorously, from the direction of the source. Beaked whales exposed to MFAS

during other phases of their dive cycle may also dive deep to avoid the source or may remain at the surface or at intermediate depths while moving away. The avoidance behavior may last for hours as an immediate response, or for days as a general avoidance of an area, while a return to foraging behavior and resumption of the typical dive cycle may take several hours, potentially leading to missed foraging opportunities. These responses seem conserved across the few species for which data have been collected. The general response also appears consistent for individuals with frequent exposures to MFAS and other anthropogenic noise as well as for individuals found in more acoustically pristine environments. The continued work with real naval sonar vessels, exposures to other MFAS sources, exposures from multiple sources simultaneously, and with additional species are all ongoing and will provide further critical insight into the nuances of these complex behavioral responses.

While our knowledge on beaked whale baseline behavior and behavioral responses to MFAS have been greatly expanded in the last two decades, there is still much to learn. More controlled exposures of different real-world MFAS sources at varying distances are needed in order to truly characterize the separate but co-mingled impacts of source type, received level, and source distance on the strength and type of responses observed. Better characterizations of prey fields both in- and outside of areas of repeated exposures would help elucidate whether movements to other areas truly represents lost or reduced foraging opportunities or could simply find whales competing for resources in a crowded prey field. Additional physiological studies in beaked whales and other deep-diving odontocetes that investigate how behavioral responses and stress hormones may impact or mediate gas exchange and nitrogen uptake is critical in furthering our understanding of DCS in these species. Collaborations among researchers in different areas of expertise as has been done in many of the above studies is the best way to continue moving the needle forward on our understanding of the impacts of MFAS on beaked whales.

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# Impact of High Marine Traffic on Harbor Porpoise: Effect on Abundance and Distribution

75

Georg Nehls, Ramunas Zydelis, Rainer Matuschek, Miriam Brandt, Ansgar Diederichs, Caroline Hoeschle, and Frank Thomsen

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**Abstract**

Increasing evidence documents effects of impulsive underwater noise on hearing and behavior of marine mammals, but knowledge about the impacts of extensive or chronic exposure of cetaceans to continuous noise from shipping is still scarce. The distribution of harbor porpoises (*Phocoena phocoena*) was analyzed in relation to shipping noise and other environmental variables in an area with intense marine traffic, the Fehmarnbelt in the Baltic Sea. Autonomous noise recorders were deployed at 19 positions within the Fehmarnbelt between September 2009 and November 2010 (bandwidth = 16 Hz–16 kHz). Harbor porpoises were surveyed from March 2009 to October 2010 using aerial visual line transect surveys (altitude = 76 m and 250 m). The acoustic data were analyzed using 10 s intervals (CI = L5, L50 and L95). Noise maps were derived using AIS data for each month in 2010 in a grid of 750 × 750 m cells and applying an empirical geometrical spreading loss model. Ambient noise in the Fehmarnbelt was dominated by shipping noise (median SPL: 109–132 dB re 1μPa), and noise levels found are typical for busy waterways. Distribution modelling was used for analyzing correlations between environmental variables and porpoise densities. The best fitting distribution model included variables characterizing water temperature at the bottom, water depth, and underwater noise; plus, temporal variable as survey year and season were also important. The model shows porpoise densities declining with increasing median sound levels to about 130 dB re 1 μPa. Thus, median shipping noise levels significantly affected the distribution of harbor porpoises in the study area, but little habitat displacement appeared as a result. This finding might reflect habituation and a trade-off, where the benefits for the porpoises to stay in the study area outweigh the disruptions of disturbance due to underwater noise exposure.

## Introduction

Growing awareness about the diverse effects of underwater noise in the world's oceans have stimulated a wealth of studies into assessing and regulating underwater noise in order to mitigate significant environmental impacts (see reviews by, e.g., Hildebrand 2009; OSPAR Commission 2009; Williams et al. 2015). Underwater noise may have harmful impacts on aquatic animals including cetaceans through interfering with their acoustic communication, orientation, and prey capturing abilities, and – at high doses – might even lead to auditory damage (Richardson et al. 1995; Southall et al. 2007; National Marine Fisheries Service 2018). Commercial shipping is a very widespread and important source of anthropogenic underwater noise. Noise propagates well under water and the noise emissions of a vessel may exceed ambient noise levels at distances of several 10 km. Although ships are not the loudest sources of underwater noise, their

sheer abundance makes shipping noise a very important source of anthropogenic noise in the oceans (Hildebrand 2009; OSPAR Commission 2009; United Nations 2018). Ambient noise levels have been rising in some regions along with an increase in shipping activity (Anderson et al. 2008; Andrew et al. 2011; Miksis-Olds et al. 2013; Kaplan and Solomon 2016). In some areas, cetaceans are frequently exposed to noise levels, which may elicit behavioral responses (see reviews by Clark et al. 2009; Hildebrand 2009; OSPAR Commission 2009; Erbe et al. 2012, 2019).

The concerns of potential ecosystem implications of rising levels of ambient noise due to shipping have found its way into regulation with the EU Marine Strategy Framework Directive (MFSD) (European Commission 2008, 2010; Tasker et al. 2010). The MSFD requires each EU member state to achieve "Good Environmental Status" (GES) in their marine waters by 2020. Trends in continuous low frequency noise as emitted by shipping is one of the indicators used to assess the GES and as of 2010, all member states have to systematically monitor this type of anthropogenic noise to assess GES and then, based on the initial assessment, define and decide on principles and objectives for GES. In the same context, the International Maritime Organization (IMO) has issued guidelines for the reduction of underwater noise from commercial shipping (International Maritime Organisation 2014). The IMO guidelines are non-mandatory but will likely influence regulation in the future (see United Nations 2018).

Disturbance from underwater noise may have complex direct and indirect effects on marine mammals, ranging from stress (e.g., Rolland et al. 2012) to habitat displacement (Rako et al. 2013). There is growing interest in analyzing how and under which circumstances this might affect their abundance (Bejder et al. 2006; Lusseau and Bejder 2007; Senigaglia et al. 2016) and cause impacts on population level (Nabe-Nielsen et al. 2018; Pirotta et al. 2018; Mortensen et al. 2021). It is an implicit assumption that increasing ship traffic and the resulting noise beyond a certain level and intensity creates conditions that could lead to population consequences. However, as even very busy waterways may still be of high importance for cetaceans (McClellan et al. 2014), there is likely no simple relationship between noise levels and cetacean abundance, distribution, and population level effects. Behavioral reactions of cetaceans to anthropogenic underwater noise are highly variable, ranging from attraction and indifference through short-term changes in behavior to short- or long-term avoidance, which could lead to long-term habitat displacement (Richardson and Würsig 1997; Nowacek et al. 2007; Senigaglia et al. 2016). Rather than simply loudness, further characteristics of a noise signal, such as its duration, its abruptness, its frequency spectrum, and the signal-to-noise ratio play an important role in stimulating an animal's response (Southall et al. 2007).

For harbor porpoises (*Phocoena phocoena*), the onset of individual responses to various noise sources was shown at widely different broadband noise levels: While harbor porpoises responded to low-frequency pile driving at sound exposure levels of 143 dB re 1 $\mu Pa^2$ s (Brandt et al. 2016), a much lower threshold of 119 dB re 1 $\mu Pa^2$ s was found when studying seal scarer noise (Brandt et al. 2013). Even lower response thresholds were shown for the reaction of captive porpoises to stimuli in their best hearing ranges (Kastelein et al. 2017; Kok et al. 2018). Tougaard et al. (2015) conclude that responses to underwater noise relate to frequency-specific

hearing abilities of cetaceans and predict a response when noise levels exceed the hearing threshold by about 45 dB. Similarly, Dyndo et al. (2015) emphasized the importance of the high-frequency components of vessel noise in eliciting avoidance reactions in harbor porpoises.

Avoidance reactions of cetaceans to shipping noise have been demonstrated mainly in relation to relatively small vessels such as whale-watching boats (Erbe 2002; Bejder et al. 2006; Gervaise et al. 2012), leisure crafts (Piwetz et al. 2012; Rako et al. 2013), and speed boats (Ng and Leung 2003; Li et al. 2015). Avoidance reactions of marine mammals to ship noise were found to increase with vessel speed (Richardson et al. 1995; Oakley et al. 2017; Wisniewska et al. 2018). While vessel speed usually correlates with increasing noise level, the movements of the ships and the perceived abruptness of the noise also seem to play a role in eliciting avoidance responses (Ellison et al. 2018). The types of observed responses range from rather subtle behavioral changes to long-term avoidance of disturbed areas.

Most studies conducted so far have looked at direct responses and short-term effects of anthropogenic noise on marine mammals (but see Bejder et al. 2006; Lusseau and Bejder 2007), and modern biologging techniques provide detailed insights into individual responses to underwater sound (van Beest et al. 2018; Wisniewska et al. 2018; Mikkelsen et al. 2019; Martin et al. 2021). Recent studies indicate that shipping noise may affect physiological status (Dey et al. 2019), habitat choice (Lin et al. 2020), and populations (Wang et al. 2020) of cetaceans in estuarine habitats, where animals are more directly exposed to commercial shipping, but still very little is known on the impacts of extensive or chronic exposure of cetaceans to underwater noise from commercial shipping in the sea. The noise emissions from commercial vessels are more intense than those of whale-watching and leisure boats, but of lower frequencies (Richardson et al. 1995; McKenna et al. 2012; Southall et al. 2017). The course of commercial vessels might be more predictable for cetaceans as compared to leisure crafts. Assessing the impact of commercial shipping on cetacean abundance needs to analyze cetacean abundance and their exposure to shipping noise on larger scales and cannot rely only on direct observations (Merchant et al. 2017). The exposure of cetaceans to shipping noise can be assessed by relating cetacean abundance and distribution to underwater noise maps. The global AIS system provides detailed online data on shipping and allows generating noise maps accounting for transmission loss in relation to environmental parameters such as bathymetry, sediment, temperature, and salinity (Maglio et al. 2016; Neenan et al. 2016; Sigray et al. 2016; Bagočius and Narščius 2018; Sertlek et al. 2019; Farcas et al. 2020). It is thus intriguing to use such noise maps, which can be easily produced – even in real time (Nehls et al. 2022) – for the assessment of potential noise impacts on marine mammals (Erbe et al. 2012; Merchant et al. 2014). However, while the exposure of cetaceans to extensive or chronic shipping noise has been mapped in different attempts (e.g., Erbe et al. 2012; Redfern et al. 2017), there is very little knowledge on how mapping of noise levels can be used to predict effects on distribution and abundance of cetaceans. In order to understand the consequences of globally increasing ship noise, this knowledge is an essential step.

The harbor porpoise is a small cetacean species, which is widely distributed and by far the most abundant cetacean in European waters. Recent numbers for the North

Sea have been estimated at about 345,000 (CV = 0.18, Hammond et al. 2017) with no changes in abundance since 1994 (289,000, CV = 0.14, Hammond et al. 2002) and 2005 (355,000, CV = 0.22 (Hammond et al. 2013). The harbor porpoises occurring in the study area Fehmarnbelt belong to the Inner Western Baltic population, which comprises about 23,000 individuals and has been stable over the last two decades (Hammond et al. 2013). The Fehmarnbelt is a relatively shallow area between the islands Lolland (Denmark) and Fehmarn (Germany), showing water depths up to ca. 35 m. At its narrowest part between the islands, it has a width of about 20 km. The Fehmarnbelt is an area with high levels of marine traffic used by ships travelling from the North Sea through the Kiel Canal and the Great Belt to and from the eastern Baltic Sea. Parts of the Fehmarnbelt have been assessed as being of ecological importance, and several marine protected areas have been included in the European network Natura 2000, including the Special Area for Conservation (SAC) "Fehmarnbelt." This SAC covers 280 $km^2$ in the middle of the Fehmarnbelt, and it overlaps with the east-west going main shipping lane and is crossed by high frequent ferry routes between Germany and Denmark (FEMM 2013).

In the present study, the distribution and the total abundance of harbor porpoises (*Phocoena phocoena*) are analyzed in relation to underwater noise levels in the Fehmarnbelt. As the response of marine mammals to acoustic signals may not be uniform but likely depending on the context of noise exposures (Ellison et al. 2011; Lin et al. 2020), habitat quality is considered as an important factor to be considered, as the extent of a response may differ in relation to habitat quality. Therefore, the distribution of harbor porpoises was modelled accounting for environmental variables including underwater noise levels derived from a simple empirical model.

## Methods

### Characterization of Ship Traffic Through Fehmarnbelt

Ship traffic in the Fehmarnbelt is characterized by mainly two sources: (1) cargo vessels and tankers moving longer distances through the Baltic Sea and (2) passenger ships sailing on two routes between Germany and Denmark.

The most frequented routes are the T-route with 35,000–45,000 (Rambøll 2011) movements annually, mostly comprised of cargo ships, and the ferry route Rødby-Puttgarden with about 35,000 movements annually, which crosses the T-route in the Fehmarnbelt. At the crossing of these two main routes, shipping intensity reaches eight passages per hour. These two routes form by far the most important part of shipping in the Fehmarnbelt study area, although ship traffic additionally occurs along some other minor routes, and the T-route splits to the east and the west of the Fehmarnbelt diverting traffic to different destinations (Fig. 1). Vessel speed in the T-route is mostly between 10 and 20 knots with container vessels usually travelling at 15–20 knots and a substantial portion (41%) of the passenger vessels travelling faster than 20 knots (Rambøll 2011). Within the focal area of this study, the main shipping lanes are situated at water depths of 20–30 m.

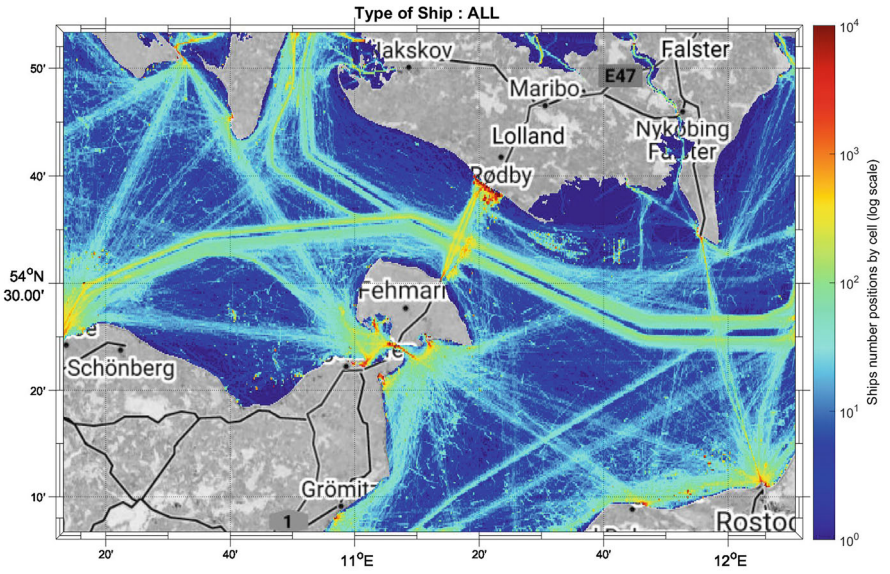



**Fig. 1** Marine traffic lanes in the Fehmarnbelt study area identified from AIS signals

## Noise Measurements

To measure underwater noise, autonomous measurement buoys, capable of recording sound for up to 4 weeks, were deployed at different locations in the Fehmarnbelt.

They were equipped with low-noise hydrophones of type B&K 8106 with built-in preamplifier. Hydrophone receiving sensitivity is nominally −173 dB re 1 V/μPa. Noise recordings cover the bandwidth range from 16 Hz to 16 kHz.

Nineteen buoys were deployed in the Fehmarnbelt (Fig. 2) from September 2009 to November 2010. With the help of a flotation body, the hydrophone was kept approximately 1.8 m above the seabed. Noise measurements were conducted with three buoys at a time. Stations were changed at monthly intervals, but one location (A6) was kept permanently throughout the study period.

Additional noise measurements were carried out during the service trips from aboard the service ship. A portable AIS receiver (automatic identification system) served to record the AIS-telegrams from all ships crossing the Fehmarnbelt. Additionally, sound speed profiles were calculated based on CTD measurements.

The AIS recordings were used to identify ships passing near the measurement buoys (Fig. 3). The combination of this data together with the recorded sound pressure allowed to evaluate the apparent transmission loss for individual ships. Apparent transmission loss is the reduction of ship sound pressure level with increasing distance from the noise source. This experimentally estimated transmission loss served to model the noise propagation and was used to compute noise maps.

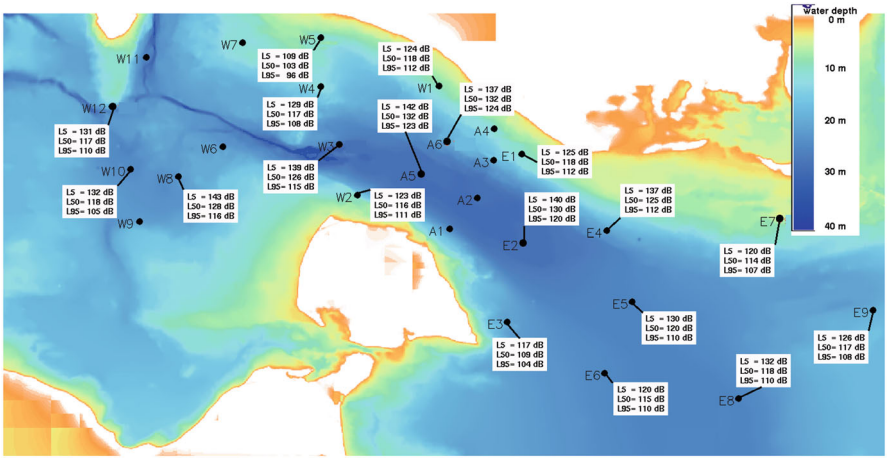



**Fig. 2** Map of the Fehmarnbelt area with color-coded water depths. Noise measurements were conducted periodically, associated to a longer-term C-POD measurement program marked by black dots (FEMM 2013). Noise measurements are labelled with their corresponding percentile levels L5, L50, and L95

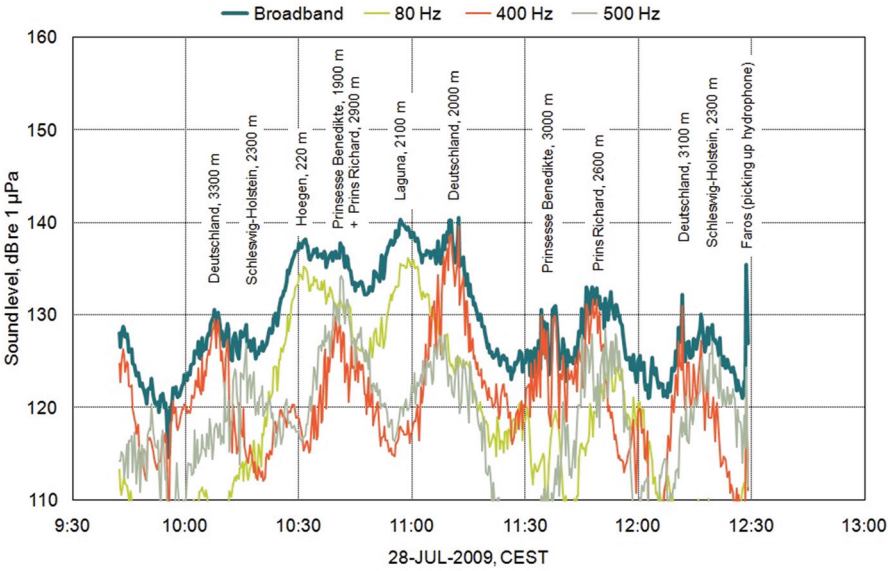



**Fig. 3** Measured sound pressure levels at a central position in the Fehmarnbelt near the ferry route. The name of the ship passing nearest as well as the distance to the hydrophone is given. Levels are given as broadband SPL, and in 1/3-octave bands for 80 Hz, 400 Hz, and 500 Hz, illustrating a variable spectrum of sound frequencies among passing ships

## Data Analysis

First, each recording was subjected to a quality control. Using a calibration signal at the beginning of each recording, the sound pressure level SPL was calculated at intervals of 10 s. After passing the quality check, the data were analyzed in terms of their percentile values (exceedance levels). Calculations were carried out for the median L50 as well as values for L95 and L5. The median sound pressure level is statistically more robust than the mean sound pressure level because the latter is dominated by the loudest period during the measurement and can therefore give a false picture of the overall noise situation. For example, the close passage of a single ship during the 30-day measurement interval may result in a very high mean SPL that does not adequately describe the overall noise situation. The L95 level is the sound pressure level that is exceeded 95% of the time.

It can be interpreted as the lower limit of the permanent ambient noise level during the measurement period at this location. Accordingly, L5 is the sound pressure level that is exceeded during 5% of the measurement time and thus contains information about the highest levels occurring. The difference between L5 and L95 thus characterizes the overall variability of the noise situation. Figure 3 shows the deployment positions for the noise data loggers together with the exceedance levels evaluated there.

A spectral analysis of the recorded ambient noise at position A6 in the loudest part of the Fehmarnbelt is shown in Fig. 4. The form of the 1/3-octave spectrum between 300 Hz and 16 kHz is very similar to the noise radiated by typical merchant ships as given by Wales and Heitmeyer (2002). It can be seen that below 300 Hz, the spectrum of the typical ship noise is higher than that of the measured median ambient noise. Measured spectral components below 100 Hz show the typical drop towards lower frequencies due to the cutoff frequency resulting from the water depth (see Urick 1983). Also shown in Fig. 4 is the hearing threshold of harbor porpoises as given by Kastelein et al. (2002). Extrapolating this threshold towards lower frequencies shows that the ambient noise, i.e., shipping noise, is below the detection threshold at about 250 Hz. Although frequency components below 250 Hz contribute significantly to the broadband level, they are not audible to harbor porpoises.

## Modelling Noise Maps

A noise map was created by relating shipping frequency to measured noise levels for the study area. AIS data for the Fehmarnbelt region were taken from Ramböll (2011) for each month in 2010. The grid for the shipping density consists of 750 × 750 m cells. Only cells containing at least 30 ships passing through per month were included for analyses and were treated as sound sources for the other cells. The source level of those sound emitting cells was computed as a fixed level + 10*lg (No. of ships). Ship type and speed was neglected in this simple model as this was already integrated in the noise measurements. The constant part (fixed level) of this source level was evaluated by comparison of the sound measured at a suitable

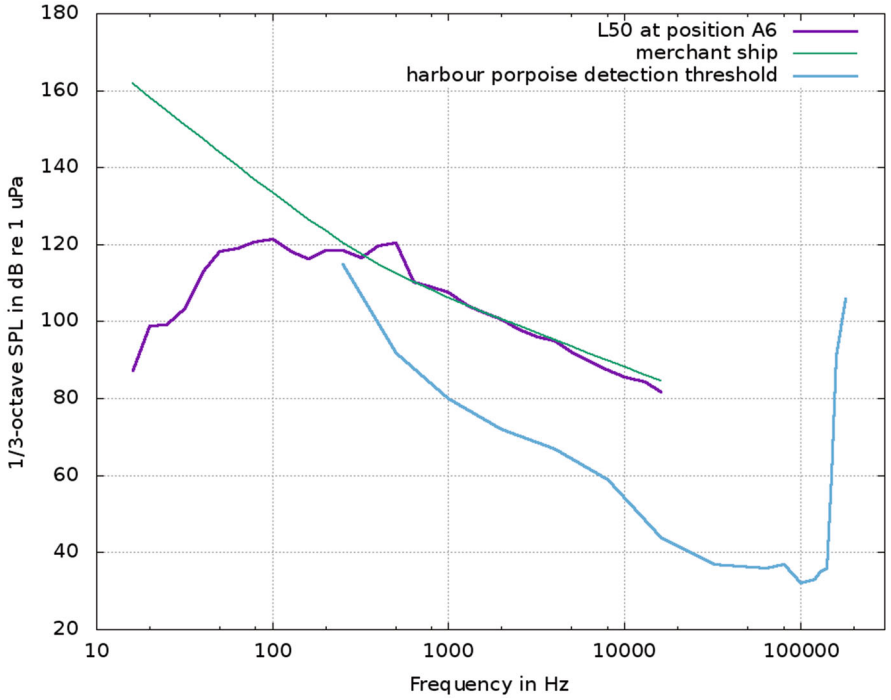



**Fig. 4** Typical 1/3-octave spectrum in the noisiest part of the Fehmarnbelt. A typical merchant ship spectrum as given by Wales and Heitmeyer (2002) shows good qualitative similarity above 350 Hz. Extrapolation of the harbor porpoise detection threshold given by Kastelein et al. (2002) towards lower frequencies will cross the ambient noise spectrum at about 250 Hz

reference station and the modelled sound pressure at this position in an iterative procedure. Station A6 was chosen for reference, as it is located directly in the shipping lane and showed the highest noise levels with L50 equal to 132 dB re 1 μPa. The constant source level per ship was evaluated to 164 dB re 1 μPa. Since no AIS data on ship type, length, or speed was available, this source level entered the model independent of those parameters. Omitting cells with less than one ship per day leads to a slight underestimation of the noise levels, but this is about 30 dB below the source levels neglected.

The decrease of the noise level as a function of distance r to the source gives the transmission loss (TL). The transmission loss usually is of the form TL(r) in dB = k * log(r/r0). Here, k is the propagation constant. Analysis of noise recordings of individual ship passages that were identified by AIS messages yielded a propagation constant of k = 22 for the model. This is similar to results obtained by van der Schaar et al. (2017).

For sound propagation in very shallow areas with 10 m water depth or less, a value of 25 was chosen for the propagation constant. This takes into account that the so-called cutoff frequency, below which sound propagation strongly diminishes,

rises with decreasing water depth. Moreover, there is a higher propagation loss because of the increase in reflections from the sediment and the water surfaces.

The influence of the sediment type on sound propagation was considered as well. Sediment data (grain size) was provided by Ramböll (2011) with the same resolution as the AIS data. The propagation loss due to absorption of sound in the seabed, especially in muddy regions is not negligible (see for example Urick 1983). This effect was incorporated in the noise maps. Bottom attenuation in dependence of grain size was modelled using data for the attenuation from Kibblewhite (1989). In summary, the model is based on the shipping density, the water depth of the source and immission cells, and the grain size of the seabed materials along the propagation path. This yields effective propagation constants k between 14 and 16 depending on the environmental conditions.

## Porpoise Surveys

Harbor porpoises were surveyed from March 2009 to October 2010 using visual aerial surveys of two designs: mammal-only survey flights (flown at 250 m altitude) and bird/mammal survey flights (flown at 76 m altitude). Each dataset was treated separately, and porpoise sightings were adjusted for availability and detection biases (see details in Grünkorn et al. 2005). After correcting for uncertain detection (availability bias and perception bias) and calculating porpoise densities, both datasets were combined, and survey effort data were prepared by splitting transects into approximately 1 km segments. Each transect segment was associated with temporally and spatially matching environmental variables. For detailed description of data collection and treatment, see FEMM (2013).

Porpoise observations were numerous in summer months and few in winter, therefore two seasons were defined in the data: summer (April to August) and winter (remaining months). Also, survey year was used as another temporal factor variable.

## Modelling Porpoise Distribution

Distribution modelling was used for analyzing correlations between environmental variables and porpoise densities. A number of environmental variables characterizing the general water environment and specific anthropogenic pressures were considered in the model. Hydrodynamic variables were derived from a hydrodynamic model developed by DHI specifically for the study area and the study period (FEHY 2013). While the number of variables that can be derived from the hydrodynamic model is very large ($>100$), we focused on the most feasible parameters: current flow describing variables (U, V, and W components of current velocity), frontal characteristics (pycnocline strength, frontal strength, and eddy activity), and water mass patterns (salinity, temperature, and density). Two sets of hydrodynamic data were considered: variables characterizing water environment near the bottom and at the surface. Water depth and seabed sediment grain size were included as static habitat

characterizing variables. Finally, four variables were used describing anthropogenic pressures: distance to land, distance to wind farms, distance to shipping lanes, and underwater sound. Estimated numbers of harbor porpoises were used as a response variable and environmental variables were used as predictors.

Generalized additive modelling (GAM) with a log link and Tweedie error distribution was used in order to account for overdispersion in the data (Candy 2004; Shono 2008). Initially, all candidate variables were fitted in single variable models and only those that were significant were further included in the full model. Further, variables were eliminated from the full model in a stepwise manner based on their significance: nonsignificant variables were excluded. A priori, decision was made to keep an interaction term between coordinates (Easting, Northing) and factor variables identifying year (2009 or 2010) and season (winter or summer) in the model. The permitted degrees of freedom were restricted to maximum 5 ($k = 5$) for continuous variables and 20 ($k = 20$) for the interaction term between coordinates. Fitting the Tweedie distribution, modelling requires selection of an index parameter determining the type of the distribution. Values between 1 and 2 were used as they fit a Poisson-gamma distribution. The value was chosen by changing it and refitting the models in order to find the index maximizing the deviance explained by the model. Predictor variables entering the final model were checked for collinearity in order to make sure that model results are not distorted by highly correlating variables (Dormann et al. 2013). Potential residual autocorrelation in the final model was tested using a correlogram. The generalized additive model (GAM) was fitted using the "mgcv" package (Wood 2015) in R v.3.4.3 (R Core Team 2017).

The fit of the final model was assessed by the deviance explained and used for creating porpoise distribution map. For mapping, the prediction grid with a resolution of 750 × 750 m grid cells was generated. The central point of each cell was associated with values of each variable averaged for the summer 2010 season. The final model was used to predict porpoise density values for each grid cell.

## Results

### Noise Maps

The exploratory analysis (see Fig. 3) showed that shipping noise dominates ambient noise levels in the Fehmarnbelt. Natural sources of sound like wind, breaking waves, and rain did not contribute much to the overall measured broadband noise levels. The noise map in Fig. 5 illustrates how noise propagations varies within the study area and provides a snapshot of the sound field produced by 19 ships based on a real AIS snapshot. All vessels were assigned a source level of 164 dB.

This simulation was carried out to illustrate how the noise maps were created. The noise sources (ships) are clearly delineated as hot spots (red). However, the resulting noise levels do not only decrease on concentric circles around the ships. Due to the different bathymetry and bottom structure, the transmission loss is strongly

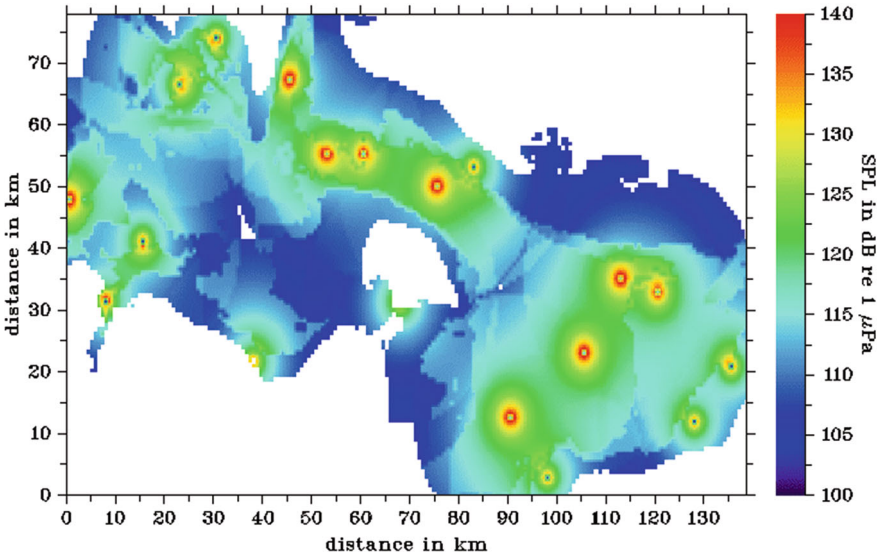



**Fig. 5** Snapshot noise map of the underwater broadband levels generated by 19 ships in the Fehmarnbelt area where high noise levels are shown in red and low noise levels in blue

dependent on the propagation path. As a result, regions with very shallow water and regions with muddy seabed sediments have lower sound levels.

The pixels where the ships are located are not part of the color code since the source level of 164 dB inside those cells clearly exceed 140 dB. Color coding starts at neighboring cells only. Their bathymetry and sediment play a crucial role in the appearance and size of the ships noise footprints.

A summarized median noise map of the mean shipping density for the summer of the year 2010 is shown in Fig. 6. For the reasons given above on sound propagation, this map is not just a reflection of the shipping density. There are areas west and east of Fehmarn with highly absorbing seabed sediments, which lead to lower sound pressure levels in the water column. Especially along the Kiel-Baltic Sea route, the model predicts variations of sound levels of 10 dB and more, primarily due to variable seabed conditions.

The quietest areas appear as the very shallow waters of the Rødsand lagoon, the coast east of Falster, and the German waters close to the island of Fehmarn and the mainland.

The modelled median noise levels show a close correlation to the noise levels measured at 19 locations at monthly intervals (Fig. 7). The deviation from measured noise levels probably arises from some temporal variation and a spatial mismatch of the model grid and the measurement locations. While the measurements at location A6 were conducted throughout a full year, and these measurements were taken as basis for the modelling, all other measurements were conducted for 1 month. There is likely some uncertainty arising from the fact that modelling is based on AIS data

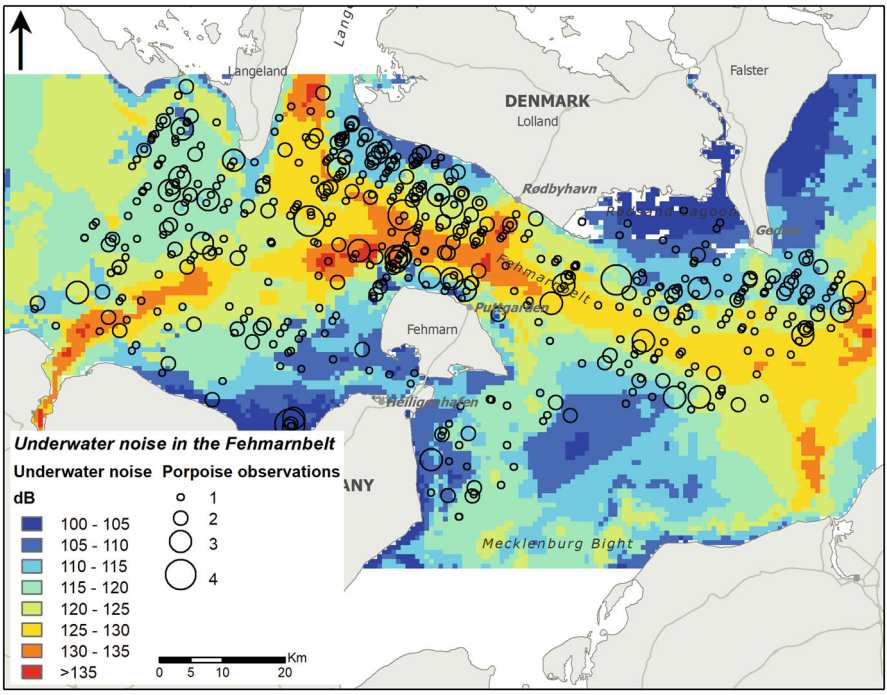



**Fig. 6** Underwater median noise map (L50) of the Fehmarnbelt with porpoise sightings during aerial surveys conducted in 2009 and 2010

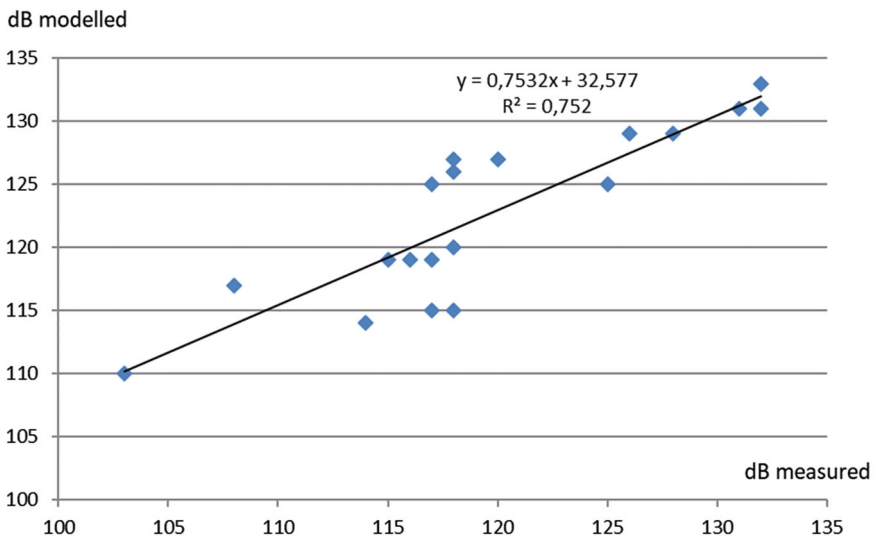



**Fig. 7** Correlation of modelled noise levels versus measured noise levels

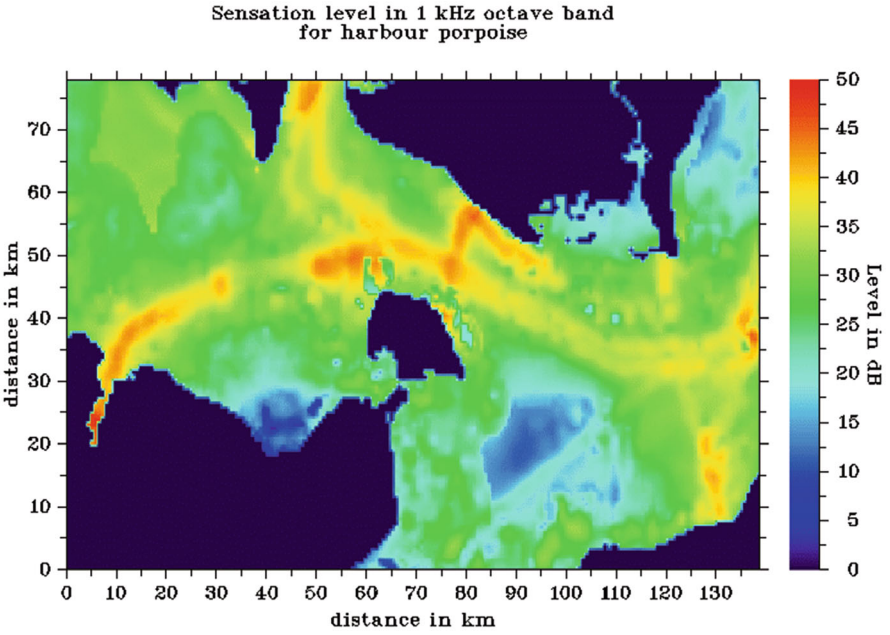



**Fig. 8** Noise map of harbor porpoise sensation level (L05) for the 1 kHz octave band for the year 2010

and thus excludes smaller leisure crafts, which have no obligation for AIS recordings. However, noise measurements indicated little seasonal variation, and as noise emission from smaller crafts are much lower as compared to commercial shipping, the overall contribution to the noise budget in the Fehmarnbelt is considered as low.

In order to highlight areas of noise levels, which might elicit disturbance responses from harbor porpoises, model noise levels for the octave band 1 kHz were calculated based on the measured 1/3-octave percentile spectra and the broadband noise propagation. For 1 kHz, the algorithm had to be modified since this frequency corresponds to a wavelength of roughly 1.5 m. At such a short wavelength, the influence of water depth on sound propagation can be neglected. The L05 noise levels for the 1 kHz octave band forms the basis for the calculation of a harbor porpoise sensation level map at this frequency (Fig. 8). Sensation level means the difference between the determined noise level in a certain frequency band and the hearing threshold of cetaceans in that band, which is set to 80 dB re 1 µPa at 1 kHz (Kastelein et al. 2002). Tougaard et al. (2015) hypothesized that noise levels exceeding the hearing threshold of cetaceans by 40 dB to 50 dB would elicit an avoidance reaction. While median noise levels do not exceed this value even at the loudest station A6 (Fig. 4), the orange to red areas of the sensation level noise map (Fig. 8) indicate those parts of the Fehmarnbelt, where such levels are exceeded in 5% of the time, so that frequent disturbance of harbor porpoise might be expected.

## Harbor Porpoise Distribution Modelling

Plotting sightings of porpoises over the noise map of the Fehmarnbelt area indicates that porpoises occur in areas distinguished by generally high underwater noise levels as well as in more silent areas. However, there was a tendency that porpoises were less abundant in the noisiest zones (Fig. 6).

Following the model selection procedure, the best fitting model included variables characterizing water temperature at the bottom, water depth, and underwater sound in addition to variables that were forced into the model: coordinates, survey year, and survey season (Table 1). All these variables were highly significant and deviance explained by the model was 10.3%. Autocorrelation function showed that no systematic correlation was left in the model residuals.

Variable response curves showed that harbor porpoises preferred water temperatures between 5 °C and 15 °C, water depth below 10 m, were more abundant in 2010, and animal numbers were higher in summer season as compared to winter. The shape of the response curve representing underwater sound indicated that porpoises generally occur in noisy environments and their density does not seem to be affected by noise up to levels of about 110 dB. At louder noise levels, the response curve decreases, indicating that porpoise densities decline with increasing sound levels starting from 115 to 130 dB (Fig. 9).

The map of predicted harbor porpoise densities created using the best fitting model shows that harbor porpoises were most abundant in the northwestern corner of the study area around southern Langeland and in the western part of the Fehmarnbelt between Fehmarn Island and the coast of Lolland, Denmark, except for the deepest part (Fig. 10).

To further evaluate possible effects of median noise levels on harbor porpoise abundance, we tested whether harbor porpoises seem to deselect noisy areas relative to their proportional extension. Application of the Jacob's selectivity index (Fig. 11) indicates that areas with median noise levels below 110 dB were strongly deselected, while areas with median noise levels between 110 dB and 125 dB were selected. Areas with relatively high median noise levels between 125 dB and 130 dB were

**Table 1** Parametric coefficients and approximate significance of smooth terms of the harbor porpoise distribution model

| Parametric coefficients | Estimate | Std. error | T value | Pr(>\|t\|) |
|---|---|---|---|---|
| Intercept | −1.843 | 0.099 | −18.684 | <0.001 |
| Year (2010) | 0.757 | 0.125 | 6.063 | <0.001 |
| Season (winter) | −0.959 | 0.168 | −5.703 | <0.001 |
| Approx. significance of smooth terms | edf | Ref.df | F | p-value |
| s(Easting, Northing) | 8.902 | 8.998 | 7.519 | <0.001 |
| s(Temperature) | 2.933 | 2.997 | 13.008 | <0.001 |
| s(Water depth) | 2.939 | 2.997 | 7.422 | <0.001 |
| S(Underwater sound) | 2.969 | 2.999 | 7.152 | <0.001 |

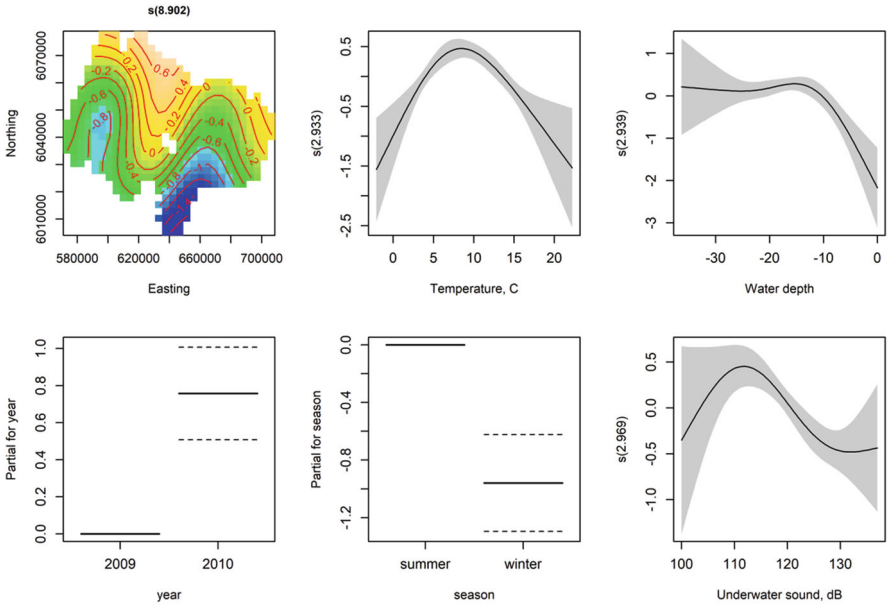



**Fig. 9** GAM response curves for the model of harbor porpoise distribution in Fehmarn Belt. The degree of smoothing is indicated in the legend of the Y-axis and for the interaction terms (Easting, Northing) in the heading. The gray-shaded areas and the dotted lines (for factor variables) show the 95% confidence intervals

deselected but no deselection signal occurred at highest levels of 130 dB to 135 dB, which are found in the main shipping lanes (see Fig. 6).

## Discussion

The results of this chapter show relatively high median noise levels within the Fehmarnbelt, which are typical for busy waterways across Europe and elsewhere. Median sound pressure levels fell between 103 dB and 132 dB re µPa with highest levels found where the main shipping lane T-route crosses the ferry route between the islands of Lolland, Denmark, and Fehmarn, Germany. Median noise levels along the shipping lanes were about 30 dB higher compared to areas at further distance to the shipping lanes. Noise levels similar to those observed in the Fehmarnbelt are found all along the main shipping routes in the Baltic Sea between Russia and Kiel Canal (Folegot et al. 2016). Measurements in the German Baltic Sea for the EC-Life + project BIAS showed average 1/3 octave sound pressure levels between 97 dB and 129 dB re 1 µPa at center frequencies of 63 and 125 Hz (Folegot et al. 2016) with highest levels close to main shipping routes. These results are in the same range compared to those presented here. The differences between methodologies (i.e., 1/3 octave band levels at certain center frequencies vs. median broadband) would in any

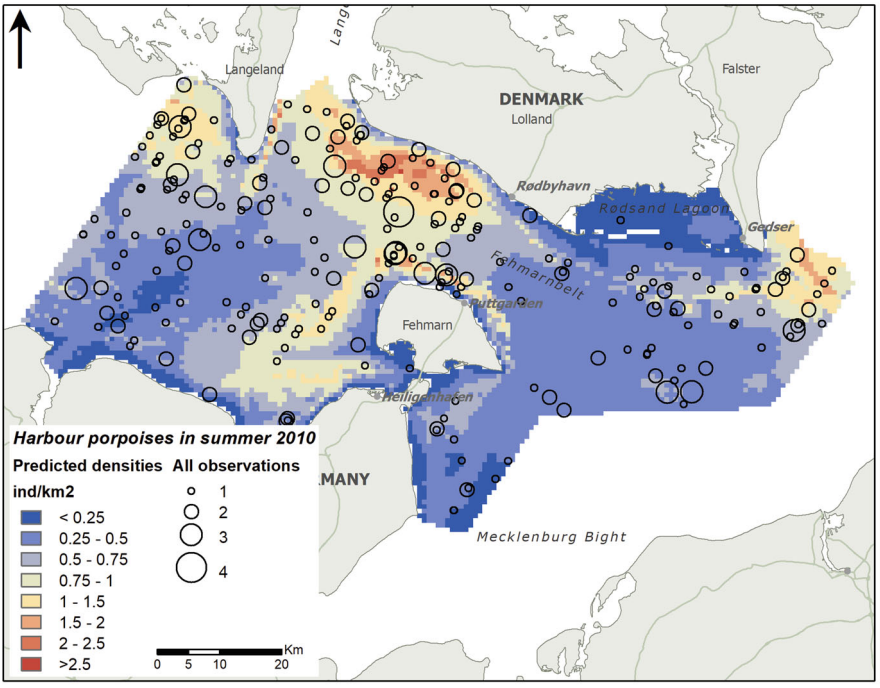



**Fig. 10** Predicted harbor porpoise density distribution in the Fehmarnbelt in summer 2010 with overlaid all porpoise observations recorded during aerial surveys

case result in values of the present study to be slightly higher compared to BIAS data, which is the case. Even considering methodological differences, the sound levels recorded in the Fehmarnbelt are similar to those reported from other areas with similar shipping activity. For example, Bailey et al. (2010) noted mean background noise levels between 104 and 119 dB re 1 μPa in a marine protected area off the coast of Scotland, rising to 138 dB re 1 μPa during times of high shipping activity (30 s average, recorded frequency range: 10 Hz–120 KHz). Nedwell et al. (2007) measured ambient noise at five offshore wind farm locations off the UK East and West Coast. They found levels to be as high as 132 dB re 1 μPa (recording range: 10 Hz–120 kHz; see also Merchant et al. 2017). The higher recording bandwidth in these latter two investigations probably resulted in broadband levels being somewhat higher compared to those of the Fehmarnbelt.

Porpoise density ranges in the Fehmarnbelt correspond well with those observed in areas of the Belt Sea population in general, although species abundance decreases along a north-south gradient leading to low densities towards the southeast. The predicted porpoise distribution map corresponds well with observations of harbor porpoises during the surveys and matches the hotspot area for the species identified by Sveegaard et al. (2011, 2018). Similar to other investigations, it was found that porpoise occurrence relates to habitat parameters. Harbor porpoise did not distribute

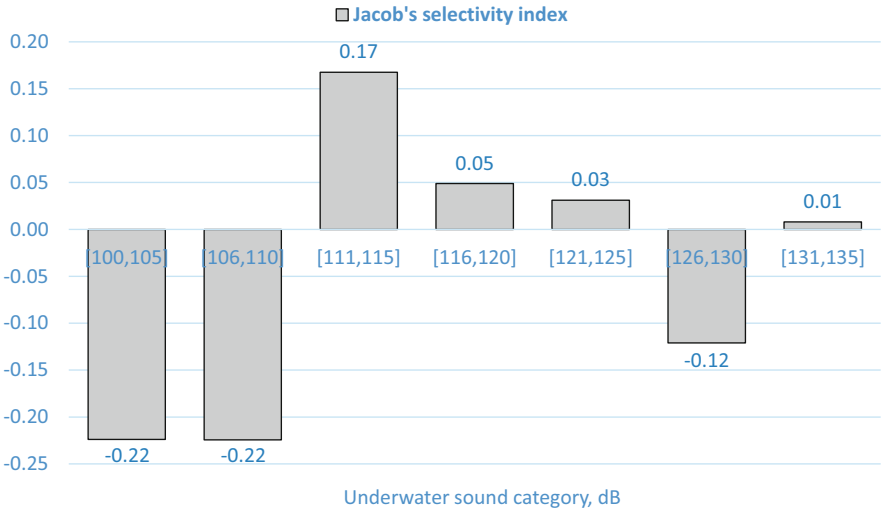



**Fig. 11** Harbor porpoise selectivity indices in the Fehmarnbelt for areas related to median noise level (Jacob's selectivity index)

evenly within the study area, and mean summer densities varied from below 0.1 to about 2 harbor porpoise/km$^2$ indicating substantial differences in habitat preferences within the area. The best fitting model describing harbor porpoise distribution in the Fehmarnbelt included parameters describing bottom current component U (though not significant as single factor), water temperature at the bottom, water depth, and underwater sound, and furthermore coordinate variables, which were forced into the model. The importance of depth and water temperature in the porpoise distribution model is consistent with results of other authors, who also found these variables to be significant (Hammond et al. 2002; Isojunno et al. 2012; Gilles et al. 2016; Lambert et al. 2016). These variables generally indicate similar patterns, suggesting that porpoises in the Fehmarn Belt areas prefer relatively shallow waters, but deeper than 10 m, and coincide in higher presences with water temperatures between 5 °C and 15 °C. In the context of this study, temperature also describes the seasonal pattern of porpoise abundance, as many animals leave the area during winter, and does not only reflect spatial variation within the area. Some authors found porpoises preferring deeper places, e.g., in Scotland, where preferred depth range was between 50 and 150 m (Booth et al. 2013). Other studies conducted over a larger spatial scale (Embling et al. 2010; Lambert et al. 2016) and/or in areas where tides and currents are pronounced (Johnston et al. 2005; Skov and Thomsen 2008; Booth et al. 2013; Jones et al. 2014) found hydrographic parameters such as current strength significantly influencing porpoise distribution. This was also the case in this study, although not very pronounced. This may be due to less variance in hydrodynamic parameters and stronger variance of other parameter in the Fehmarnbelt study area. Furthermore, the scale of the analyses certainly determines what patterns may be

found and what effects can be detected. The relatively low temporal resolution of porpoise abundance data in the present study, when compared to, e.g., continuous acoustic recordings of porpoises, means that it is somewhat limited in detecting effects of hydrodynamic variables, which show high temporal variability. To investigate effects of these hydrodynamic variables, passive acoustic monitoring is therefore more powerful, because of its better temporal resolution due to continuous recording and such investigations in the Fehmarn Belt indeed found a stronger correlation between hydrodynamic parameters and porpoise activity (FEMM 2013).

Harbor porpoises require a frequent and high-energy input due to their small size, limited body fats, and high-energy expenditure (Wisniewska et al. 2016; Rojano-Doñate et al. 2018), and environmental variables explaining porpoise distribution probably only partly reflect animal physiological needs and more likely are proxies of temporal and spatial dynamics in prey distribution. Thus, it can be assumed that their distribution follows the distribution of the most abundant prey species, and when direct information is available on food resources, they are often identified as being an important predictor of harbor porpoise abundance. Gilles et al. (2016) found porpoise density to decrease with distance to fishing hotspots for sand eel, which serves as an important food source in the North Sea. Main food sources for the Baltic porpoises are herring and cod (Sveegaard et al. 2012; Andreasen et al. 2017), and accordingly, porpoise distribution was shown to reflect herring distribution on a larger scale in the northern Baltic Sea, Skagerrak, and Kattegat (Sveegaard et al. 2012). However, for none of the fish species that form the main prey species of harbor porpoise there are data available at a sufficiently high resolution for meaningful analyses, and no clear connections of temporally and spatially variable prey occurrences with any hydrodynamic variables have been described.

With respect to underwater noise, it is demonstrated that, concerning the emitted frequencies, a large part of the shipping noise in the Fehmarnbelt is clearly audible to harbor porpoises. Though shipping noise is mainly of low frequency, the analysis shows that the higher frequency components are likely of sufficient level to elicit temporary avoidance responses in substantial parts of the Fehmarnbelt, most pronounced in the main shipping route to and from Kiel Canal and in the ferry route between the island of Fehmarn and Lolland (Fig. 8). For the assessment of significant avoidance responses, it has been assumed that a response of harbor porpoises to broad band ship noise is elicited at noise levels above 144 dB (FEMM 2013) corresponding to response distances of up to 1 km for larger vessels and ferries. Recent studies of Dyndo et al. (2015) and Wisniewska et al. (2018) also found that harbor porpoises may show initial responses at considerable distances to ship noise, likely responding to noise characteristics rather than just noise levels. Similarly, during a study on wind farm construction effects on porpoise occurrence, Benhemma-Le Gall et al. (2021) found porpoise detections to decrease with shipping intensity within a 5 km radius, depending on the mean distance of the ships to the POD. They also found porpoise detections outside piling activities to decrease by 17% as the mean SPL increased from 102 to 159 dB (Benhemma-Le Gall et al. 2021).

It appears that noise significantly influenced the distribution of harbor porpoise in the Fehmarnbelt. According to the multivariable habitat distribution model, a decrease in porpoise densities was found above received median sound pressure levels of 117 dB re1 μPa and up to median sound pressure levels of 130 dB re 1 μPa. At levels below 117 dB and above 130 dB, the model could not detect a clear response, as uncertainty at these noise levels was high, partly due to limited observations at these noise levels. Comparing the results of the distribution model with the selectivity index reveals further insight into the interaction among basic habitat features and anthropogenic pressures. While the multivariable distribution model indicates a negative impact of median noise levels from about 117 dB to 130 dB, the Jacob's selectivity index indicates a positive selection of harbor porpoises for areas exposed to median noise levels between 115 dB and 125 dB followed by a deselection only for noise levels of 125–130 dB. The finding that porpoises are more abundant than average in areas of elevated noise levels does, however, not contradict the finding that underwater noise affects the presence of harbor porpoises in such areas. Abundance in good quality habitat (based on prey abundance or abiotic habitat features) may be lower with increasing noise when compared to a baseline situation, but it may still be higher than average abundance or when compared to areas of differing habitat characteristics. The findings simply highlight the importance of including habitat parameters when analyzing the significance of impacts of underwater noise on marine mammals in the absence of a baseline situation; a simple comparison of abundance may not allow detecting the significance of noise impacts.

Despite the high frequency of ships passing through the Fehmarnbelt, showing subareas experiencing up to about eight ship passages per hour and rather high noise levels, the survey data show that harbor porpoises occur frequently in the shipping lanes and in areas where noise levels are highest. Concurrent with the aerial surveys, direct observations from ferryboats servicing between Rødby and Puttgarden provided frequent sightings of harbor porpoises at relatively close distance and expected high noise levels of more than 150 dB. During 1 day sailing back and forth with the ferry, up to 53 sightings were obtained (FEMM 2013) giving evidence of frequent porpoise occurrence within the most busiest and noisiest shipping lane. In a parallel study using passive acoustic monitoring (FEMM 2013), "distance to shipping lane" was a significant parameter for modelling overall porpoise presence up to a distance of about 1 km, indicating that shipping has a small-scale effect on the presence of harbor porpoises. However, when analyzing porpoise abundance from aerial survey data, "distance to shipping lane" was not significant. If avoidance responses of harbor porpoise – and in the same way of other marine mammals – entail effects of significance from a habitat use and fitness perspective, it depends not only on the range of a disturbance effect but also on the frequency of disturbance and the subsequent relocation time, i.e., the time it takes for a disturbed individual to return to its original location. We consider it as a fair assumption that the distance of avoidance and relocation time will depend on both the status of the individual and the habitat quality, thus the context of the disturbance (Ellison et al. 2011). In the case of the Fehmarnbelt, there is evidence for habitat preference of harbor porpoise

clearly overlapping areas affected by high median underwater noise levels from shipping. Observations and modelling show that areas at water depths of 15–25 m, which are preferred by harbor porpoise in the study area, over large parts coincide with the main shipping lanes in the shallow waters of the Fehmarnbelt. This makes disturbance effects less visible.

Furthermore, it has to be kept in mind that the modelled distribution of harbor porpoise was obtained from aerial surveys conducted under relatively calm weather conditions, usually at wind speeds not exceeding 3 Bft. At such low wind speed, ship noise is the dominating noise source in the underwater soundscape. Higher wind speed will not only increase ambient noise and thus decrease the signal to noise ratio, but it will also lead to a decrease in sound propagation of anthropogenic noise due to more scattering at a rougher sea surface and higher sound absorption by air bubbles in the upper water layer (Thiele and Schellstede 1980; Jones et al. 2009; Dragon et al. 2016). The distribution of harbor porpoises as derived from aerial survey data during calm weather conditions thus accentuates conditions favorable for detecting effects of anthropogenic noise.

The available data thus indicate that shipping noise affects the distribution of harbor porpoise in a busy shipping area, though animals remain abundant even within the busiest areas. This highlights an apparent trade-off between individual costs (e.g., reduced feeding time, reduced short-term well-being) and benefits (e.g., value of the habitat) of an avoidance response. In other words, anthropogenic sound might be unwanted, principally disturbing, and/or initially unpleasant, but the rewards from staying in or returning to the habitat may under some circumstances and for some individuals outweigh the costs of avoiding it (McGregor 2007).

## Conclusion

The results show noise levels within the Fehmarnbelt that are typical for busy waterways in Europe and elsewhere. The findings of the distribution modelling lead to the conclusion that porpoise distribution within the study area relates to a number of specific habitat parameter, such as those reflecting the chance to encounter prey as also shown in other investigations. However, complete habitat displacement did not occur and most likely, harbor porpoise behavior in the Fehmarnbelt represents a trade-off where the benefits to stay in the overall area outweigh the costs due to anthropogenic pressures such as shipping noise.

The present study demonstrates how shipping noise in an area of variable and intense ship traffic affects the distribution of a cetacean species in relation to their habitat choice (Simard et al. 2008; Campana et al. 2015). While modern biologging techniques more recently allow studying individual instantaneous responses of marine mammals to anthropogenic noise in more detail (e.g., van Beest et al. 2018; Wisniewska et al. 2018; Mikkelsen et al. 2019), noise and density mapping provide essential information to assessing the significance of responses on a larger scale. Both approaches are considered as important for comprehensively analyzing

possible significant population effects of increasing anthropogenic ocean noise on marine mammals.

So far, no conclusions can be drawn as to the extent of a potential reduction in porpoise numbers in the area nor about potential population level consequences; however, the data point at noise impacts that reach well beyond the core areas of the shipping lanes. Considering that substantial parts of the western Baltic Sea are exposed to shipping noise at similar levels as those in the Fehmarnbelt (Folegot et al. 2016), it is of high interest to further analyze possible impacts on numbers, distribution, and habitat use of the subpopulation of harbor porpoises in this area. Further investigations following the approach of this study are thus recommended to disentangle the complicated relationship between human pressures and cetacean distribution and habitat use. It is proposed to apply the method used in this investigation to a larger area where more diverse conditions of preferred habitats exposed to different noise levels should provide more insight into how shipping noise affects numbers and distribution of harbor porpoise, which, as being a rather abundant cetacean, are considered as a useful model species for cetaceans in general.

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# Frequencies of Humpback Whale Song Units (*Megaptera novaeangliae*): Adjustments to Small Boat Noise

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**Abstract**

Anthropogenic noise has been shown to impair hearing and elicit behavioral changes among marine animals. Humpback whales are known for their complex vocal displays (i.e., song) which can be masked by vessel noise. In the Gulf of Tribugá, a part of the breeding grounds for humpback whale Stock G, a marine port construction project was proposed, and, if constructed, underwater noise levels in the Gulf would significantly increase.

To better understand how humpback whales cope acoustically with increases in vessel noise, humpback whales in the Gulf of Tribugá were studied to examine changes in the high and low frequencies of their song units *during* and/or *after* boat noise compared to *before* it was present.

Out of 38 high- and low-frequency comparisons, the same song units had frequencies that were significantly different *during* boat noise 5 times and *after* boat noise 4 times (as compared to *before* boat noise). Out of 20 bandwidth comparisons, the overall bandwidth of the units narrowed 11 times, broadened 9 times, shifted higher 6 times, and shifted lower 2 times. These results give insight into how humpback whales are capable of altering their song structure when they may be responding to small boat noise.

**Keywords**

Acoustic communication · Anthropogenic noise · Vessel noise · Vocal response · Humpback whale song · Noise pollution

## Introduction

Underwater noise encompasses a variety of different sources such as biological, geological, and anthropogenic ones. At present, significant changes in the underwater acoustic environment are occurring at a fast rate due to increases in anthrophony sources and correlated changes in biophony. Since the Industrial Revolution, increases in shipping and infrastructure development have contributed a great deal to overall anthropogenic noise level increases in the ocean while contributing to reduced levels in sound-emitting animals (Duarte et al. 2021). One pervasive way that anthropogenic noise can affect marine mammals is through auditory masking. Masking refers to when noise in the environment reduces an animal's communication space which interferes with an animal's ability to detect and perceive a sound of interest (Erbe et al. 2016). Therefore, with the overall increase in anthropogenic noise in our oceans and since many marine animals rely on sound for survival, understanding anthropogenic noise impacts on them is important for management efforts. While increases in large-scale anthropogenic activity have worldwide implications, small boat activity has implications on a more local scale. This chapter aims to outline the current impacts of small boat noise on humpback whales in an area

with low levels of anthropogenic noise before a potential large-scale anthropogenic activity occurs in this breeding ground – the building of a marine port.

Acoustic communication serves as a critical role in the social behavior of marine mammals. Many of these animals have evolved to use sound as the primary modality by which to communicate in their aquatic environment. During crucial social interactions, the animal emitting acoustic signals relies on having optimal detection and perception by their targeted receiver (Dunlop 2016). Therefore, the ability to successfully send and receive acoustic signals is vital to critical life functions for marine animals (Cholewiak et al. 2018). When sending acoustic signals in the ocean, the signal experiences propagation loss underwater, and the sender must compete with that attenuation, as well as background noise, along the paths the sound waves take. One persistent contributor to background noise is from anthropogenic sources. This includes a wide variety of sources that extend over a broad range of frequencies (1–100,000 Hz) (Frisk 2012). Most notably, the number and density of large ships in the ocean have increased globally, thus contributing to an overall increase in low-frequency background noise levels in the ocean (Frisk 2012; Tournadre 2014).

As anthropogenic noise continues to increase in our oceans, studies continue to show that this noise negatively impacts marine animal communication by compromising hearing ability and eliciting physiological and behavioral changes in marine animals (Duarte et al. 2021). Some animals have been shown to change the frequencies and source levels of their vocalizations to shift away from the potential effects of masking (Rey-Baquero et al. 2021). Humpback whales are vulnerable to low-frequency shipping noise due to their characteristic vocalizations (called song) having fundamental frequencies concentrated at lower to moderate ranges between 0.03 and 8 kHz (Perazio and Mercado 2018). Their songs are sung by males predominantly in winter breeding grounds, but are also heard along spring/summer feeding grounds and migration routes (Mattila et al. 1987; Clapham and Mattila 1990; McSweeney et al. 1989; Vu et al. 2012). The function of these songs, while debated, is commonly hypothesized to be related to mating behavior – attracting females or mediating male-male interactions (Herman 2017). The Gulf of Tribugá, located in the northern Colombian Pacific in the Chocó Department, is a part of the breeding grounds for humpback whales of Stock G from May through December (Avila et al. 2020).

Chocó, Colombia, consists of roughly a dozen afro-indigenous fishing villages, each consisting of about 50–8000 people along 200 km of coastline. The main form of transit between villages is by boat due to few existing paved roads. When traveling between villages, performing SCUBA, fishing, and/or providing whale watching tours, families use dug-out canoes or small fiberglass boats with outboard motors. Additionally, larger supply vessels (~20 m in length) bring fuel and cargo roughly every week and harvest shrimp during the spring and summer months. Because of this, the Gulf currently possesses a relatively pristine soundscape compared to other gulfs across the globe because it contains a small variety and low density of motorized vessels. However, a large infrastructure concern to local conservation groups in 2020 was the major marine port construction project that

was proposed in the Gulf. Although permits for the port have been denied, it has been an ongoing but fluctuating effort by the Colombian government since the 1990s to build the port. If ever approved in the future, underwater noise would significantly increase due to both construction and operational shipping lanes, and the acoustic environment would permanently be altered.

When both vessel traffic noise and humpback whale song are primary contributors to the soundscape, like in the Colombian Pacific during breeding season, overlap in time and frequency exists between the two sound sources (Rey-Baquero et al. 2021). This overlap likely leads to masking of the whale song which could potentially lead to behavioral changes. Additionally, humpback whales in this breeding area are historically exposed to lower background noise levels and wider communication spaces compared to other studied breeding grounds (Duennebier et al. 2012; Bejder et al. 2019). Thus, new and increased vessel noise, which would come with the establishment of shipping lanes from a marine port, could have a more sudden impact on them compared to the whales in other areas that are more habituated to vessel noise. This is why it is especially important to monitor humpback whales' potential acoustic behavioral changes in the Gulf of Tribugá's current soundscape which has relatively low levels of noise by today's standards.

A before-after control-impact (BACI) study design was implemented to measure the acoustic behavior of humpback whales to better understand how they may cope with a transient increase in vessel noise that they could adapt to spontaneously. The typical BACI framework involves the measurement of an ecological variable of interest at a site that is expected to be affected by an anthropogenic, management, or other assumedly disturbing activity before and after the presence of the activity (Kerr et al. 2019). The BACI framework was implemented to identify potential changes in song units in response to noise disturbance from small boats.

However, noise from boats does not usually have sudden onset and offset. Rather, a passing boat will be heard at low levels from a distance, increase to highest levels at the closest point of approach to the receiver, and then have decreasing noise levels as it fades away into the distance. Therefore, an amount of time exists when the song of a whale could overlap in time and space with the fluctuating noise levels of the passing boat. Instead of assigning before and after times of the BACI as delineated by the closest point of approach, the BACI framework was adjusted to include samples of humpback whale song units sung during the active presence of boat noise. This framework has been named a BA(D)CI study where D stands for "during." The before time range ends the moment a boat engine was detectable in a spectrogram, at which time the time range became the during range. Once the boat engine either turned off or disappeared from detectability in the spectrogram, the during time range changed to the after time range. This adjustment was implemented to observe the possibility that rapid responses in humpback whale song could occur upon the immediate presence of boat noise (*during*) as opposed to slower responses that may be observed in the units sung at various time lags after boat noise has subsided (*after*). This may give important information regarding how quickly a whale is able to make crucial song adjustments if trying to avoid masking in real time.

## Metrics

Frequency measurements of the song units sung *before* boat noise that were detected on the passive acoustic monitoring device were classified as the "Control" part of the BA(D)CI framework, while frequency measurements of the song units sung *after* and *during* boat noise were both classified as the "Impact." In addition, to consider any natural variability in the whale's song units between different whales, songs that were recorded without coincident recording of boat noise present were analyzed to serve as an additional Control variable. These non-boat Control (NBC) song units were recorded from different hours and days as the songs that were interrupted by boat noise, so it should be noted that comparing the *NBC* song units to *after*-boat Impacted units would most likely be comparing two different whales and are not an appropriate comparison for determining a response to boat noise. Comparing the *NBC* song units to the *before*-boat Control song units, though, can provide insight into how much variability exists between individuals. Then the amount of variability in an individual without boat noise present can be compared to the amount of variability in an individual singer while reacting to boat noise.

## Hydrophone Deployment

Acoustic data were collected at Morro Mico from October 22 to November 28, 2018, at the coordinates 5°52.101′N, 77°19.007′W using an ecological acoustic recorder (EAR) on loan from Oceanwide Science Institute. Morro Mico is a small rock island located just north of all the coastal villages in the Gulf of Tribugá and is just south of a national marine sanctuary (Utría National Natural Park), so it experiences less boat traffic than areas between villages where people are transiting more often. The EAR was deployed at 25 m depth at high tide by SCUBA divers attaching it to an anchor. It recorded at a 15.625 kHz sampling rate for a duty cycle of 33.3% (recording occurred for 10 min every half an hour). Data was provided by the PHySIColombia Project database (Fig. 1).

## Before, During, and After Comparisons

Acoustic data from the EAR was downloaded and converted to .wav files using custom MATLAB code (L. Munger). For a more comprehensive look into the data analysis, please refer to the sister study (Rey-Baquero et al. 2021).

Data were analyzed in a total of 29 10 min audio files using Raven Lite to annotate humpback whale song units on a spectrogram. This was to measure the high and low frequencies, and thus shifts in bandwidth, for each song unit type. Since frequency is a parameter that can be affected by propagation, faint songs (roughly $<6$ dB above background noise levels) were not analyzed to reduce the possibility that a shift in high-frequency elements of the song was affected by attenuation and not the animal's behavioral change. Each audio file analyzed was considered one part of a song from one individual whale.




**Fig. 1** A map of the field site (Map was created in R using "ggmap" package (Kahle & Wickman 2013)) and location of the hydrophone deployment. The hydrophone was deployed at Morro Mico into the Gulf of Tribugá in the Chocó region of Colombia in October to November, 2018

Of the 29 audio files analyzed, 20 had boat and whale presence, and 9 had only whale presence. For each audio file, song units were characterized based on their contours and then classified into one of the ten different unit types (see Table 1 for a list of all unit types) regardless of the min, peak, or max frequencies or length. Specifically, for audio files containing both boat and whale presence, the contours of the song units were required to remain the same *before* vs. *during* or *before* vs. *after* boat noise (Fig. 2) to be considered comparable. Due to the timing of when boat noise was interrupting the song in the 20 audio files with boats, 20 *before* vs. *during* comparisons and 18 *before* vs. *after* comparisons were collected. That is to say, if boat noise was at the beginning or end of an audio file, no *before* or *after* time range, respectively, was available. In the 9 audio files containing only whales, 20 *before* vs. *NBC* comparisons were characterized (Table 1).

The null hypothesis stated that no significant structural changes in the song units occurred *during* nor *after* exposure to boat noise compared to song units *before* exposure. The first alternative hypothesis stated that the high, low, or both the high and low frequencies in song units were sung significantly differently *during* boat noise compared to *before* it was present. The second alternative hypothesis stated that the high, the low, or both frequencies in song units were sung significantly differently *after* boat noise compared to *before* it was present.

## Frequency and Bandwidth Analysis

The high and low frequencies, duration, and annotated labels for each song unit were extracted from Raven Lite and opened with Excel for statistical analysis. In Excel,

**Table 1** The results of the Kruskal-Wallis significance tests displaying the low frequency (LF) and high frequency (HF), sample size, and p-value for each unit type. Each of the three temporal classes (*during*, *after*, and non-boat Control (NBC)) was compared to the units *before* the boat noise. The shaded boxes represent *p*-values that are significant ($p < 0.05$). NA represents samples that were not present so could not be compared

| | Non-boat control | | During | | After | |
|---|---|---|---|---|---|---|
| Unit type | *n* | *p* value | *n* | *p* value | *n* | *p* value |
| **Unit A** | | | | | | |
| LF | 172 | 0.268 | 87 | 0.983 | 67 | 0.015 |
| HF | 172 | 0.001 | 87 | 0.357 | 67 | 0.004 |
| **Unit A1** | | | | | | |
| LF | 71 | 0.000 | 36 | 0.012 | 30 | 0.007 |
| HF | 71 | 0.000 | 36 | 0.174 | 30 | 0.464 |
| **Unit B** | | | | | | |
| LF | 241 | 0.000 | 122 | 0.118 | 98 | 0.711 |
| HF | 241 | 0.283 | 122 | 0.949 | 98 | 0.611 |
| **Unit B1** | | | | | | |
| LF | 138 | 0.000 | 93 | 0.707 | 95 | 0.996 |
| HF | 138 | 0.971 | 93 | 0.009 | 95 | 0.000 |
| **Unite C** | | | | | | |
| LF | 21 | 0.072 | 12 | 0.053 | 12 | 0.133 |
| HF | 21 | 0.590 | 12 | 0.086 | 12 | 0.197 |
| **Unit CD** | | | | | | |
| LF | 64 | 0.026 | 45 | 0.391 | 40 | 0.333 |
| HF | 64 | 0.003 | 45 | 0.065 | 40 | 0.305 |
| **Unit CU** | | | | | | |
| LF | 10 | 0.037 | 7 | 0.053 | 5 | 0.083 |
| HF | 10 | 1.000 | 7 | 0.053 | 5 | 0.248 |
| **Unit D** | | | | | | |
| LF | 156 | 0.001 | 24 | 0.006 | 82 | 0.108 |
| HF | 156 | 0.000 | 24 | 0.051 | 82 | 0.552 |
| **Unit D1** | | | | | | |
| LF | 46 | 0.000 | 98 | 0.270 | 59 | 0.527 |
| HF | 46 | 0.901 | 98 | 0.138 | 59 | 0.423 |
| **Unit D2** | | | | | | |
| LF | 169 | 0.090 | 90 | 0.000 | NA | NA |
| HF | 169 | 0.090 | 90 | 0.000 | NA | NA |

the Kruskal-Wallis test was used to calculate the significance level ($p$-value $<0.05$) for each of the comparisons between unit types (Fig. 1). Shifts in the bandwidths of each unit were analyzed by:

1) Overall bandwidth: comparing the difference in the high and low frequencies between *before* vs. *during* and *before* vs. *after* for each song unit type.

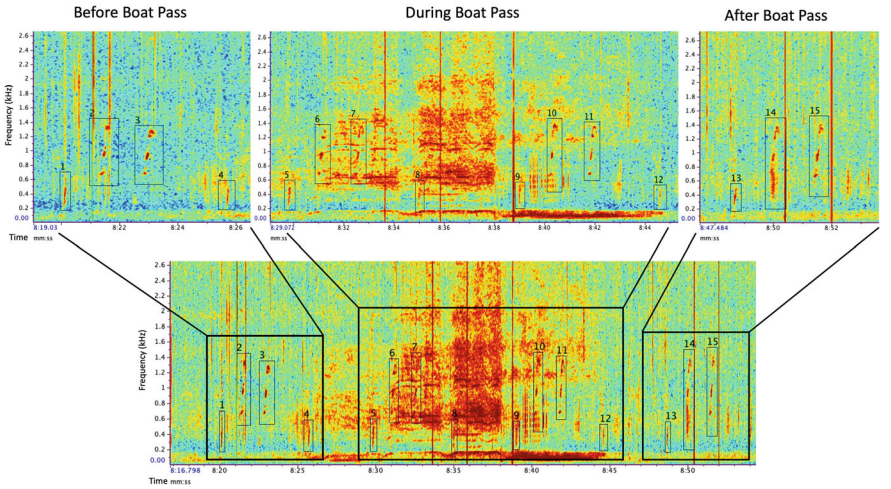



**Fig. 2** An example of the same song units *before*, *during*, and *after* a boat pass from one individual singer. Each box and subsequent number represent an individual song unit. Units with the same contours were classified into a unit type based on their contours (e.g., unit type B1 and unit type D1)

2) Narrowing/broadening of bandwidth: comparing the median frequencies of all samples (both high and low collectively) for during and after boat noise and then comparing these to the *before* samples.

## Noise Effects on Humpback Whale Song

### Effects of Noise on High and Low Frequencies of Humpback Whale Song

Out of the 19 audio files containing both boat activity and whales, there were 18 comparisons of *before* vs. *after* boat noise and 20 comparisons of *before* vs. *during* boat noise. Out of these comparisons, the high and low frequencies of song units were significantly different *before* boat noise compared to after it was present 4 of the 18 times. For the *during* comparisons, the high and low frequencies of song units were significantly different *during* boat noise compared to *before* it was present 5 of the 20 times. When song units from *before* boat noise were compared to the same unit type from audio files without any boats present (a.k.a. comparisons between different whales), they were significantly different 11 times of 20.

### Effects of Noise on Overall Bandwidth of Humpback Whale Song

Bandwidth results were acquired for each of the ten-unit types due to the shifts in the high and low frequencies of song units. For bandwidth, there were ten *before*

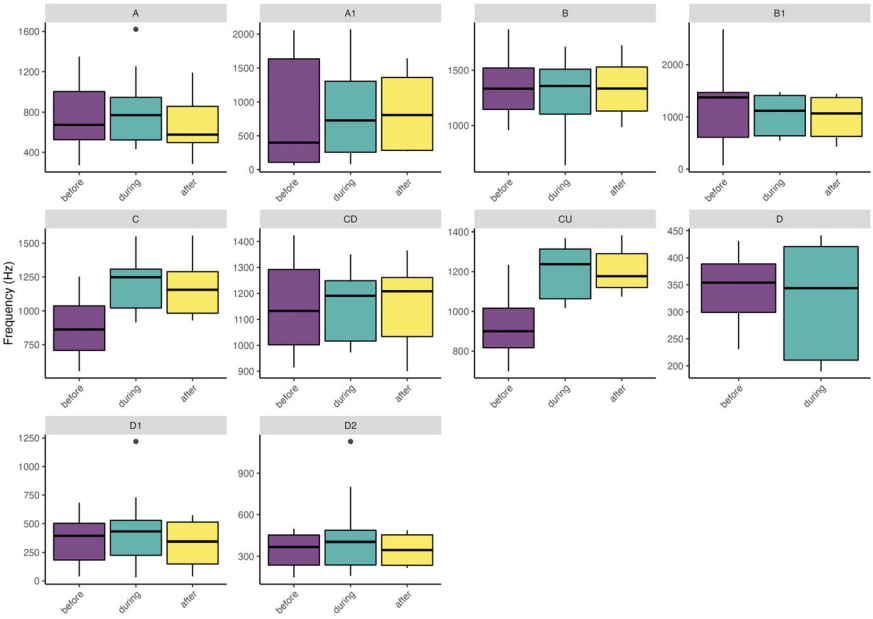



**Fig. 3** Box and whisker plots (displaying the median, minimum, and maximum values, the first and third quartiles, and the extreme singular outliers) of the frequency changes of song units in the presence of boat noise. *Before* represents units that have yet to be exposed to boat noise in the audio file; *during* and *after* represent units in the presence of or following boat noise in the audio file. Units were identified from individual singers and compiled based on unit type for analysis

vs. *during* comparisons, nine *before* vs. *after*, and nine *before* vs. *NBC* comparisons. Among the *during* comparisons, the bandwidth narrowed five times, broadened five times, shifted higher seven times, and shifted lower three times. Among the *after* comparisons, the bandwidth narrowed five times, broadened four times, shifted higher four times, and shifted lower five times. Among the *NBC* comparisons, from the files with boats, the bandwidths were narrower four times, broader five times, higher seven times, and lower two times (Fig. 3).

## Assessing Song Unit Adjustments

In the Gulf of Tribugá, whale watching is an important seasonal livelihood for local fishermen who rely on the high tourism demand during the winter breeding season. Although humpback whales are exposed to less vessel noise in this area than others studied in the Pacific, humpback whales are still annually exposed to whale watching vessel noise where their acoustic field is masked when in close proximity with these vessels (Rey-Baquero et al. 2021). Considering this, adjusting song unit structure, as seen in other marine mammals, could help to avoid the most prevalent effects of masking.

## Variations in Song Units

Some humpback whales showed changes in the spectral structure of their song units when assumed to be in audible proximity to boat noise. The high and low frequencies from the song units *before* boat noise from one singer were significantly different than the high and low frequencies of song units from another singer when no boat noise was detected (*NBC*) in an audio file 11 of 20 times. For example, the high and low frequencies from song unit type A in sound file 00000727 were compared to the high and low frequencies from the same song unit type (unit type A) in sound file 00001207. These comparisons are assumed to be comparisons between two different whales singing the same unit type (further explanation provided in methods) due to the mobility behavior of whales. Additionally, song unit frequencies in the same song from the same individual (the song analyzed in 1 audio file) significantly changed 9 times between *during* and *after* comparisons in the 20 files with boats analyzed. Of these nine significant changes, four were *during* and five were *after* boat noise presence. Therefore, significant frequency differences between whales existed more times than significant frequency changes in the same whale's song.

Although it is common to see conformity to a single song type within a population, it is also typical to observe variations in the same song sung between different individuals as seen in the results of Garland et al. (2013) and Payne and McVay (1971). Therefore, the 11 of 20 (about half of the time) significant differences observed between the *NBC* units and the *before* units are likely representative of individual variation between different whales and not of any response to boat noise. Comparatively, the changes in the same unit type from the same singer (potentially in response to boat noise) were also significantly different about half of the time. This demonstrates that humpback whales exhibit variability in their songs between one another and that they are capable of individual elasticity, changing the frequencies of their song units if needed.

Since the changes in the *during* and *after* cases as compared to *before* cases were temporally around the presence of boat noise, it is reasonable to assume that the significant variations observed were correlated to the boat noise. A study by Thomas F. Norris (1995) found that humpback whales changed their unit and phrase structure in response to boat noise. He found that phrases were shorter in the presence of boat noise and that frequencies of some units were altered with the occurrence of large vessel noise (Norris 1995). Similarly, fin whales have been shown to alter the frequencies of their songs in response to vessel noise. Fin whales lowered the bandwidth, peak frequency, and center frequency of their sounds in response to increased background noise from large vessels (Castellote et al. 2012). Like the frequency shifts observed in the two studies just mentioned, the humpback whales currently being studied may have intentionally shifted the frequencies of their song units when exposed to boat noise. Further, these changes could in fact be a causal result of the humpback whales using their flexible voices in an attempt to shift away from the effects of masking.

## Changes in Bandwidth

Humpback whales were observed changing the frequencies, and thus the bandwidth of their vocalizations, both *during* and *after* the presence of boat noise as compared to *before* it. One such bandwidth shift was to higher bandwidths which could help to avoid the loudest part of the boat because boat noise is concentrated at lower frequencies. A second shift was to lower bands which may help those units propagate further (especially if they are below the main boat noise frequencies) as lower frequencies propagate further underwater than higher frequencies. A third shift was narrowing the bandwidth which may help to focus more acoustic energy into one section of the spectrum and increase the source level of the peak frequency of a unit. Finally, broadening the bandwidth allows songs to contain a wider frequency range. With this wider range, hopefully more of the song's frequencies will transmit sufficiently enough, or be masked less, than more narrow-banded songs so that any vital information contained within can reach its intended target. As these interpretations illustrate, it is possible that any bandwidth shift could yield a benefit to the singing whale. Since humpback whales have also been previously observed increasing the source levels of their vocalizations in response to background noise levels (Dunlop et al. 2015; Fournet et al. 2018), they may employ multiple strategies to reduce masking effects, with frequency changes being one of them.

## Variations in Song Units with Exposure to Varying Noise Levels

A study by Tsujii et al. (2018) found that humpback whales off the coast of Japan altered part of their singing behavior due to large shipping vessel noise. The whales studied reduced their sound production or stopped singing temporarily in response to the noise (Tsujii et al. 2018). In contrast, the humpback whales currently being studied showed that they often continue to sing throughout small boat noise. One potential explanation for this difference could be that Tsujii et al.'s study site had large cargo ships and most likely higher noise levels compared to the Gulf of Tribugá which is currently void of shipping lanes. This may indicate that humpback whales lack the necessary adaptations to cope with louder vessel noise that comes with increases in shipping lanes and rather than attempt to adjust their song units, they rely on temporal masking release or saving their energy to strategically sing only when the surrounding noise seems low enough to be worth the communication effort. If there is a gradation of responses for adapting to increasing levels of vessel noise from sometimes adjusting frequency to reducing singing until only the most opportune times, this would be especially concerning in areas that are more heavily trafficked. Whale song being increasingly masked by escalating anthropogenic noise levels could correlate with more extreme, different, or more various acoustic behavioral changes.

## Conclusions

A variation of the BACI study design was implemented in order to quantify humpback whale acoustic behavioral reactivity to small boat noise in a gulf currently void of noise from shipping lanes. Significant changes in song unit structure were found between different individuals and in a single individual's song in the presence of boat noise from small vessels.

A study conducted by Maria Paula Rey-Baquero and Laura Valentina Huertas-Amaya analyzed how whale watching boat noise effects humpback whale song in the Gulf of Tribugá (Rey-Baquero et al. 2021). Their study found temporal aspects of singing activity were significantly different when boat noise was present compared to when it was absent. There were fewer numbers of longer units sung in the presence of boats and longer inter-unit intervals (the time between units) which they hypothesized could be a strategy to escape the masking (Rey-Baquero et al. 2021). The combination of these two studies contributes to our understanding of how humpback whales may attempt to cope with anthropogenic noise sources in a relatively low-disturbance region. This is especially important to document the current behavioral reactivity of humpback whales to present noise conditions and aim to describe these behaviors prior to the potential increase in noise levels caused if the port is ultimately constructed in the Gulf of Tribugá. Further, both studies contribute to a larger effort to record the baseline soundscape in a place with relatively less coastal infrastructure than many parts of the world. In this form, the Gulf of Tribugá can serve as a reference point for what louder marine soundscapes may have sounded like before their primary disturbance was developed.

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# Acoustic Models to Consider Noise Pollution from a Possible Port on Humpback Whales in the Gulf of Tribugá, Colombia

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**Abstract**

The Gulf of Tribugá is a relatively pristine acoustic environment in the Colombian north Pacific. Since the 1990s, many administrations have tried to build a deepwater port in Tribugá. While the construction permits were most recently denied in September 2020, they could be applied for again in the future. If built, the Gulf's terrestrial and marine environments would change forever. One strategy to promote the conservation of these marine and coastal habitats is to visually illustrate how much ocean space that ship noise could occupy if a seaport were built. Thus, noise footprints of possible configurations of shipping lanes were simulated. Acoustic propagation models were made for a "control" shipping lane configuration and two predictive lane configurations to compare the spatial coverage of their noise footprints that had levels above the threshold of humpback whale audibility. Differences in the spatial extent of these configurations were found, with the deepwater trench and the random configurations more similar to each other than the control configuration, and both non-control ones had noise occupying more space than the control configuration.

**Keywords**

Colombian Pacific Ocean · Noise pollution · Shipping lanes · Humpback whale · Audibility · Acoustic propagation modeling

## Introduction

The humpback whale (*Megaptera novaeangliae*) is a cosmopolitan species, migrating long distances between feeding grounds at high latitudes and breeding grounds in tropical waters. Humpback whales are grouped into 14 distinct population segments (DSPs) worldwide (Fisheries 2020, 2022). The Southeast Pacific DPS is also called "breeding Stock G" and is the focus of this research. Stock G breeds along the southwestern Pacific (Peru, Ecuador, Colombia, Panama, Costa Rica) during the austral winter (June–October) and feeds in southern Chile and the Antarctic Peninsula during the austral summer (December–April) (Acevedo et al. 2017; Avila et al. 2021).

Both sexes of this species and all age classes are documented to communicate with acoustic signals or "social calls" (Mattila et al. 1987; McSweeney et al. 1989; Norris et al. 1999; Dunlop et al. 2008; Stimpert et al. 2012; Seger et al. 2015; Rey-Baquero et al. 2021; Gascón 2021) to facilitate life functions such as foraging, courtship, communication, reproduction, and calf protection (Richardson et al. 1995; Merchant

et al. 2015; Haver et al. 2019). In addition to social calls, humpback whales have two other types of vocalizations, including songs and repetitive tones (Payne and McVay 1971; Norris et al. 1999; Rey-Baquero et al. 2021; Gascón 2021). Song refers to a series of "units," which are structured vocalizations, that create increasingly complex combinations known as "phrases," "themes," "songs," and "singing sessions." These songs are only produced by males, mostly during the breeding season (Payne and McVay 1971; Clapham 1996; Clapham 2000; Darling and Berube 2001; Mercado et al. 2005; Au et al. 2006; Stimpert et al. 2012). Repetitive tones have only yet been documented off the coast of Colombia (Gascón 2021).

All of these vocalizations propagate through the ocean to reach conspecifics and facilitate communication. But they compete with other sounds, particularly noise from vessels, in the environment to propagate effectively. For the remainder of this chapter, the biological sounds of interest are vocalizations by humpback whales, meaning they are the "signals" referred to in this work. Furthermore, any sound source that is not a biological signal will be referred to as "noise." particularly the sounds produced by vessels.

Noise from vessels and animal vocalizations are subject to the same physical limitations in how they propagate underwater. Oceanographic variables like bathymetry, sediment type, water temperature, and salinity have an influence on the propagation, dispersion, attenuation, and absorption of sound in the water column (McKenna et al. 2012). Additionally, the source characteristics of a sound also affects how it will travel. For vessels, traffic density, propeller type, length, and speed all affect their source characteristics. Even though noise is a by-product of vessels traveling at sea, it can impact the acoustic environment around it and affect the communication strategies of marine mammals that inhabit it. Too much noise could disrupt the signal transmission of humpback whales and thereby affect a population's viability (Cholewiak et al. 2018; Rey-Baquero et al. 2021). When noisy zones intersect with critical habitat areas (i.e. areas occupied by species with physical and biological characteristics essential to the ecosystem or at risk and requiring conservation or protection (Fisheries 2022), the effects of the noise on the vital functions of the animals must be a focus of analysis to promote conservation goals.

In 2002, the U.S. National Marine Fisheries Service (NMFS) Ocean Acoustics Program compiled data and developed methods to assess and quantify noise exposure levels for different anthropogenic sources, including Temporary Threshold Shift (TTS) and Permanent Threshold Shift (PTS) and incorporating the auditory exposure criteria described in Southall et al. (2008). Threshold shifts that recover within hours, days, or weeks after exposure are called a TTS. A TTS occurs when the cells of the inner ear become fatigued, making it difficult to detect and interpret sounds. The probability of a temporary threshold shift occurring increases proportionally with sound level and duration of exposure (Weilgart 2007). A permanent threshold shift (PTS) occurs when hearing does not completely normalize after noise exposure and the ability to hear has been permanently altered or lost (Ryan et al. 2016; Putland et al. 2018). In general, fish and marine mammals tend to be affected by permanent threshold shifts when exposed to particularly high levels of anthropogenic noise (Putland et al. 2018).

A third category of exposure criteria pertains to behavioral changes, which are assumed to be more common than PTS and TTS (Southall et al. 2015). The sound level that is expected to elicit a behavioral change in marine mammals is 160 dBrms (Ainslie 2010; Heaney and Seger 2020). If noises above 160 dB elicit a behavioral change often enough to interrupt vital functions, it can lead to deleterious effects such as reduced foraging, difficulty maintaining cohesion, difficulty navigation, and increased stress (Ainslie 2010; Rolland et al. 2012; Heaney and Seger 2020).

A sound that reaches a minimum sensitivity or is loud enough for an average individual of a certain species to hear it is an audible sound (Erbe 2002). Audibility is a fourth category in how animals receive sound pressure and the main category of focus for this study. For humpback whales, no studies have been conducted that measure audibility levels directly because such research is commonly conducted with smaller species and in confined spaces. There are studies, however, that extrapolated audiograms from other species to estimate a humpback whale audiogram (Houser et al. 2001; Erbe et al. 2014). These audiograms provide a description of hearing capabilities of humpback whales at various frequencies. It is reasonable to assume that if an animal can hear a sound, it is likely to generate a reaction if that sound is deemed threatening. Because ship noise is neither impulsive nor as loud as some noises (e.g. pile driving), its audibility level and what spatial scales different shipping lanes create above this level is a focus of this research.

A fifth and final category discussed here is a combination of audibility and the received level of a noise source, called the masking threshold. Masking is when the sound pressure level of a sound necessary to be audible is drowned out by a different and unwanted sound source, a.k.a. "noise." If a noise is easily detected by an animal, it is possible that it has the capacity to superimpose itself on the animal's signal and interrupt it to such an extent that it is more difficult to detect and interpret by a conspecific. When experiencing masking, it is likely that an animal may suspend emitting its signals to avoid being interrupted, could change key signal characteristics, or could attempt to increase its vocal effort to improve its audibility (Lombard effect) so it can be heard by the intended receiver (Knowlton 2017). These and other masking release strategies could carry an energetic cost to the species (Noren et al. 2012). While many vessels do not reach source levels high enough to exceed the TTS and PTS thresholds, and few even exceed the behavioral change threshold, they can reach marine species' audibility thresholds. Therefore, this study focuses on received levels from shipping traffic above published audibility thresholds for humpback whales (Houser et al. 2001; Erbe 2002) and assumes that any audible sound may have the potential to do some masking on humpback whale signals and/or elicit a behavioral response from some individuals.

Globally, several studies have used acoustical models to better understand the effects and/or impacts caused by increased noise produced by different types of vessels (McDonald et al. 2006; Hildebrand 2009; Aulanier et al. 2017). One such study was conducted in the Canadian Arctic, where shipping as a dominant noise source (SNL = Shipping Noise Level) of high traffic density (>110 dB re 1μPa$^2$) was observed to cause a transformation of soundscapes, masking marine mammal signals and triggering behavioral responses (Aulanier et al. 2017). Similarly, in the

Celtic Sea, ship and seal tracks were coupled with an acoustic modeling system to assess noise exposure level. It was found that the animals which experienced steep changes in sound levels up to ~20 dB at a frequency of 125 Hz and ~10 dB on average over 10–1000 Hz would dive through the thermocline, particularly during the summer months (Chen et al. 2017). An additional study, conducted in the Dutch North Sea, used sound maps averaged over time and frequency to reveal that most of the human noise in the soundscape came from ships, followed by seismic sources and, lastly, by seismic explosions (Sertlek et al. 2019). Most of the anthropogenic sound energy from all three sources was concentrated between 100 Hz and 1 kHz, which are frequencies important to many marine mammal species. Therefore, this research is also using models to determine how much space is occupied by ship-specific noise frequencies that overlap with peak frequencies in humpback whale song. The models are based on current types of large ships transiting through the waters off western Colombia to understand where in this space the noise levels are or could be high enough to be heard by a humpback whale from Stock G.

For comparison to the current noise footprints off the Colombian coast, this research has also made predictive scenarios of likely shipping lanes to understand how much more space the shipping noise could occupy should human infrastructure increase in the location of the Gulf of Tribugá where conservation goals have recently collided with large-scale construction goals.

The Gulf of Tribugá is located in the northern region of the Pacific coast of Colombia, specifically on the western side of the department of Chocó. Its southern limit is Cabo Corrientes at 5°53′N, 77°28′W, while its northern end is at 6°11′N, 77°09′W. Its approximate area is 103,110 ha (Díaz et al. 2016). This Gulf contains a series of geographic features such as the coves of Arusí, Coquí, Tribugá, and Utría – all of which are framed by the Serranía del Baudó, which is a mountain range parallel to the westernmost mountain range of the Andes and the coastline (Díaz et al. 2016). The eastern foothills of the Serranía del Baudó correspond to a tertiary formation of unconsolidated marine sediments that are located at great depths and have thicknesses of ~4000 m (Díaz et al. 2016). Average annual sea surface temperatures (SSTs) in the Gulf of Tribugá can range spatially from 24.5 °C to 28 °C, and the largest variance for any given month is in March, from 24.79 °C to 28.68 °C. This temperature range is ideal for supporting coral reef patches, especially in the Utría Cove. Salinity in the Gulf is rather consistently 33.2 ppm (INVEMAR 2007; Díaz et al. 2016) throughout the year. An important fact to consider in this study is that salinity and temperature are both quite uniform over depth, leading to a high likelihood of many areas having isovelocity sound speed profiles, at least near the surface.

Most of the human population along the coast of the Gulf of Tribugá are Afro-Colombian and Indigenous people living in several villages. The total number of municipalities that surround the Gulf of Tribugá have a population of 127,633 inhabitants (ANLA 2020). The municipality of Nuquí has few roads as a method of access. Its main accessibility is by a small airport connected with Medellín, Bogotá, and Quibdó. Another option to reach Nuquí is through maritime transportation, by taking an overnight trip on small fuel or goods shipment boats that

connect with Buenaventura, or by paying for a taxi-style boat from Bahía Solano. Due to the lack of roads, it is not possible to drive between the villages, which is why navigation by residents is primarily done by river and sea. This lack of connection is why Nuquí tends to have problems with high costs of food products and sometimes shortages in many types of supplies and medical care. Therefore, the Gulf has developed and relies heavily on an economic infrastructure revolving around ecotourism, artisanal fishing, small-scale cattle ranching, and agriculture (Díaz et al. 2016).

The study area has a wide diversity of flora and fauna. Specifically, in terms of marine fauna, the Pacific shelf has a great variety of species such as sharks, fish, coral formations, and baleen and toothed whales (Díaz et al. 2016). It is one of the most important fishing reserves in the eastern tropical Pacific for local Colombian economies (Díaz et al. 2016). The Gulf of Tribugá is an essential ecosystem for humpback whales since it is one of Stock G's primary breeding grounds. Currently, it preserves great biodiversity of both marine and terrestrial species as it is a place less affected by human infrastructure on land and in the sea compared to more commercial coastlines in other countries. Thus, underwater noise pollution in Tribugá, as compared to other baleen breeding grounds in North and South America (Rey-Baquero et al. 2021), is thought to be relatively lower. As a result, many conservation groups consider it essential to protect this ecosystem from further human encroachment (DMRI 2020), and efforts like naming it a Hope Spot in 2020 are trying to foster its protection.

Recent research supports the assumption that boat noise in the Gulf of Tribugá, despite it being at lower levels than many places around the globe, still evokes behavioral responses from humpback whales. For example, Rey-Baquero et al. (2021) reported temporal differences in song units produced by humpback whales in the Gulf of Tribugá before as compared to after boat noise occurred. Additionally, a study that used the Gulf of Tribugá's passive acoustic monitoring dataset at the Morro Mico site found that humpback whales changed high and low frequencies of their songs about half the time after a boat pass occurred (Brown et al. 2021). Ethogram studies have also documented behavioral changes, like in Bahía Málaga where 44% of the humpback whales sighted in the area were approached by ships and subsequently modified their behavior, displaying faster travel speed and decreasing resting time (Avila et al. 2015). Such foundational research highlights the variety of reactions of different individuals in Stock G responding to vessel presence, acoustic or otherwise. Much of the acoustic energy from ships is concentrated at low frequencies, which increases the likelihood of masking acoustic signals from marine species that communicate in the same low-frequency range, such as humpback whales. Considering that songs are characterized by source levels reaching 151 dB re 1μPa$^2$ in Tribugá Rey-Baquero et al. 2021), and ships transiting between ports can have source levels higher than this (MacGillivray and Jong 2021), masking of large-scale merchant traffic could have negative implica-

tions on the communication of marine species in the aquatic ecosystem along shipping routes and for many kilometers around them (McKenna et al. 2012).

The construction of a multipurpose seaport in the area has been on the Colombian government's agenda since the 1950s. Construction and operation of this port would introduce all levels (PTS, TTS, behavioral, and masking) of noise exposure to the Gulf of Tribugá (Barrios and Plazas 2015). If built, the port would be similar in size to the port of Buenaventura, but smaller than the port of Long Beach, CA, USA. It would be intended to transport goods to other national and international ports (Colombia port of Buenaventura 2019; Port of Los Angeles 2020). Such characteristics typical of other similar-sized ports, as well as plans of the Tribugá port released by Arquímedes, were considered for the predictive modeling of this research.

Understanding the impact of the Tribugá Port, if it were constructed, is the major reason for doing this study. While the Gulf itself would see a lot of additional noise from ships coming in and out of the port, the larger ocean area along the Pacific coast of Colombia, Ecuador, and Panama would also be affected by the new shipping lanes. Therefore, the study area for this project not only considers the Pacific Coast of Colombia, but a larger portion of the Southeastern Pacific along much of the northwest corner of South America that overlaps with a larger portion of Stock G's migration and breeding grounds.

## Research Objectives

The current work (1) used acoustic propagation modeling to calculate how much area off the West Coast of Colombia is currently ensonified by large vessel traffic for a single "control" scenario, (2) simulated two possible shipping lane configurations and used acoustic propagation modeling to estimate the area that noise from each configuration could ensonify off the West coast of Colombia, (3) compared the increase in noise area between the current scenario and the future simulations, and (4) calculated how much area that boat noise could be audible to humpback whales in each configuration. Typical databases for the oceanic variables were used in the acoustic propagation modeling. General characteristics of the vessels that transit from the port of Buenaventura were used to predict the types of ships that could be transiting from the Port of Tribugá, if it were to be built. The JOMOPANS-ECHO model (MacGillivray and Jong 2021) was used to calculate the source levels of the ships given these general characteristics. The best possible current audiograms for humpback whales (Houser et al. 2001; Erbe 2002) were used to set audibility thresholds. Combining these factors allowed for the calculation of the area of the ocean over humpback whale audibility levels so that if there exists a more optimal navigation route, i.e. the one that would affect the communication space of humpback whales the least, it could be determined.

## Construction of Control Map and Predictive Configurations

The construction of the shipping lane configurations was based on a number of assumptions. First, there is currently only one shipping route, as shown on MarineTraffic.com (Fig. 1), several hundred kilometers off the coast of Colombia. With a new port based in Tribugá, it is assumed that this route would become more densely transited and that more routes would need to be created. Where those new routes are placed could have various effects on the underwater soundscape since sound waves propagate so differently depending upon the temperature, salinity, and depth of the water column they are traveling over. Therefore, the larger oceanic area of the NW coast of South America was considered in this acoustic propagation project, including the Gulf of Tribugá out to and beyond the current shipping route, up to the Panama Canal, and down to the Port of Buenaventura. Since sound travels hundreds of miles in the ocean, the study area is larger than just the coastal waters of Colombia. Instead, boundaries were selected to include as much sound as possible from ships traveling from the Panama Canal to the Port of Buenaventura.

The four corners of the study area are, beginning at the northwest corner and going clockwise, NW (7.879604°, −83.054129°), NE (8.986911°, −77.639609°), SW (0.982849°, −81.979942°), and SE (1.935371°, −76.528972°) (Fig. 2). The present work considered three configurations: the "control map" (the current ships from MarineTraffic.com on May 15, 2021), additional ships organized into a "random map," and additional ships organized to travel over the "deepwater trenches."

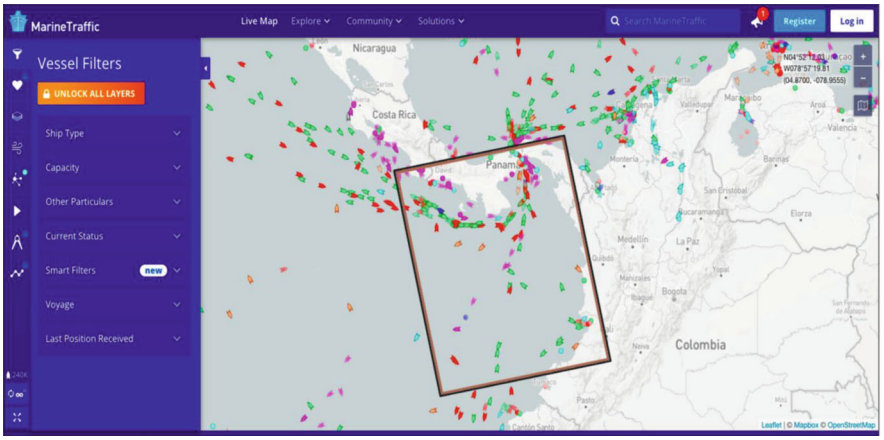



**Fig. 1** Automatic Identification System (AIS), geographic coordinates, information and/or variables for each vessel for May 15, 2021. (Source: MarineTraffic – Global Ship Tracking Intelligence (www.marinetraffic.com))

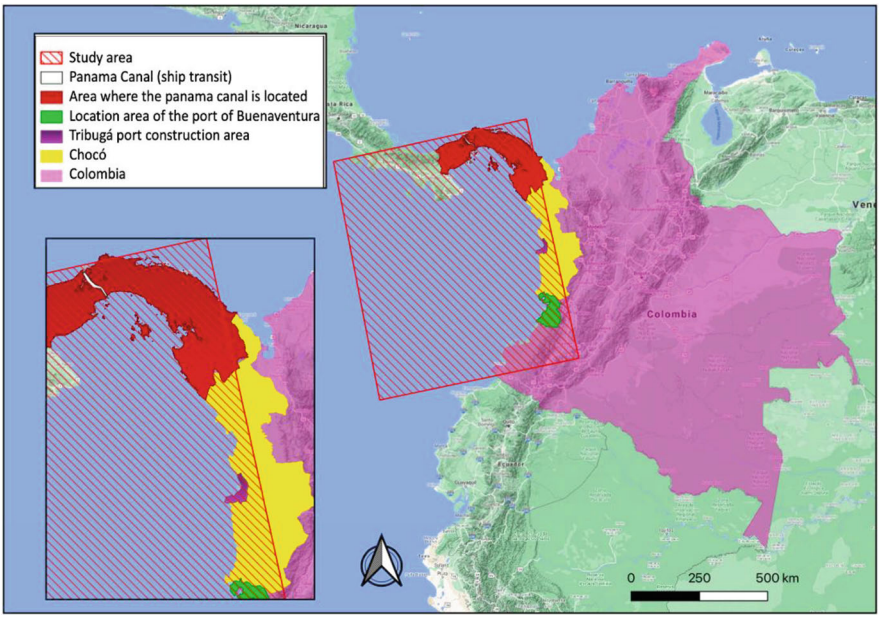



**Fig. 2** The study area is centered in the Colombian North Pacific, reaching from the Southern border of Colombia to the Panama Canal. The current main shipping lane cuts largely through the middle of this area

## Control Map

The following information was collected from MarineTraffic.com from ships within the study area on May 15, 2021: name/number, type, length, draft, location, and speed. Not all of this information was readily available, so if the vessel had only a name listed, further information about it was searched online. Several vessels were only assigned numbers, types, and locations. For these cases, data were interpolated from known vessel parameters. Distributions of lengths, drafts, and speeds of each ship type were generated. Then, the standard deviations of these distributions were used as limits in a random number generator (Thijssen 2013) to fill in the needed parameters for each vessel lacking such information. In this way, all variables needed for the JOMOPANS-ECHO model (MacGillivray and Jong 2021) could later be calculated from a range of realistic values.

## Construction of Predictive Maps

For the random and deepwater maps, new shipping lanes were drawn and ships were placed along these routes. The new hypothetical shipping lanes were assumed to have destinations that would serve the largest and closest international ports of surrounding countries and continents. These five destinations

included: the Panama Canal, the Port of Long Beach (California, United States), Chinese ports like Shanghai or Shenzhen, the Port of Buenaventura (Colombia), and the Port of Buenos Aires (Argentina).

To estimate the number of ships that could be leaving and entering the hypothetical Port of Tribugá, the LCA (Logistics Capacity Assessment) tool was used to obtain the count of ships entering the port of Buenaventura in 2019 (the last year in which the count was estimated), giving a total of 1458 entrances. This total, when divided by the 365 days of the year, gives a maximum total of 4 ships per day entering the port of Buenaventura (Colombia port of Buenaventura 2019). In comparison, the Port of Long Beach in 2019 had a total of 1867 vessel arrivals, so 5.11 boats per day (Port of Los Angeles 2020).

The Port of Tribugá is estimated to handle roughly an average amount of ship arrivals ranging between those at the ports of Buenaventura and Long Beach. Therefore, it was assumed that over 5 days, approximately 5 boats per day would enter and 5 would exit Tribugá. The total number of ship entrances and exits over each 5-day span was actually set at 47 ("almost" 5 per day) with the following weightings: 10 ships would be in transit to/from China, Long Beach, and South America (30 ships in total); 9 ships would be in transit to/from the Panama Canal (since there is a limit to how many ships can pass through the Panama Canal), and 8 ships would be in transit to/from Buenaventura (based on the Colombia port of Buenaventura 2019).

Because sound waves travel well in deepwaters, especially if they can enter the SOFAR channel, a deepwater map was drawn to route vessels over the deepwater trenches that spread out from the Gulf of Tribugá. The rhetoric around the port consistently cited the deepwater trenches being a reason for building the port so that the area would not need to be dredged as commonly as the Port of Buenaventura does (Barrios and Plazas 2015). For this reason, the deepwater shipping lanes may be a more reasonable hypothetical map than the random map's shipping lanes. To do this, ships were placed into lanes over the deepwater trenches in route to the aforementioned five destinations until they had to shift out from over a trench to a more direct path. The spacing between ships was evenly drawn between ports of call and was similar to the current spacing of ships along the route from the Panama Canal to the Port of Buenaventura using marinetraffic.com. However, spacing between ships over the deepwater trenches was also shortened as compared to the control map to account for the higher density of ships given the additional 47 in this scenario.

As described above, it was assumed during the construction of the random map configuration that roughly 5 vessels would depart and arrive each day, with roughly one going to/coming from each of the 5 assumed destinations. The same additional 47 ships were added to the study area, starting with the deepwater configuration. Nonetheless, to increase the sense of randomness with the random map, several ships were moved from a shipping lane and placed in other areas of the study area. For example, one of these may be a supply ship diverting to the Galapagos.

The randomization decisions were made intentionally to move ships into areas of the ocean that might be void of large ship noise between the deepwater model lanes to make it as different as possible to that scenario. Finally, the locations of

every ship in each map were downloaded as three .kmz files for later ingestion into the acoustic propagation model and GIS software.

## Calculating Source Level for All Ships in All Maps

Once all variables were obtained for use in Eqs. (1) and (2) of the JOMOPANS-ECHO (J-E) model of MacGillivray and Jong (2021), source levels were calculated. The J-E model uses a reference speed per vessel class measured in knots ($V_c$) so our maps used only vessels in these classes.

Equation (1) in the J-E model solves for source level:

$$LS_f\ \ J-E(f,\ \ V,\ \ l\ \ ,C) = Ls_{f,0}(f,C) + 60\log_{10}(V/V_c)\,dB + 20\ \ \log_{10}(l/l_0)dB \tag{1}$$

where $LS_f$ refers to the spectral density of the source level, $f$ represents frequency in Hz, $V$ indicates velocity in knots, $l$ refers to the length of the ship in ft. (feet), $l_0$ equals 300 ft., $C$ represents ship class, and $V_c$ indicates reference speed (MacGillivray and Jong 2021).

However, to solve Eq. (1) one must first calculate $LS_{f,0}(\hat{f}, C)$, using Eq. 2. Eq. 2 states that $\hat{f} = f/f_{\text{ref}}, \hat{f} = 480$ Hz * $(V_{\text{ref}}/V_c)$, $f_{\text{ref}} = 1$ Hz, $V_{\text{ref}} = 1$ kn, $K = 191$ dB, and $D = 3$ for all classes except $D_{\text{cruise vessel}} = 4$. Considering the above values, proceeded to solve Eq. (2):

$$Ls_{f,0}\left(\widehat{f,C}\right) = K - 20\ \log_{10}\left(\hat{f}_1\right)\,dB - 10\log_{10}\left(\left(1-\frac{\hat{f}}{\hat{f}_1}\right)^2 + D^2\right)\,dB \tag{2}$$

where $K$ stands for the average sound level of a ship and $D$ stands for ship class (MacGillivray and Jong 2021). In other words, $K$ is a typical source level (SL) for many ships and the rest of the equation is a large extension of "20 $\log_{10}$ ($\hat{f}_1$)," which represents spherical spreading. Then, with the result of the calculation for $LS_{f,0}(\hat{f},\text{C})$ from Eq. (2), it leads to the calculation of Eq. (1) and thus to the generation of the source level of each ship based on aspects unique to each ship. The $f$ (or frequency of choice) that we used was 350 Hz because this is a known peak frequency used in common song units of Stock G (Perazio and Mercado III 2018), and since it's also a common harmonic in ship noise, masking could occur in this band.

## Calculating the Acoustic Propagation Models

With source level (SL) calculated for each ship in each map using the J-E model, we could use maps of transmission loss (TL) for each to make maps of received levels (RLs) at each point in the study area. It is of note that the additional 47 ships in the deepwater map are the same as the additional 47 ships in the random map so that

differences in noise area were not a result of varying ship types, but only a result of ship locations in the study area. RLs were calculated to be the sound levels from combined ambient and ship noise that humpback whales would be experiencing if they were located in a particular spot in the study area grid. The TL fields were computed with the acoustic propagation model "Seahawk," a split-step Padé parabolic eq. (PE) model (Heaney and Campbell 2016) developed by Applied Ocean Sciences.

The PE model requires ingestion of the following variables: water column temperature and salinity (taken from a CTD cast or from World Ocean Atlas (Boyer et al. 2018), the latter of which was used in these models) (Locarnini et al. 2018), bathymetry, sediment type, depth of cargo vessels in the water, average humpback whale swimming depth, a frequency band of interest, and locations of sound sources (taken from marinetraffic.com) (Rey-Baquero et al. 2021; Soares et al. 2015; McKenna et al. 2012).

This study used the following values for these variables:

Because it is assumed that other marine mammals are like humans and sense acoustic energy in a logarithmic way, the frequencies between 300 and 400 Hz that were incoherently summed together to represent this audible band of boat noise were not linearly spaced; rather they were the decidecade bands that are non-linearly spaced (Miksis-Olds et al. 2021). The exact central frequencies of the decidecade bands that acoustic transmission loss was calculated and incoherently summed for were 300, 308, 317, 327, 336, 346, 356, 366, 377, 388, and 400 Hz. The noise from the boats was modeled to be emitted at a depth of the hauls from each vessel that had such information in the MarineTraffic.com database or were determined by random draw from a realistic distribution of haul depths. Finally, because the part of the water column where humpback whales are usually found singing and breathing is near the surface (Pack et al. 2017), the received levels at depths of the receivers in the model were incoherently summed every 10 m from 0 to 30 m.

These boat-specific, whale-specific, and environmental parameters from Table 1 were fed into the parabolic equation model (Applied Ocean Sciences) for each ship. The transmission loss fields of noise from all ships were then incoherently summed across the entire study area. The form of the SONAR equation used was:

$$\mathrm{SL} - \mathrm{TL} + \mathrm{NL} = \mathrm{RL}$$

where SL is the Source Level of the boats, TL is the Transmission Loss of each boats' noise, NL is the Noise Level of the ambient acoustic environment, and RL is the resulting Received Level field of acoustic energy across the study area. For NL, the value from a 2018 passive acoustic monitoring study in the Gulf of Tribugá that measured median ambient noise levels (NL) was used: 48 dB re 1 $\mu Pa^2$ (Rey-Baquero et al. 2021). This uniform background level was applied to the calculations after the TL was calculated for each ship's SL and incoherently summed to create a single TL field of all the ships combined.

This resulted in a single field of RL values across the study area. The RL field can be interpreted as the amount of sound that a given humpback whale within the top

**Table 1** Variables used in the acoustic propagation model

| Variable | Value | Rationale |
|---|---|---|
| Frequency | 300–400 Hz | Peak song frequency is 350 Hz; a band avoids pure tone so is more realistic |
| Humpback whale depth | 0–30 m | Typical part of the water column occupied by humpback whales (Pack et al. 2017) |
| Bathymetry | GEBCO 2019 | Best bathymetry map available for the region |
| Sound velocity profiles (water column temperature and salinity) | World Ocean Atlas | Best available database of sound velocity profiles |
| Sediment grain size | $\phi = -2$ (Wentworth table) | Value representing an average of sand/stony bottom, known as common bottom type from SCUBA surveys in Tribugá |
| Propagation range | 400 km | Likely range to capture most of the TL for a SLs of the ships |
| Ambient wind noise | 48 dB re 1 $\mu Pa^2$ | Average ambient wind noise for the Gulf of Tribugá recorded in Tribugá in 2018 (Rey-Baquero et al. 2021) |
| Receiver depths | 0–30 m every 10 m | Pack et al. (2017) |

30 m could receive from the collection of ships within a detectable range around it in the study area for a particular shipping lane configuration. Finally, this gridded RL field, a.k.a. a georeferenced matrix representing the amount of sound (in dBs) from a combination of ship noise and ambient sound levels, was exported as a NetCDF file to be ingested into GIS software.

## Determining the Audibility Threshold

It is important to note that in the gridded map of RLs, not every pixel in the map would have levels high enough to be heard by a humpback whale because they ranged down to 48 dB (baseline ambient noise but below the audibility threshold). For that reason, the pixels in the maps were color-coded as either "inaudible" or "audible" (depending on the audiogram) to best interpret how much area in which a humpback whale could detect ship + ambient noise. Published audiograms of humpback whales were used to establish these audibility thresholds to delineate audible vs inaudible (Houser et al. 2001; Erbe 2002). While 300–400 Hz was the bandwidth used in the PE model and considered in the audiograms, it was not completely compatible with the two humpback audiograms (Fig. 3). Therefore, this study considered two humpback whale audibility thresholds (51 dB and 80 dB) to add nuance to the interpretation of the maps. The 51 dB threshold is more conservative (because it is a lower threshold of audibility) and would assume that more whales would have better hearing and be more capable of detecting lower levels of ship noise. The 81 dB threshold is less

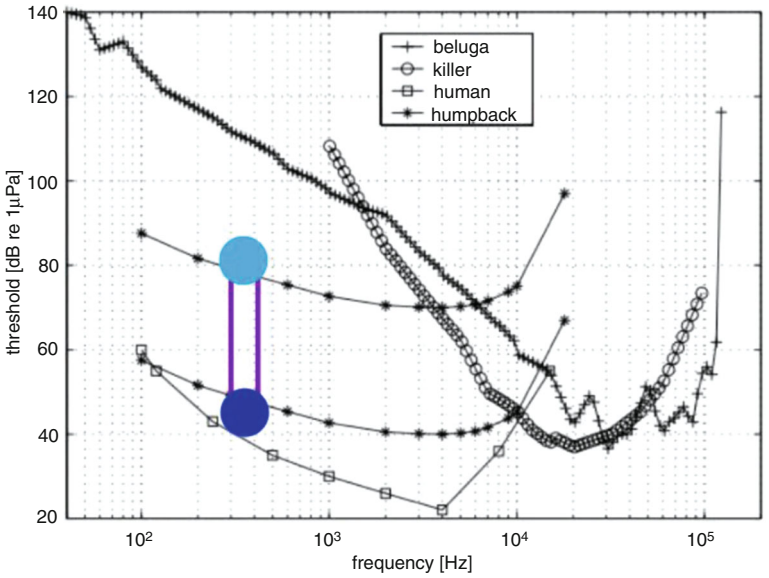



**Fig. 3** The humpback whale audiograms (asterisk lines) from Erbe (2002) based on humpback whale anatomical data and predictive models. The purple box indicates the thresholds that were considered for the 300–400 Hz frequency band under consideration. Shared with permission of Cristine Erbe

conservative. At this higher audibility threshold, it would assume an average whale had less sensitive hearing and therefore would be less likely to hear lower levels of ship noise. It is likely that the actual average audibility threshold of humpback whales lies somewhere between these two values.

## Evaluating Received Level Maps in ArcGIS

ArcGIS was used to ingest the results from the PE models and to calculate spatial parameters of the study area. ArcGIS is a geographical information system (GIS) that allows for handling and visualizing geographical statistics through layer-building maps (Shaktawat 2020). This tool was necessary to visualize and to calculate the audibility range category for each configuration. First, conversions were made to ensure accurate calculations. To calculate distance, bearing, and to convert the grid pixels to square area, the Movable Type Scripts (MTL) website was used, taking specific latitude and longitude points. According to MTL, the distance between the northern and southern edges of the study area was 1541 km, and the distance between the eastern and western edges was 1188 km (Chris 2022). Using these geographic reference points, it was possible to determine how many kilometers from N-S and E-W each pixel represented throughout the study area.

To convert .nc files to raster .tiff files, Qgis was used. Then, the coordinate system was defined in the "Transverse Mercator" projection. By using this coordinate system projection on a georeferenced NetCDF file, it was possible to calculate how many square kilometers each pixel represented. All .tiff files were uploaded to ArcGIS to find the unique values (pixel count per value) for the three selected audibility ranges (48–50.9 dB, 51–80 dB, and 80.1–123.4 dB) for each .tiff file. A spatial analysis reclassification was then carried out to vectorize the raster files into polygons. Each pixel in the raster file did not represent the same spatial area, but the conversion from pixel to square kilometers, as described in the previous paragraph, was done to accurately calculate the total area of the audibility ranges in each map so they were comparable to each other. Then, the pixels in each audibility range category were summed to calculate the total area occupied by each audibility category. These final spatial areas in each audibility class were then compared across the three maps.

## Maps of TL and RLs of Shipping Lane Configurations

A total of 79 vessels transited inside the study area on May 15, 2021. Of all 79 ships, the ship that produced the highest intensity source level was a merchant ship named Tianjin with 162.1 dB re 1 μPa @ 1 m, followed by a natural gas tanker (LNG) with an SL of 157 dB re 1 μPa @ 1 m. In third place was Conti Contessa, a merchant ship with an SL of 156 dB re 1 μPa @ 1 m. The ship with the least intense SL was number 42 with 122 dB re 1 μPa @ 1 m. The only merchant ship with (barely) enough energy to pass the "behavioral change" threshold (160 $dB_{rms}$) in marine mammals was "Tianjin," whose SL at 162 dB would have attenuated below 160 dB rather quickly.

The spatial area of noise from all of these 79 ships was depicted in planview (bird's eye view) maps. The .nc files from the RL matrices were projected in Google Earth to visualize the area that each boat's noise occupied and how noise from nearby boats combined. Notably, the control map showed a more pronounced accumulation of sound energy near the coastline. Sound energy was represented on a jet color scale with warm colors such as red, orange, and yellow representing relatively louder zones, while areas with cool colors such as dark blue and light blue represented less acoustic energy and therefore being relatively quieter. Additionally, the dark blue tones were placed in the matrix for <48 dB (or average ambient noise levels). These dark blue areas are all far away from any sound sources, since they are the portions of the study area where energy from the ships had significantly attenuated. Dark blue areas or rings close to the ships were likely over deepwaters as they are places where sound waves were not in the top 30 m of the water column, but rather were traveling in deeper waters until reflected off the bottom or other bathymetric feature to return again to the surface. It is this reflection of sound waves between the surface and bottom that creates the rings of sound energy in our planview maps, since only acoustic energy in the top 30 m of the water column is shown. The reflection paths, however, can be seen in

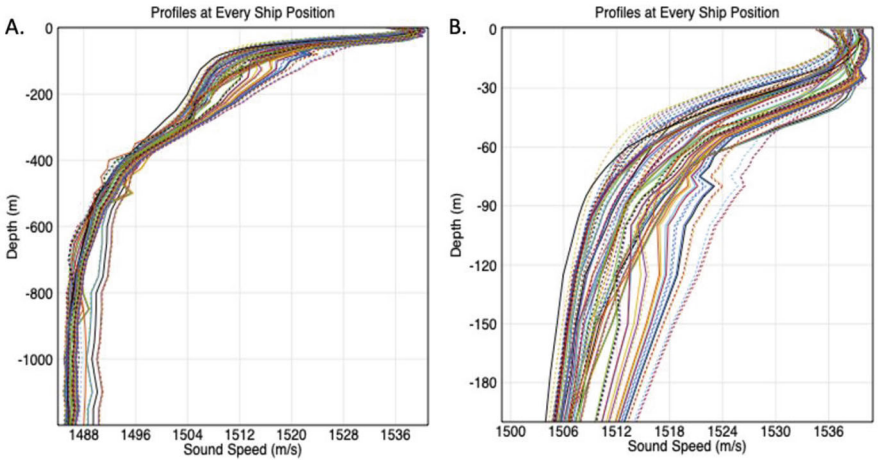



**Fig. 4** Sound speed profiles (SSPs) from World Ocean Atlas for vessel positions taken from marinetraffic.com from the surface until (**a**) 1200 m depth and (**b**) until 200 m depth. (Courtesy of John Boyle)

the slice view figures, such as Fig. 7b. In order to compare sound propagation patterns between ships, the sound speed profiles at the location of each vessel were plotted (Fig. 4). The sound speed profiles plotted on top of one another show not only the range in profiles across all 79 vessels, but also a clear pattern of how sound travels as the water column deepens over the study area. In general, the sound speed is about 1544 m/s at the surface and reaches a sound speed minimum of about 1482 m/s around 800 meters that stabilizes until the ocean bottom. When looking closely at the top 30 m, there is a sound speed maximum of about 1538 m/s near 15 m deep, but then sound speed quickly slows down until 90 m deep. It is between 90 and 200 m that the most variability in sound speed occurs over the study area. This 30 m sound speed maximum is interestingly the layer where humpback whales tend to sing between breaths (Pack et al. 2017). This layer of the ocean, in the study area, where sound travels the fastest, possibly creates a slight surface duct. This means that humpback whale songs could travel most effectively at the depth at which males sing. Ship noise, however, also travels effectively in the same layer, meaning masking in this layer could be particularly problematic.

## Predictive Maps

For the two simulated maps (random and deepwater) the highest and lowest source level of any simulated ship was 161.3 dB re 1 μPa @ 1 m and 122 dB re 1 μPa @ 1 m, respectively. Furthermore, while the source levels of the simulated set of boats were the same, their positions changed. The two configurations are shown as

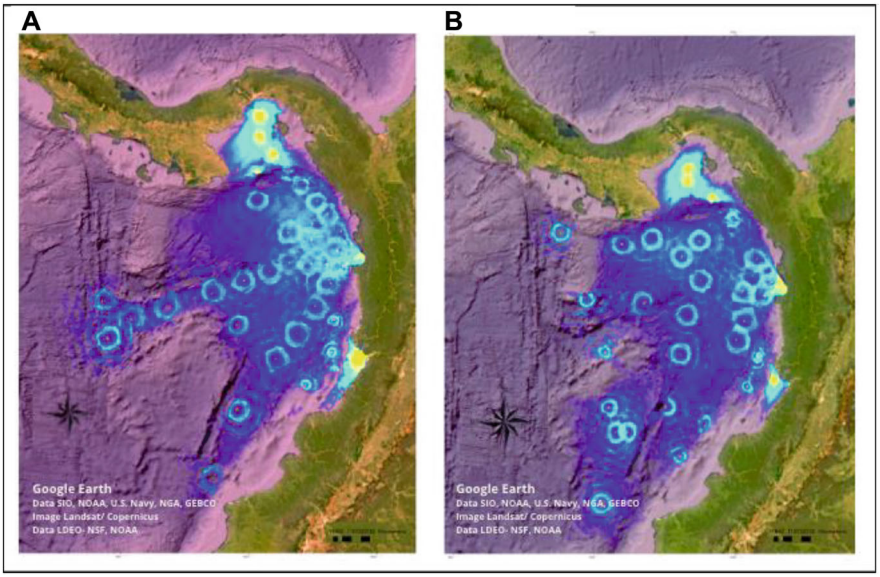



**Fig. 5** Predictive maps of the planview modeling results of two simulated shipping lane configurations: deepwater (left) and random (right). The colors represent the relative intensity of the sound around each ship. Both maps only represent frequencies at 300 Hz at 30 m depth and are without the effect of the wind to create a better visualization of the distribution of the ships. (Background image source modified from Google Earth)

planviews in Fig. 5. Recall that the 47 simulated ships in these two predicted shipping lane configurations were added to the 79 original ships from the control map. As the results will show, the addition of this many ships makes the overall noise area significantly larger in both simulated configurations as compared to the control. This increase is realistically what could happen if a port were ever built, so the comparison of more future ships to fewer current ships is a valid concern for conservation groups, management groups, policymakers, and the like.

## Interpreting Transmission Loss Figures

The three following figures show the main ways that transmission loss (TL) occurred from various ships with different source levels (SLs) in different spots along the shipping routes. The scale is in decibels lost during transmission, thus the negative values. Warm colors(−40 dB to −60 dB) represent sound levels with higher intensity because they have traveled less distance and/or attenuated less, therefore have experienced less transmission loss. Each figure represents a different TL planview (left-side subplots) and sliceview (right-side subplots) to exemplify some interesting examples of how bathymetry, temperature, and pressure combine to propagate or attenuate sound in various locations across the study area. The three

chosen TL planview and sliceview figures illustrate how sound can accumulate in shallow areas closer to the coast compared to the greater dissipation of sound as it travels to the depths in areas farther from the coast. While bottom sediment can rapidly absorb sound waves, shallow waters can also reflect sound waves between the bottom and surface more often, further ensonifying the water column over shorter distances until the absorption into the bottom wins out.

Because sound waves from Ship 30 went into the depths and were entrapped in the SOFAR channel quickly, its TL planview does not exhibit a typical concentric ring structure as many planviews of ships do (like Fig. 7a). Rather, Ship 30's emitted sound meets the seamounts, which have depths exceeding a little more than 2900 meters (Fig. 6b). The sound waves being mostly blue in color by 40 m from Ship 30's location show about 80 dB of TL or a rather rapid attenuation of the emitted energy with distance compared to ships in shallower waters of the study area. Even after 5 km from the point of emission, at a depth of 400 m, the noise has attenuated by 42 dB (warm colors), and by 15 km from the point of emission, TL is 77 dB (cold colors). Reasons for this could be (1) the amount of distance sound waves are traveling to reach the ocean bottom create space over with to be attenuated, (2) the type of sediment found at depth could be highly absorptive, and/or (3) temperature below about 400 m is colder than at the surface so sound travels slower and has more time to dissipate its energy.

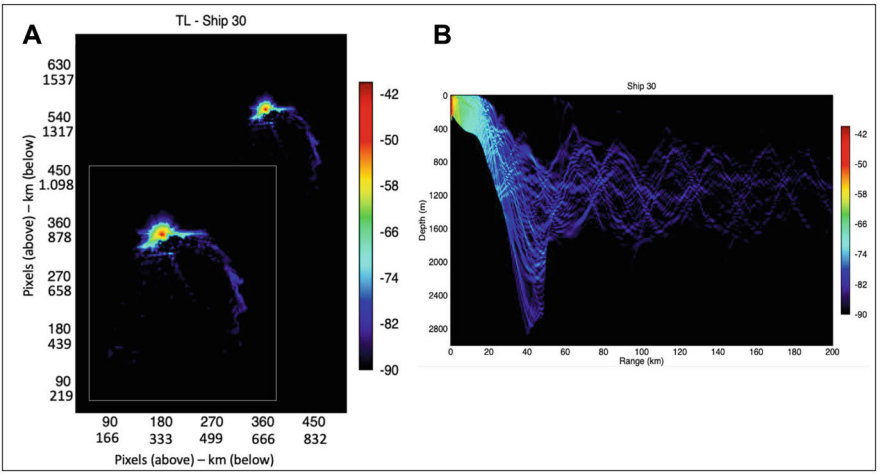



**Fig. 6** A deepwater subsurface duct example from Ship 30 as an incoherent average of the TL in the 300–400 Hz band. TL planview (A-left; zoomed out and zoomed in) that displays the top 30 m of these oscillating reflections, and the TL slice (B-right) that runs 180 degrees from the vessel position (north to south). Note the reflection off of the ocean bottom (blue rays in the sliceview) that bounce off the black bathymetry contour and then become entrapped in the SOFAR channel about 1 km deep. The blue tail in the planview is likely rays reflecting off the bottom that do not become entrained in the SOFAR channel and reach the top 30 m of the water column throughout their reflections. Units in the color bar are transmission loss (TL) in dB re 1μPa$^2$ where warmer is less TL and cooler is more TL

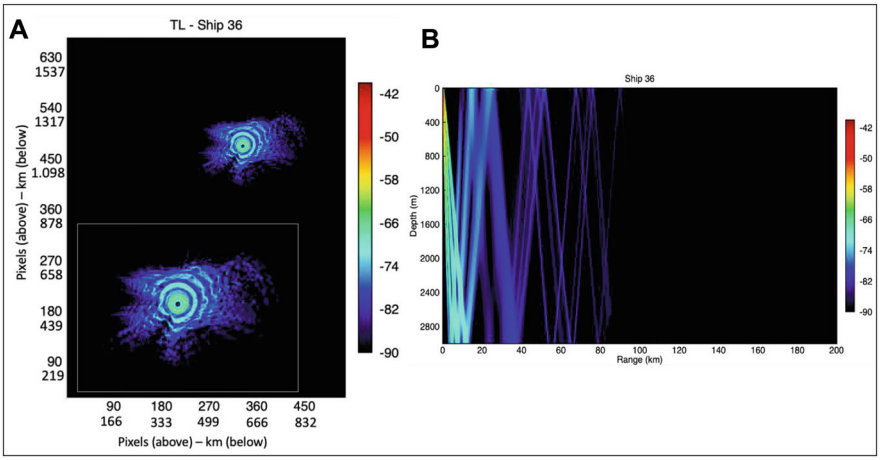



**Fig. 7** A deepwater bottom-to-surface reflection path example from Ship 36 as an incoherent average of the TL in the 300–400 Hz band. TL planview (A-left; zoomed out and zoomed in) displaying the top 30 m of the water column, and TL sliceview (B-right) going south from the ship's location. Note the propagation of sound around the boat is similar in many directions (sound propagation rings in the planview). Also, note the reflection of sound waves as they strike the surface and bottom of the ocean (blue rays in the sliceview). Units in the color bar are transmission loss (TL) in dB where warmer is less TL (less negative values) and cooler is more TL (more negative values)

Figure 7 illustrates how the sound waves of a ship at Ship 36's location propagate from the surface immediately to the sea floor at 3 km deep. Over these 3 km, from the first surface to bottom transit, about 70 dB are attenuated. When reflecting off the bottom, it is noted that another few dB are lost, and depending on the type of bottom sediment at different locations, this absorption of sound energy could be more or less, causing sound energy to dissipate faster or slower. Over the next four reflections against the surface and bottom, the TL is only about another 10 dB, so more attenuation is occurring at the beginning near the ship and less attenuation is occurring at the end of the sound wave's path, farther away from the ship.

Acoustic propagation from most of the ships followed either of these two examples (Figs. 6 and 7): getting entrained in a subsurface duct or reflecting between the surface and bottom. The final way sound propagated from ships across the study area was in shallower waters (Fig. 8). Figure 8 illustrates how the sound waves of a ship at Ship 04's location propagated from the surface immediately to the sea floor at 50 m deep.

## Final Maps of Transmission Loss (TL) and Received Levels (RLs) from Control and Simulated Shipping Lane Configurations

The final maps of the random and deepwater simulated configurations (Figs. 10 and 11) show the positions of the ships and their noise prints in TL (A-left subplots) and in RLs with the 48 dB NL included (B-right subplots). The map

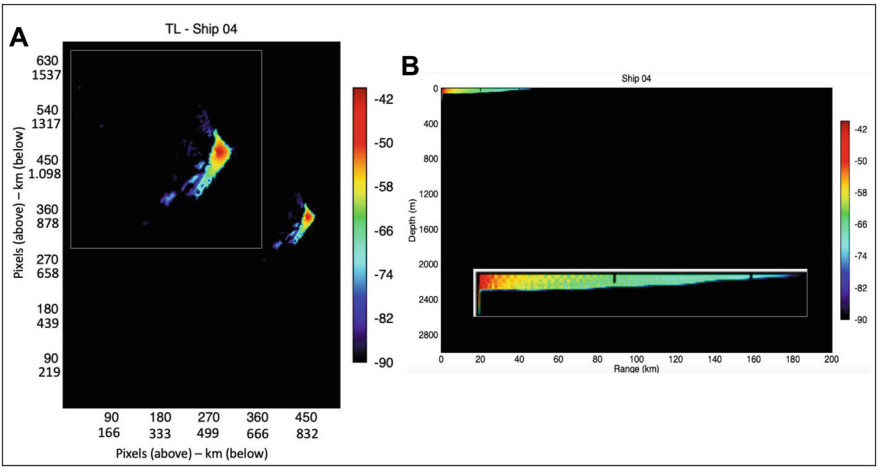



**Fig. 8** A deepwater bottom-to-surface reflection path example from Ship 04 as an incoherent average of the TL in the 300–400 Hz band. TL planview (A-left; zoomed out and zoomed in) displaying the top 30 m of the water column, and TL sliceview (B-right) going south from the ship's location. Note the propagation of sound fills up the shallow water column completely from surface to bottom (**b**) and toward the east until the coastline, but sound waves likely drop below 30 m and continue toward the west (**a**). Units in the color bar are transmission loss (TL) in dB where warmer is less TL (less negative values) and cooler is more TL (more negative values)

format makes it easy to see how ship noise propagates throughout the study areas in the Colombian Pacific Ocean. In the TL maps (A-left), the red color(−30 dB to −45 dB) represents the levels with less loss (i.e., they are more audible), while the blue color(−85 dB to −90 dB) represents the levels with more loss (i.e., they are less audible). In other words, the areas with blue shades(−85 dB to −90 dB) represent areas where the whales would likely be unaffected by the noise emitted by the ships because the received levels are below their audibility threshold. It is in these blue areas that we can assume they may communicate more effectively, assuming no other sources of interruption exist.

To determine exactly which colors represent sound levels that are and are not audible to whales, the SONAR equation must include noise level (NL) to create a more realistic map of RLs. The resulting map of RLs (Fig. 9b) represents the incoherent summation of ship noise with the 48 dB of average ambient sound known to be a typical sound level in the Gulf of Tribugá at Morro Mico (Rey-Baquero et al. 2021). Therefore, this map shows the total modeled sound energy from ships and ambient noise levels that the whales could receive at each point in the study area. In some places, the ambient sound is louder than the attenuated noise that traveled long distances from the ships. Nonetheless, over a large portion of the map, the ship noise is louder than (or in excess of) the ambient noise (MacGillivray and Jong 2021). It is important to say while there may be a significant amount of sound energy at depth, it is not represented in these maps, so the concentration of sound energy along

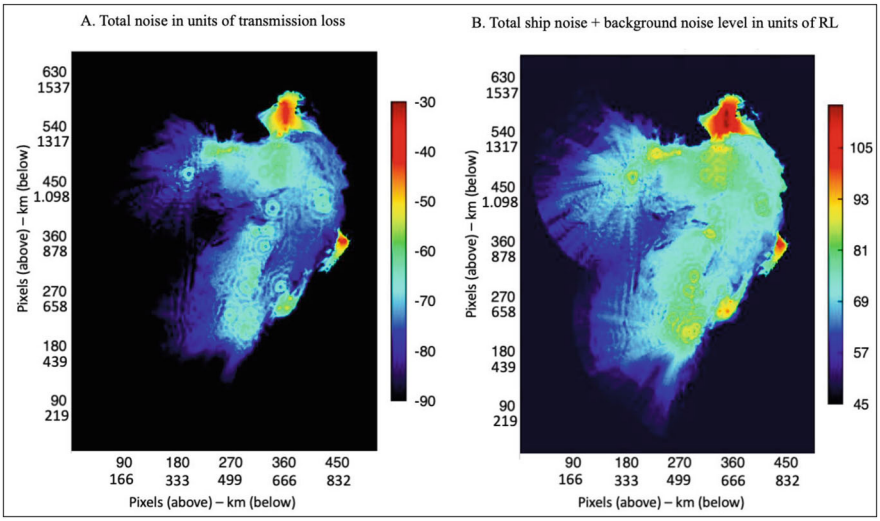



**Fig. 9** Planview maps for TL in dB re 1 μPa$^2$ (left) and RL in dB re 1 μPa$^2$ (right) of noise emitted by 47 ships of the control map in the Colombian Pacific Ocean study area

the coastlines is only in the area of the water column that literature shows humpback whales tend to swim and sing in (0–30 m in this study).

Warmer colors represent less transmission loss and higher intensity sound, while cooler colors represent more transmission loss and less intense sound. Figures made in the parabolic equation modeling software owned by Applied Ocean Sciences. The color scale used in the subplot (Fig. 9b) is 45–110 dB, so places without any noise from the ships in our models will appear black, making them easier to see. Capping the range at 110 dB allows the noise level of 124.5 dB to be dark red, making the highest ship noise levels more visually obvious. Maps are of the full 30 m surface summation and integrated over 300–400 Hz bandwidth. The same TL and RL planview maps (an integration of sound levels only in the top 30 m of the water column) were made for the deepwater trench configuration and the RL noise field can be seen in next Fig. 10. The most intense noise levels (91–123.4 dB) in the deepwater maps were found along coastal areas such as Buenaventura and the coastal strip near the Panama Canal (Fig. 10c). The central area of the study area tended to have more intermediate noise levels (73–90 dB) in yellow and greenish hues, while the areas farthest away from the coasts held the lowest noise levels (45–72 dB), shown in dark blue and light blue hues (Fig. 10c).

The second hypothetical configuration was the "random map" (Fig. 11b), which adjusted the 47 additional ships from the deepwater configuration to be more randomly arranged within the study area. As before, the control map (Fig. 11a) was combined with the additional 47 ships of the simulated random configuration (Fig. 11b) and a flat ambient noise level to create the combined total noise field of the projected random configuration (Fig. 11c). Ships in this

configuration tended to be closer to the coast (Fig. 11b). In this random configuration, the highest noise levels (91–123.4 dB) accumulated in areas closer to the shores (Fig. 11c). The central area of the study area tended to have more intermediate noise levels (73–90 dB) in yellow and greenish hues, while the areas farthest away from the coasts held the lowest noise levels (45–72 dB), shown in dark blue and light blue hues (Fig. 11c).

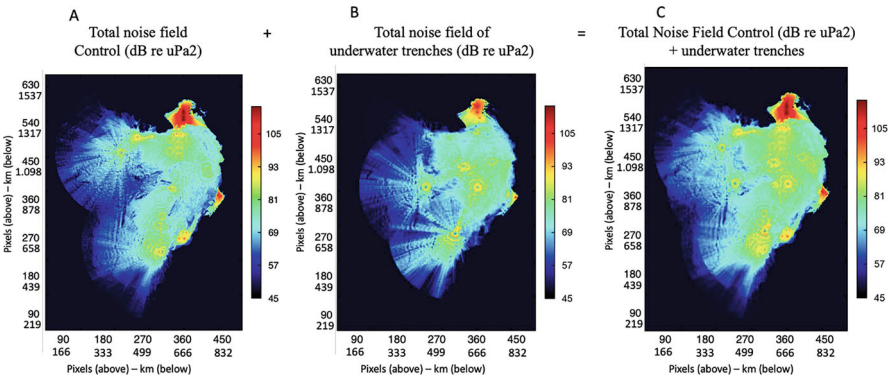



**Fig. 10** Total noise field of deepwater and control configuration. (**a**) The control configuration in RL (dB re 1 μPa$^2$), (**b**) total noise of 47 ships in the deepwater simulated configuration, and (**c**) the incoherent sum of the noise from the ships in the control map plus the noise from the 47 additional ships. Figures made in AOS' PE software. Color scale in dB of RL and include NL = 48 dB re 1 μPa$^2$

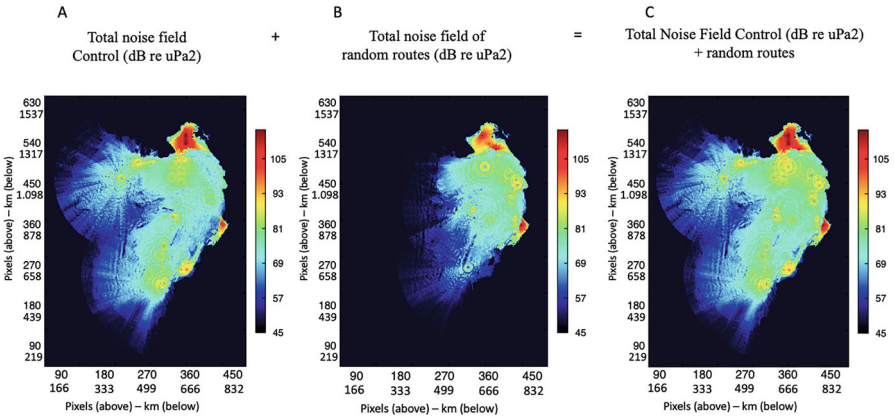



**Fig. 11** Total noise field of random and control configuration. (**a**) The control configuration in RL (dB re 1 μPa$^2$), (**b**) Noise map of 47 ships randomly arranged around the Colombian Pacific Ocean, and (**c**) the incoherent sum of the noise from the ships in the control map plus the noise from the 47 additional ships of the random configuration. Figures made in AOS' PE software. Color scale in dB of RL and include NL = 48 dB re 1 μPa$^2$

The main visual difference between subplots A and B in Fig. 9 is that the circles do not appear as clearly in Fig. 9b. This is because the ambient sound contribution at all locations has a received level of 48 dB, so the RL map has a more diffuse effect that is more realistic as it includes an average amount of ambient noise throughout the environment. On the other hand, the primary spatial comparison between the control map and the deepwater and random maps come from the fact that the control map (Fig. 10a) is added to both (Figs. 10b and 11b) to simulate current shipping traffic plus future projected traffic if there was a port (Figs. 10c and 11c).

## Scenario of Audibility Zones

Before comparing the spatial extent of all noise levels in the three maps, audibility ranges were delineated. Figures 12 and 13 are visual representations of the audibility ranges with color indicating audibility zones considering the humpback whale audiograms from the literature (Houser et al. 2001; Erbe 2002). Red represents the portions of the study area with the highest audibility (80.1–123.2 dB) where the possibility that whales can detect ship noise is high. Dark blue indicates portions of the study area that contain noise levels between the two published audiograms for the bandwidth in question (51–80 dB). This is an audibility range where humpback

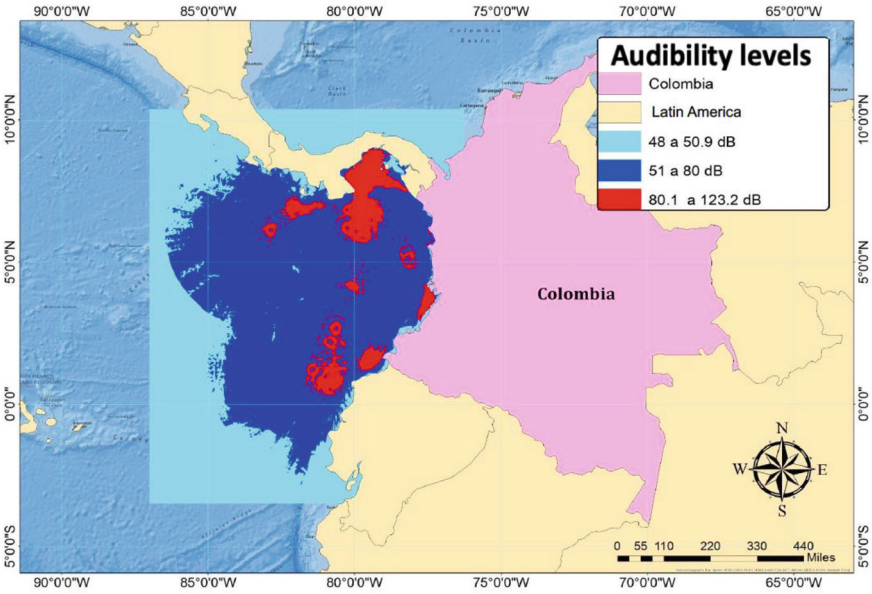



**Fig. 12** Map of received levels grouped by audibility ranges in the study area for the control map configuration. Minimum audibility/lowest sensitivity threshold (80.1–123.2 dB) in red, intermediate audibility/medium sensitivity threshold (51–80 dB) in dark blue, and maximum audibility (48–50.9 dB)/highest sensitivity threshold in light blue

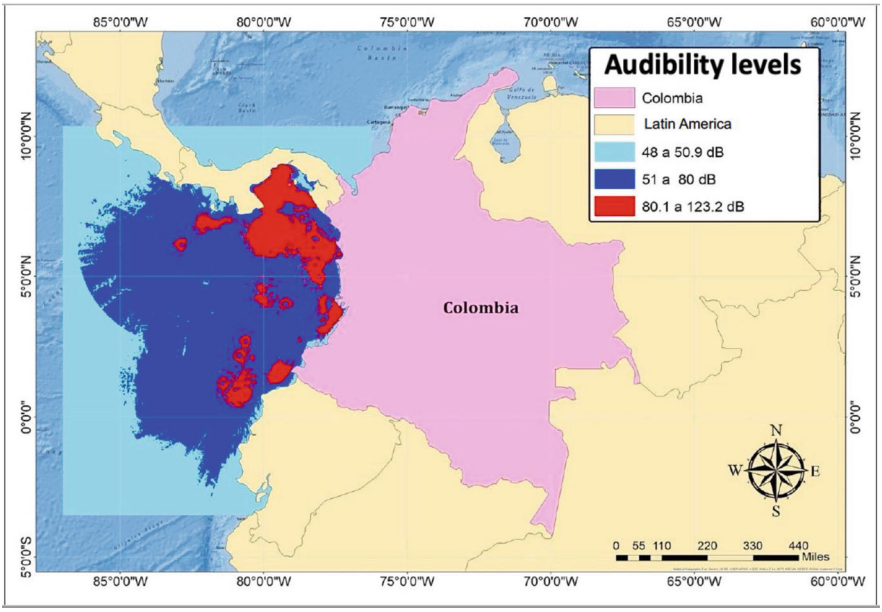



**Fig. 13** Map of received levels grouped by audibility ranges in the study area for the random map configuration. Minimum audibility/lowest sensitivity threshold (80.1–123.2 dB) in red, intermediate audibility/medium sensitivity threshold (51–80 dB) in dark blue, and maximum audibility (48–50.9 dB)/highest sensitivity threshold in light blue

whales could still hear the ship noise, but less certainly so, or where fewer could. This is also the portion of the study area that is very susceptible to change, as ship noise could easily shift to higher audibility levels with the addition of just a few more ships, ships with louder source levels than considered in this study, or from slightly different routing over areas with more effective propagation. The portions of the study area in light blue correspond to the most sensitive audibility threshold (48–50.9 dB) for humpback whales. It is possible that some whales could still hear the shipping lane traffic noise in these portions of the study area, but only those individuals with particularly sensitive hearing.

Similar to the noise field planview for the random configuration, the red areas that fall within the highest audibility range (the areas where humpback whales with even low hearing sensitivity could probably detect boat noise) are closer to the coasts with an accumulation of noise in areas closer to the Panama Canal, the port of Buenaventura, and southern coastal areas (like Tumaco and Ecuadorian coastal areas) (Fig. 13).

The deepwater configuration, as predicted from the planview noise field (Fig. 14), has portions of the study area that surpass the highest audibility threshold (reddish colors from 80.1 to 123.2 dB), particularly toward the center. The central portion of the study area actually tends to have a noisy channel that is mostly filled in between the Port of Buenaventura and the Panama Canal.

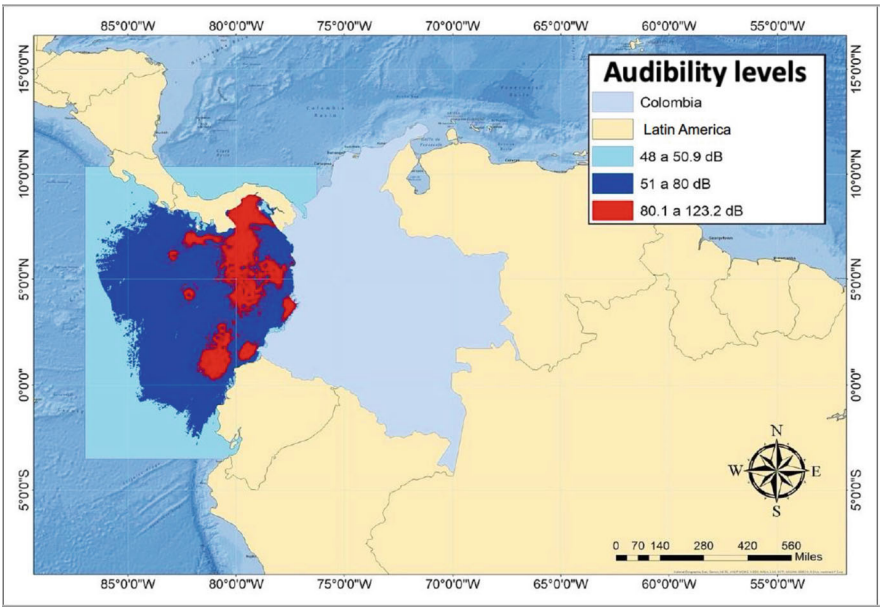



**Fig. 14** Map of received levels grouped by audibility ranges in the study area for the deepwater trenches map configuration. Minimum audibility/lowest sensitivity threshold (80.1–123.2 dB) in red, intermediate audibility/medium sensitivity threshold (51–80 dB) in dark blue, and maximum audibility (48–50.9 dB)/highest sensitivity threshold in light blue

**Table 2** Areas associated with the three audibility ranges of humpback whale hearing for all three map configurations

| | Area in $km^2$ | | |
|---|---|---|---|
| Audibility ranges in RLs (dB re 1 $\mu Pa^2$) | Control | Random | Deepwater trenches |
| 48–50.9 | 1,086,991 | 1,084,276 | 1,070,432 |
| 51–80 | 676,602 | 629,083 | 615,889 |
| 80.1–123.2 | 89,734 | 139,965 | 167,003 |

The calculations for spatial coverage of each colored (each audibility range) polygon in each of the configurations (random, deepwater, and control) are presented in Table 2. The total area for all the configurations was 1,854,500 $km^2$. The control configuration yielded the largest area (1,086,991 $km^2$) for the lowest audibility range (48 at 50.9 dB) which is thought to be a level too low for most humpback whales to detect sounds (Houser et al. 2001). The control configuration also had 676,602 $km^2$ with levels between 51 and 80 dB or where RLs would be relatively average for humpback whales to detect sounds. Finally, the control configuration had the smallest area (89,734 $km^2$) occupied by noise levels between 80.1 and 123.2 dB which are received levels assumed to be sufficiently loud for most humpback whales to detect sounds. The deepwater configuration had the largest area (167,003 $km^2$) within noise levels in the highest audibility range, while the random configuration

**Table 3** Spatial area percentages of audibility ranges in each of the control, random, and deepwater configurations

| | Percent area (%) | | |
|---|---|---|---|
| Audibility ranges in RLs (dB re 1 μPa$^2$) | Control | Random | Deepwater trenches |
| 48–50.9 | 59 | 58.53 | 57.76 |
| 51–80 | 37 | 33.91 | 33.23 |
| 80.1–123.2 | 5 | 7.56 | 9.01 |

had a more moderately sized area (139,965 km$^2$). This moderately sized area, however, is only 27,038 km$^2$ less than the deepwater configuration and 50,231 km$^2$ larger than the control map. Therefore, the random configuration does not maintain the control configuration's less intense noise field: it is actually closer to the more disruptive noise field of the deepwater configuration.

Table 3 shows the percentage of space in the study area occupied by the three audibility ranges of ship noise in each of the configurations. The control configuration had the highest percent area for the 48–50.9 dB audibility range at 59% of the total study area with respect to the other configurations, while the deepwater configuration had the lowest percent area at 57.76%. The control configuration also had the highest percent area for the audibility range of ship noise between 51 and 80 dB at 37%. Similarly, the deepwater configuration had the lowest percent area at 33.23%. Conversely, the deepwater configuration had the highest percent area (9.01%) of the audibility range between 80.1 and 123.2 dB (the noisiest levels), while the control configuration had the lowest at 5%.

The total area and percent of area results show that the control configuration is the most suitable for facilitating humpback whale communication, but that was expected. Intuition would suggest that placing any additional ships in the study area would increase the vessel noise print overall, so the control configuration being the best-case scenario of the three configurations was a good sanity check. However, when more ships were added, the random configuration was more suitable for humpback whale communication than the deepwater configuration, at least for the audibility ranges chosen in this study.

Adding the 47 ships to the study area resulted in a couple of interesting observations. First, for the audibility range of 80.1–123.2 dB, the deepwater configuration had almost twice (9%) the percent area being occupied by noise at these levels than the control configuration (5%). Placing 47 ships over the deepwater trenches nearly doubled the amount of area that noise from the original 79 ships produced. Therefore, a 59% increase in ship traffic led to an 80% increase in the percent area covered by noise in the highest audibility range. For the random configuration with only ~7.5% of the study area instead of 5% occupied by noise levels in the highest audibility range, a roughly linear relationship between additional ships (59% increase) and additional noise (a 50% increase) does exist. This is strong evidence that ships traveling over areas of the ocean with effective propagation input more noise to the near-surface acoustic environment than ships traveling over areas of the ocean with less effective propagation features. Even though deepwater ports

arguably do not need to dredge their shipping lanes as often (meaning less noise in the environment over time from dredging), these results show that having ships travel over the trenches may offset the noise reduction benefits associated with less dredging, as it results in noisier soundscapes.

## Propagation Models Estimations

This study estimated the noiseprints of potential shipping routes that could service the proposed Port of Tribugá if it is ever built. Using parabolic equation propagation models, areas and percent coverage of the study area were calculated for two audibility levels to better understand how increased ship noise could be perceived by humpback whales. Two predictive shipping route configurations were calculated and compared to understand the variability in propagation environments shaped by natural features that exist off the Pacific coast of Colombia. Although the proposed port would undoubtedly increase sound levels, the exact effect of factors such as the location, size, and a number of additional ships on the soundscape remains unknown but were predicted as best as possible. Modeling the noise pollution from various possible shipping lane configurations could help managers mitigate some of the noise that would inevitably result from increased ship traffic. While there is no configuration of additional ships that will prevent further ensonification of the ambient acoustic environment, it could be possible to determine which configurations are relatively less disruptive than others for humpback whales to communicate effectively.

## Understanding the Colombian Pacific Soundscape

Acoustic propagation models are commonly used to determine how sound spreads through a given environment. They can later be used to pinpoint where the noisiest and quietest parts of that area are likely located given a certain configuration of sound sources. When the ships' sound sources in the current models were close to shore, shallow coastal waters commonly became very noisy. This could be the case if the sediment is not very absorptive because sediments more absorptive than the grain size we chose would dampen the noise more quickly. In contrast, when the ship sources were over deepwater, the sound attenuated quickly, as it could dissipate in many directions, so the acoustic energy was spread out over a larger volume of water. If the sound became entrained in a submarine duct, though, it did travel very far distances in some directions from the sources. Overall, the propagation environment depends on parameters such as the sound speed profile and sediment type that allow sound to refract, absorb, scatter, or otherwise dissipate and attenuate slower or faster across the soundscape (Ainslie 2010; Erbe 2011). The results of the present investigation show that sound propagation has little attenuation near the coast of Colombia as compared to its offshore, deeper water areas. This could be because the sediment that predominates in shallow water is known to be sandy or silty from

current knowledge of SCUBA divers ($\Phi = 2$ in the Wentworth table), resulting in waves that reflect off the seafloor more than if the sediments were mud or clay, but absorb better than if the sediments were rock or limestone (Ainslie 2010; Erbe 2011).

The features of the ships also affected propagation. The sound emitted by ships is not isotropic (i.e., it does not scatter equally in all directions (e.g., Arveson and Vendittis 2000)) and not all frequencies emitted by ships are as intense as others. Since propagation depends on the source frequency of the environment in which the ships travel, if the ships' source frequencies match those of the environment, then they would naturally propagate better than others, potentially traveling hundreds of kilometers. In addition, changing speed, cargo type, size, and other ship characteristics affect the dispersal patterns of their noise (e.g., Arveson and Vendittis 2000). These complexities could explain the vertical mode transmission loss figures in the present study where sound radiated with different frequencies and directions, generating a complex and dynamic noise field. Furthermore, in an example transmission loss model, Erbe et al. (2012) also indicated that as a vessel travels over different environments from coastal to offshore waters, its noise field changes. In shallow water, the propagating noise repeatedly interacts with the surface and seafloor where it is reflected, scattered, and partially absorbed (Erbe et al. 2012). The directionality of the noise in deepwater tends to be dipolar and interactions with other sound waves are reduced by coupling quickly to the SOFAR channel, with little energy loss. This entering and propagating of sound through a submarine duct (like the SOFAR channel) was observed for some of the ships in this study. It mainly occurred over deeper waters, illustrating how the paths of the sound waves can change according to the bathymetry. With this, it can be said that humpback whales, independent of where they are, can experience quite different noise fields even at the same distance from the same ship (Erbe et al. 2012).

This discussion also illustrates why changing the position of a ship inherently changes a number of variables that affect acoustic propagation. Variables like sound speed profile, distance to shore, depth, etc. are inherently tied together in the ship position variable and cannot be teased apart without doing an intense Monte Carlo investigation that was beyond the scope of this study. For an example of determining which inherent variable encased in the position of a sound source most affects acoustic propagation within and between sites, see Sect. 6 of Heaney and Seger (2020).

## Determining the Best Potential Shipping Configuration for Humpback Whales

There is a submarine duct in the study area that was a common channel for the modeled sound to become entrained in and spread boat noise effectively at depth. According to the transmission loss propagation models (e.g. Figs. 7 and 8), initially, the direct interaction of the acoustic signal between the bottom and the surface occurred. But after a few reflections, considerable acoustic energy refracted into a subsurface channel. Conduits like these have been hypothesized to allow humpback

whales, which usually occupy shallower waters (<80 meters) (Pack et al. 2017), breathe at the surface, and then submerge and communicate with other individuals over longer distances than if they were to sing at other depths with less propagation potential. However, acoustic propagation does not discriminate between boat noise and animal sounds. The same mechanism that whales can use to project their signals will also allow noise from boats and other manmade sources to propagate far across the oceans. For this reason, the deepwater trenches configuration was included in the set of models to evaluate. It was unknown whether noise emitted by the ships aligning with the trenches would cause more sound energy to enter the deep acoustic channel or not, since depth is not the only nor strongest factor in affecting propagation distance. Therefore, the idea that depth could increase propagation distance needed to be tested against a more random configuration. Also, the deepwater trenches configuration was important to evaluate since one of the reasons that the Colombian government (Arquimedes project S.A 2015) claimed to want to build a port in Tribugá was for its deepwater accessibility (which would reduce the amount of dredging needed for maintenance of the port). Therefore, shipping lanes that followed the paths of the deepwater trenches would be a realistic simulation.

After evaluating three models (control, random, and deepwater), the "control map" configuration (Fig. 12) was the simulation with the smallest audible area covered by boat noise. This is because it was a real-world scenario where only the ships that transited on May 15th over the Colombian Pacific were considered and no simulated additional boats were added. Any port or increase in shipping activity in the area would most likely create a larger area of audible noise, so it was a good sanity check that our control model was the one with the smallest acoustic impact area. This would also be the main reason why there are such noticeable differences in the area occupied by boat noise between this control map and the two other configurations.

The two hypothetical configurations (random and deepwater) showed increases of 50,231 $km^2$ and 77,269 $km^2$ compared to the control occupied by audible boat noise, respectively. Thus, it can be said that in the Colombian Pacific Ocean, if a port were built in Tribugá, there would be an increase in the area where noise would be audible to humpback whales and, therefore, could cause a possible reduction in their communication space. Any spatial increase in boat noise could confound signals more often for a species that, during their reproductive season, rely heavily on acoustic communication. Meaning, larger noiseprints from ships could generate possible degenerative effects in their biological processes and behavior (Ovide 2017).

## Potential Effects on Humpback Whales with Respect to the Models Made

Depending on how much acoustic energy a vessel is emitting, its impact on cetaceans will vary in severity, from PTS to TTS to behavioral and masking effects. Where the noise is traveling also varies and matters. For example, when sound

waves emitted from a ship propagate through the surface or within a subsurface duct, they could be heard by humpback whales, disrupting and masking their calls and/or songs (Erbe 2011). Even if all ship noise became entrained in a deep SOFAR channel like in Fig. 6, that would move the noise out of the water column depths where humpback whales usually swim, but would move the noise into the depths where other species, like sperm whales, vocalize and communicate. Therefore, the interpretation of impacts from noise on marine mammals needs to be species-specific. Since the most common baleen whale species in the Gulf of Tribugá is the humpback whale, it was chosen as the focal species for this project. But it is unknown which other species besides humpback and Bryde's whales might be present and how many of them inhabit the areas far offshore of Colombia's Pacific Coast (Vallejo et al. 2022). Therefore, the results of this study should not be extrapolated to impacts on other species. When considering other species, new acoustic models that are tuned to their diving depths, hunting depths, audiograms, and calling frequencies would need to be made.

A physiological effect on baleen whales as a result of noise is another impact to consider. According to Erbe (2011), stress is a physiological response in which heart rate and gas exchange increase and more blood flows to the brain, thus activating a fight or flight response. It is not well known whether a fight or flight response is activated by ship noise in all whales, but it has been documented in large whales that boat noise can elicit behavioral changes that could be indicators of fight or flight (Avila et al. 2015; Ovide 2017). This work shows that larger areas with audible anthropogenic noise would result if more vessels, regardless of configuration, are added to the waters off the Colombian Pacific. It is therefore reasonable to assume that an increase in the occurrence of stress responses to noise by humpback whales could also increase as the area ensonified by boat noise grows (Fair and Becker 2000). If these humpback whales are in areas with high noise levels for extended periods, this repetitive or prolonged stress exposure could negatively affect their long-term health (Erbe 2011).

Stress response to audible noise and its overlap with a humpback whale's communication space is the most likely effect to occur in the study area. This is because all shipping lane configurations yielded sound levels below the PTS threshold for the low-frequency marine mammal hearing group. In fact, most source levels of ships did not even exceed the TTS threshold. Only in a few rare cases in our models would a humpback whale have experienced TTS conditions, and only then if they were swimming alongside a vessel. Therefore, the interpretation of this work should focus more on constant noise exposure and how it could have long-term effects due to corrosive impacts on whale hearing when noise exposures are experienced over time, thus generating a chronic effect on an individual's health (e.g., Bejder et al. 2006; Clark et al. 2009; Ellison et al. 2012).

For a contrary interpretation, the possible resiliency of the humpback whale to human noise should be considered. Ellison et al. (2012) reported mysticete cetaceans to inhabit or often settle in areas where anthropogenic noise is abundant. This suggests the capacity for behavioral adaptation or plasticity to fluctuating environments when a dose of noise is applied (Ellison et al. 2012). The whales in the Gulf of

Tribugá may be equally as adaptive to added noise in their habitat, so it is not guaranteed that they would be severely impacted by some increase in noise. Instead, the increase in ships could lead to different impacts on the humpback whale population, like more ship strikes, which are fatal. Regardless, due to the paucity of data on stress response and its long-term effect on the immune system, it is reasonable to consider that the area surrounding the Gulf of Tribugá, being that it has little anthropogenic noise currently, could be serving as an acoustic haven for Stock G during their lifespans. Therefore, while science currently cannot document whether ensonifying a breeding ground with human noise could cause irreversible damage or not, it is a worthy research endeavor to understand better in the future and the precautionary principle should be implemented when possible.

## Conclusions

The shipping lane configurations created for the hypothetical mega port in the Gulf of Tribugá resulted in areal noise increases of 50,231 $km^2$ within the study area for the "random" configuration, and 77,269 $km^2$ for the "deepwater trenches" configuration as compared to the control map. Although the random configuration (Fig. 13) moved ship noise sources out from over some deepwater trenches to determine if that was enough to make a significant decrease in the audibility area of the shipping lanes, it did not result in a large difference compared to the deepwater configuration. Specifically, the deepwater configuration had an excess area of 27,038 $km^2$ where boat noise was between 80.1 and 123.2 dB (the area of most audibility). Nevertheless, increases of 13,194 $km^2$ and 13,844 $km^2$ in random configuration were detected for the 50–80 dB and 48–50.9 dB classes, respectively. Therefore, while the random configuration would be the better shipping lane management strategy of the two in terms of audible ship noise area, it is not that much of an improvement, and further exploration into other configurations could yield an even better management option.

If a port were to be built in the Gulf of Tribugá, it is not known exactly what type of vessels would transit, nor what cargo would be transported. Additionally, there is not much information on the potential routes that would be used for merchant shipping. However, the present research showed that the addition of more ships in the ocean would significantly increase noise levels in the water, so there would be potential for more masking of acoustic signals emitted by humpback whales. Studies like this one would allow the government and/or related entities to regulate, mitigate, and take the necessary measures to anticipate a decision that could be detrimental to areas that are highly diverse and therefore must necessarily be conserved.

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# Susceptibility of Harbor Porpoise Hearing to Intermittent Sound Exposures

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Alexander M. von Benda-Beckmann, Ron A. Kastelein, Frans-Peter A. Lam, Christ A. F. de Jong, Paul J. Wensveen, and Darlene R. Ketten

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**Abstract**

The effects of intermittent sound exposures on the hearing of marine mammals are not well understood. This chapter explores a model to predict growth of Temporary Hearing Threshold Shift (TTS) considering level and duration of the sound exposure in harbor porpoises, a species which has vulnerable hearing. This model was evaluated using TTS growth data obtained for harbor porpoises with different sound exposures (intermittent and continuous sonar, and intermittent pile driving and air-gun sound).

An adapted version of a modified-power-law (MPL) model was introduced, which attempted to overcome some shortcomings of the original model, proposed by Humes and Jesteadt (J Acoust Soc Am 85:1285–1294, 1989)) that required growth and recovery from individual exposures to be measurable. Fitting of the data to this model resulted in parameters that could explain TTS growth observed in a wide range of sonar exposure conditions (various sound pressure levels, 1–2 kHz, 10% duty cycle vs continuous exposure, and various exposure durations), but underestimated the TTS growth of exposure to impulse sound (air guns and pile driving) and sonar with higher (6–7 kHz) frequencies. These results suggested that development of a single TTS growth model for different fatiguing sound exposures is promising, but requires further development and validation, as well as additional data on TTS growth and recovery due to different signal parameters of the fatiguing sound.

**Keywords**

Temporary threshold shift · Auditory impact · Harbor porpoises · Underwater sound · Modified power law model

## Introduction

Management of man-made underwater noise to protect marine life requires guidelines for predicting risk of hearing damage. Noise exposure criteria have been proposed to predict the onset of auditory effects in various marine mammal species (Southall et al. 2007; Tougaard et al. 2015; Finneran 2016; Southall et al. 2019).

There is a general understanding that impulsive sounds are harmful to the mammalian hearing. Currently different thresholds are applied to pulsed and nonpulsed signals, but especially the effects of intermittent sound exposures on the hearing of marine mammals are not well understood (Finneran 2015).

The sensitivity to hear a sound is typically expressed in terms of the hearing threshold (i.e., the lowest level of sound required to detect the sound with a specified probability). When the hearing system is overexposed to sound, a raising of the hearing threshold is observed, which is called a threshold shift (TS). A threshold shift that recovers to the baseline hearing threshold after some duration is called a "temporary threshold shift" (TTS) and is called "permanent threshold shift" (PTS) when it does not fully recover. Prediction of when PTS is induced in marine mammals relies on the ability to extrapolate from low levels (6 dB) of TTS to high levels (40 dB; Southall et al. 2019).

A recent study carried out explored different approaches to improve predictions of TTS growth in harbor porpoises (*Phocoena phocoena*). First, the literature of mechanisms and models explaining TTS growth and recovery in humans and terrestrial mammals was reviewed to identity promising model approaches. Then two TTS growth models were investigated: a kurtosis-based method and an adapted version of a modified power-law (MPL) model, proposed by Humes and Jesteadt (1989). The MPL model was applied to TTS data collected with harbor porpoises exposed to a range of different sound exposures. The kurtosis-based approach is described in von Benda-Beckmann et al. (2022).

## Literature Review

### TTS Growth Studies in Human and Terrestrial Animals

TTS growth and recovery have been reported for a wide variety of exposure conditions and species. A number of papers provide extensive reviews on these studies (e.g., Melnick 1991; Finneran 2015). TTS growth typically follows an exponential growth with SPL (Finneran 2015), possibly followed by a plateau when the TTS reaches the so-called asymptotic threshold shift (ATS; a maximum induced TTS that does not increase even when exposure duration is increased, Melnick 1991). In some instances, a reduction in TS during the exposure (cochlear toughening) can be observed (Hamernik et al. 2003). The exposure level associated with the turning point where the TTS growth changes from increasing logarithmically with SEL to a linear increase is referred to as the "critical level." It has been argued that this constitutes a point at which mechanical fatiguing processes start to dominate over metabolic processes (Price 1981; Hamernik et al. 1987). For humans, TTS growth and recovery can be predicted well as function of the percentage of the ATS is induced by the exposure (Melnick 1991). However, ATS has not been observed in marine mammals, due to avoiding the long exposure durations employed in terrestrial mammal laboratory studies required to reach an ATS, such

as those on chinchillas and primates (4–36 h) (Bohne and Clark 1982; Clark 1991; Yoshida and Liberman 2000; Finneran 2015).

In laboratory studies, terrestrial mammals have commonly been exposed to fatiguing aerial sounds of 15 mins or longer, often for multiple hours. These sounds, with A-weighted SPLs over 100 dB re 20 μPa, elicited shifts of 40–50 dB with recoveries considered as acceptable after 2 or more days (Miller et al. 1963) and in some cases up to 1 month postexposure. These studies typically induce larger initial TTSs than are reported for marine mammals exposed to underwater sound and, similarly, have complete recovery times that are longer for these larger shifts. It is also noted in Price and Kalb (2018) that at the very highest exposure levels recovery is delayed and often results in PTS (Hamernik et al. 1987; Luz and Hodge 1971; Price 1983).

Key points of consensus that have emerged across experiments in multiple mammal species about the nature of threshold shift risk are as follows:

**Risk of noise-induced hearing loss**: (a) Damage is roughly proportional to total sound energy (duration X intensity); (b) intermittent exposures are less dangerous than continuous; (c) low frequencies are generally less dangerous than higher frequencies at similar sound levels; (d) narrow-band sounds are more dangerous than broad-band; and (e) large inter-subject variability in noise resistance (susceptibility to TTS/PTS).

**Growth, recovery, and pattern**: (a) Hearing loss deteriorates to an asymptote during continued exposure; (b) recovery begins after termination of the exposure; (c) above a certain "critical level," hearing loss grows more rapidly; (d) hearing can recover over long periods (up to 1 month postexposure, possibly even longer); (e) the maximum TS due to narrow-band sound exposure occurs near the exposure center frequency at lower SPLs; (f) the frequency with maximum TS shifts upward in frequency to about 1/2 octave above (and sometimes even higher) the exposure center frequency with increasing SPLs; and (g) the maximum TS due to sound exposure that is broad-band compared to the hearing range occurs at mid frequencies within the hearing range.

## Mechanisms for Hearing Loss and Recovery Due to Impulsive Sound Exposure in Terrestrial and Marine Mammals

Mechanisms for TTS and PTS have been reviewed in several studies (Nordmann et al. 2000; Heinrich and Feltens 2006; Ryan et al. 2016; Kurabi et al. 2017; Liberman and Kujawa 2017). These indicate several key observations: (a) Different mechanisms underlie TTS and PTS; (b) different mechanisms may mediate TTS; (c) hair cells of the organ of Corti are the most vulnerable to permanent noise damage; (d) most PTS (>50 dB loss) is due to damage to the stereocilia of the hair cells; (e) severe TTS causes little overt pathology: changes are submicroscopic; (f) inner-hair cell (IHC) damage results in loss of transduction; outer-hair cells

(OHC) damage raises thresholds; and (g) hair cell loss leads to retrograde neuronal death over time.

Clearly, mechanisms underlying TTS must be to some extent recoverable. Many of these are still being investigated. The majority of the mechanisms that are considered to be related are fatigue, swelling, buckling, or recovery of structures that are submicroscopic, such as the fine hair-like projections atop the hair cells (stereocilia) and strands between them (tip-links) (Heinrich and Feltens 2006; Liberman and Kujawa 2017; Ryan et al. 2016).

Mechanisms that produce PTS related with permanent damage to hair cells, or their afferent neurons, are likely mediated by biochemical processes that occur within the cells themselves (Liberman and Kujawa 2017; Kurabi et al. 2017). One recent hypothesis for noise-induced PTS is that an excess of glutamate in the system produces "excitotoxicity" and an imbalance of calcium ions that are fundamental to degeneration of inner ear structures observed after noise exposure (Kujawa and Liberman 2009; Liberman and Kujawa 2017).

In the case of blast exposures, in addition to the potential for immediate mechanical destruction of some ear structures (dislocation of ossicles, blow-out of inner ear membranes), the exceptional peak levels of the associated sound wave can produce acute damage to inner ear components, such as disruption of hair cells, stereociliary arrays, and supporting cells, with collateral damage to adjacent cells from the flood of intracellular fluids (Liberman and Beil 1979; Slepecky 1986; Patuzzi et al. 1989; Kurabi et al. 2017). These processes all result in PTS.

Sound exposure at levels that are low enough to not cause or contribute to TTS (a concept known as "effective quiet") has been investigated primarily in humans (Ward et al. 1976). Effective quiet has not been measured in marine mammals (Finneran 2015). Due to the lack of data on effective quiet, current models tend to accumulate all exposures over a long period (e.g., 24 h), which may result in an unrealistic high accumulated exposure.

## Self-Suppression of Hearing Sensitivity

Several papers (e.g., Nachtigall and Supin 2013, 2014; Nachtigall et al. 2018; Finneran et al. 2015; Finneran 2018) have reported the ability of trained, captive odontocetes of several species to reduce their auditory brain stem response to loud sounds. These studies have documented the "self-mitigation" response in these species but were not geared to identifying explicit mechanisms resulting in the onset and control involved in diminishing the auditory response.

Any one or a combination of mechanisms (such as an efferent reflex or stapedial reflex), simultaneously or sequentially, are feasible for marine mammals based upon the current knowledge of their highly developed auditory structures that contain all of the components known to be involved in reducing sensitivity to sound in other mammals.

## Models for TS Onset and Growth of Intermittent and Impulsive Sound Exposures

### Equal Energy and Frequency Weighting

In a recent review of TTS studies on marine mammals, Finneran et al. (2015) summarizes the empirical models for predicting the TTS growth of intermittent sound exposures and how these apply to marine mammals. The review discriminates between two types of models: "exposure summation" methods and "TTS summation" methods.

The most common exposure summation method used is the "equal energy" rule in which the TTS-onset is solely dependent on the cumulative (frequency-weighted) sound exposure level (SELcum), regardless of how the exposure is distributed over time. This method does not account for potential recovery between exposures. An equal energy rule is currently used widely to predict noise-induced hearing loss for humans (NIOSH 1998) and forms the basis for marine mammal exposure criteria (Southall et al. 2007; Finneran 2016; Southall et al. 2019). In these cases, to account for differences in the frequency sensitivity of marine species, a frequency weighting is applied to the sound exposure level, which emphasizes the increased risk of hearing loss at the frequencies of best hearing (Finneran 2015; Houser et al. 2017). Frequency weighting was first proposed by Fletcher and Munson (1933) for variations in the perception of loudness of low-level sounds rather than a relation to susceptibility to damage. Over the years, A-weighting (one of five weightings for humans developed for different sources) has become a standard for estimating workplace exposures, with trade-off of time versus intensity. The appropriate use of frequency weighting is still debated for impulse sound (Finneran 2015; Houser et al. 2017) as well as the applicability to nonhumans exposed to a variety of sound sources, although it is lately being applied in marine mammals (Southall et al. 2007, 2019).

### Modified Power Law Growth and Recovery During Intermittent Exposures

For humans, equal energy models do not explain the differences in TTS between continuous and intermittent noise exposures, which is likely because these models fail to account for recovery. The modified-power law (MPL) model has been proposed for predicting TTS growth or over multiple exposures (induced either by sound, or by medicine). This "TTS summation" method has been shown to provide good agreement to experimental data for human and terrestrial mammals after exposure to intermittent sounds (Ward 1973; Humes and Jesteadt 1991; Macrae 1993, 1994). In this model, the TTS from individual exposures is raised to some power, and it also can account for recovery between exposures (Finneran et al. 2010; von Benda-Beckmann et al. 2022). The MPL model proposed by Humes and Jesteadt (1989) is

$$\left(10^{\frac{T_N}{10\ \mathrm{dB}}}\right)^P = \left[\left(10^{\frac{T_1}{10\ \mathrm{dB}}}\right)^P + \left(10^{\frac{(T_{N-1}-T_r)}{10\ \mathrm{dB}}}\right)^P - 1\right], \quad (1)$$

with $T_N$ the amount of TTS induced after $N$ exposures, $T_{N-1}$ the TTS caused by the ($N$-1)-th exposure, and $T_1$ the amount of TTS induced by a single exposure. The recovery term $T_r$ represents the amount of recovery of TTS that occurred since the (N-1)-th exposure (Finneran et al. 2010), and $P$ is a compression term. For humans, the factor $P$ was found to lie in the range of 0.1–0.3 (Humes and Jesteadt 1991).

Finneran et al. (2010) showed that the model can be used to fit TTS growth in dolphins exposed to intermittent tonal noise. They found a best fit of P=2.4 in dolphins (Finneran et al. 2010). In section "Adapted Modified Power Law Model," the possibility is explored of applying the MPL model for predicting growth of TTS in a harbor porpoise due to intermittent exposures, with relatively short intervals (10–20 s), such as those used in air-gun or sonar operations.

### Mechanical Models for Modeling Auditory Injury from Impulse Sounds

Models of the mechanics of the middle and inner ear exist that relate acoustic stimulation to hearing damage in humans and cats (Price 2007a; Zhang and Gan 2011; Zagadou et al. 2016; Walilko et al. 2019). The US Army Auditory Hazard Assessment Algorithm for Humans (AHAAH) model (Price 2007a, b; Price and Kalb 2018) was developed specifically to estimate permanent hearing loss (PTS) from shot noise and blasts in humans (with derivative modeling in cats).

The AHAAH model assumes that the risk from loud impulse sounds is related to the number of upward deflexions of the basilar membrane (which is assumed to put the stereocilia tip links of the hair cells in tension, which Price and Kalb (2018) speculated to be the most likely mechanical failure mode). Frequency-dependent displacement of the basilar membrane is computed using an electric analogy that models the outer ear, middle ear, and inner ear. The model also includes two nonlinear elements: middle ear muscle response (stapedial reflex) and peak-clipping due to the limited dynamic range in displacement allowed in the middle ear (displacement of the annular ligament of the stapes).

The AHAAH model expresses the hazard in terms of Auditory Risk Units (ARU), which are defined as the sum of the maximum amplitude squared of the upward displacement $D$ (in μm) of the basilar membrane for each cycle $\mathrm{ARU} = \sum D^2$, which are summed over all cycles (interval between zero crossings). The maximum ARUs over all frequencies were found to be related to the amount of compound threshold shift (CTS – amount of hearing threshold shift observed which may be combination of TTS and PTS) observed at 30 minutes after exposure in cats by

$$\mathrm{CTS} = 26.6 \ln\left(\frac{\mathrm{ARU}}{1\mu\mathrm{m}^2}\right) - 140.1 \tag{2}$$

For these intense stimuli that result in mechanical damage, the amount of CTS also has a linear relation to the amount of PTS induced and hair cells lost, according to Price (2006), who found that approximately 60–80% of the CTS would become PTS (Price and Kalb 2018). Note that this can be rewritten as

$$\mathrm{CTS} \approx 10 \log_{10} \left( \frac{\sum_{\mathrm{cycles}}^{\square} D^2}{1 \mu \mathrm{m}^2} \frac{1}{10^{\frac{140.1}{60}}} \right)^6 \mathrm{dB}, \tag{3}$$

Hence, the combination of two exposures, causing $\mathrm{CTS}_1$ each, will generate a combined $\mathrm{CTS}_2$ of

$$\left(10^{\frac{\mathrm{CTS}_2}{10\ \mathrm{dB}}}\right)^{\frac{1}{6}} = \left(10^{\frac{\mathrm{CTS}_1}{10\ \mathrm{dB}}}\right)^{\frac{1}{6}} + \left(10^{\frac{\mathrm{CTS}_1}{10\ \mathrm{dB}}}\right)^{\frac{1}{6}}. \tag{4}$$

This equation is therefore essentially a modified power-law relation, with $P = 1/6 \sim 0.16$, which lies well in the range of $P$ values observed in humans (Humes and Jesteadt 1991). This suggests that a MPL model can also apply to impulsive sound exposures. However, the exposures to continuous and impulse sound may have different growth and recovery mechanisms, so the same set of parameters are not expected to explain both TTS growth from exposures with continuous and impulse sounds.

The ARUs have shown to better predict risk of CTS than growth models based on the equal energy rule for frequency-weighted sound exposure level (see Price and Kalb 2018). The AHAAH also accurately predicted the frequency-encoding locations along the basilar membrane that were affected by hair cell loss in cats (Price 2003; Price and Kalb 2018).

A practical drawback of this AHAAH model is that it is complex and consists of 69 parameters and relies heavily on knowledge on middle ear and inner ear properties, which have been well studied in cats and humans, but are currently less well established for many marine mammals (e.g., Miller et al. 2006; Ketten 2000; Tubelli et al. 2012, 2018; Tubelli and Ketten 2019).

## Recovery Models

The form and degree of recovery depends considerably on the underlying mechanism for TTS growth. The dominant mechanism for the observed TS may change according to the type and degree of exposure (differently for SPL or exposure duration). That is, if the TTS is caused by tip-link breakage, it is time for tip-link to regenerate. For longer duration exposures, an alternative recovery mechanism may be the regulation of chemical processes related to excitotoxicity that lead to damage. It is likely that several mechanisms operate simultaneously, with relative importance changing with exposure conditions, making it impractical at this stage to create a bottom-up predictive model for the recovery.

There have been some developments in describing the exponential short-term (timescales of few milliseconds) adaptation of hair cells (Charasse et al. 2003; Zylani et al. 2009), sometimes also power-law relations used for modeling the long-term (timescales of 10s of seconds) adaptation of discharge rates in IHCs (e.g., Zylani et al. 2009). However, so far these recovery models deal with saturation of hair cells

and the resulting forward masking (brief elevation of hearing thresholds after sound stimulation, i.e., Harris and Dallos 1979) and do not incorporate the effects of noise overexposure.

Recovery of TTS is typically measured on timescales of minutes to hours, or even longer (depending on the elicited TS). It is generally found that one or a combination of logarithmic decay functions can properly describe the recovery of TTS with time (Ward 1960; Finneran 2015). Melnick (1991) notes that when expressed in terms of the ATS, the percentual change follows an exponential function, as $TTS(t) = ATS \cdot \exp.(-t/\tau)$. Short-term recoveries of OHC responses on the timescales of hundreds of milliseconds to seconds have been measured indirectly using cochlear microphonics (CM) techniques (Price 1974), suggesting that a logarithmic recovery with time may be appropriate for these small timescales. However, it has been debated to what extent CM techniques directly measure the OHCs activity (e.g., Perez et al. 2007).

### Summary

Figure 1 represents the key mechanisms that are involved in the occurrence and mitigation of hearing loss induced by sound exposure. Mechanisms underlying TTS can affect both the structure and integrity of the inner ear, producing changes in hair cell responsivity, possibly through alterations of the stereocilia (e.g., broken tip-links), through intracellular chemical changes affecting synaptic transmissions at the base of hair cells and alterations in hair cell conformation with swelling and degeneration of synaptic ribbons that transmit electrical signals from the cochlea to the brain.

The summary of mechanisms underlying TTS and PTS presented here served to show the complexity of the mechanisms for TTS growth, and recovery. Although significant advances have been made in the understanding of these processes in humans and terrestrial mammals, the exact functioning is still not well understood. It is likely that several mechanisms are at play with changing importance depending on exposure duration (metabolic exhaustion) and increasing sound pressure (mechanical fatiguing). This makes it not yet feasible to make a direct link to biomechanically motivated growth and recovery functions. For this reason, the next sections explored the MPL model to empirically relate sound levels to TTS growth for different exposure conditions.

## Adapted Modified Power Law Model

### TTS Growth from Individual Exposures, and Timescale for TTS Growth

For the MPL model (Eq. 1) to be useable, the amount of threshold shift caused by a single exposure $T_1$ and the recovery of the threshold shift at the time of the next exposure $T_r$ need to be measured directly (Humes and Jesteadt 1991;

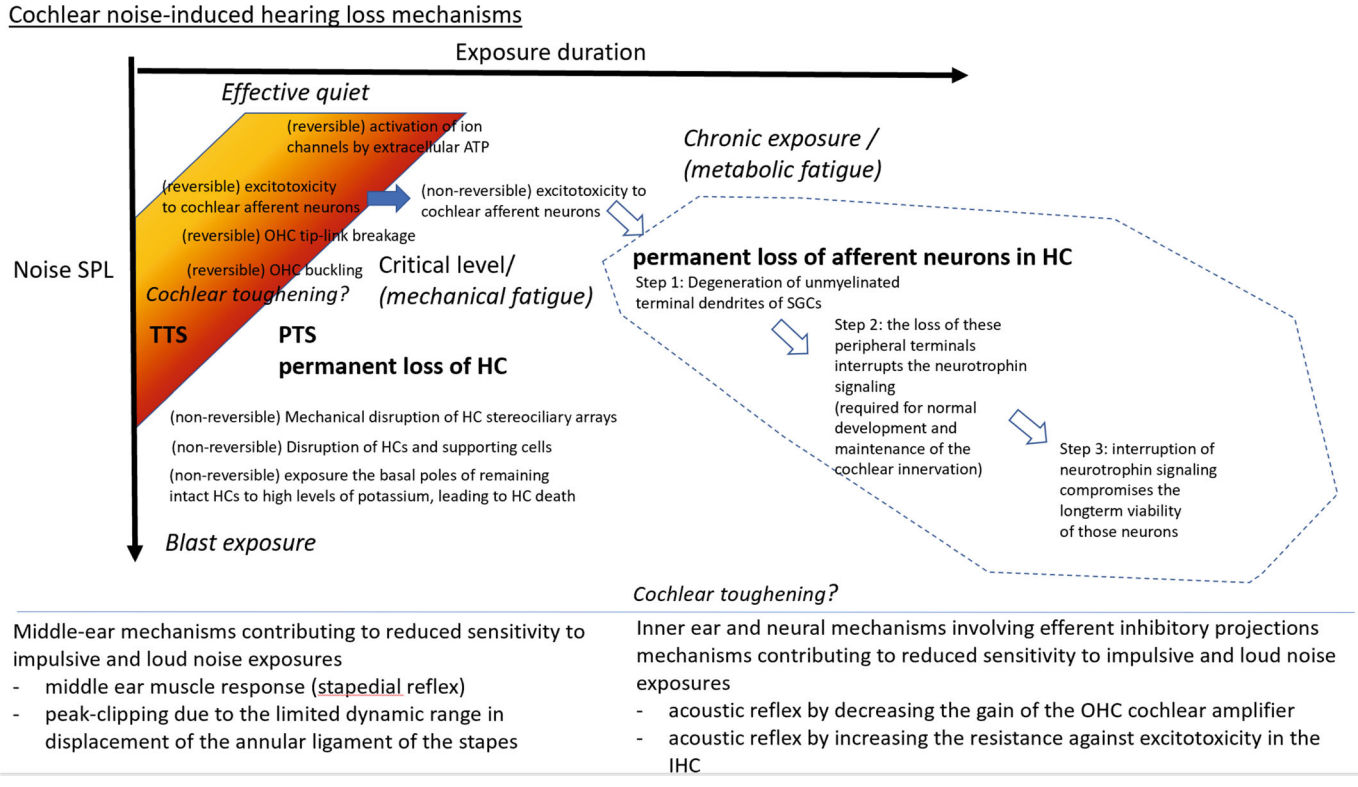



**Fig. 1** Schematic summary of underlying mechanisms causing noise-induced TTS and PTS and their dependencies on the level of the noise and exposure duration, and middle ear and inner ear mechanisms that mediate hearing sensitivity and vulnerability of animals

Finneran et al. 2010). However, in the case of harbor porpoises exposed to repeated, intermittent air-gun and sonar sounds, TTS was only measurable after multiple exposures (Kastelein et al. 2014a, b, 2015, 2017); hence, $T_1$ could not be directly measured. A second challenge was that seismic, piling, and sonar exposures typically have short repetition times, in the order of 5–20 s. Recovery rates at those timescales are hard to measure directly (see section "Recovery Models"). In order to make use of Eq. (1), therefore, variables $T_1$, $T_r$, and $P$ were estimated by fitting the model to data of TTS measured after multiple exposures. To allow such a fit, relationships between exposure conditions and TTS growth and recovery have to be hypothesized.

The MPL model describes how to add the threshold shifts caused by individual exposures, but it does not specify what causes the TTS. Exposure durations differ between the stimuli considered here (few 100 milliseconds to seconds for intermittent air guns and sonar) and become somewhat arbitrary defined for continuous noise exposures. It is therefore not obvious what constitutes the "single exposure" that is added by the MPL model. In this study, harbor porpoise TTS data was used to explore whether the MPL model can be generalized to exposure conditions with arbitrary duty cycles, by representing a continuous sound exposure as a series of consecutive "sub-exposures" with a 100% duty cycle (i.e., no recovery between subexposures).

In the proposed adapted MPL model, the measured exposure is related to its contribution to the TTS growth ($T_1$ in the MPL model) on a chosen unit reference timescale $t_{ref}$ (Fig. 2). In principle, the exposure can be quantified in different ways, e.g., either SEL within this timescale, but other parameters such as the peak sound pressure level or SPL could be used as well. For instance, the AHAAH model (section "Mechanical Models for Modelling Auditory Injury from Impulse Sounds") relates the CTS growth to the maximum displacement of the basilar membrane, which is likely more related to the frequency-weighted sound pressure measured in a smaller frequency band over the relevant timescale of the hearing system.

Several studies showed that the TS typically increases exponentially with SEL (Finneran 2015). Therefore, the amount of additional TTS induced was related to the sound exposure level, $L_{E,p,1}$, measured on a short reference timescale $t_{ref}$ using the following relationship,

$$T_1 = a \log_{10}\left(1 + 10^{\frac{L_{E,p,1}-b}{10\ \mathrm{dB}}}\right), \tag{5}$$

with parameters ($a$, $b$) to be fitted from data.

A time duration for the individual intermittent exposures in the continuous sound needs to be chosen. Possible options would be to use a timescale related to the hearing integration time, or shorter timescales such as the dynamic timescales of neurons (~ 1.4 ms, as in Zylani et al. 2009), or per cycle of the basilar membrane (BM) vibrations as in the AHAAH model (Price 2007a). Here, the integrated energy within the integration time of the mammalian auditory system was considered as the base unit of exposure. For this, an approximate average of the measured integration times for signals below 10 kHz: $t_{ref} = 200$ ms (Kastelein et al. 2010) was adopted.

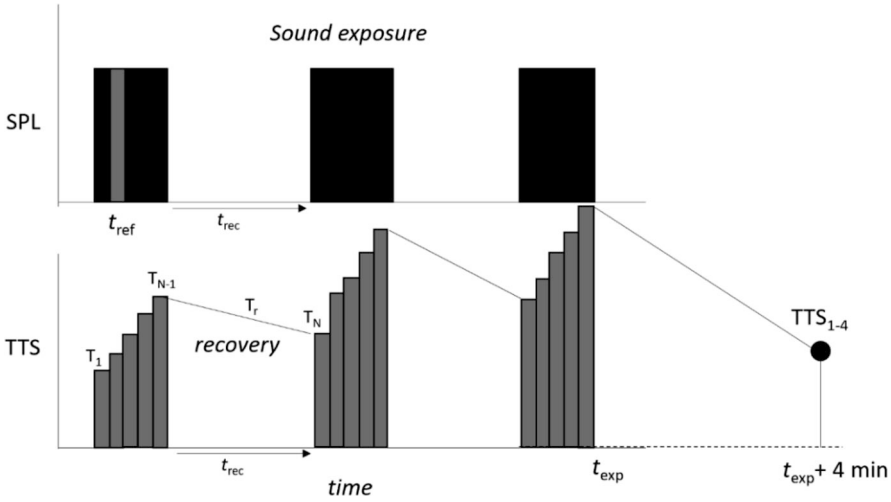



**Fig. 2** Schematic representation of the TTS growth model for an intermittent sound exposure based on the adapted MPL model. The model considers a unit reference time ($t_{ref}$), over which the contribution to TTS is computed, which are summed over multiple time units using the MPL model (Eq. 1) resulting in the measured $TTS_{1-4}$ (black circle). The TTS is assumed to recover during the silence periods of duration $t_{rec}$ between individual exposures

$L_{E,p,1}$ is the sound exposure level within that time window (Fig. 2). The effect of choice of reference timescale is discussed in section "Discussion."

## Compression Factor *P* in the MPL Model

In this analysis, a range of possible compression factors were considered in the range of $P = 0.1$ to 3 (the observed range for human TTS studies, Humes and Jesteadt 1991) and dolphins (Finneran et al. 2010).

## TTS Recovery Between Exposures

For the recovery term between pulses, a single logarithmic decay with time after exposure was assumed, with decay rate $m$ that was dependent on total amount of TTS caused by previous exposures (as generally observed in marine mammals, Finneran 2015):

$$T_r = m \log_{10}\left(\frac{t_{rec}}{t_{ref}}\right) \text{dB}, \qquad \text{for } t_{rec} \geq t_{ref} \tag{6}$$

$$T_r = 0 \text{ dB}, \qquad \text{for } t_{rec} < t_{ref}$$

with $m = c_1 \cdot T_N + c_2$, $t_{ref} = 0.2$ s, and $t_{rec}$ the duration of silence between exposures. It was assumed that no recovery occurred for consecutive exposures periods when signal durations exceeded the integration time of the unit exposure ($T_r = 0$ dB). After the final exposure, $T_r$ was computed for a period of 4 min (up to the point at which TTS was measured) (Fig. 2). Here the parameters ($c_1$ and $c_2$) were obtained from fitting the model to TTS growth data.

### Fitting of the Adapted MPL Model

Statistical model fitting (Eqs. 1–3) was carried out using Bayesian inference, with a Gibbs sampling routine (in JAGS through R), and the model was fitted to the 1–2 kHz exposure data alone. Uniform priors were used for the estimated parameters: P ~ uniform (0.1, 3); a ~ uniform(0.0001, 1); b ~ uniform (140, 180); c1 ~ uniform (0, 0.01); and c2 ~ uniform (0, 0.01). This dataset was used because it had the widest range of exposure conditions at fixed exposure frequencies. The model predictions based on TTS 1–2 kHz exposures were compared to the TTS measurements for exposures to 6–7 kHz sonar, air-gun, and pile-driving sound.

## Results

### Fit of the MPL Model to 1–2 kHz Sonar Data

The fitted MPL model resulted in reasonable predictions of the growth functions for the 100% and 10% duty cycles, and for the varying duty cycle (Fig. 3). The lower growth rates for 10% duty cycle, as well as the similar growth rates for different exposure conditions (combinations of SPL and exposure duration), were predicted with the MPL model. Posterior mean estimates of the MPL model parameters (Fig. 4 and Table 1) resulted in a compression factor $P = 0.31$.

### Comparison with Other Exposure Types

After the model was fitted to the 1–2 kHz sonar data, it was investigated how well it could predict TTS growth induced by different frequency exposures.

#### Sonar Exposure (6–7 kHz) (Kastelein et al. 2015)

Kastelein et al. (2015) showed that the 6–7 kHz sonar signals induced a maximum TTS at 9.3 kHz. When compared to this TTS data (measured at 9.8 kHz; Kastelein et al. 2015), the model predicted lower growth than observed, especially for the continuous sound exposure (Fig. 5).

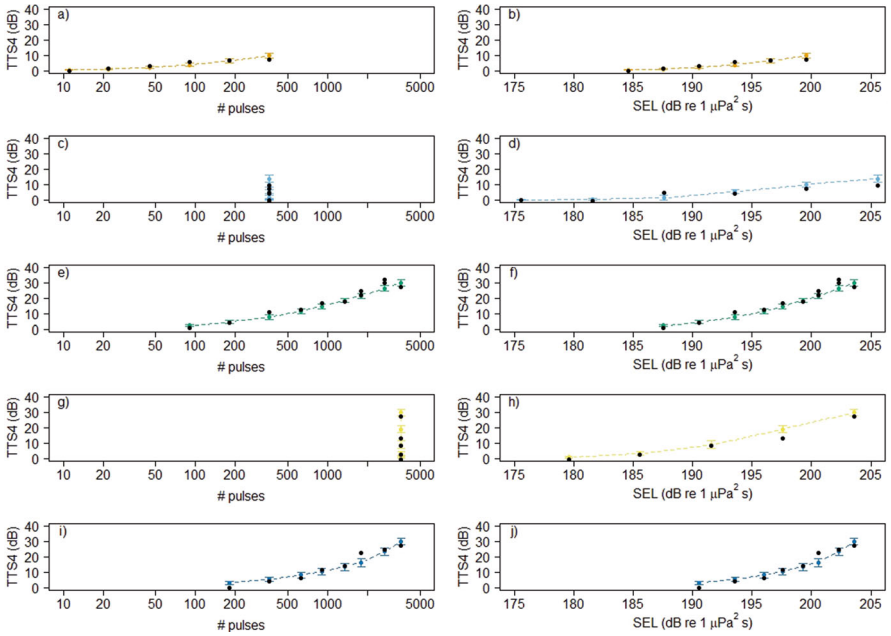



**Fig. 3** Measurements (black) and model predictions (color; posterior mean estimates with 95% credible intervals) of TTS growth as a function of number of pulses (left) and SELcum (right) for various exposure conditions: intermittent (panels a-b, e-f, and i-j) and continuous sonar exposure (panels c-d, g-h) using either fixed SPL (168 or 174 dB re 1 µPa) or varying SPL. Exposure signals were 1-s duration 1–2 kHz HFM down-sweeps, with 5–100% duty cycle). (Data from Kastelein et al. 2014a)

## Impulsive Sound Exposures

The adapted MPL model was applied to TTS growth data obtained for porpoises exposed to piling playback sounds for periods of 15–240 minutes (Kastelein et al. 2016). The low growth with exposure duration as observed in Kastelein et al. (2016) is predicted using the exposure model from Eq. 1, although the predicted TTS levels are somewhat lower (Fig. 6). The MPL model was compared to TTS growth measurements obtained for porpoises exposed to 1–10 air-gun exposures (Kastelein et al. 2017). When comparing the model predictions to the TTS measurements at the frequency with the highest TTS (4 kHz), the model fitted to the 1–2 kHz sonar TTS growth predicted smaller TTS than those observed for air-gun exposures (Fig. 6).

## SEL Predictions at TTS Onset as Function of Exposure Duration

Interestingly, the MPL model predicted that the SEL onset threshold for TTS (6 dB) was dependent on the exposure duration, both for intermittent as well as continuous sound exposure (Fig. 7). This appears similar with an observation for dolphins

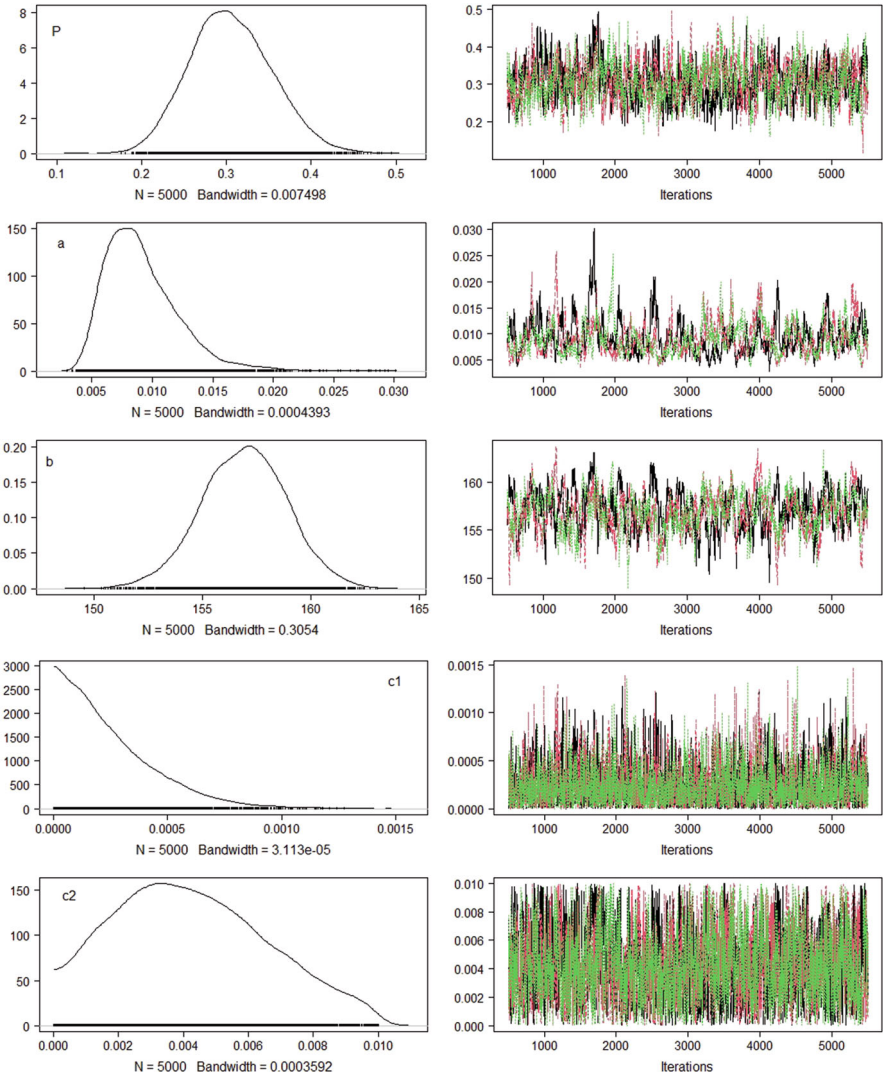



**Fig. 4** Posterior probability distributions (left) and trace plots (right) for parameters in the MPL model fitted to the TTS growth data for 1–2 kHz (LFAS) continuous and intermittent sonar. The model was fitted using Bayesian inference with a Gibbs sampling routine

exposed to sonar sound, where the SEL onset threshold appeared to decrease with increasing exposure duration (Mooney et al. 2009). For shorter duration exposures, thresholds for TTS onset were predicted to be higher, but for longer duration exposures, the model converged to an equal-energy rule (constant SEL, independent on duration over which the SEL was delivered).

**Table 1** Statistics of the estimated posterior probability distributions (mean, standard deviation (SD), and 25 and 75 percentiles) for the parameters of the adapted MPL model fitted to the 1–2 kHz sonar TTS growth data

| Model parameter | Mean | SD | 25% | 75% |
|---|---|---|---|---|
| $P$ | 0.31 | 0.05 | 0.27 | 0.33 |
| $a$ (dB) | $9.1 \times 10^{-3}$ | $3.1 \times 10^{-3}$ | $6.9 \times 10^{-3}$ | $1.1 \times 10^{-2}$ |
| $b$ (dB) | 157 | 2.0 | 156 | 158 |
| $c_1$ | $2.6 \times 10^{-4}$ | $2.1 \times 10^{-4}$ | $8.9 \times 10^{-5}$ | $3.6 \times 10^{-4}$ |
| $c_2$ | $4.3 \times 10^{-3}$ | $2.3 \times 10^{-3}$ | $2.5 \times 10^{-3}$ | $5.9 \times 10^{-3}$ |

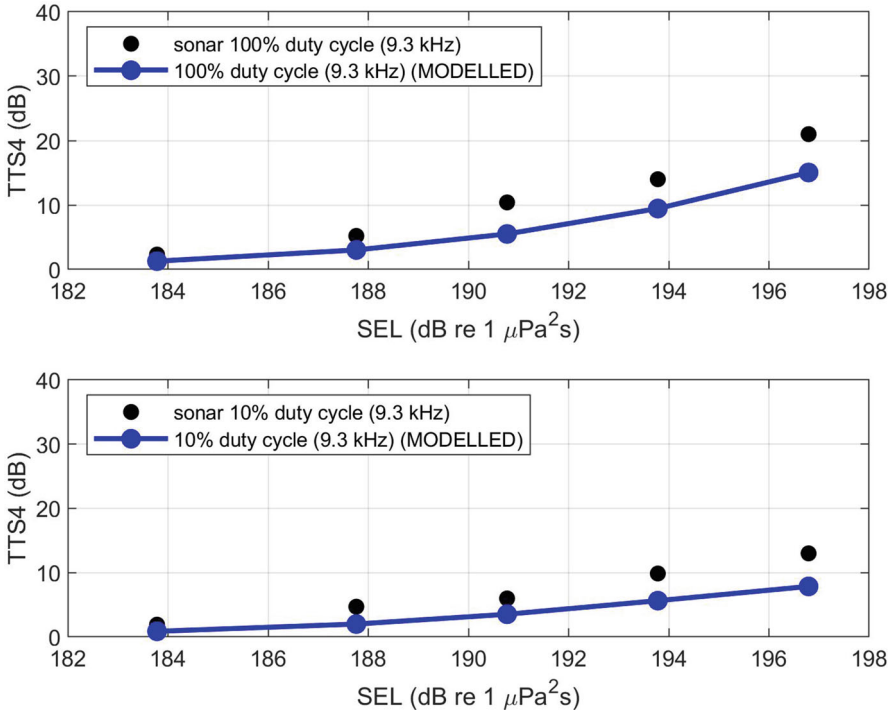



**Fig. 5** Posterior mean predictions of the MPL model fitted to 1–2 kHz sweeps (blue) compared to observations of TTS growth for 6–7 kHz sweeps (Kastelein et al. 2015) and continuous (top; black) or intermitted exposure at 10% duty cycle (bottom; black)

## Discussion

Based on the literature review, the modified power law (MPL) model was explored as a model for TTS growth in different exposure conditions, as it allowed for accounting for recovery of TTS between exposures.

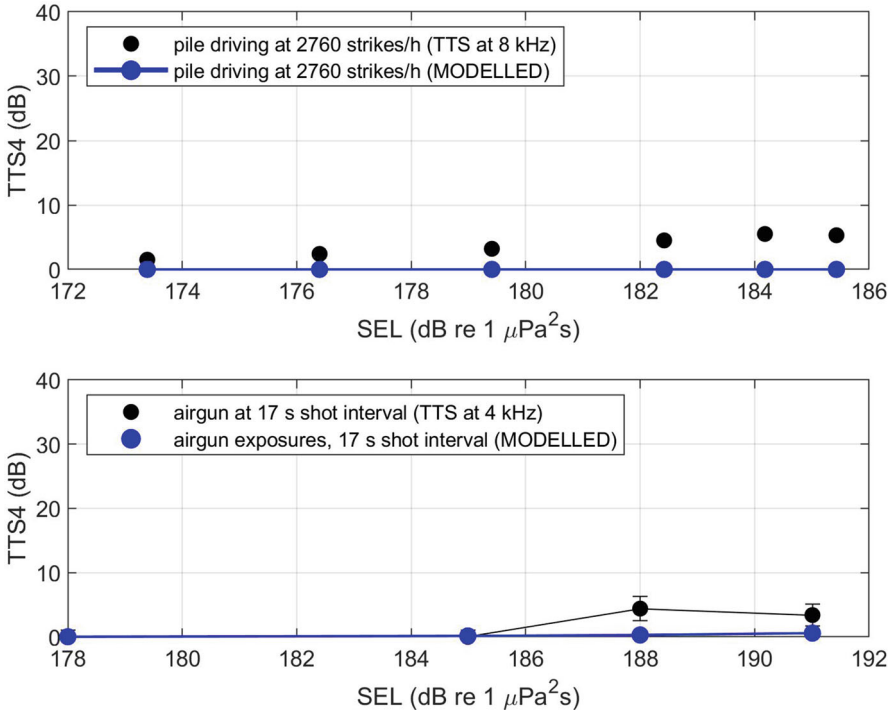



**Fig. 6** Top: predictions of the adapted MPL model for TTS growth for pile-driving exposure (blue) compared to observed TTS growth from Kastelein et al. (2016), using the parameters fitted to the 1–2 kHz sonar growth rate data (Table 1). That study used an average broadband SELss = 145 dB re 1 μPa$^2$s for each exposure, and a duty cycle of 2760 strikes per hour (i.e., exposure duration determined SELcum). Exposure durations ranged between 15 and 240 minutes. Bottom: posterior mean predictions using the MPL model (blue) for TTS growth compared to air-gun exposure (up to ten shots with a double air gun, Kastelein et al. 2017) with mean duty cycle of 17 s

In this study, an adaptation to the MPL model was proposed such that it could be applied to porpoises exposed to intermittent and continuous sound exposures, to investigate whether it explained differences in TTS growth for exposure durations and exposure sound pressure levels (SPLs). The objective was to fit these growth functions to harbor porpoise data in which animals were exposed to intermittent and continuous sonar sounds. This study built on the MPL as applied for bottlenose dolphins by Finneran et al. (2010), by considering observed generic relationships of TTS growth and recovery rates with exposure conditions (Finneran 2015) and then parameterizing these relationships and fitting them to available harbor porpoise data. For this purpose, the original MPL model needed to be extended to make it generally applicable to a wide range of exposure conditions (intermittent as well as continuous).

The success achieved in this study was that the model could match the growth of a reasonably sizeable dataset of TTS growth in porpoises for sonar sounds. However,

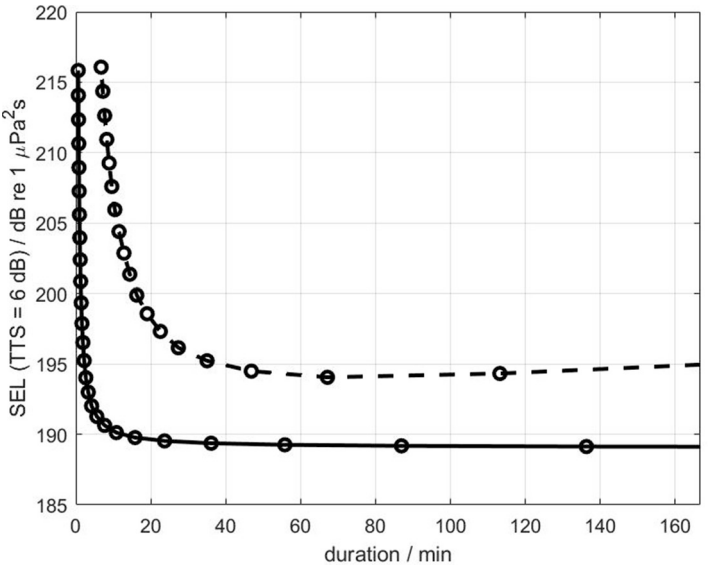



**Fig. 7** Posterior mean prediction of the SEL TTS onset (TTS = 6 dB) threshold as a function of exposure duration for continuous (solid line) and 1 s sonar transmissions at 10 s intervals (dashed).

the modeling results presented in this section should be considered with some caution and only as a first attempt to combine and compare intermittent and continuous exposures of different types (tonal vs impulse) into one comprehensive model.

It was found that the model could reasonably predict TTS growth over a wide range of exposure conditions (SPL and duration) when fitted to a broad set of 1–2 kHz sonar exposures. However, the model underpredicted the growth of TTS observed for sonar at higher frequencies (6–7 kHz), as well as for impulsive air-gun and piling playback exposures. Given what is currently known about the response of mammalian ears to impulse sound and sensitivity at different frequencies, it is not surprising that the model-fit to 1–2 kHz sonar data does not yet yield a good fit to the air-gun and pile-driving TTS data.

The best-fitting compression factor P = 0.31 in the MPL model was found to be consistent with the range of values reported for humans (0.1–0.3, Humes and Jesteadt 1991; Price 2007), and the range of expected values based on the compressive mechanisms that are believed to underlie the growth mechanism. However, these numbers are smaller than $P = 2.4$ found to best explain the TTS growth for dolphins exposed to intermittent sonar sounds (Finneran et al. 2010).

According to Humes and Jesteadt (1991), the MPL model "maintains that threshold sound levels, or their corresponding sound intensities, are not themselves linearly additive. Rather, simple addition holds after the threshold has been nonlinearly transformed to a quantity corresponding to the internal effect associated with

the threshold-level stimulus." This "internal effect" referred to by Humes and Jesteadt (1991) can be interpreted as a reduction in sensitivity, or gain, and from an elevation of the baseline above which sound pressure becomes effective in driving the auditory system (Neubauer and Heil 2004).

There are different mechanisms that lead to a nonlinear relationship between input stimuli and hearing threshold shift. A compressive relationship exists underlying the process resulting in firing patterns of neurons (Heil and Neubauer 2010). Heil et al. (2011) demonstrated that the spike rates in auditory-nerve fibers (ANFs) (leading to perception of the sound) from the IHC on the SPL of acoustic stimuli follow a power-law of $R \sim P^{\beta}$. They find the best fitting value for $\beta = 3$ when the auditory system is driven linearly (i.e., low SPL). Such a relationship implies a $P = 1/3$ relationship in the MPL model. Heil and Neubauer (2010) argue that different relationships reported between the rate and sound pressure ($\beta = 1$–5) may be explained by saturation of neurons which leads to an apparent change in this relationship, which would imply a change in P. Another important factor that causes compression to occur is the amplification by OHCs (Robles and Ruggero 2001). The displacement of the basilar membrane, $D$, is nonlinearly related to the exposure SPL, $L_{p,exp}$, by $\log_{10}(D) \sim 1/B\ L_{p,exp}$ (Robles and Ruggero 2001). In different mammals (chinchillas, cats, and Guinea pigs), it is found that for low SPL the relationship is typically linear ($B = 20$), but for higher SPL the basilar membrane displacement scales with SPL are $B \sim 40$–$60$ (Robles and Ruggero 2001), and that $B$ also depends on the location along the basilar membrane, where compressive nonlinearity is stronger near the base (higher frequencies) and less near the apex (lower frequencies). This implies that the compression phenomenon along the BM is approximately $P = 1/3$ (for $B = 60$) to 1 (for $B = 20$). Since both mechanisms (compression by BM dynamics and the nonlinear relationship between BM displacement and neuron firing rates) can act at the same time. It can therefore be expected that the compression factor is not necessarily a constant, but may lie in the range of values for $P$.

To better understand the compression factor, TTS summation experiments (as in Humes and Jesteadt 1991), or two-tone masking experiments (as for Humes and Jesteadt 1989), should be carried out to independently establish the value of $P$ in porpoises.

Some adaptations could be considered in further development of the model. Currently, the SEL measured on the timescale of the hearing integration time (200 ms) was considered to contribute to the TTS growth in the MPL model. The choice of reference time will have a different effect on how much the sound on this timescale adds to the TTS growth (Eq. 4.2). For the continuous 1–2 kHz sonar exposure with constant SPL, the choice of $t_{ref}$ (200 ms vs shorter timescales of 1.4 ms, corresponding to the dynamic timescales of neurons; Zylani et al. 2009) did not affect the TTS growth in the MPL model for the 1–2 kHz sound exposures. However, for impulse sounds such as pile driving and air guns with large variations in amplitude over short timescales (compared to the hearing integration time), as well as frequency of the sound exposure, the choice of a smaller $t_{ref}$ can lead to different TTS growth. Future studies should therefore consider the model sensitivity on adopted timescale for TTS growth in the MPL model.

Other exposure metrics describe the characteristics of the sound, such as frequency-weighted and band-filtered peak sound pressure on a shorter timescale, which more directly relate to the dynamics of the basilar membrane and may possibly better explain the difference between the TTS growth of different impulsive sounds and higher frequency sonar exposures. For instance, the AHAAH model (Price 2007a, see Sect. 2.4.4) assumes an increase in risk that is proportional to the number of cycles, and the amplitude of displacement of the basilar membrane. For high-amplitude sound, nonlinear effects within the ear can affect the amount of energy reaching the inner ear; hence, the time-dependence of the pressure waveform becomes relevant. Considering success of the AHAAH (Price 2007a) for predicting the TTS and PTS in humans and cats, it is worthwhile exploring the possibility to use biomechanical models (Tubelli and Ketten 2019) for predicting the dynamics of the basilar membrane, which according to the Price (2007a) model are a good predictor for hearing loss for loud impulse sounds. Such models will be informative for understanding the response of the porpoise hearing to impulse stimuli, which helps to inform and motivate appropriate frequency weighting functions for impulse sound and can be used to understand the role of middle ear responses on the sensitivity of porpoises to sound exposure.

The current model did not assume TTS growth rates from individual exposures to be frequency dependent. This may explain the slower predicted growth compared to the 6–7 kHz sonar exposure. Including a frequency weighting may help alleviate this mismatch, although recent measurements of TTS susceptibility at higher frequencies (Kastelein et al. 2019a, b) suggest that TTS growth may not grow faster at even higher frequencies, and hence the frequency dependence of TTS growth should be investigated further.

Currently, the posterior mean estimates of the parameters for recovery ($c_1$ and $c_2$) suggest a recovery rate on the timescales of the silences considered in this study (10–20 s) that is much smaller than the recovery rate extrapolated from observed longer timescale (minutes to hours) recovery in porpoises (see Kastelein et al. 2015; Finneran 2015). In principle, this is not unexpected, as the $\log_{10}(t / t_4 \text{ min})$ term that is typically fitted to long-term recovery (Finneran 2015) will diverge for small timescales, which seems physically unrealistic. Future approaches should aim to match the recovery relation from short scales to longer timescales in order to be consistent with measured recovery rates for longer timescales at which TTS is actually measured.

More measurements of TTS growth under different exposure conditions are clearly required to obtain more confidence in this model. The benefit of a model framework as the one presented here is that it allows to build hypothesis that can be tested. For instance, the model suggests a deviation from the equal energy rule for relatively short exposures (a few minutes), which can be tested experimentally. The model can also be applied to other species (such a delphinids) for which TTS growth information is available. However, there are likely some inherent limitations to the MPL model. For instance, in other species TTS sometimes shows a delayed growth (i.e., continued growth after the end of the exposure, e.g., Luz and Hodge 1971; Finneran 2015), and processes such as cochlear toughening (Hamernik et al. 2003), middle-ear responses, and afferent feedback are not explicitly included in this model approach.

## Conclusion

Based on the literature review, different possible approaches for incorporating the impulsive and intermittent sound into a generic TTS growth model were identified. An adaptation to the modified power-law (MPL) model was used to explain TTS growth data collected for harbor porpoises for intermittent and continuous sound exposures. The MPL model was considered a promising method to predict TTS growth in different exposure conditions, as it can include recovery of TTS between pulses. The model was adapted to apply to intermittent as well as continuous exposures, by cumulating the TTS over subsequent exposure periods at a timescale corresponding with the auditory integration time. The study explored how to apply the model to TTS growth in harbor porpoises.

Some adaptations were necessary to overcome two issues: (i) the TTS induced from single exposures were too low to cause observable amounts of TTS; (2) recovery on the timescales of the shot interval was not directly measurable.

Based on generic parametric relationships between TTS and exposure parameters, the model was fitted to measure TTS growth in harbor porpoises. It was found that the model could reasonably predict TTS growth of intermittent and continuous 1–2 kHz (LFAS) sonar sounds over a wide range of exposure conditions but did not yet successfully explain the (limited) dataset of TTS growth for 6–7 kHz (MFAS) sonar and intermittent impulse sounds. Although the adapted version of the MPL model considered here should be considered preliminary, the reasonable fit to the observed data for a wide range of exposure conditions looks promising and should be explored further.

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# Interference of Communication and Echolocation of Southern Resident Killer Whales

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Rianna E. Burnham and Svein Vagle

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**Abstract**

Human use of aquatic environments is increasing, elevating ambient sound levels through the introduction of more man-made noise. The implications of marine mammal species being subject to increased noise levels can range from masking of ecologically important signals to injury and even death. Successful mitigation relies on characterizing the risk and understanding the level of disturbance on the focal species. Considering the noise-induced changes in effective conspecific communication and echolocation by cetaceans, for example, could be a useful metric for managers when considering both the level of effect and efficacy of control or prevention measures. The estimated reduction in range from a maximum, under minimum ambient or historical conditions, gives a means to quantify impact and make comparisons over time and space. This may add to evidence-based means to design and test mitigation measures aimed at reducing acoustic disturbance for cetacean species.

**Keywords**

Masking · Acoustic range · Southern resident killer whales · Vessel noise · Commercial shipping traffic

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## Introduction

Underwater soundscapes are dynamic. The acoustic landscape of an area encompasses all the biological, geophonic, and anthropogenic noise sources in the area, which differ on temporal and spatial scales. Soundscapes are shaped by contributions of natural abiotic and biotic sounds and human-derived noise. Abiotic sources such as wind, waves, and precipitation contribute substantially to the ambient noise levels. Biological sources such as invertebrate choruses and whale calls also define an area's soundscape. Anthropogenic additions are quickly becoming a ubiquitous contribution to aquatic sound fields. Determining the components of a soundscape and the relationship between natural and human-derived noise is critical for understanding the impacts it may have on animals using the area (Pijanowski et al. 2011).

Increases in ambient noise levels in comparison to pre-industrial conditions have been reported globally. In areas particularly impacted by commercial shipping, increases in excess of 12 decibels (dB) have been reported in the low frequencies (Hildebrand 2009). This increase in noise is concomitant to an increase in size and number of vessels in the global merchant fleet. These inputs are focused around shipping routes, as well as in waters around coastal cities and near major ports which also experience greater tourist-based and recreational traffic (Andrew et al. 2002; Frisk 2012; New et al. 2015). Smaller, often non-commercial, vessels also contribute to the mid- to high frequencies.

## Anthropogenic Noise as a Stressor

Elevated underwater sound levels from man-made sources are increasingly being recognized as a stressor for marine mammals that use acoustics as their principal means to send and receive information. Acoustic cues, received passively or via the echo of the animals' own signals, guide orientation and navigation, aid foraging and prey capture, and maintain group cohesion or social contact. The ambient sound level forms the acoustic scene that is interpreted and determines the range over which acoustic signals are effective.

Elevation of ambient sound may elicit a physiological and/or behavioral response in individual animals or groups. The impact on the animal, and, therefore, to some degree the response, is dependent on the received characteristics of the sound. Aspects such as the frequency, duration, intensity/amplitude, area impacted, and whether the noise is impulsive or continuous can instigate different responses. A scale of possible responses and physical effects has been described (see Erbe et al. 2019; Southall et al. 2019; Southall 2021).

On an individual level, the increase in noise may initiate modified calling or swimming behaviors. These may include the cessation of important behaviors or habitat use, thus altering energy budgets; increased physiological stress; a change in hearing sensitivity; and physical injury, especially in the hearing apparatus, or, in extreme cases, lead to the death of the animal (Erbe et al. 2019; Southall et al. 2019, 2021). Richardson et al. (1995) suggested four zones of sound propagation that

describe the spatial extent of acoustic signals and the implications of disturbance. First, and the most spatially extensive, is the zone of audibility, which is the area within which a sound exceeds an animal's hearing threshold and is detectable above background noise. This has also been termed the animals' "reverberation space" (Clark et al. 2009) and could also encompass the zones of detection and discrimination described by Erbe et al. (2016). Next, the zone of responsiveness is the region within which a reaction to the sound may occur, this is where signal recognition and interpretation occurs (Erbe et al. 2016), and the receiver acts in the way that was intended by the signaler. The zone of masking (Richardson et al. 1995) reflects the area in which the exchange of biologically significant sounds could be impacted by external noise. Within this is the zone of hearing loss, discomfort, or injury, where the sound level causes a threshold shift or hearing damage. The extent of these zones depends on characteristics of the sound and its receiver, as well as acoustic transmission properties of the area (Richardson et al. 1995; Johnson and Tyack 2003).

Sub-lethal effects include noise interfering, or masking, acoustic signals that are being sent and received by marine mammals. Cues from conspecifics, prey, predators, and those used as navigational aids may be obscured (Simmonds et al. 2004; Erbe et al. 2016, 2019). The simultaneous presence of a noise at the same frequency as an animal's signals can reduce the efficacy of the signal, decreasing the range over which a call may be detected. For communication and socially derived calls, the foreshortening of the communication space (Clark et al. 2009; Hatch et al. 2012; Burnham et al. 2023) hinders contact calling between conspecifics. The calls, whistles, burst pulses, and "clicks," for example, which animals utilize in communication, are critical. This includes social calls used for cooperative behaviors, breeding, or mother-young recognition and reduces success of navigation, aggregation, group coordination, and foraging. Typical acoustic responses to noise by marine mammals are to call louder, longer, or more often (NRC, 2003). The Lombard effect describes the noise-triggered adaptation of signal production, drawing on the phenotypic plasticity of acoustics use in vertebrates, to increase the amplitude of call to try and overcome the interference (see Kunc et al. 2022). This response has been noted in several cetacean species (e.g., Holt et al. 2009, 2011, 2015; Dunlop et al. 2014; Guazzo et al. 2020; Helbe et al. 2020; Thode et al. 2020). Also, the timing and direction of calling can be modified to allow for spatial, temporal, or co-modulation masking release (Erbe et al. 2016). However, these strategies of altered or increased vocal output can alter energetics of calling, which can have fitness consequences for the individual.

The amplitude and frequency of the noise are pertinent for the masking calculations, but also the rate of onset, intensity, rise time, length of exposure, repetitiveness, and spatial and temporal scale of the noise also shape the physiological response and whether it is seen on an individual or population level (Castellote et al. 2012). Near-distance and short-term reactions to acoustic disturbance have been described through observation and experimentation for several marine mammal species. These include longer dive times, shorter surface intervals, evasive movements to shield young, and increased swimming speed (e.g., Norris 1994; Gordon and Moscrop 1996; Frankel and Clark 1998; Croll et al. 2001). Avoidance behaviors

and displacement from critical breeding, rearing, and feeding areas have also been attributed to noise exposure. Marine mammals have changed vocalization rate and composition in response to noise. Alteration in call patterns includes frequency shifts, changes in energy levels of vocalizations, or making signals longer or more repetitive (e.g., Nowacek et al. 2007; Weilgart 2007; Tyack 2008; Rolland et al. 2012; Janik 2014). Modulation in the timing and directional components of calls has also been noted, with animals seeking temporal or spatial masking relief mechanisms by calling when noise levels are reduced or in orientations away from the source (Erbe et al. 2016). In some cases, calling can cease for extended periods (weeks to months), which may have serious behavioral implications.

Anthropogenic additions to ambient noise have been listed as a key threat to marine mammal populations and one of the factors that could hinder their success, survival, or recovery (see Weilgart 2007). For animals that rely so heavily on sound, the implications of masking, one of the lesser impacts of increased noise levels, have wide-ranging implications for both immediate- and long-term survival by reducing the ability to send and receive important information (Holles et al. 2013; Wale et al. 2013; Voellmy et al. 2014).

## Masking as a Metric

Noise additions can partially or fully reduce the audibility of signals over extensive spatial scales, where even a remote noise source could obscure a faint signal (Erbe et al. 2022). Although masking is a sub-lethal effect, it could underlie the behavioral and calling modifications noted in response to noise introductions. It reduces the capacity for cetaceans to communicate with conspecifics (e.g., Blackwell et al. 2013) and to gather acoustic information that could enhance their long-term success and survival (e.g., Holles et al. 2013; Wales et al., 2013; Voellmey et al. 2014). The extent of the masking is dependent on the spectral, temporal, and spatial relationship between signal and the masking noise (Southall et al. 2007).

In the absence of audiogram data, it is assumed that cetacean hearing is the most sensitive in the frequencies in which they call. Also, it is assumed that the impacts of masking will be highest in these frequencies, with noise sources interfering with the effective use of acoustic signals. Therefore, to be able to take the "whale perspective" (see Burnham 2017; Popper et al. 2020), frequencies pertinent to the focal species should be considered, with noise sources in these frequencies eliciting the greatest concern. This would mean considering the frequencies of calls in the species' full repertoire, from social conspecific calls to echolocation signals.

Conspecific communication calls transfer information between senders and receivers (Bradbury and Vehrencamp 2011). The range over which these calls are effective is defined by the distance over which they can be detected and acted upon by the intended receiver. The maximum range, however, is the distance over which these call types are discerned above minimal background noise levels. Echolocation range is determined by the distance over which a signal or "click" can be transmitted from a whale to a target and the returning signal be received at a level exceeding the

background noise and at an amplitude great enough to be both identified and interpreted. Masked echolocation signals and, therefore, the reduction of the echolocation range (Au et al. 2004) mean that the distance over which a signal can be transmitted and the returning signal be discerned and deciphered is reduced as a result of an obscuring noise. Both the communication and echolocation ranges are defined by the signal emitted and the ambient noise levels of the sound field that it is being projected into. The soundscape, therefore, defines the area or range over which these signals can be successfully employed.

The potential of a noise to be a source of masking can be estimated through empirical data. The calculations consider the co-occurrence of whales' calls and noise and the overlap between the signal and noise frequency and amplitude parameters. The change of effective communication or echolocation signals for a cetacean is expressed as a proportional reduction of the maximum range, calculated under historic, ideal, or minimum ambient sound levels as a reference (see Clark et al. 2009; Hatch et al. 2012; Burnham et al. 2023). This minimum sound level is used to describe the maximum range that communication calls and echolocation occurring in differing noise scenarios can then be expressed as a percentage of. Conveyed this way, masking becomes a metric of potential impact. This metric is a useful and intuitive means to express the impact of acoustic disturbance to managers and policy makers while taking an animal-centric approach. Calculating proportional rather than absolute changes allows comparisons over time and space. In this way, the masking metric could indicate areas most affected by the noise sources and where mitigation strategies are most needed. This expression of masking could also be used to represent the amount of time or space over which the whales are impacted by noise.

## Masking and Its Implications of Acoustic Disturbance

The masking effect on whale species from anthropogenic noise was first highlighted by Payne and Webb (1971) while considering the use of low-frequency calls by baleen whales to retain conspecific contact. The maximum range of 20 Hz fin whale (*Balaenoptera physalus*) calls was estimated under three noise scenarios. These were a moderate noise level, used to represent the low-frequency sound levels in the oceans at the time the work was undertaken when commercial shipping was active, and average and quiet pre-propeller ambient sound levels. These calculations showed substantial reductions in call range under the moderate noise compared to the quiet condition (Payne and Webb 1971). This study highlighted not only the importance of the acoustic modality to whales to retain their "herd" but also the impact that altered soundscapes can make on the efficacy of their calling. Further, Hatch et al. (2012) quantified the loss of communication space for the North Atlantic right whale (NARW, *Eubalaena glacialis*) at current shipping levels compared to pre-industrial ambient sound levels. This was defined by a difference of 10 decibels (dB) between the historic and current sound fields in the low frequencies where contact calls of NARW are focused (71–224 Hz; Urazghildiiev and Clark 2006;

Hatch et al. 2012). Recordings showed the ambient sound levels to be elevated up to 20 dB since the mid-twentieth century (Scheiefele and Darre, 2005; Hatch et al., 2008, 2009). This resulted in a 63–67% loss on average in area over which calls between conspecifics would be audible and a maximum of 87% lost communication space during noisy periods compared to the historic levels (Hatch et al. 2012).

For toothed whale species, the calculation of call range has refined researchers' perception of call function. Calculations by Jacobs (2019), for example, demonstrated that sperm whale (*Physeter macrocephalus*) coda calls were most likely used to communicate to nearby conspecifics as a within-group social display, rather than a longer-distance display of group membership which was the previously held presumed use (see Rendell and Whitehead 2003). This too can inform researchers on the social structure and group dynamics of odontocete species. The implications of commercial shipping noise on communication calls and echolocation signals for killer whales (*Orcinus orca*) have been shown by Burnham et al. (2023), comparing noise levels resulting from the current levels of commercial shipping to a derived minimum ambient sound level, where vessel and abiotic (wind and wave) noise is absent.

Both the work by Hatch et al. (2012) and Burnham et al. (2023) consider the implications of commercial shipping on an endangered whale species in ecologically important areas used for foraging. The focal areas of both works represent critical whale habitat, yet also have considerable commercial and recreational vessel traffic. Stellwagen Bank National Marine Sanctuary (SBNMS), the main area of the work by Hatch et al. (2012), is a highly productive area in the southern portion of the Gulf of Maine. The area is also, however, one of the most intensely used by commercial and recreational fisheries and encompasses the shipping lanes to Boston Harbor. Similarly, the Salish Sea, the waters around southern British Columbia and northern Washington State, the area of focus for the work by Burnham et al. (2023), sees commercial vessels transiting to ports in Vancouver, Seattle, Victoria, Nanaimo, Port Angeles, and Tacoma, and represents a major shipping corridor between Asia and North America. The shipping lanes in this area overlap greatly with known feeding areas for the endangered southern resident killer whales (SRKW). Whereas the work by Hatch et al. (2012) is a comparison over time, Burnham et al. (2023) use data from the same time period to look at the additions of noise from commercial vessels directly to the ambient soundscape. The principal consideration in this analysis was the potential implications to foraging behaviors resulting from these additions. By extending the calculations from purely reporting the difference in sound levels in decibels for these two conditions to a more biologically relevant metric, and expressing one as a proportion of the other, allowed Burnham et al. (2023) to empirically describe the acoustic disturbance to the whales in this area and discuss the change in soundscape in terms of its possible ecological impact.

Aggregate noise models using automatic identification system (AIS) vessel data and sound transmission models have shown that the Salish Sea and surrounding areas have the highest ambient noise levels along the British Columbia coast (Erbe et al. 2012). Commercial vessels and their rate of passage are consistent throughout the year, emitting persistent noise into the ambient sound field. Bulk carriers or

container ships are the most frequent vessels (Veirs and Veirs 2006; Erbe et al. 2012; Veirs et al. 2016), with each passage shown to elevate the sound levels approximately 20 dB re 1 μPa in frequencies below 10 kHz relative to ambient levels (Veirs et al. 2016). A similar, but more fine-scale, model than that used by Erbe et al. (2012) highlighted noise increases in the shipping lanes, in shallow water, and in regions where vessels turn or maneuver within the Salish Sea. This model also allowed the consideration of the changes in sound levels through the water column, considering typical swimming depths and diving depths of whales to be able to better understand their perceived soundscape. An upper water column estimate represented traveling behavior, a mid-water depth was used for prey search and capture depths, and 100 m reflected the maximum repeatable foraging dive depth for SRKW as logged by animal-borne tags (see Wright et al. 2017; Baird 2000, Baird et al. 2005; Tennessen et al. 2019a, 2019b). This allowed a more comprehensive understanding of the impacts of vessel noise on the whales feeding in this area. Changes in the sound levels in frequencies that reflected communication calls and echolocation signals were considered. Conspecific communication calls of SRKW include whistles (2–17 kHz; Ford 1989; Thomsen et al. 2001) and burst pulse sounds (1–15 kHz; Ford 1989) as social signals. These pulsed calls may also have modulated higher- or lower-frequency components (Miller and Bain 2000). The frequency range of 1–40 kHz was considered to encompass the full repertoire of social signaling among SRKW, with the distance possible for conspecific calling estimated from parameters and source levels described by Holt et al. (2009, 2011). Ultrasonic echolocation signals of SRKW in the 15–100 kHz range (Heise et al. 2017) are for foraging and navigation (Ford 1987, 1989, 1991). Au et al. (2004) previously considered the implications on echolocation from changes in sea state and the addition of abiotic noise. For their calculations, Au et al. (2004) used a 50 kHz echolocation signal to find a model average-sized chinook salmon (*Oncorhynchus tshawytscha*) under differing sea state and rain conditions. The same candidate frequency of 50 kHz, signal source levels, and target strength of the average chinook salmon were replicated from the definitions used by Au et al. (2004) to estimate the signal interference from commercial vessels (Burnham et al. 2023). Greater range reductions over a greater spatial extent were apparent for the communication call frequencies compared to the echolocation candidate frequency at each depth considered and through time. For the communication frequencies, reductions were greatest October to March, when sound transmission was most effective (Vagle et al. 2021). During this period, calculations suggested a 90% reduction in the conspecific call range. The impacts of commercial vessels at 50 kHz were confined to the shipping lanes, with the distance over which signals could be accurately sent, received, and detected by an individual reduced in excess of 50% at all times and in all areas of the Salish Sea due to commercial vessel traffic. During the winter months, the model outputs suggested that communication calls and echolocation signals would not exceed background noise levels from commercial vessel transits. The reductions for both signaling types were greatest at the 50 m depth, representing the typical swimming depth of chinook salmon prey (Wright et al. 2017). Also, the areas most impacted by noise, quantified by the greatest reduction especially in the

communication frequency range, were foraging areas (Olson et al. 2018; DFO 2021).

A more reduced acoustic range indicates the increased likelihood that the source of noise would create a negative effect. Being expressed as a proportional reduction allows direct comparison between both noise sources and frequency ranges considered and allows a comparable examination of noise impacts through time and space. The analysis on SRKW by Burnham et al. (2023) showed the noise resulting from the presence of commercial vessels would be of greatest concern during the winter months. However, knowledge on SRKW presence and distribution during this time is poor, and so the full implications of the noise relative to whale habitat use cannot be defined. Movements of SRKW and their timing are tied to prey. The sightings in the Salish Sea between April and September are linked to the migration of chinook salmon, whereas winter movements may follow chum salmon (*Oncorhynchus keta*) runs (Osborne 1999; Vélez-Espino et al. 2014), but the correlative strength is yet to be determined. When interpreting the acoustic impact over horizontal and vertical space, it is necessary to keep the potential changes in habitat use and prey in mind. Reductions in communication call range could hinder the coordination in prey location and capture and then the sharing of fish post-kill (Ford and Ellis 2014; Wright et al. 2016). The reduction of range in echolocation reduces the area over which prey could be identified and pursued, reducing prey availability (Au et al. 2004; Sato et al. 2021).

Applying these same methods of modeling to calculations of reduced acoustic ranges to wind speed data, to represent wind and wave noise in the soundscape, showed that even at the highest wind speeds and at times or in locations where wind noise was substantially elevated, the natural noise source would not meet or exceed that of commercial vessel noise. The greatest reduction in wind noise scenarios was to 40% of the maximum range, which was consistent with theoretical estimates calculated by Au et al. (2004) in higher sea states or periods with heavy rain. The excess of the vessel noise was particularly highlighted for SRKW echolocation frequencies, where the anthropogenic source outweighed wind noise in the shipping lanes. This simple subtraction showed the anthropogenic component of the Salish Sea soundscape is the dominant input over time and space and the main acoustic disturbance to SRKW signaling and sound reception. Existing variability and gradients in ambient noise levels are important aspects to consider when designing management plans. The consideration of wind-derived noise is one means of considering the natural, non-biological sound field components. The clearer the understanding of which component of the soundscape is dominant over various spatial and temporal scales, the more effective the mitigation measures of human-derived noise will be.

## Masking as a Metric for Management

The expression of anthropogenic noise impacts through masking quantified as a reduction in effective signaling range could be beneficial for managers designing mitigation strategies. It could suggests avenues to reduce the disturbance or

highlights regions where measures should be prioritized. At their simplest, these range reductions are calculated using the maximum detection range of calls, with this defined by a signal-to-noise ratio (SNR) that exceeds zero, under different noise scenarios. This, however, may not acknowledge the need to have a greater SNR to clearly decipher the information encoded in the calls or signals. This does also not consider the hearing thresholds from audiogram data that is available for some species. A variable representing hearing sensitivity could be included in future iterations of agent-based modeling techniques to better estimate the impact of the noise sources on whales and would add a correction factor to the absolute range estimations. As weighting may be given to different frequencies in the vocal or listening range, the results of the model output could also be weighted by the importance of the area to the whales, for example, if it was applied to critical habitat representing a foraging or breeding area, or weighted in terms of the observation rate or residency time of whales in a locale.

Plasticity in calling may also allow for mitigation of noise on the use of acoustics. Inclusion of altered amplitude of communication calls (e.g., Foote et al. 2004; Holt et al. 2009, 2011, 2015; Dunlop et al. 2014; Guazzo et al. 2020; Helbe et al. 2020; Thode et al. 2020), altered inter-pulse interval in echolocation has been noted (e.g., Lammers et al. 2004; Madsen et al. 2005; Morisaka et al. 2011), or modification in timing and direction of calling to allow for spatial, temporal, or co-modulation masking release (Erbe et al. 2016) to model calculations may help develop more realistic scenarios. Currently, without these refinements, models represent the worst-case scenario for masking by commercial traffic with much refinement still needed.

The potential acoustic impact from recreational vessels is also being increasingly realized (e.g., Hermannsen et al. 2019). The presence of these vessels is much harder to quantify for many areas as they are not mandated to carry an AIS transceiver. However, they can have a notable influence on the soundscape, especially in the higher frequencies. The inclusion of small vessel traffic into models will likely increase the impact of vessel noise and reduce effective acoustic ranges even further than has so far been demonstrated.

The approach discussed in this chapter advocates for taking a species-centered or even individual approach to understanding disturbance of ecologically relevant signals and is a better means to quantify the potential effect of noise additions than soundscape descriptions alone. Several analyses in masking (e.g., Clark et al. 2009; Hatch et al. 2012; Burnham et al. 2023) demonstrate that increases in anthropogenic sound sources reduce the extent over which these acoustic signals may be effective. This has implications for successful navigation, group cohesion and foraging, and prey capture. Particularly in areas like SBNMS (Hatch et al. 2012) and the Salish Sea (Burnham et al. 2023), which have been identified as critical habitat for species and encompassing important foraging regions, vessel noise threatens acoustical acuity for prey location, pursuit, capture, and sharing. For more social species like SRKW, masking limits the area over which prey is captured. Also, the number of individuals and/or distance between the group members over which the prey might be shared would also be lessened, reducing the overall successful outcome of foraging. This may then have health implications for individuals and the success of populations.

Metrics expressed as a percent of habitat or time individuals or populations are impacted by a certain level of acoustic disturbance could be a useful tool to present to management agencies. For example, expressing the amount of designated critical habitat that would be subject to 50% or greater reduction in communication or echolocation signals at a particular time or the proportion of time spent in the critical area that a whale is subject to a certain level of range reduction gives a direct dose-response type of figure to a manager. These form a tangible presentation of effect and clearly tie the noise additions to a definable impact. This might be most usefully applied to migrating species, for example, when the ecological relevance and the time period that whales are present are more strictly defined.

It should also be noted that the masking metric adds more evidence and quantification of acoustic disturbance to already documented changes in swimming and calling behaviors for whale species in the physical and acoustic presence of vessels (e.g., Holt et al. 2009, 2011, 2015, 2021; Dunlop et al. 2014; Williams et al. 2014; Guazzo et al. 2020; Helbe et al. 2020; Thode et al. 2020). For many species, there are undoubtedly implications for the cumulative effects from multiple stressors or the same stressor repetitively that could be present in much of a species' range that have yet to be fully described (e.g., Christiansen et al. 2015; Montes et al. 2020; Silber et al. 2020). Also, masking precedes auditory injury or changes in hearing ability through auditory fatigue and temporary threshold shift (TTS) or a permanent change in sensitivity or threshold shift (PTS). A masking metric could help guide mitigation measures to the areas most needing noise reduction before anthropogenic additions meet or exceed the levels that might instigate TTS, PTS, or physiological stress responses. The dose-response relationships, ultimately leading to changes in distribution and success as a result of ocean noise, have yet to be conclusively determined for cetaceans. The ultimate response of habitat abandonment may be a direct result of acoustic disturbance or an indirect impact through changes in prey availability. This could be through startle or displacement responses of the whale, or equally by its prey, or reduced detection of prey as a result of increasing ambient noise levels (e.g., Sato et al. 2021). This would make the effect of acoustic disturbance on foraging possibly hard to disentangle the behavioral responses from prey decline.

Analyses using masking as a metric of impact serve as one of the building blocks for scenario development and noise-level predictions. This could aid evidence-based management actions. Predicted near-future increases in vessel traffic and capacity internationally will heighten the risk of acoustic disturbance for many cetacean species, and techniques like those discussed in this chapter may help determine their effect.

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# Hearing Impact on Marine Mammals due to Underwater Sound from Future Wind Farms

80

Frank Thomsen, Uwe Stöber, and Joanna Sarnocińska-Kot

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**Abstract**

Offshore wind farms are part of the transition to renewable sources of energy, and both the total numbers and sizes of wind turbines are rapidly increasing. While the impact of underwater sound related to construction work has been in the focus of research and regulation, few data exist on the potential impact of underwater sound from operational offshore wind farms (OWF). Here, the knowledge gained on operational noise from OWF and its potential effects on marine mammal behavior is synthesized. In a case study, effects of a 10- and 20-MW direct drive OWF turbine on the hearing of marine mammals were assessed using a simple model spreadsheet based on geometrical spreading loss and hearing types as defined by NMFS/NOAA. Effects over 24 h exposure from a single turbine ranged between a few meters and a little over 700 m depending on the effect (permanent threshold shift (PTS)/temporary threshold shift (TTS)) and hearing group investigated. In general, impact ranges for PTS were small (i.e., up to 50 m from the sound source) and are thus likely negligible. The same was true for TTS impact ranges for 10 MW. However, impact ranges for TTS from a 20 MW

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turbine could reach a little over 700 m for low-frequency cetaceans which could bridge distances between wind turbines. Thus, impact areas from single turbines might overlap and the whole wind farm can be considered an impact area. It is concluded that for larger size wind turbines, operational noise needs to be considered in sufficient detail as a part of the environmental impact assessment in the wind farm planning phase. In addition, further observations and modeling efforts are necessary to increase the accuracy of the estimates.

**Keywords**

Offshore wind farm · Noise · Fish · Marine mammal · Operational noise · Construction noise · Cumulative impacts

## Introduction

Within the fast-growing research area of aquatic noise, the effects of marine renewables, especially offshore wind farms (OWF), have received special attention. This is because plans to develop OWF have become ever more ambitious, not only in Europe where the industry has established itself in the past 20 years but also in many other areas such as off the US East coast, off the coast of Taiwan, Australia, and China. As the industry grew, so did the need to understand the impacts on marine life. Research on offshore wind farm-related noise and its effects on marine mammals has been undertaken since 20+ years with a focus on European waters and species that are most abundant there such as harbor porpoises (*Phocoena phocoena*) and to a certain extent harbor and gray seals (*Phoca vitulina* and *Halichoerus grypus*). Up until recently, the scientific consent was that construction noise is the main issue if impact pile driving is being used due to the relatively high sound intensities and consequently large impact ranges associated with this activity. Operational noise was not considered an important topic due to the much lower sound intensities during that phase (see reviews by Madsen et al. 2006; Thomsen et al. 2006, 2016). These findings were corroborated by field studies which found little adverse impact of existing OWFs on porpoises and seals (Thomsen et al. 2016). However, recently, Stöber and Thomsen (2021) found indications that behavioral impact areas from larger size wind turbines, such as those planned to be installed in the future, could overlap and the whole wind farm might be considered an impact area. They concluded that for larger size wind turbines, operational noise needs to be considered in sufficient detail as a part of the environmental impact assessment. Yet, Stöber and Thomsen (2021) also pointed out that further observations and modeling efforts are necessary to increase the accuracy of the estimates.

One issue that was largely left out in the study by Stöber and Thomsen (2021) was the effect of underwater noise from future turbines on the hearing of marine mammals in the form of temporary and permanent hearing threshold shifts (TTS and PTS). Intuitively, this was not perceived as an important issue since operational noise levels from future turbines are still relatively low compared to the impact

criteria TTS and PTS (NMFS 2018; Southall et al. 2019). However, as proven in a milestone investigation on harbor porpoises in captivity, even relatively low-level exposure to continuous noise can lead to TTS if exposure duration is long (Kastelein et al. 2012). Noise from OWF turbines in operation is more or less permanent, and some investigations indicate that porpoises might get attracted to wind farms perhaps because they represent refugees from fishing activities or due to higher fish abundance resulting from an artificial reef effect (see review by Thomsen et al. 2016). Thus, it is worth exploring whether the concerns about behavioral impacts from offshore wind farms in operation are also valid for hearing impairment.

This chapter builds on the study by Stöber and Thomsen (2021) to include the potential of operational noise from offshore wind farms for hearing impairment in the form of TTS and PTS. First, the recent findings by Stöber and Thomsen (2021) are summarized. This is followed by a case study estimating impact ranges for PTS and TTS for different marine mammal hearing groups for 10 and 20 MW OWF turbine. The chapter closes by discussing the results with an emphasis on pointing out knowledge gaps and priorities for further investigations.

## Summary of Recent Findings

Previous reviews of state of knowledge about operational noise have been published by Madsen et al. (2006), Thomsen et al. (2006), and Thomsen et al. (2016). Two more recent investigations by Stöber and Thomsen (2021) and Tougaard et al. (2020) have performed a systematic analysis that should be summarized here quickly:

- OWF operational sound is originating in the nacelle with most energy below 1 kHz. Another source of sound is wind-induced vibration of the tower. The sound is transmitted through the tower and then radiated from the foundation into the water. Thus far, different foundation types (i.e., monopiles, tripod, jackets, and gravity-based) do not seem to lead to systematically different sound profiles (Tougaard et al. 2020).
- The broadband rms SPLs normalized to 1 m distance from the source ranges from 129 to 166 dB re 1 μPa m (nominal power = 0,45–6,15 MW; Stöber and Thomsen 2021). Similarly, Tougaard et al. (2020) found turbine noise levels 10–20 dB below the received levels measured from ships for the same distance.
- Emitted sound levels increase with turbine size (Tougaard et al. 2020: 13.6 dB/decade in meters) and wind speed (Tougaard et al. 2020: 18.5 dB/decade in m/s).

Tougaard et al. (2020) concluded that the noise footprint of individual turbines is small compared to other sources such as ships. However, they also demonstrated that cumulative noise levels from an entire wind farm could be elevated above background noise up to a few kilometers from a wind farm under very low ambient noise conditions. Based on the existing data, Stöber and Thomsen (2021) extrapolated a broadband source level of 170 dB re 1 μPa m for a 10 MW turbine with a gearbox.

The broadband source level of a 10 MW direct drive turbine was assessed to be 10 dB quieter (160 dB re 1 μPa m). Using existing regulatory criteria, a 10 MW direct drive turbine was expected to cause behavioral response in marine mammals up to 1.4 km distance from the turbine, compared to 6.3 km for a turbine with gearbox (Stöber and Thomsen 2021).

## Potential for Hearing Impairment (TTS and PTS)

Building on the work by Stöber and Thomsen (2021), a case study on the potential of future OWF turbines (10 and 20 MW) for hearing impairment in the form of TTS and PTS was undertaken. For the 10 MW direct drive turbine, the broadband sound pressure level of 160 dB re 1 μPa m as derived by Stöber and Thomsen (2021) was used. For the estimation of the source level for a 20 MW turbine, the data by Tougaard et al. (2020) was reevaluated and new fits of the dependence of the SPL on the power and wind speed were derived considering only the monopile datasets. Applying these pure monopile trends, a source sound pressure level of 182.4 dB re 1 μPa m was derived. Then 10 dB were subtracted to arrive at an estimate for a direct drive turbine, which leads to a final source level of 172.4 dB re 1 μPa m for a 20 MW for this turbine type.

The impact ranges for TTS and PTS for the different marine mammal hearing groups were calculated using the NOAA/NMFS spreadsheet tool (available from https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-acoustic-technical-guidance). According to the supplied information in the spreadsheet tool, a weighting factor adjustment of 2 kHz was applied. The propagation loss coefficient was set to 15 which is between special and cylindrical spreading and close to transmission loss characteristics in many coastal waters where OWFs are being planned (see, for example, Madsen et al. 2006; Thomsen et al. 2006). The exposure duration was set to 24 h.

Effects over 24 h exposure from a single turbine ranged between a few meters and a little over 700 m, depending on effect (PTS/TTS) and hearing group investigated (Table 1). In general, impact ranges for PTS were small (i.e., up to 50 m from the sound source) and are thus likely negligible. The same was true for TTS impact ranges for 10 MW. However, impact ranges for TTS from a 20 MW turbine could reach a little over 700 m for low-frequency cetaceans which could be larger than distances between wind turbines. Thus, impact areas from single turbines might overlap and the whole wind farm can be considered an impact area.

## Discussion

This chapter indicates that underwater noise from 10- to even 20-MW OWFs in operation will likely not lead to any significant impacts due to auditory injury in marine mammals even if the animals stay on the wind farm area for a day. Even temporary hearing impairment might not be an important issue for the future 10 MW

**Table 1** Impact ranges in isopleth to threshold (meters) for a projected 10 MW and 20 MW OWF turbine for different marine mammal hearing groups (PTS and TTS thresholds, see NMFS 2018; Southall et al. 2019)

| Hearing group | Low-frequency cetaceans | Mid-frequency cetaceans | High-frequency cetaceans | Phocid pinnipeds | Otariid pinnipeds |
|---|---|---|---|---|---|
| **10 MW** | | | | | |
| **PTS isopleth to threshold (meters)** | 4.9 | 0.3 | 4.3 | 2.6 | 0.2 |
| **TTS isopleth to threshold (meters)** | 105.6 | 6.0 | 92.6 | 56.5 | 4.1 |
| **20 MW** | | | | | |
| **PTS isopleth to threshold (meters)** | 32.9 | 1.9 | 28.8 | 17.6 | 1.3 |
| **TTS isopleth to threshold (meters)** | 708.6 | 39.9 | 620.9 | 379.2 | 27.6 |

turbines. However, for larger turbines, the noise generated can lead to effects that need consideration because sound levels from individual devices might overlap, making the whole wind farm area a zone of impact. Acoustic interference between sounds from different turbines was not investigated, but it is likely that the cumulative sound field resulting from interference and then emitted from an OWF is larger than individual sound fields just added up. It is theoretically possible that impact ranges for cumulative TTS (i.e., over 24 h) are larger than those for behavioral avoidance. And then animals might stay in the area and experience TTS. There is also the possibility that marine mammals are attracted to the wind farm area due to artificial reef effects and that they prioritize feeding against moving away because of the operational noise exposure. Looking at the vast amount of planned OWF in some regions and the considerable size of some project areas (see, for example, Global Offshore Renewable Map | 4C Offshore), this potential issue needs further investigation. This is especially true if we consider that many future wind farms are being planned in sea areas that overlap with important habitats of low-frequency large baleen whales for which the impact ranges in this case study were the highest compared to all other hearing types.

However, the results obtained with the simple model applied here come with considerable uncertainties. Looking at the relatively large distances traveled by many marine mammals on any given day, it could appear unlikely that these animals will stay in the relatively small project areas for 24 h. Thus, the estimated TTS areas seem theoretical. Animal movement modeling, for example, using individual-based modeling has great potential to make impact assessments more realistic, leading to more accurate predictions of animals exposed to a pressure (see Thomsen et al. 2019).

It is also worth pointing out that changing any of the variables of the applied model leads to considerable differences in the results. For example, reducing the weighting factor adjustment from 2 kHz to 1 kHz will lead to a considerable

reduction in impact ranges for high-frequency cetaceans. Local sound propagation might also be very different from the 15 log (R) spreading law chosen here leading to completely different impact ranges. Furthermore, the simplistic model approach used here does not account for acoustic properties of the seabed and local bathymetry which hugely influence sound transport. It is recommended to apply numerical modeling to improve the precision of the assessment.

One important concept that was not applied is the "effective quiet" as defined by Finneran (2015) as the highest SPL that would not produce a significant TTS or affect recovery from a TTS produced by a prior, higher-level exposure. Finneran (2015) then indicate that for harbor porpoises, this value could be 124 dB re 1 μPa and cites the study by Kastelein et al. (2012) mentioned above in support of this conclusion. Kastelein et al. (2012) did not investigate directly into "effective quiet" but rather showed that very low sound exposures can lead to significant TTS in porpoises when exposure duration is long. The 124 dB re 1 μPa was the lowest sound level that led to TTS. To the authors' knowledge, effective quiet is unknown for low-frequency cetaceans and applying just an arbitrary value would be very speculative.

## Conclusion

This study confirms earlier work by Stöber and Thomsen (2021) who pointed at potential effects of operational OWFs on marine mammals that should be considered more thoroughly in regulatory management. But it is also evident that the presented results should be interpreted with caution. They should represent a starting point for a necessary discussion about environmental costs of OWF facilities and nothing more.

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# Impulsive or Non-impulsive: Determining Hearing Loss Thresholds for Marine Mammals

81

David G. Zeddies, Samuel L. Denes, Klaus Lucke, S. Bruce Martin, and Michael A. Ainslie

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**Abstract**

It is well known that short loud sounds and longer quieter sounds can produce hearing loss. The idea that equivalent sound energy results in equivalent hearing loss founds our thinking about hearing loss, including regulations to protect hearing (e.g., NIOSH 1998). But it is also recognized that similar level sounds with different temporal characteristics may result in different amounts of hearing

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loss. This is codified for regulatory purposes (e.g., NMFS 2018) as lower thresholds for predicting hearing loss in marine mammals exposed to impulsive sounds compared to non-impulsive sounds. Currently, a qualitative approach is used to classify sound sources, which is problematic because sources can produce various sounds, some sources do not neatly fit into either category, and sound characteristics change with propagation.

Kurtosis is a measure of the probability distribution of received sound levels. For hearing, kurtosis appears to better predict hearing loss from exposure to some sounds with differing amounts of transients. The kurtosis of a signal can be used to adjust the received level, allowing for a smooth transition between impulsive and non-impulsive sounds. The adjustment focuses on sounds instead of sources, allows for the consideration of propagation effects, and includes the animals' hearing frequency sensitivity.

**Keywords**

Kurtosis · Impulsive · Non-impulsive · Hearing loss · Marine mammal · Regulatory thresholds

## Introduction

It has long been known that short loud sounds may cause hearing loss as well as longer quieter sounds (reviewed by Akay 1978). This observation led Eldred et al. (1955) to suggest that exposure to equivalent sound energy results in equivalent hearing loss, the equal energy hypothesis (EEH). The EEH has formed the basis of our thinking about hearing loss, including regulations to protect hearing in humans (e.g., NIOSH 1998).

It has been shown repeatedly, however, that the EEH does not accurately predict the extent of hearing loss – in general, exposure to short loud (impulsive) sounds are thought to result in more hearing loss than longer duration sounds of equivalent sound energy (Ward 1962; Hamernik et al. 2003; though see Mooney et al. 2009 for a counter example). In their study on chinchillas, Hamernik et al. (2003) investigated how the amplitude distribution of received sound affects hearing loss. While keeping the total sound energy and spectra the same, Hamernik et al. (2003) presented sounds with Gaussian-distributed amplitude levels and non-Gaussian distributed amplitude levels produced by inserting high-level transients into otherwise Gaussian-distributed noise. The non-Gaussian noise with high-level transients produced greater hearing loss.

With respect to hearing loss in marine mammals, Southall et al. (2007) recognized that temporal aspects of sound may result in different amounts of hearing loss and proposed lower thresholds for "impulsive" sounds compared to "non-impulsive" sounds. The current regulatory thresholds for predicting hearing loss in marine mammals are 13–18 dB lower (for auditory frequency-weighted sound exposure level, SEL, thresholds in the different hearing groups) for impulsive sounds than

non-impulsive sounds (Finneran et al. 2017; NMFS 2018). However, how to determine whether a sound is impulsive or non-impulsive is not defined.

A qualitative approach is currently taken to classify sound sources (NMFS 2018). For example, impact pile driving and seismic air guns are assumed to always produce impulsive sounds, while sonar-type sources, including multibeam echosounders, are assumed to always produce non-impulsive sounds (NMFS 2018). The qualitative approach with guidance by exemplar is problematic, however, when considering the variety of sounds that sources can produce and that some sources do not neatly fit into one category, such as down-the-hole hammering that consists of drilling (non-impulsive) and hammering (impulsive) (Guan et al. 2022). The problematic nature of the qualitative approach is further evident when noting that sound characteristics change with propagation. Sound tends to spread out in time as it moves away from the source. Sounds, such as those produced by air guns, may be pulsatile near the source but, because of differences in multipath arrival time, may be a nearly constant rumble far from the source (e.g., see Martin et al. 2017).

## Defining Impulsive and Non-impulsive Sounds

### Qualitative Definitions

ISO (2016) defines "impulsive sound" as "sound characterized by brief bursts of sound pressure." NMFS (2018) defines impulsive sources as those that "produce sounds that are typically transient, brief (less than 1 second), broadband, and consist of high peak sound pressure with rapid rise time and rapid decay" and defines non-impulsive sources as those that "produce sounds that can be broadband, narrowband or tonal, brief or prolonged, continuous or intermittent) and typically do not have a high peak sound pressure with rapid rise/decay time that impulsive sounds do." In these definitions, NMFS (2018) cites standards for measuring sound levels in air (NIOSH 1998; ANSI/ASA S1.13-2005 (R2010); ANSI S3.20-1995; ANSI/ASA S12.7-1986) but removes quantitative criteria, which may result in different interpretations. For example, ANSI (S1.13-2005 (R2010)) does not preclude harmonic signals (including frequency modulated sounds) from being classified as impulsive. The qualitative definitions used by the Finneran et al. (2017) are similar to NMFS (2018) but do state that harmonic signals with more than ten cycles are considered a steady state (non-impulsive). And there are other classification approaches, such as TG Noise (van der Graaf et al. 2012), which defines an impulsive sound as one "for which the effective time duration of individual sound pulses is less than ten seconds and whose repetition time exceeds four times this effective time duration."

NMFS (2018) calls out impact pile driving and seismic air guns as examples of impulsive sources and active sonar as non-impulsive. Sonar sources, such as multibeam echosounders (MBES), however, can produce high-level sounds with shorter-duration signals and a more rapid rise to high pressure than impact pile driving or seismic air guns (Fig. 1). MBES are classified by NMFS (2018) as non-impulsive

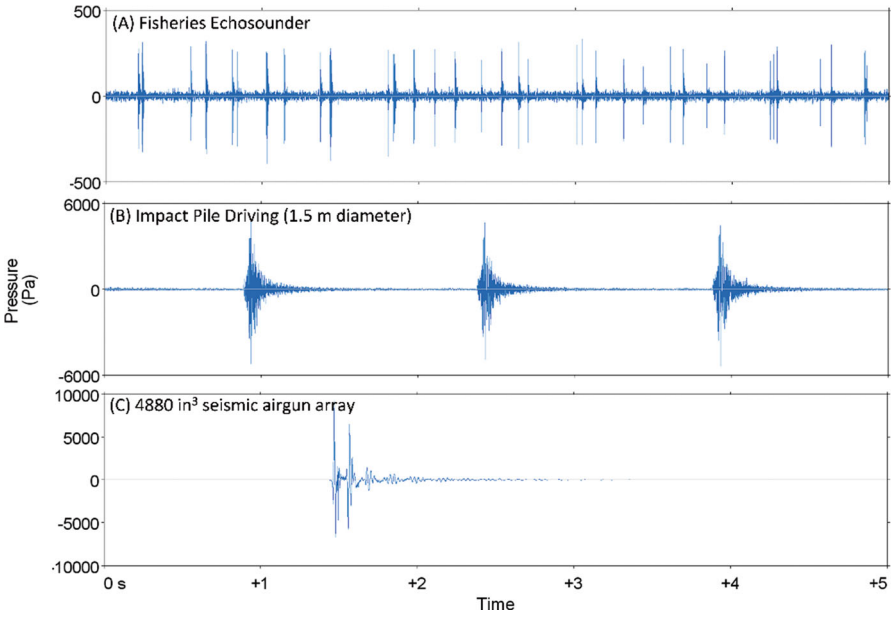



**Fig. 1** Five-second sound pressure time sequences of three anthropogenic sources: (**a**) a fisheries echosounder recorded at ~500 m distance; (**b**) impact pile driving (1.5 m diameter pile) at 541 m distance; and (**c**) a 4880 $in^3$ seismic air gun array recorded at ~700 m distance

and use the non-impulsive thresholds for predicting permanent hearing threshold shifts (PTS) despite producing signals that fit within the qualitative descriptions of impulsive sources.

## Quantitative Definitions

Quantitative definitions to classify signals as impulsive or non-impulsive to better gauge their potential impacts on hearing have been suggested. Harris (1991) suggested an impulse factor that is easily calculated as the sound level measured using impulse weighting (35 ms decay constant) minus the sound level measured with either fast or slow time weighting (0.125 or 1 s time constants, respectively) on standard in-air sound level meters. Based on Harris (1991), Southall et al. (2007) recommend an impulse factor (the difference in sound level meter measurements using continuous and impulse settings) of greater than 3 dB to distinguish between impulsive and non-impulsive sounds. Using the Harris (1991) approach of comparing sound levels measured using impulse weighting to fast weighting in the examples in Fig. 1, the fisheries echosounder is 11.3 dB, impact pile driving is 18.3 dB, and seismic air gun is 22.9 dB.

Kurtosis, the over- or underrepresentation of data in the tails of a distribution, has also been suggested as a proxy for the impulsiveness of sound. For hearing, kurtosis (ISO 2017) is a measure of the probability distribution of received sound levels (Müller et al. 2020) and has been successfully used to predict hearing loss in terrestrial mammals from exposure to sounds with differing amounts of transients (Qiu et al. 2020). In marine mammals, there are few data available on the effectiveness of using kurtosis for predicting hearing loss. von Benda-Beckmann et al. (2022) found that for harbor porpoise exposed to intermittent sounds with silent intervals of 1–13 s, kurtosis-corrected frequency-weighted SEL accurately predicted TTS (for TTS <10 dB) but kurtosis-correction was not predictive of TTS for exposure to continuous sounds.

The kurtosis of a signal with Gaussian-distributed received levels (white noise), which can serve as a baseline, has a kurtosis of 3. Qiu et al. (2020) found that signals with greater kurtosis are associated with greater hearing loss (hair cell damage), with the amount of hearing loss asymptoting for signals with kurtosis greater than 40 (using a 40 s window for calculating kurtosis). A sound signal that has a kurtosis of 0–3 could sensibly be classified as non-impulsive for regulatory purposes, while a sound signal with a kurtosis greater than 40 could be classified as fully impulsive. The kurtosis in the Qiu et al. (2020) examples is 7.2 for the fisheries echosounder, 58 for impact pile driving, and 494 for the seismic air gun. Using kurtosis, the source classifications differ somewhat from using the Harris metric, because the echosounder is only marginally impulsive.

## A Kurtosis Adjustment?

Using qualitative definitions to classify sources as impulsive or non-impulsive leads to inconsistently classified received sounds. With PTS thresholds for evaluating potential impacts differing by 13–18 dB (depending on hearing group), misclassification could result in missed predictions of injury or injury predictions when none would occur.

Using a binary classification scheme (impulsive and non-impulsive) further complicates the situation. For sources that can produce various sounds, should all their sounds be classified the same? And, conversely, should identical sounds that are classified differently because they come from different sources be assessed using different thresholds? The transients that Hamernik et al. (2003) added to the noise were not necessarily impulses, and the overall sound signal was not necessarily impulsive. There was an overrepresentation of high sound levels in the presentations that resulted in greater hearing loss compared to baseline noise with the same sound energy. Increasing the proportion of high-level sounds in a signal smoothly increases the amount of hearing loss (up to a point) as opposed to a step change in hearing loss that would suggest discreet classification into two categories.

To better predict hearing loss in individuals exposed to signals with overrepresentation of high sound levels, Goley et al. (2011) suggested adjusting the reported sound level based on the logarithm of the ratio of the signal kurtosis, $\beta$, to the kurtosis of a Gaussian signal, $\beta_G$, multiplied by a scaling factor, $\lambda$, as shown in the equation below:

$$L'_{E,W} = L_{E,W} + \lambda \log_{10} \frac{\beta}{\beta_G} \text{ dB},$$

where $L_{E,W}$ is the reported sound level and $L'_{E,W}$ is the adjusted level.

Taking, for example, the SEL threshold for high-frequency (HF) cetaceans exposed to non-impulsive sounds with an HF-weighted SEL of 173 dB re 1 μPa$^2$·s and impulsive sounds as HF-weighted SEL of 155 dB re 1 μPa$^2$·s (NMFS 2018), there is an 18 dB difference. The kurtosis of a Gaussian signal ($\beta_G$) is 3. Assuming a signal with a kurtosis of greater than 40 is fully impulsive, dividing 18 dB by the logarithm of the kurtosis ratio (40/3) gives a $\lambda$ of 16.0. Substituting into the equation above, the equation below could be used to adjust SEL for HF cetaceans:

$$L'_{E,\text{HF}} = L_{E,\text{HF}} + 16.0 \log_{10} \frac{\beta}{3} \text{ dB}, \quad 3 \le \beta \le 40 \, .$$

To evaluate the potential of a sound to cause hearing loss (PTS) in, e.g., HF cetaceans, the process is to: (1) calculate the kurtosis of the received sound, $\beta$, (limiting $\beta$ to be between 3 and 40); (2) use the equation to find the adjusted SEL; and (3) compare the adjusted SEL to the non-impulsive SEL threshold for PTS (NMFS 2018). Only the non-impulsive threshold (HF-weighted SEL of 173 dB re 1 μPa$^2$·s for HF cetaceans) is needed because adding to the SEL is equivalent to reducing the threshold. In other words, scaling the metric by the kurtosis adjustment precludes the need to scale the threshold by the "impulsiveness" of the signal. Gaussian distributed signals with kurtosis of 3 or less will receive no adjustment to SEL, so their effective threshold is equal to the non-impulsive threshold (SEL of 173 dB re 1 μPa$^2$·s for HF cetaceans). Pulsed signals with kurtosis of 40 or higher will have an effective PTS threshold equal to the impulsive PTS threshold (HF-weighted SEL of 155 dB re 1 μPa$^2$·s). Sounds with kurtosis between 3 and 40 will have effective thresholds between the non-impulsive and impulsive PTS thresholds. In this way, there is a smooth transition between signal types. Sounds with some impulsiveness (intermediate kurtosis) will incur some penalty but would not necessarily be considered impulsive.

Changes in the spectral content of a signal affect the kurtosis calculation. Because the best hearing frequency range varies with species, species-specific weighting functions should be applied before the kurtosis calculation. In this way, the kurtosis value, and the apparent impulsiveness of a signal, is specific to the hearing group. Martin et al. (2020) calculated kurtosis for real signals for different hearing groups. Martin et al. (2020, their Fig. 6b) found that a fishing vessel with an echosounder has a kurtosis value of ~3 when unweighted or when weighted for low-frequency (LF) cetaceans. When weighted for HF cetaceans (classified as "very high frequency

cetaceans" according to the nomenclature of Southall et al. (2019)), the kurtosis is ~20. When the recorder was in the beam of the echosounder, kurtosis approached 20 for LF weighting but was greater than 100 for HF weighting.

## Summary

Binary categorization of sound sources as "impulsive" or "non-impulsive" based on qualitative definitions may lead to misclassifying received sounds and inaccurately assessing potential impacts on animals. Physiological studies of terrestrial mammals find that differences in hearing loss resulting from exposure to differently structured sounds of the same total energy is predicted by the distribution of received levels and quantified by the signal's kurtosis. The impulsive and non-impulsive thresholds for predicting hearing loss in marine mammals (e.g., NMFS 2018) establish the sound types that may affect hearing as end points but not the transition between the types. Using an adjustment factor based on kurtosis allows for a smooth transition between the end points (Fig. 2), recasts the focus to the received sounds instead of the sources, and accounts for the hearing sensitivity of animals (hearing groups).

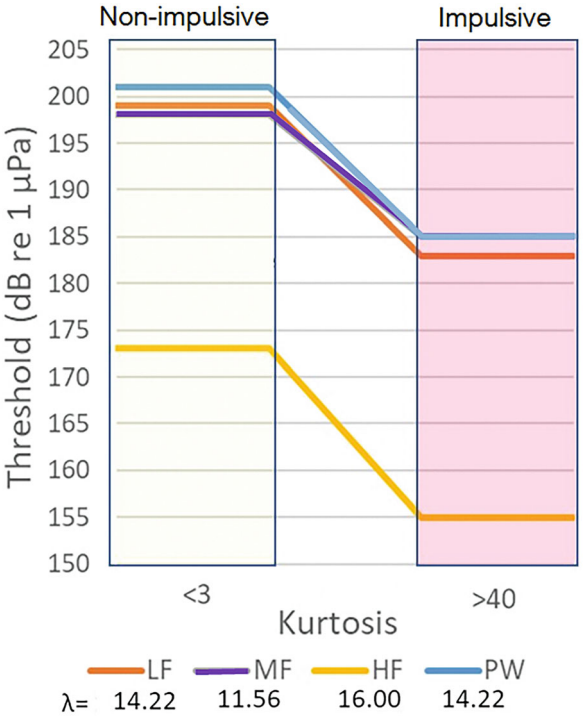



**Fig. 2** The NMFS (2018) sound exposure level (SEL) thresholds for predicting marine mammal hearing loss using a kurtosis adjustment for transitioning between impulsive and non-impulsive received sounds. Hearing groups: *LF* low-frequency cetacean, *MF* mid-frequency cetacean, *HF* high-frequency cetacean, and *PW* pinniped in water

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# Impact of Mobile Offshore Drilling Units on Odontocetes

82

S. Bruce Martin, Katie Kowarski, and Julien Delarue

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**Abstract**

Anthropogenic sounds can negatively impact marine mammals, limiting their ability to effectively forage or communicate, displacing them from a portion of their range, or causing physical harm. To date, potential impacts of sounds associated with mobile offshore drilling units (MODUs) on cetaceans have not been investigated. Acoustic recordings were collected at 1–2 km and 20–40 km from three MODUs to investigate variation in the acoustic occurrence of odontocetes with distance from the MODUs.

The *West Aquarius* MODU, *West Hercules* MODU, and *Stena Forth* MODU acoustic measurements lasted for at least 2 months each and included at least 1 min in 20 with high enough sampling rates to record the echolocation clicks of delphinids, beaked whales, and sperm whales. Recordings in the general area of the operations from previous years were also available. A comparison of the odontocete acoustic presence near the operations to the acoustic presence at longer distances showed a substantial difference in acoustic occurrence for dolphins, pilot whales, and beaked whales that could not be attributed to masking of detections by sounds from the operations.

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**Keywords**

MODU · Odontocete · Sonar · Avoidance · Ultrashort baseline pinger

## Introduction

Mobile offshore drilling units (MODUs) are large platforms used to drill oil and gas wells in the ocean for exploration and extraction. For deep water operations, there are two common forms – semisubmersible rigs and drill ships. In waters greater than ~200 m the MODUs typically hold their position over the wells using dynamic positioning (DP) thrusters. During environmental assessments of drilling operations, the underwater sound from the drill rigs and their support vessels are considered as possible threats to marine life, along with the sounds from vertical seismic profiling (see Further Readings).

Drilling operations typically take 2–4 months at each well. Over the past several years off Eastern Canada, JASCO Applied Sciences (JASCO) has made acoustic measurements of deep-water MODU drilling operations by the semisubmersibles *West Aquarius* and *West Hercules* as well as drill ships *Stena Forth* and *Stena IceMax*. The acoustic monitoring was required by the Canadian regulators during the drilling of the first well for each project (see Further Readings). To verify predictions for sound levels from the drilling programs, measurements were made close to (1–2 km) and relatively far from (20–40 km) the MODUs, the latter being where the sound pressure level was expected to drop below 120 dB re 1 $\mu Pa^2$.

In all cases it was noted that ultrashort baseline (USBL) beacons signals at 25–27 kHz were a prominent component of the acoustic footprint of the MODU operations. It is understood that the beacons are used to help the dynamic positioning systems keep the drill rigs positioned above the well head. Comparable sounds in the USBL frequency band are sonar and echosounders which have been associated with avoidance, or at least a cessation in echolocation, by beaked and sperm whales (Cholewiak et al. 2017; Stanistreet et al. 2022).

The purpose of this analysis was to compare the acoustic presence of odontocetes near the MODUs with their acoustic presence far from the MODUs and, when possible, comparing occurrence before and during MODU presence. The goal was to determine if there was a measurable effect of MODUs, similar to that reported from sonar and echosounder sounds.

## Methods

### Measurements

For drilling projects undertaken by BP Canada Energy Group ULC (Project 1), ExxonMobil Canada Ltd (Project 2), and the Chinese National Offshore Oil Corporation (Project 3), JASCO collected acoustic recordings at 1–2 km and 20–40 km

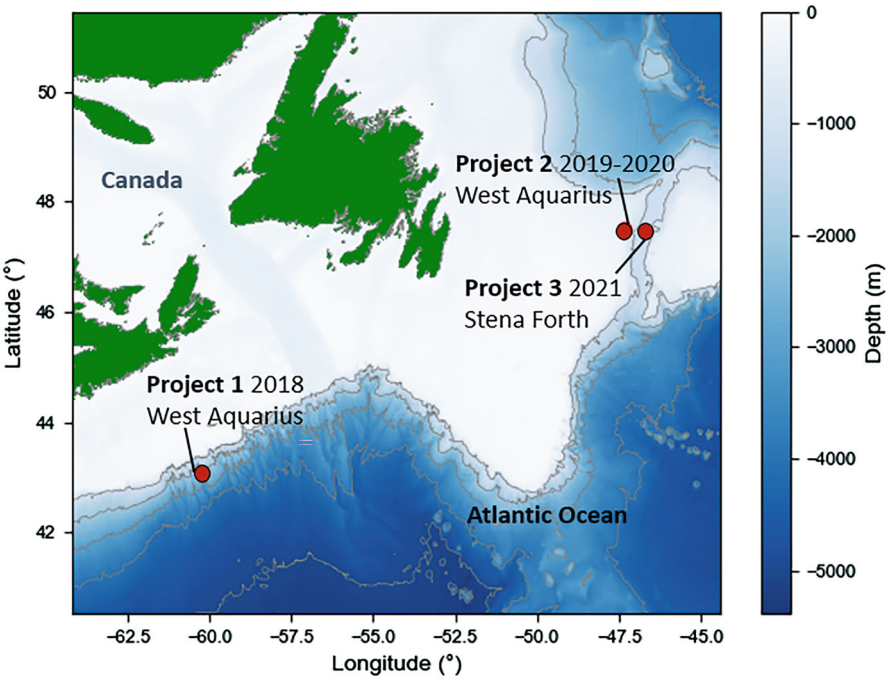



**Fig. 1** Project sites considered in the odontocete analysis

**Table 1** Overview of the recording programs: sampling rates, duty cycles, recording time frame, distance to the MODU, and water depths. Data were also collected at a lower sampling rate on Projects 1 and 2 but are not considered here

| | Sampling Rate (duty cycle) | Deployment period | Distance from MODU (km) | Water depth (m) |
|---|---|---|---|---|
| **Project 1** | 375 kHz (2 of every 10 min) | April–Aug 2018 | 2 | 2384 |
| | | | 20 | 2060 |
| **Project 2** | 512 kHz (1 of every 20 min) | Sept 2019–May 2020 | 2 | 300 |
| | | | 33 | 875 |
| **Project 3** | 128 kHz (continuous) | April–July 2021 | 1 | 1157 |
| | | | 40 | 666 |

from MODUs to investigate variation in MODU sounds and cetacean acoustic occurrence with distance from the MODUs (Fig. 1 and Table 1; see Further Readings for additional information on the projects).

All recordings were made using autonomous multichannel acoustic recorders (AMARs; JASCO, Dartmouth, NS, Canada) and M36 hydrophones (Geospectrum Technologies Inc, Dartmouth, NS, Canada, sensitivity approximately −164 dBV/Pa). The hydrophones were calibrated on deployment and retrieval using G.R.A.S. pistonphones. The hydrophones were located near the seabed using short float-on-a-rope-style moorings.

## Data Analysis

The basic analysis applied to all the data followed the Welch (1967) method: time series audio data were converted to power spectral densities using the fast Fourier transform (FFT) with a 50% overlap and a Hann window. A frequency-dependent calibration curve for the hydrophone and recorder was applied to the spectra. The spectra were then summed to generate decidecade sound pressure levels as needed for the analysis of sound pressure levels and sound exposure levels (Heaney et al. 2017). Sound exposure levels weighted for the hearing sensitivity of marine mammals from duty cycled recorders were computed using the methods from Martin et al. (2019a).

The acoustic presence of marine mammals was determined by (1) applying a Suite of automated marine mammal detectors; (2) based on the automated detector results, selecting a subset of the full dataset to be manually analyzed for marine mammal presence; and (3) comparing automated and manual results to assess and optimize automated detector performance. An automated contour detector was employed to identify periods of tonal whistles in the data (Martin et al. 2014). The automated contour detector operates on a per-frequency basis to detect signals that stand out from the background, then joins adjacent time frequency cells to create a contour that is compared to known species-dependent contour shapes. The automated contour detector identified two types of whistles. Whistles above 5 kHz were categorized as dolphins and were believed to be produced by one or more species of small dolphins (e.g., *Lagenorhynchus acutus* and *Lagenorhynchus albirostris*). Whistles with a fundamental frequency below 5 kHz were categorized as pilot whales (*Globicephala melas*) or killer whales (*Orcinus orca*).

Odontocete clicks were identified using an automated time series detector classifier (Martin et al. 2019b). This detector employs a short time span (0.333 millisecond) long Teager-Kaiser detector to identify possible echolocation clicks (Kandia and Stylianou 2006). The possible clicks are analyzed to measure three time series properties: the number of zero crossings in the waveform (i.e., click duration); the time between zero crossings (i.e., click center frequency); and whether the time between zero crossings is uniformly increasing (characteristic of beaked whale clicks). By computing the Mahalanobis distance between the measured and known properties of each species, the click was classified to a species.

A subset of acoustic files was manually validated for odontocete acoustic occurrence to assess and optimize automated detector performance for each AMAR deployment (Kowarski et al. 2021).

When automated detectors performed poorly, defined here as a Precision less than 0.75, the automated results were not included in the present overview of potential MODU impacts. The occurrence results were summarized in terms of the distributions of the percent of days in which the species were detected at the close and far recording stations.

## Results

A 30-s spectrogram of a typical period of USBL communications is shown in Fig. 2. The power spectral density distributions at each of the recording sites is shown in Fig. 3. The −3 dB bandwidth of the signal was 24–27 kHz, with sound extending out to 22–29.5 kHz. Figure 2 shows that there are periods of multiple seconds between the communications where detection of odontocete clicks would not be masked by the USBL pulses. The background sound levels are almost identical to those at the distant recorders when the communications are not occurring. The detection of odontocete whistles in the 5–15 kHz band is expected to only be modestly affected by the MODUs based on the slight increase in the power spectral densities in that band closer to the MODUs.

The sound levels shown in Fig. 3, particularly for Project 2, were substantially elevated near the MODUs. It was hypothesized that the Project 2 levels were higher than for Projects 1 and 3 because Project 2 occurred in winter, or perhaps due to the

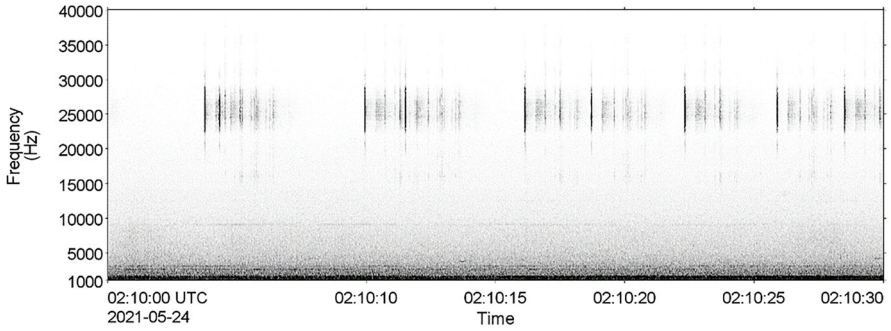



**Fig. 2** Thirty seconds of data from Project 3 in May 2021 showing typical USBL pinger communications. Note that there are periods of many seconds without USBL activity

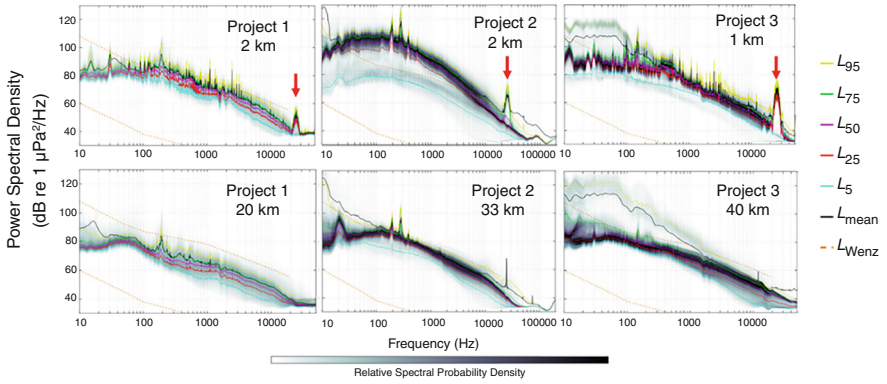



**Fig. 3** Power spectral densities of the soundscape recorded at each of the sites. Top row: the recorders close to the MODUs; bottom row: the recorders at 20–40 km from the MODUs

shallower water depths (Table 1). The levels at Project 2 were high enough that they exceeded the low-frequency cetacean temporary threshold shift levels on most days (Fig. 4, assuming the animals did not leave the area over that 24-hour period). The low-frequency sound levels increased at the end of Project 3 due to the commencement of a nearby seismic survey. For all three projects, the auditory frequency weighted daily SELs hovered near the high-frequency cetacean temporary threshold shift (TTS) threshold for auditory impairment, but were significantly below the mid-frequency cetacean TTS threshold (Fig. 4, using the thresholds defined in NMFS (2018) and assuming animals did not leave/avoid the area around the MODUs).

Analysis of the data revealed the acoustic occurrence of many odontocetes including dolphin whistles (whistles from unknown small dolphin species), delphinid clicks (clicks from small dolphins, killer whales, and pilot whales), pilot and killer whale whistles (these can be difficult to differentiate and there may be some errors in their identification), sperm whales (usual clicks; *Physeter macrocephalus*), narrow-band high-frequency (NBHF) clicks of porpoise (*Phoecena phoecena*) or Kogia (*Kogia breviceps*, *Kogia sima*), northern bottlenose whales (*Hyperoodon ampullatus*), True's (*Mesoplodon mirus*) or Gervais' (*Mesoplodon europaeus*) beaked whales (these signals could not be confidently differentiated to the species level, but are likely True's based on location), and Cuvier's beaked whales (*Ziphius cavirostris*). Automated detector performance varied across datasets, with some instances where manually validated results were considered a more reliable indication of species presence, even though only a subset of data was analyzed.

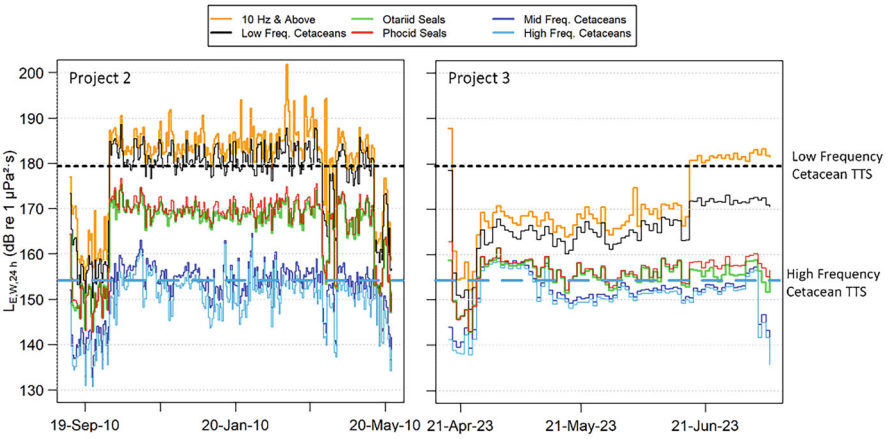



**Fig. 4** Daily auditory frequency weighted sound exposure levels for Projects 2 and 3 (see NMFS 2018 for details on the frequency weightings). The mid-frequency cetacean TTS threshold is 1 dB below the low-frequency cetacean threshold. The low-frequency group includes the large baleen whales. The mid-frequency group includes most dolphins, the beaked whales, sperm, and killer whales. The high-frequency group includes porpoise, kogia species, and some dolphins

The range of automated detector performance metrics was considered when interpreting results. For example, Fig. 5 provides evidence that delphinid clicks were more common at 40 km (present on 95% of days) than 1 km (present on 42% of days) from the MODU. It must be considered that at 40 km, the *per file* Recall was 10% higher than at 1 km, meaning that the detector missed 10% more true calls there. If one assumes that this performance differential transfers to *daily* Recall (which is certainly an overestimate) and consequently increase the occurrence at 1 km to make it artificially more comparable to 40 km, delphinid clicks become present at 1 km on 52% of days. With this artificial increase in occurrence, the acoustic occurrence of delphinid clicks remain substantially higher at the 40 km site. This exercise of considering whether Recall of automated detectors was skewing the interpretation of results was completed for each species and the overall trends (where the signal was more common) remained unaltered.

There was evidence that acoustic signals were more common further from the MODU for dolphin whistles, delphinid clicks, True's/Gervais' beaked whales, and Cuvier's beaked whales (Figs. 6 and 7). Trends in the occurrence of dolphin whistles and delphinid clicks are arguably most convincing given the consistency across all three projects (Fig. 6). No evidence of masking was found to explain these trends. In addition, there was some evidence of more frequent occurrence further from the MODU for both killer and pilot whale whistles (Fig. 6). However, the strength of these observations was weakened as they were often based on manual results and there was one instance where detections were more common close to the MODU (Fig. 6; pilot whale, Project 2).

Some odontocete acoustic signals either did not show trends related to distance from MODU or their acoustic occurrence was too low to reasonably assess impacts.

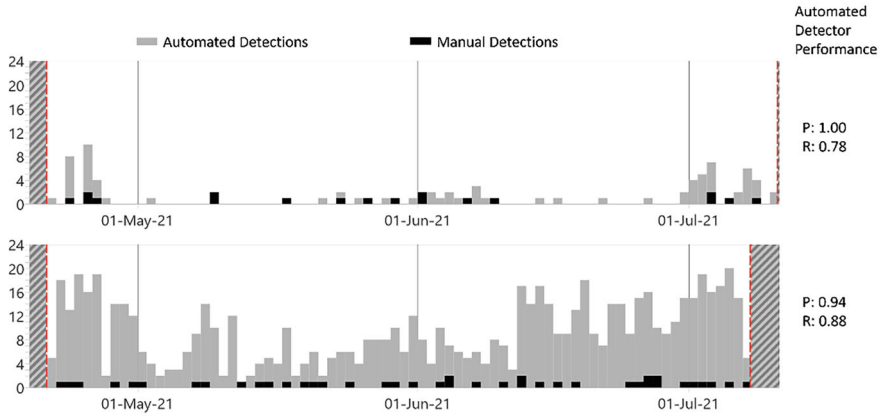



**Fig. 5** Number of hours in each day with delphinid clicks recorded at (top) 1 km and (bottom) 40 km from April to July 2021 (Project 3) with automated detector performance metrics per 10 min included along right side. The grey areas indicate hours of darkness from sunset to sunrise (Ocean Time Series Group 2009). Hashed areas indicate when there was no acoustic data and red dashed lines indicate the start and end of recordings

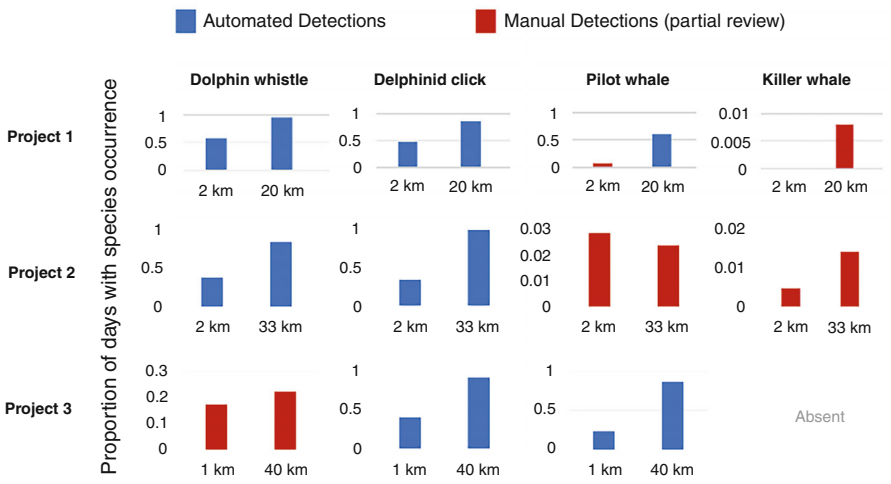



**Fig. 6** Bar plots showing the proportion of recording days that delphinids were manually detected (red) or automatically detected (blue) at recording station 1–2 km and 20–40 km from the MODU. Manual results cannot be directly compared to automated results given that only a proportion of data was reviewed while all data had automated detectors applied. Note that *y*-axis are different among species to show patterns between stations and years

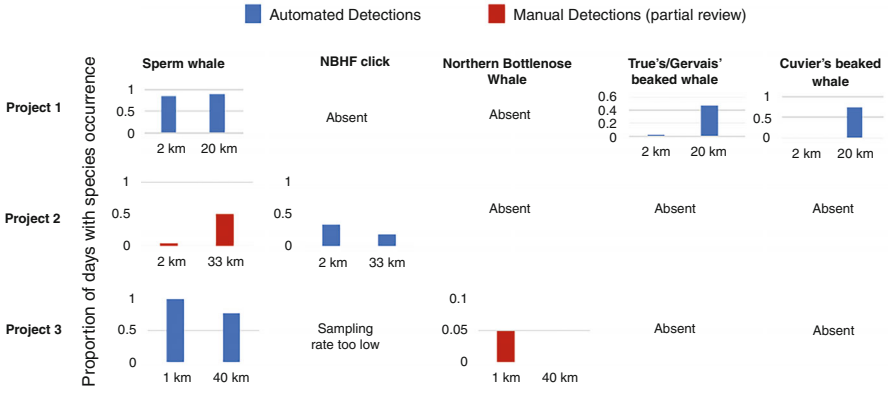



**Fig. 7** Bar plots showing the proportion of recording days that sperm whales, NBHF, and beaked whales were manually detected (red) or automatically detected (blue) at recording station 1–2 km and 20–40 km from the MODU. Manual results cannot be directly compared to automated results given that only a proportion of data was reviewed while all data had automated detectors applied. Note that *y*-axis are different among species to show patterns between stations and years

Sperm whales did not show consistent evidence of days with detections trending with distance from MODU (Fig. 7). There was low occurrence of both NBHF and northern bottlenose whales in the acoustic data, limiting the ability to assess trends, though the data available suggests no trend regarding distance to MODU (Fig. 7).

## Conclusion

The results indicate that MODUs impact nearby marine life. Evidence was particularly strong for impacts to delphinids, followed by True's/Gervais' and Cuvier's beaked whales. These species either reduced their vocal activity in the presence of the MODU, remaining in the area and potentially experiencing auditory impairment (if the remained close enough to the MODU), or the animals avoided the area and were displaced from habitat.

Impacts must be better understood and quantified for effective species management and future project planning. We recommend that studies be conducted to quantify the scale of avoidance of USBL pingers by different species of odontocetes. Questions of interest include which species are affected, how far from a USBL pinger is avoided, and the time scale after cessation of the pinger activity needed for the species to return to baseline activity levels. As a minimum, it is recommended that future PAM around MODUs include data from before, during, and after operations, ideally with pre-operation's information from more than 1 year. A structured autonomous exposure experiment could also be considered where a USBL source is deployed at the seabed, with autonomous recorders at ranges of 1, 3, 10, and 30 km. The USBL pinger would be programmed to cycle on and off for a day at a time, then 2 days, 4 days, 8 days, and 16 days to examine return rates to baseline activity. Controlled exposure experiments could also be considered with tagged animals and/or acoustic directional tracking of individuals and groups that would provide valuable insight into the species behavioral responses. Information could be collected and made available to researchers by marine mammal observers stationed onboard MODUs. For example, they can estimate the distance sighted animals are from the MODU, the animal's behavior, and report how long they remain in the area. This would allow an estimate of whether auditory impairment is reached.

MODU operators should consider whether the use of USBL can be reduced or mitigated in any way, such that levels fall below risk to auditory impairment of marine mammals or operations occur when and where fewer species are expected to be impacted.

## Further Readings

For information on the Canadian Environmental Assessments related to the drilling activities, and acoustic modeling in particular, see e.g., Appendix E of the Environmental Impact Statement for Project 3 at Environmental Impact Statement (iaac-aeic.gc.ca)).

The monitoring requirements for Project 3, which is similar for all projects, can be found at Sect. 3.12.3 of CNOOC_DS_Final (iaac-aeic.gc.ca)).

For further information on Project 1, see https://www.bp.com/content/dam/bp/country-sites/en-ca/canada/home/documents/nova-scotia/acoustic-monitoring-during-scotian-basin-exploration-project-summer-2018.pdf

For further information on Project 2, see Exxon Mobil Flemish Pass Exploratory Drilling Operations (squarespace.com).

For further information on Project 3, see https://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=&ved=2ahUKEwib-P-Co9H7AhVQkIkEHcdzDBYQFnoECBMQAQ&url=https%3A%2F%2Fcnoocinternational.com%2F-%2Fmedia%2Fcnooc-images-and-files%2Foperations%2Famericas%2Fcanada%2Fatlantic-canada%2Fca-exp-ac-dwd-rg-00028-rp-01%2D%2D-flemish-pass-accoustic-monitoring-report_rev-2.ashx%3Fla%3Den%26hash%3D5CE605F74BF4B87EA68CEBFE548644D7&usg=AOvVaw2aWWwGPsQJZhvXJKp3NDwp

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# Effects of Shipping Noise on Migrating Ringed Seals

83

Muhammad Saladin Prawirasasra, Mart Jüssi, Mirko Mustonen, and Aleksander Klauson

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**Abstract**

The emergence of ambient sound due to ship noise has adverse effects on pinnipeds. This chapter is focused on analyzing potential disturbance behavior of inhabitant animals when exposed to sound from ferry vessels. Three adult free-ranging ringed seals were tagged with GPS tracking devices to reveal their transiting tracks in the Gulf of Riga. Analysis of the telemetry data demonstrates their regular routes are between the Gulf of Riga and Väinameri to forage and rest. During transiting, the seals are likely being exposed to underwater anthropogenic

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noise emitted from the ferry vessels. Impact of loud continuous anthropogenic noise and very close presence of fast-moving vessels on the seals are possibly predicted from the disruption patterns on regular dive profiles. Disturbed behaviors can be noticed when the seals dive deeper or perform multiple short dives and surfacing coincide to close encounters with the ferries. Received levels (RLs) that cause reactions on the seals are estimated from modeled underwater sound propagation.

**Keywords**

Underwater anthropogenic noise · Modeled underwater sound propagation · Disturbed behaviors · Dive profiles · Ringed seals

## Introduction

One of the inhabitant marine mammals that can be found in Estonian waters is ringed seals (*Phoca hispida*). The seasonal presence of ice makes this region a safe place for them to live specifically to provide protection for their pups from predators (Sundqvist et al. 2012). The tendency of the world's increasing temperatures to reduce the area of ice cover is considered as one factor that is decreasing the chance of pups to survive, classifying this species under the category of vulnerable and protected (Helcom 2013). The vast increase of ship traffic and offshore constructions are also considered as negative factors for the seals in the Baltic Sea (Syrjälä et al. 2020; Tougaard et al. 2020).

The low-frequency radiated noise from anthropogenic activities is known to have adverse effects for marine biota including pinnipeds (Erbe et al. 2019). Previous studies have demonstrated that either free-ranging seals or seals in captivity communicate through calls (Mizuguchi et al. 2016; Stirling 1973). The simultaneous presence of anthropogenic noise and their calls could potentially reduce their communication space and experience auditory masking (Clark et al. 2009; Erbe et al. 2016). Moreover, research has shown unusual reactions of phocids to strong sounds. For example, experiments with harbor seals (*Phoca vitulina*) showed an increase in the median distance from the location of the sound source that simulated broadband wind generators (Koschinski et al. 2003). Both gray seals (*Halichoerus grypus*) and harbor seals show aversive behavior when stimulated with various sound signals including 0.5 kHz sine. The obvious response was they were moving away from the sound source when exposed to sound pressure levels of 144 dB (Götz and Janik 2010). It was also noted that masking thresholds were revealed on two ringed seals in captivity when they were exposed to tonal stimuli of 102 dB. The use of tagging devices together with sound level measurements in gray and harbor seals revealed their uncommon reactions when they were having close encounters with ships. Gray seals terminated their ascending dive prior to completion of the surfacing as the observed disturbance reactions. On the other hand, harbor seals reacted differently by diving deeper and staying in deeper places before the sound pressure reduced. These phocids' reactions and their behavioral hearing threshold (BT)

(ISO 2017) can be found in Mikkelsen et al. (2019a, b). Yet, the predicted levels that cause onset disturbance behaviors of seals are still lacking.

The objective is to find the impact of low-frequency continuous anthropogenic noise on the ringed seal. The analysis focuses on diving profiles of tagged, transiting ringed seals while crossing a noisy ferry route. The disturbance behaviors are assessed by comparing diving profiles that are situated in two different acoustical situations. Avoidance reactions of the seals' movements to the strong acoustical sources are also further analyzed. Estimates were made of the disturbance levels that resulted in interrupted diving profiles using sound propagation modeling.

## Methodology

### Study Location

The study took place in the narrow strait, Moonsund, Estonia, which is situated between Muhu Island and the mainland as depicted in Fig. 1. The location is very shallow water considering the maximum depth within the vicinity does not exceed 20 m. Also, it is usually freezing during winter time, which provides a good protection for the ringed seals' pups from predation. In addition, the acoustical monitoring using passive recorders revealed the presence of the gray seals in the region (Prawirasasra et al. 2019, 2021).

Two shipping lanes cross the monitoring location. One has moderate ship traffic levels through Moonsund that connects the Baltic Proper in the north and the Gulf of Riga in the south. Another lane is a 7-km-long busy ferry route that connects Virtsu and Kuivastu. The latter can be categorized as busy traffic considering more than 24–30 daily crossings occur from each direction with durations between crossings varying between 0.5 and 5 h depending on the hours. During the study, there were three operating ferries of M/S Regula, Viire, and Scania sailing in these ferry routes. According to the schedule, most of the ferries cross the strait during the day and two ferries are traveling from the opposite harbors at the same time with the average sailing speed being around 9–10 knots.

### Tagging Devices and Seals' Handling

A device tag that was used in the observation was a GPS Phone tag made by Sea Mammal Research Unit (SMRU), St. Andrews, UK. The telemetry tag collects geolocations onboard at 20-min increments with Fastloc® GPS. Ambient seawater pressure sensors measured the dive depth with 0.1 m resolution. Therefore, the threshold of dive detection was set to 1.5 m to separate between diving and time spent at surface. Dive profiles were constructed from nine recorded depth positions during the diving, which were saved from the measurement at the end of each dive (McConnell et al. 2004).

The tagging devices were attached to three free-ranging adult ringed seals who were previously caught with custom designed tangle nets. Glue was used to attach

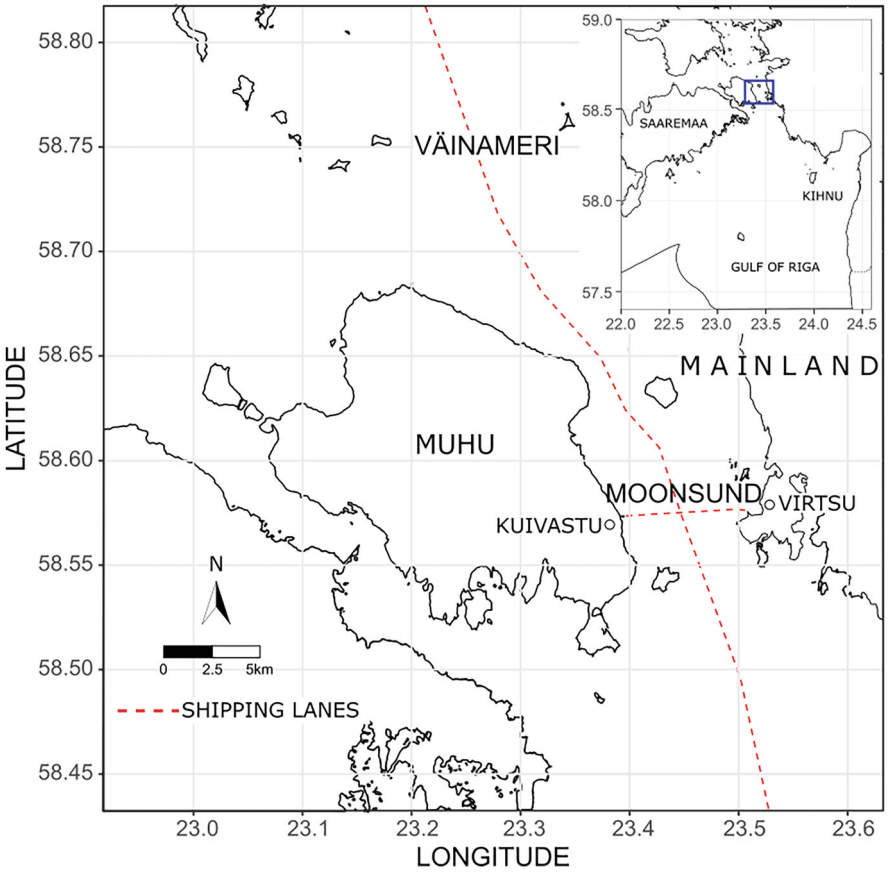



**Fig. 1** A map of the observation area, Estonia. The area is highlighted by a blue rectangle on a larger map in the top right corner inset figure. (Figure is adapted from Prawirasasra et al. (2022))

the device in the upper neck area using quick setting epoxy resin. The captured seals were released in the shortest possible time after the monitoring devices were DEPLOYED. Location of the captures was in the ringed seals' haul out in Väinameri in May 2009. The adult ringed seals were expected to have had prior experience to being exposed to the ship noise. Thus, their naive reaction to a new unnatural event was unlikely to happen.

## Estimation of Ferry's Source Level (SL) and Underwater Sound Propagation Modeling

The underwater sound propagations radiated from the moving sound sources were modeled using Quonops Online Service (Guelton et al. 2013). The acoustical source

of continuous anthropogenic noise was predicted by the RANDI model (Breeding et al. 1994). The modeled SL was verified from measurement of sound radiation of M/S Regula in June 2018. Radiated noise from the ferry was recorded at the closest point of approach (CPA) of 800 m to the hydrophone. The hydrophone was submerged underwater at 15 m depth. However, an accurate measurement of a ship's source spectrum can be performed when requirements are fulfilled such as deep location of the measurement (ISO 2019). Thus, the SL of the ferry can be estimated by predicting the transmission loss (TL) in an isovelocity sound speed profile using the following equation:

$$\mathrm{TL} = 20\ \log 10\ (H/1\ m) + 10\ \log 10\ [(R - H)/H] \tag{1}$$

With $R$ in meter is horizontal radius from ship to the hydrophone and $H$ is depth in meter of the water column. By substituting $R$ and $H$ with the measurement conditions, SL of the M/S Regula can be estimated after the prediction of TL value. The comparison analysis between modeled and estimated SL resulted in a fairly good agreement between them. Therefore, the RANDI model of M/S Regula could be used as a SL for acoustical sources in the underwater sound propagation modeling. In addition, the modeled SL of M/S Regula was also suitable for other operating ferries considering similar physical dimensions and sailing speed.

Underwater sound propagation in shallow water is different from deep water due to higher limiting or cutoff frequencies. Previous observation of RL spectra of ship noise in the vicinity demonstrated the existence of strong attenuation at lower frequencies (Jensen et al. 2011; Prawirasasra et al. 2021). It shifts the dominant frequency of shipping into higher frequency compared to suggested frequencies by Marine Strategy Framework Directive (MSFD) (Decision 2017). A frequency of 500 Hz one-third octave band (TOB) was used for underwater sound propagation modeling instead.

Snapshots of Quonops noise map were generated to predict the radiated ambient noise levels at particular positions of the acoustical sources. Natural ambient noise was approximated by selecting a point at a large distance from the ship with less acoustical impact from noise radiation from the acoustical sources. Excessive sound pressure level (SPL) was determined by comparing RLs at the position of the seals and the natural ambient noise levels.

## Dive Profiles

The telemetry data were analyzed by subsetting based on geographical coordinates extending from 1.5 km to north and south of the ferry route. The dive profiles were constructed from varying recorded depth positions in time. Meanwhile, the use of automatic identification system (AIS) of ship location reports and geographical coordinates of tracking data from each tag was possible to calculate the horizontal

distances between ships and the seals. The "regular" patterns of the dive profiles were referred to undisturbed behaviors of the seals during the crossings.

Irregular dive profiles were determined when compared to previously defined "regular" dive profiles. The assessment of the mismatch patterns was done by visual comparison. Irregularity of dive profiles in the vicinity of the ship might be an indication of disturbed behavior of the seals. The possible reactions can be variables in durations of dive/surfacing or dive depths. In Mikkelsen et al. (2019a), the regular dive profiles were interrupted in the presence of loud noise. The noted disruptions can be identified when the gray seals dived deeper and remained underwater. Another possible reaction were drastic changes in either diving speeds or dive directions. However, the current tracking data consists only of the averaged speed values between two sequential surfacing; therefore sudden responses are less probable to be noted.

## Result

### Analysis of Tracking Data

The monitoring periods of tagged seals started from May 2009 and ended in October 2009 without covering their breeding period in February and March. Each tag was identified by "A," "B," and "C." Analysis on tracking data showed that journeys of the tagged seals ended either in the Väinameri or the Gulf of Riga through Moonsund. Those destination locations were most likely associated with their resting and foraging activities (Halkka and Tolvanen 2017). The trajectories of their movements during monitoring periods are shown in Fig. 2.

Altogether, the three ringed seals crossed the ferry route at least 36 times with most of the travels being performed during the day. Time series analysis of their tracking data revealed that the tagged seals spent most of their time in the Gulf of Riga in the summer compared to autumn months. The transit corresponds to the Gulf of Riga (57.5°) and Väinameri (59°), which are represented in a time series of latitudes as depicted in Fig. 3.

### Regular Dive Patterns of Ringed Seal

A regular dive profile could be identified when the seals were separated more than 500 m away from the ships. The low RL of radiated ship noise seemingly did not disturb the crossing seals as their dive profiles remained regular. Notable features of regular dive profiles consisted of comparable durations between sequential diving and surfacing periods that continued for 3–5 min and 30 s, respectively (Kelly and Wartzok 1996). Another observed feature was dive depths. The transiting seals tended to dive close to the sea bottom while crossing (Crawford et al. 2019). The examples of undisturbed behaviors of the crossing seals are illustrated in Fig. 4.

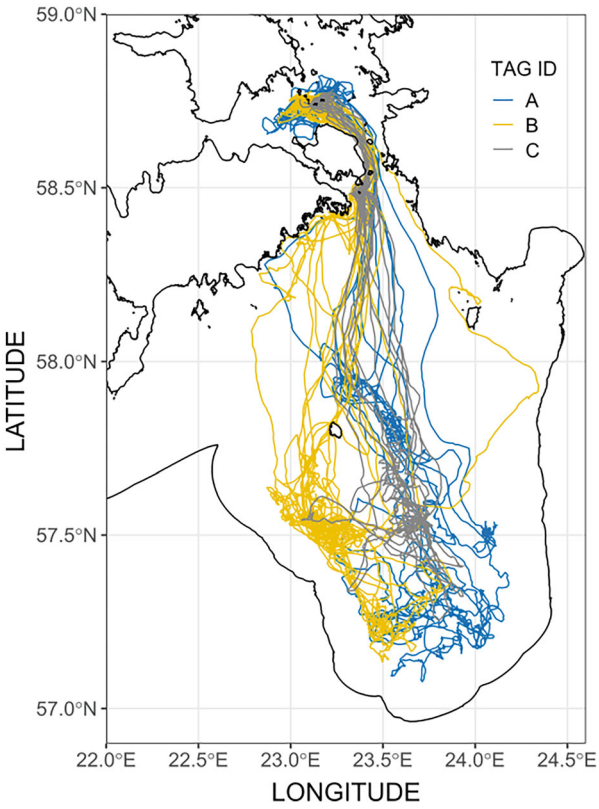



**Fig. 2** Trajectories of three tagged ringed seals in the whole period of monitoring. Blue, yellow, and gray color lines indicate "A," "B," and "C." (Figure is adapted from Prawirasasra et al. (2022))

## Changes in Dive Profiles

Despite having similar crossing time with the ferries, the collisions between them rarely occurred. Reconstructions of their movements showed that they only had three close encounters with distances less than 50 m. In the first case, seal "C" had a close encounter with the M/S Regula when traveling to the Gulf of Riga. Analysis on the dive profile showed there were no reactions of "C" to the approaching ferries as the regular dive profiles remained observed. On the other hand, two other cases that were experienced by "B" and "C" showed potential disturbance reactions of the tagged animals. Changes in dive profiles corresponding to varying distances from the ferries are illustrated in Fig. 5. "B" traveled to the north and was at least 800 m away from Viire preceding to having close encounters with the M/S Regula. In such distances, radiated noise from Viire did not affect the transiting seals. At the CPA of M/S Regula, "B" dived to the bottom of the sea to avoid the noise source, while previously "B" had a longer surfacing. Such reactions can be considered as a natural response in the presence of danger and it can be interpreted as an indication of

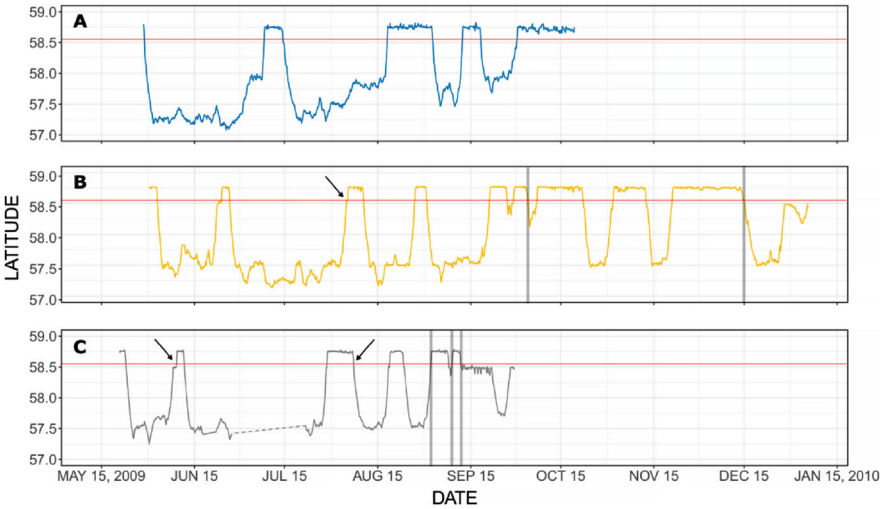



**Fig. 3** Plot of time versus latitude coordinates of transiting seals (**a**) "A," (**b**) "B," and (**c**) "C." The latitude coordinates of the ferry route are indicated by the horizontal red lines. Performed journey during nights is signified by the vertical gray bars. Arrows indicated the close encounters of the seals with the ferries. Dashed line shows the missing transmitted data. (Figure is adapted from Prawirasasra et al. (2022))

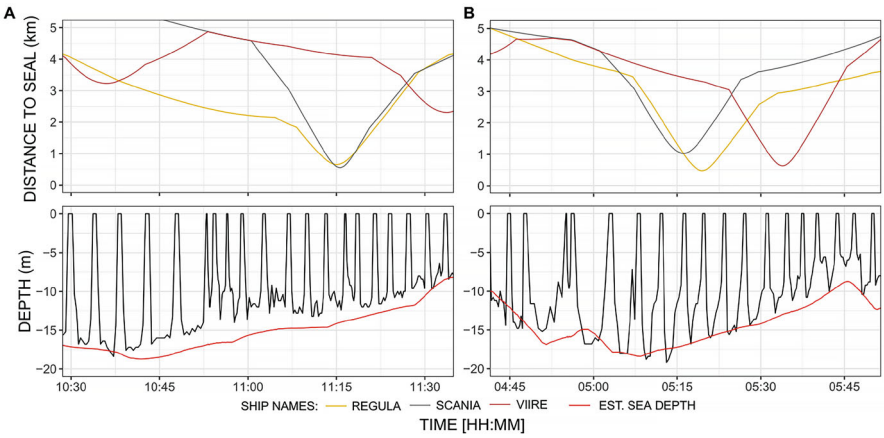



**Fig. 4** Identified regular dive profiles of (**a**) "A" and (**b**) "C" crossing seals. Undisturbed behavior remains observed when the ferries were at least 500 m away from the seals. The bathymetry is shown by the red line

behavioral reaction (Fig. 5a). More prominent reactions were shown in another case as depicted in Fig. 5b. "C" was situated at a distance of 35 m to M/S Regula and started to do multiple short dives and surfacing in very short durations. The tagged seals continued underwater for 32–48 s and went to the surface for 4–12 s. Thus,

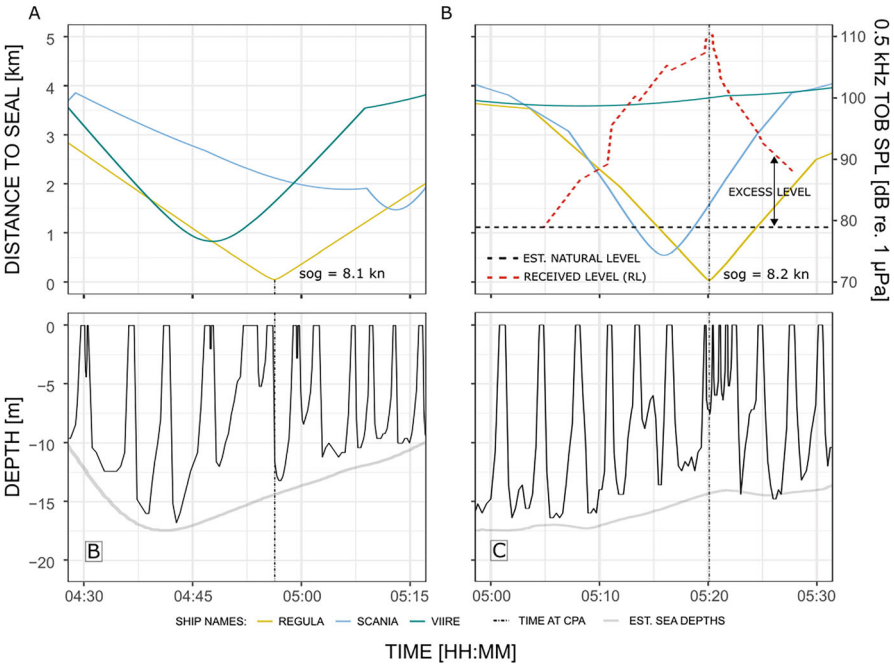



**Fig. 5** Observed potential disturbance reactions of the seals to approaching ferries. No indications of disturbed behaviors when the seals are at least 500 m away from the ferry. The interrupted dive profiles can be noted in the form of diving deeper in (**a**) and more prominent reactions in multiple short dives and surfacing are shown in (**b**). The latter happens when the tagged seal is being exposed to a sound excess of 110 dB at 0.5 kHz TOB. M/S Regula's close encounters to the tagged seals along with their reactions within 50 m to "B" and 35 m to "C" are depicted at (**a**) and (**b**). Upper figures indicate the distances of the ferries to the seals and dive profiles of the crossing seals are shown in lower figures. The varying modeled RLs at the "C" positions due to the traveling ferries are indicated in red-dashed line at (**b**). The estimated natural sound is shown in black-dashed line. The sound excess is defined as the relative levels between RLs and natural sound. The yellow, blue, and dark green lines show the distances of M/S Regula, Viire, and Scania ferries to the tagged seals. The sailing speeds of M/S Regula at the CPAs were 8 knots. Time of CPAs are shown by vertical black dash-dotted lines. (Figure is adapted from Prawirasasra et al. (2022))

these reactions can be suggested as disturbance behaviors of the seals when being exposed to loud ship traffic noise and having near collision with the ships. While in the previous encounters with the Scania at larger distances, no reactions were shown.

Meanwhile, the varying RLs at the positions of the tagged ringed seals corresponding to changing relative distances to the ferries can be approximated. The BT can be estimated at the moment when the irregularities of the dive profiles were starting to be noticed. In this case, the reactions of the ringed seals were observed after the seals were stimulated by 110 dB at 0.5 kHz TOB or when the sound emerged 30 dB above the natural sound. The approximation of RLs and sound excess are illustrated at the upper right figure in Fig. 5b.

## Discussion

The tracking analysis has shown that the tagged ringed seals crossed the Moonsund 36 times with only 3 close encounters with the ferries. However, the irregularities in dive profiles can be observed only in two cases. The interruptions might be associated with their reactions to the ship traffic noise. In the first case, the reactions of the tagged seals dived deeper in the presence of loud anthropogenic noise. The seals continued underwater until the sound pressure returned to acceptable levels. In the second case, the occurrence of more pronounced irregular dive patterns happened. The seals traveled by performing multiple short dives and surfacing that occurred in very short durations. The surfacing reactions might be noise mitigation to reduce the intensity when being exposed to loud anthropogenic noise (Sills et al. 2015).

Other observed variables, traveling direction, and swim speed were analyzed from the two close encounter cases with disturbances. However, no indications of flight or significant avoidance reactions were indicated by the seals toward the very close, fast-approaching vessels. Even more, the seals kept swimming at constant paces. Possibly the objects have been well habituated in the presence of the ship traffic along with low resolution of recorded data where only average speed can be predicted without showing sudden reactions.

The assessment of BT that resulted at 110 dB in 0.5 kHz TOB can be compared to previous investigation with phocids (Mikkelsen et al. 2019a). This study demonstrated that the reaction of the gray seal can be noted when being exposed to 92 dB in 0.5 kHz TOB or 113 dB in broadband of continuous anthropogenic noise. In another study with harbor seals as documented in Kastelein et al. (2006), the reaction of the seals can be notified at level of 107 dB at 24 kHz of sound stimuli. The flight reactions of the seals seem not possible to happen considering available anthropogenic sound sources in the region as the estimated BT was considerably lower than referenced in Götz and Janik (2010). Moreover, the observed reactions matched with one of listed behavioral responses in Southall et al. (2021), and it can be categorized as low reaction (1) on the severity scale.

## Conclusions

Telemetry tags have been used to monitor the transiting route of three ringed seals. The tracking data demonstrated the regular transiting routes of the seals between Väinameri and the Gulf of Riga corresponding to their rest and forage activities. The Moonsund can be considered as an important transiting area for the ringed seals as they crossed it twice in a month. The ferry traffic that connects Kuivastu and Virtsu and crosses the Moonsund can be considered as the main contributors of anthropogenic noise in the location.

There is no solid evidence of seals' flight reactions when they are being exposed to loud ship noise. The exposed seals keep their swim direction and speed even when receiving the loudest possible noise. The lack of notable changes might indicate the tagged seals have been habituated with the presence of ferry traffic noise. In addition,

the reactions possibly do not appear considering the available data. Only average speed can be estimated; thus no abruptness can be identified.

Underwater continuous noise emitted from ferries seemingly has a low impact on traveling seals. According to the spatial movement analysis from 3 tagged seals, exposure to louder noise occurs only in 2 out of 36 crossings resulting in identified possible disturbance behaviors. The crossing seal swam with several short dives and surfacing that happen in very short durations. It is observed when the seals receive 110 dB at 0.5 kHz or when the ambient noise exceeds 30 dB at 0.5 kHz above the natural level. The obtained results are quite similar with the cited literature even though the number of cases is very small. The complete report of this chapter has been published in Prawirasasra et al. (2022).

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# Section XI

# Noise Effects: Other Vertebrates

# Temporary Threshold Shift in Turtles

84

Lena Marie Mannes, Magnus Wahlberg, and
Jakob Christensen-Dalsgaard

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**Abstract**

Temporary threshold shifts (TTS) have been studied in a variety of aquatic organisms from fish to marine mammals, but never in sea turtles. Here, turtle TTS was measured in air, and the in-air to underwater sensitivity difference was used to estimate TTS in water in an aquatic turtle. Red-eared sliders (*Trachemys scripta elegans*) were exposed to broadband noise in air for 24 h, and auditory sensitivity was assessed by auditory evoked potentials from the brainstem before exposure, immediately after exposure and during the following recovery period. All animals showed a depression of sensitivity immediately after exposure and full recovery 3–5 h after exposure. Based on differences in hearing thresholds in air and water, the exposure would be equivalent to 111 dB $L_{eq}$ (re 1 μPa) for aquatic sound, resulting in a sound exposure level of 160 dB re 1 $\mu Pa^2 s$ from the 24 h exposure. It is likely that long-term underwater sound level exposure at such levels will cause underwater TTS in turtles.

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**Keywords**

Sound · Turtle · Auditory brainstem response · Temporary threshold shift · Noise exposure

## Introduction

Temporary threshold shift (TTS) is an intermittent reduction in auditory sensitivity caused by sound exposure that has been studied in a variety of terrestrial as well as aquatic organisms. TTS is probably caused by intense exposure of inner-ear processes, for example, in the hair cells, the hair cell synapse, or in the auditory nerve fibers (Ryan et al. 2016). Studies of TTS are important both for understanding the control of sensitivity of the ear and for environmental regulation, since the onset of TTS is a convenient estimate of the threshold for physiological effects of noise (Saunders and Dooling 2018). However, recent studies of TTS in some mammalian species caution that even though sensitivity (measured by audiogram thresholds) recovers after noise exposure, there might still be permanent physiological damage, leading to cochlear synaptopathy and loss of auditory nerve fibers (Liberman and Kujawa 2017). Thus, an understanding of the mechanisms underlying TTS is necessary in order to assess the potential environmental threat of intense anthropogenic sounds.

For turtles in general, and particularly for the seven marine species, there are no data on TTS thresholds. This is unfortunate, as sea turtles are endangered, raising concerns about the added impact of environmental noise (Martin et al. 2012). In fact, very little is known about hearing in sea turtles, and especially their underwater hearing (Willis 2016). In-air and underwater hearing thresholds have been measured in green sea turtles (*Chelonia mydas*, Ridgway et al. 1969; Piniak et al. 2016), loggerhead turtles (*Caretta caretta,* Bartol et al. 1999; Martin et al. 2012; Lavender et al. 2014), and leatherback turtles (*Dermochelys coriacea,* Piniak et al. 2016) using physiological (auditory evoked potentials) or behavioral assays. In addition, in-air thresholds have been obtained from Kemp's ridley (*Lepidochelys kempii,* Ketten and Bartol 2006). Sea turtles have a similar frequency range of in-air and underwater hearing between 50 Hz (the lowest frequency studied) and 1000 Hz and most sensitive frequencies around 300–500 Hz. This frequency range is similar to in-air hearing abilities in all turtles and tortoises measured so far (e.g., Wever 1978).

Sensitivity thresholds are difficult to compare due to different experimental setups and stimulation methods (e.g., sound in air and water, or direct vibration of the tympanum). As shown in Fig. 1, the lowest thresholds to underwater sound, based on auditory evoked responses, is reported at 80 dB re 1 μPa for *C. caretta* (Ketten and Bartol 2006), and 85 dB re 1 μPa in *C. mydas* (Piniak et al. 2012) and juvenile *D. coriacea* (Piniak et al. 2016). Audiograms measured by auditory evoked potentials usually show higher thresholds than behavioral audiograms, and in one case where auditory evoked response audiograms are compared to behavioral audiograms in one loggerhead turtle the shape of the audiograms is similar, but behavioral thresholds on average 8 dB lower (Martin et al. 2012). Even though differences in methodologies may preclude a detailed comparison with between studies, these data

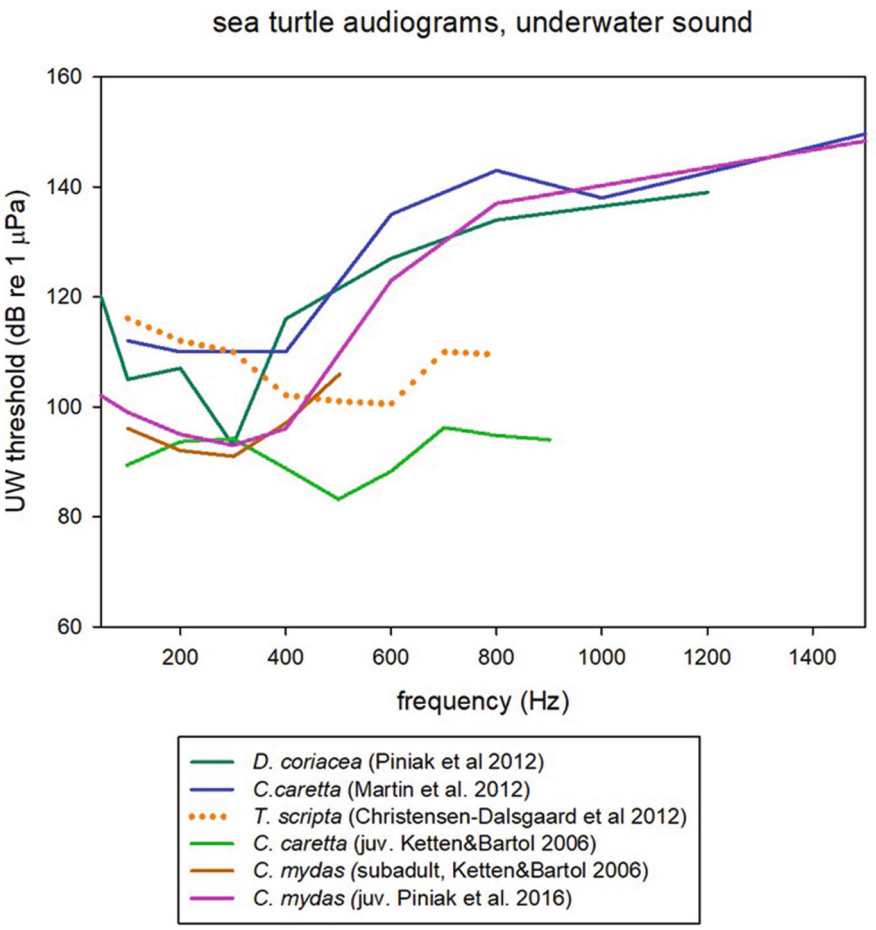



**Fig. 1** Underwater audiograms of sea turtles. Sensitivity to underwater sounds of different species of sea turtles, based on data from Piniak et al. (2012), Martin et al. (2012), Ketten and Bartol (2006 and 2008), and Piniak et al. (2016). An underwater audiogram of *Trachemys scripta*, based on data from Christensen-Dalsgaard et al. 2012, is shown for comparison

indicate that marine turtles are almost as sensitive to underwater sound (within their limited frequency range) as are some of the most well-studied fish (Popper and Fay 2011) and pinnipeds (Reichmuth et al. 2013). Turtles are therefore potentially affected by anthropogenic sound, just like some species of fish and pinnipeds. In addition, there are behavioral indications that marine turtles are disturbed by aquatic sound: Loggerhead turtles will actively avoid or dive in response to airgun sounds (O'Hara and Wilcox 1990; DeRuiter and Larbi Doukara 2012).

This chapter investigated TTS in a freshwater aquatic turtle, the red-eared slider (*Trachemys scripta elegans).* Although the slider is more amphibious than the marine species, the basic configuration of its ear and auditory sensitivity (Fig. 1) is similar to marine species (Ridgway et al. 1969; Christensen-Dalsgaard and Manley 2013). It has

been shown that the red-eared slider's ear is adapted for underwater hearing (Christensen-Dalsgaard et al. 2012), and it is by far the most well-studied turtle with respect to hearing. Like other turtles and tortoises, it has a tympanic middle ear, where the vibrations of the cartilaginous tympanic disk are coupled to the inner ear by the single middle ear ossicle, the columella (homologous to the mammalian stapes). In air, the turtle is less sensitive than other tympanate vertebrates, but the specializations of the middle ear, most notably an air-filled middle ear cavity, make the slider sensitive to underwater sound pressure in the 200–1000 Hz range, with best sensitivity around 400 Hz and lowest thresholds at 80 dB re 1μPa. The audiogram has the same shape in air and water, and the sound pressure thresholds are 5–12 dB higher in water than in air (Christensen-Dalsgaard et al. 2012).

The inner ear of the red-eared slider has been the subject of intensive studies (reviews in Manley 1989; Fettiplace 2020) facilitated by the extraordinary ability of turtle nerve tissue to survive under low oxygen tension. The auditory papilla (basilar papilla) is short, containing approximately 1100 hair cells, innervated by 1350 afferent fibers (Manley 1989) and covered by a tectorial membrane. Electrical resonance of the hair cells caused by negative feedback between repolarizing K-currents and depolarizing Ca-currents account for most of the frequency tuning in the papilla (Fettiplace 2020). Thus, the slider would be an excellent model to study the effects of TTS on the hair cells.

In this study of TTS in red-eared sliders, turtles were exposed to noise in air for 24 h (Gaussian white noise, approx. 80 dB Leq at the turtle). The auditory sensitivity was measured by recording auditory evoked potentials from the brainstem in lightly anesthetized turtles (Christensen-Dalsgaard et al. 2012) before exposure, immediately after exposure, and during the following recovery period. In all animals there was a depression of sensitivity immediately after exposure and full recovery 3–5 h after exposure.

## TTS in the Red Slider

Auditory evoked responses were measured in six red-eared sliders stimulated with transient "clicks" (a half-cycle 4 kHz sinusoid) and tone-masked clicks using the difference between the response to the clicks and to the masked click as a measure of the sensitivity to the tone (mABR, see Brandt et al. 2018; Christensen-Dalsgaard et al. 2012). An example of the click response from one animal, before noise exposure, is shown in Fig. 2. From such measurements, the threshold could be calculated as a fixed signal-to-noise ratio of the brain stem response.

The initial threshold shift after their 24 h noise exposure was measured in each turtle right after sedation. During the 24 h exposure the turtle was placed into a bin with a bit of water to prevent dehydration (but without water level covering the eardrums) and food. The light in the chamber was turned off during the exposure.

The difference in thresholds (dB before and after noise exposure) is shown in Fig. 3, both showing the difference in threshold when stimulated with a transient "click" and when using 500 Hz. For the click response the median difference was 13 dB, and at 500 Hz the median difference was 20 dB. For both stimuli, threshold

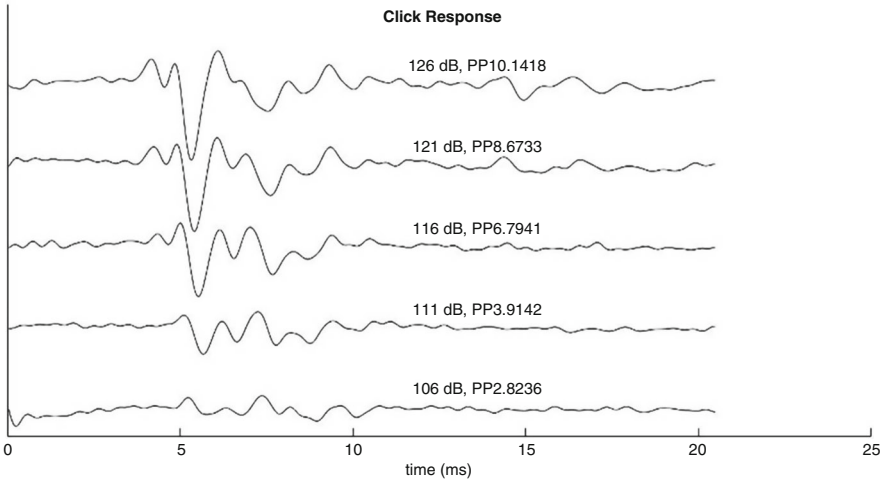



**Fig. 2** ABR with click stimulation. Example of the auditory brain stem response of a turtle to a transient "click" (half-cycle sinusoid at a nominal frequency of 4 kHz) showing traces at five stimulus levels (air-borne sound, dB re 20 μPa, peak equivalent). The numbers on the traces are the peak-peak amplitude of the brain stem response

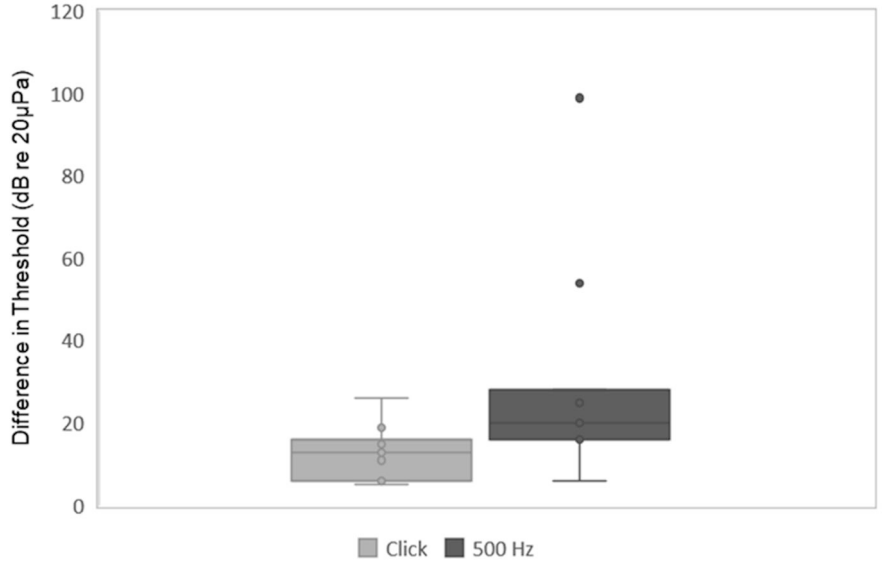



**Fig. 3** Difference in click and tone threshold directly after noise exposure. Threshold shift of auditory brain stem responses immediately after 24 h noise exposure, showing both threshold shift of click and a 500 Hz tone

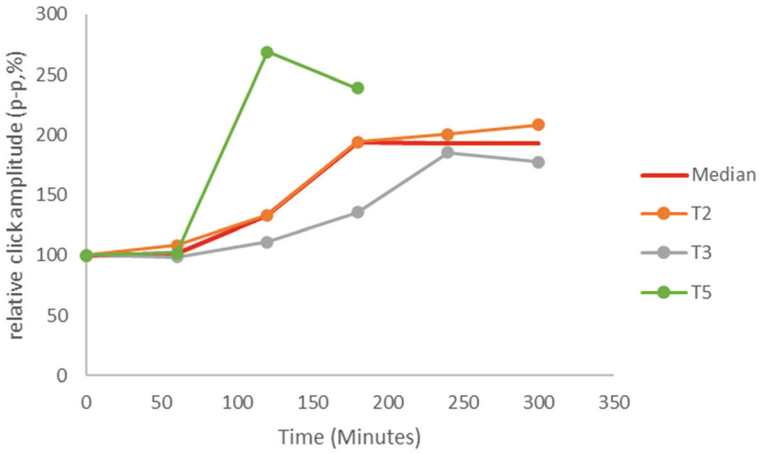



**Fig. 4** Recovery from noise exposure. Recovery of auditory brain stem response to click stimulation after noise exposure, shown for three individuals (T2, T3, and T5) and the median response. The peak-to-peak brain stem responses are measured relative to the response immediately after noise exposure (in units of %)

differences were statistically significant (Click: $p = 0.005$; 500 Hz: $p < 0.01$). There were considerable individual differences in the threshold shifts, especially for tonal stimulation at 500 Hz, with for one individual reaching 99 dB.

After recording the initial threshold shift, the recovery time of each individual was measured by hourly measurements of sensitivity to stimulation by clicks. The click amplitude (percent) relative to the initial click amplitude postexposure was plotted in Fig. 4. An amplitude increase was observed over time in three turtles, with a median of 4 h for the turtles to recover to a preexposure amplitude (approximately 200% of the initial postexposure amplitude). In all animals, thresholds were reduced after initial noise exposure. While three of the turtles recovered in the 5 h observation period, the other turtles brain stem response either did not increase or increased only slightly throughout the measurements. In all animals the thresholds had recovered to preexposure levels a month after exposure.

## Implications of Noise Exposure

The study reported here clearly show that a moderate noise exposure at 80 dB $L_{eq}$ in air produces a measurable TTS, on average 13 dB for click thresholds and 20 dB for tone thresholds, in red-eared sliders.

Although the turtles were exposed to air-borne sound, which is easier to control than underwater sound, these preliminary results may indicate how turtle hearing is affected by underwater sound exposure. The pressure thresholds are approximately 5–12 dB higher in water than in air for this species (Christensen-Dalsgaard et al. 2012). We may therefore assume that the in-air sound exposure level used in this

study (80 dB $L_{eq}$ re 20 μPa) corresponds to an underwater sound exposure of 111 dB Leq re 1 μPa, assuming a 5 dB higher pressure threshold in water and compensating for the different reference pressures used in air and underwater. The corresponding sound exposure level during 24 h exposure in water is 111 dB + $10\log_{10}(24{\cdot}60{\cdot}60)$ = 160 dB re 1 $\mu Pa^2s$. Note that the noise levels reported here are not weighed by the turtle audiogram and therefore the exposure level within the turtle's hearing range resulting in TTS is probably lower.

Harbor seals (*Phoca vitulina)* and harbor porpoises (*Phocoena phocoena*) are some of the most well-studied marine species regarding TTS, and they have TTS thresholds ranging from 140–200 dB re 1 $\mu Pa^2s$, depending on stimulus frequency (Tougaard et al. 2022). For several species of fish, exposures at 180 dB re 1 $\mu Pa^2s$ resulted in up to 20 dB TTS (Popper et al. 2005), but fish data are very variable, both depending on the species and the experimental methodology. The results of especially marine mammals are thus comparable (even though the range is huge) with this study's finding from turtles. When addressing concerns for anthropogenic noise affecting marine animals, it is important not only to address fish and marine mammals but also any potential effects on turtles. Many sources of anthropogenic underwater sounds generate exposure levels above 160 dB re 1 μPa. This may include, for example, nearby ships, pile driving while constructing wind farms, and air guns while searching for oil and gas (Richardson et al. 1995). Since marine turtles have underwater hearing abilities comparable to the red-eared slider (Figure 1), sea turtles should be a concern for protective measures.

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# Atlantic Green Turtles and Hawksbill Turtles: Behavioral Responses to Sound

85

Ron A. Kastelein, Arie Smink, and Nancy Jennings

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**Abstract**

Effects of sounds on the behavior of sea turtles were investigated, to (1) determine whether sounds can be used to deter turtles from areas of human activity and (2) inform Environmental Impact Assessments by evaluating how anthropogenic sounds affect sea turtles. Two Atlantic green turtles *Chelonia mydas* and two

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hawksbill turtles *Eretmochelys imbricata* in a pool were exposed to synthetic sounds, an impulsive sound, Helicopter Long-Range Active Sonar (HELRAS) down-sweeps, and recordings of killer whale *Orcinus orca* vocalizations. Typical responses were increased swimming speed and/or changes in swimming pattern or direction, but the pure tones, impulsive sound, HELRAS, and killer whale sounds elicited no response. Strong responses occurred to series of 0.2–1 kHz up-sweeps with 100% duty cycle, duration 0.5 s, and sound pressure level >151 dB re 1 μPa. Above this threshold, the spectrum, frequency modulation, and temporal pattern of the sounds, and the behavioral state of the turtles, determined their response. Most responses occurred during exposure to sounds with frequencies between 0.2 and 1 kHz, within the hearing range of turtles. This pilot study suggests that turtles respond to high-amplitude, low-frequency anthropogenic sounds and that deterring sounds should be low in frequency and temporally and spectrally complex.

**Keywords**

Behavioral response · Environmental impact assessment · Hearing · Sound · Marine turtles · Sea turtle

## Introduction

Sea turtles consist of seven species in the superfamily Chelonioidea, of which six are threatened (IUCN categories Vulnerable, Endangered, and Critically Endangered; https://www.iucnredlist.org) and one is Data Deficient. Sea turtles have streamlined shells, large paddle-shaped fore flippers adapted for propulsion, and rear flippers for steering. Except when laying eggs on beaches, they remain at sea (Gulko and Eckert 2003; Lutz et al. 1996).

Turtles are incidentally injured or killed during industrial, fishing, and recreational activities at sea (Eckert and Eckert 2005; Lester et al. 2013; Ramirez et al. 2017; Swimmer and Brill 2006), so methods to deter them from areas where these activities take place are needed. High-amplitude sounds have been used to deter other marine animals (Thompson et al. 2020), and knowledge of turtle hearing and responses to sound could be used to generate sounds that deter turtles. Such knowledge could also be used to avoid accidental deterrence of turtles from ecologically relevant areas, such as those used for resting, breeding, and feeding, by anthropogenic sounds.

Little is known about the importance of sound in the ecology of turtles, but turtle ear anatomy is most suited to hearing under water via bone conduction in the skull (Christensen-Dalsgaard et al. 2012; Lenhardt et al. 1985; Willis et al. 2013). The hearing of three of the seven species has not been tested at all (hawksbill turtle *Eretmochelys imbricata*, olive ridley sea turtle *Lepidochelys olivacea*, Australian flatback sea turtle *Natator depressus*). In the remaining four species, underwater hearing sensitivity has been quantified by using auditory evoked potentials (presenting an acoustic stimulus and recording neural responses from electrodes on the

**Table 1** Summary of published information on underwater hearing in the four sea turtle species that have been subject to relevant research. Most sensitive hearing is defined as within 10 dB of maximum sensitivity

| Species | Hearing frequency range | Most sensitive hearing | Source |
|---|---|---|---|
| Atlantic green turtle *Chelonia mydas* | 0.1–0.5 kHz | 0.2–0.4 kHz | Bartol and Ketten 2006 |
| Loggerhead sea turtle *Caretta caretta* | 0.024–1 kHz | 0.2–0.5 kHz | Lavender et al. 2014; Lenhardt et al. 1983; Martin et al. 2012 |
| Kemp's ridley turtle *Lepidochelys kempii* | 0.1–0.5 kHz | 0.1–0.2 kHz | Bartol and Ketten 2006 |
| Leatherback sea turtle *Dermochelys coriacea* hatchlings | 0.05–1.2 kHz | 0.1–0.4 kHz | Piniak et al. 2012 |

head; Table 1; see Bartol and Ketten 2006; Lavender et al. 2014; Lenhardt et al. 1983; Martin et al. 2012; Piniak et al. 2012). Popper et al. (2014) proposed sound exposure criteria for sea turtles based on the very few published data and recommended increased research on the effect of sound on sea turtles.

So far, only a few attempts have been made to identify sounds that could deter sea turtles from dangerous areas, and all involved loggerhead sea turtles *Caretta caretta*. O'Hara and Wilcox (1990) investigated the avoidance response of loggerhead sea turtles to low-frequency high-intensity sounds generated with two types of airgun, firing at two pressures. Most of the energy of the sounds was at 0.025 kHz, but energy was present up to 0.7 or 1.0 kHz, depending on the airgun type. The turtles mainly reacted to the airguns firing at the higher pressure (140 kg/cm$^2$) by moving ~30 m away from the sound source. Moein et al. (1995) investigated the use of airguns with spectra 0.1–1.0 kHz to repel juvenile loggerhead sea turtles from hopper dredges. Avoidance of the airguns was observed upon first exposure, but after three exposures, the turtles became habituated to the stimuli. DeRuiter and Doukara (2012) observed the responses of 164 wild loggerhead sea turtles to broadband impulsive sound produced at very high levels during a seismic survey. Turtles that were close to the airguns (median, 130 m; maximum, 839 m) experienced relatively high exposure levels (~191 dB re 1 μPa [peak] at 130 m and ~175 dB at 839 m) and were observed to dive. At least six individuals dove immediately following an airgun shot, often showing a startle response. Diving was interpreted as an avoidance response to airguns. Lenhardt (2002) developed a device aimed at deterring sea turtles from dangerous areas: a vibrating bar, producing light and sounds, that can be placed under a ship. However, this device has not been tested.

The present pilot study assesses the effect of sounds on the behavior of four individuals of two sea turtle species. The results will indicate whether sound stimuli can be used to deter sea turtles from potentially harmful locations and facilitate the development of strategies to reduce turtle mortality or injury. The results are also a step toward the assessment of the effects of anthropogenic sounds on the behavior

**Table 2** Characteristics of the two male Atlantic green turtles *Chelonia mydas* and one male and one female hawksbill turtle *Eretmochelys imbricata* that participated in the sound experiments, including their Archival Resource Key (ARK) numbers

| Turtle species | ARK no. | Sex | Years at Blijdorp Zoo, Rotterdam, the Netherlands | Age (years) | Weight (kg) | Max. curved shell length (cm) |
|---|---|---|---|---|---|---|
| Atlantic green turtle | 703938 | Male | 12 | >12 | 93 | 95 |
| Atlantic green turtle | 703714 | Male | 14 | >14 | 112 | 95 |
| Hawksbill turtle | 703458 | Male | 15 | 15 | 59 | 90 |
| Hawksbill turtle | 705138 | Female | 6 | 40 | 57 | 94 |

and ecology of sea turtles, required to inform Environmental Impact Assessments of human activities which produce underwater sound at sea.

## Materials and Methods

### Study Animals and Location

The study was conducted with two male Endangered Atlantic green turtles *Chelonia mydas* and one male and one female Critically Endangered hawksbill turtle *Eretmochelys imbricata* (Table 2).

The study took place in an indoor pool at Blijdorp Zoo, Rotterdam, the Netherlands, in 2010. The pool was normally used as a quarantine area for the aquarium department and was not open to visitors or zoo staff, so no disturbance occurred during the experiments. The pool was oval (7.6 m × 5 m × 1 m deep; Fig. 1) and made of plastic. To reduce sound conduction from the floor, it was placed on a layer of Styrofoam. The water level was kept constant. The water was clear so underwater visibility was very good, even over the length of the pool. The water temperature was 24 °C, and the salinity was around 34.5‰. There was no current in the pool during the experiments as the water circulation pump was switched off. A tent was erected over the pool to retain heat and reduce disturbance; the air in the tent was heated to 24.5 °C. The equipment used to produce the sound stimuli and video recording equipment were housed out of sight of the turtles, in a research cabin just outside the tent.

### Video Equipment

The turtles' behavior was filmed from above by aerial camera 1 (Mariscope, model Micro) with a wide-angle lens, which was placed on a pole 3 m above the water

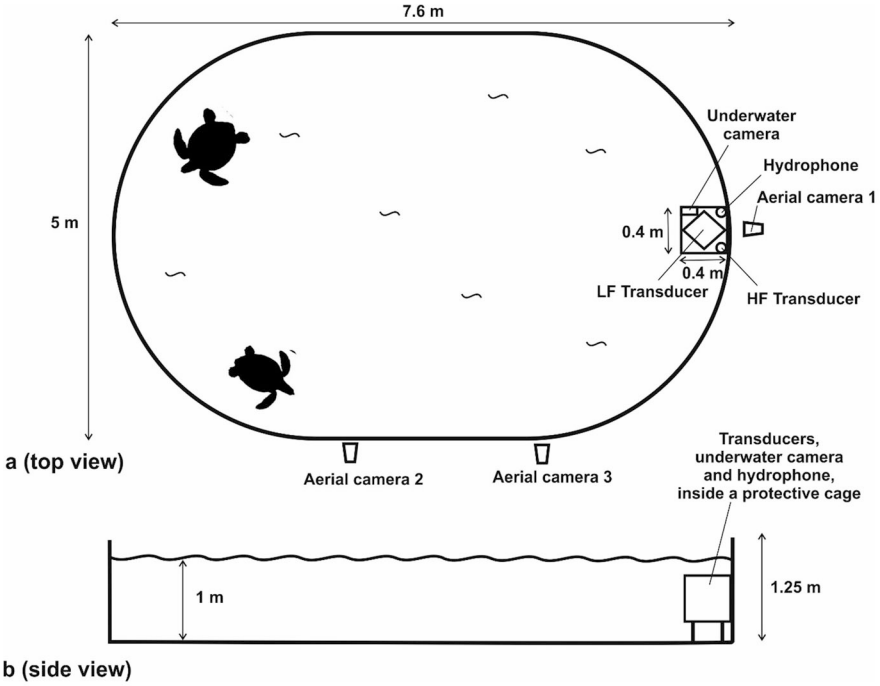



**Fig. 1** The pool at Blijdorp Zoo, Rotterdam, the Netherlands, in which the acoustic experiments were conducted, showing the locations of the low-frequency (LF) and high-frequency (HF) transducers, hydrophone, and aerial and underwater cameras, to scale (**a**, top view; **b**, side view). Sea turtle silhouettes in (**a**) are also to scale: top, Atlantic green turtle *Chelonia mydas*; bottom, hawksbill turtle *Eretmochelys imbricata*

surface at one end of the pool (Fig. 1). The entire surface of the pool was captured on the video image. Two additional aerial cameras (2 and 3) were placed on the long side of the pool. One underwater camera (Mariscope, model Micro) was placed next to the low-frequency transducer inside a protective cage (Fig. 1). The output of the cameras was fed through a video quad splitter and digitized with an analog-to-digital converter (EZ Grabber, Vista version) and then stored on a computer (Medion; Fig. 2). The images from all four cameras were visible to the operators on monitors in the research cabin.

The background noise and the stimuli were recorded via a recording hydrophone (Labforce, model 90.02.01) and a conditioned charge pre-amplifier (SEAMARCO, model CCAMS1000-1). The output of the pre-amplifier was digitized via the analog-to-digital converter and recorded on the computer in synchrony with the video images. The output was also fed to an amplified loudspeaker, so that the operators in the research cabin could monitor the background noise and the test sounds during sessions. Via a microphone and pre-amplifier, comments could be added to the video recordings.

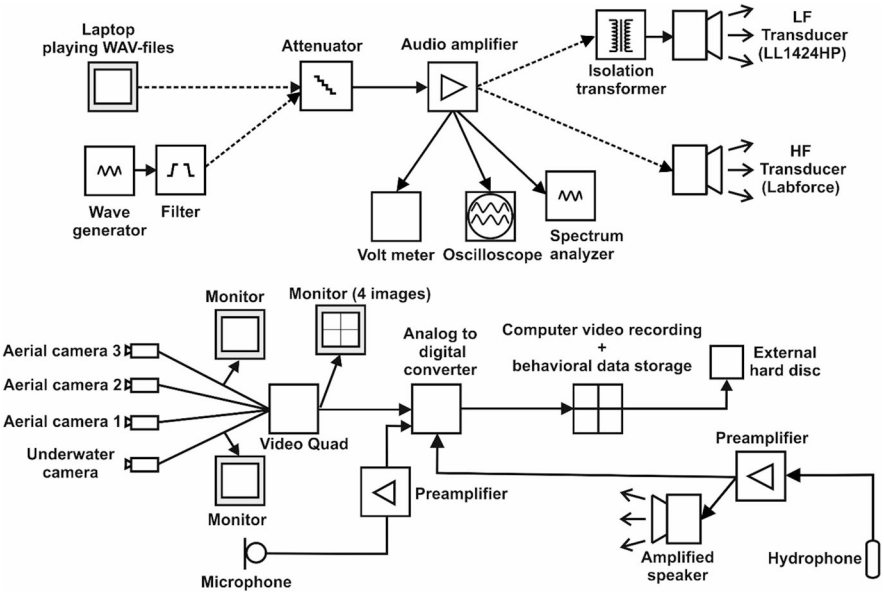



**Fig. 2** Block diagram of the sound generation and control systems (top) and the listening and recording equipment (bottom) used in the turtle behavioral response study

## Sound Equipment and Stimuli

The turtles were exposed to synthetic sounds, an impulsive sound, Helicopter Long-Range Active Sonar (HELRAS) down-sweeps, and recordings of killer whale *Orcinus orca* vocalizations (Table 3).

Of the synthetic sounds, the pure tones, square waves, triangle waves, sawtooth waves, frequency-modulated tones, and white noise bands were generated with a wave generator (Hewlett Packard, model 33120A) and a digital band-pass filter (Krohn-Hite, model 3384; Fig. 2). The other synthetic sounds were created as WAV files in Adobe Audition (version 3.0) and played on a laptop computer with durations 10 s and duty cycle 100%.

For the impulsive sound, the $t_{90}$ signal duration in the pool was 10 ms, and the zero-to-peak sound pressure level (SPL) was 120 dB re 1 µPa. The HELRAS down-sweeps were recorded at sea (see Kastelein et al. 2019). Killer whales are potential predators of leatherback sea turtles *Dermochelys coriacea* (Pitman and Dutton 2004), so killer whale vocalizations that were recorded at Point Retreat, Alaska, USA, were used (frequency range, 0.5–8 kHz; one-third-octave source level, 177 dB re 1 µPa@ 1 m; mean one-third-octave SPL in the pool, 173 dB re 1 µPa). The impulsive sound, HELRAS down-sweeps, and killer whale vocalizations were also played from the laptop computer.

From the laptop, the signals went to an audio power amplifier (Velleman HQ VPA2200MBN), the output of which was controlled with a custom-built digitally controlled attenuator (SEAMARCO AS 2009-01; 1 dB steps; Fig. 2). The sounds were projected under water via two transducers in a protective cage at a depth of 50 cm at

**Table 3** Thirteen sound types and the fundamental frequencies at which they were produced for the behavioral response experiment, comprising 80 combinations of sound type and frequency in total. The turtles were exposed to synthetic sounds (indicated with *), an impulsive sound, Helicopter Long-Range Active Sonar (HELRAS) down-sweeps, and recordings of killer whale *Orcinus orca* vocalizations. Most sounds generated harmonics and thus also contained energy at higher frequencies than those listed here. Details of the 18 sounds (shown here in bold) that elicited responses in the turtles are shown in Table 4. For the synthetic sounds, exposure durations were 10 s, and duty cycles were 100%. Only the sine wave up-sweeps indicated with † were produced with the high-frequency transducer. The sound types are shown in the order in which they were tested. Within each sound type, frequencies were tested in order from left to right and from top to bottom

| Sound type | Fundamental frequencies (kHz) and details |
|---|---|
| Pure tone (continuous wave)* | 0.2, 0.4, 0.6, 0.8, 1, 2, 4, 8, 16, 25 |
| Square wave* | 0.2, 0.4, **0.6**, 0.8, **1**, 2, 4, 8, 16, 25 |
| Triangle wave* | 0.2, **0.4**, 0.6, 0.8, 1, 2, 4, **8, 16**, 25 |
| Sawtooth wave* | 0.2, 0.4, **0.6**, 0.8, 1, 2, 4, 8, **16**, 25 |
| Frequency-modulated tones* | Center frequencies: **0.2, 0.4**, 0.6, 0.8, **1**<br>(modulation bandwidth +/− 100 Hz, modulation rate 10 Hz) |
| Up-sweeps (sine wave)* | 0.2–0.3, 0.3–0.4, 0.4–0.5, 0.5–0.6, 0.6–0.7, 0.7–0.8, 0.8–0.9, 0.9–1.0, **1–2**, 2–3, 3–4, 4–6, 6–10, **0.2–0.7**, 0.7–1.0, 16–20†, 20–25†, 25–30†, 30–40† (all with signal duration 0.5 s)<br>0.2–1.0 (signal duration 0.3 s)<br>**0.2–1.0** (signal duration 0.5 s)<br>0.2–1.0 (signal duration 1 s) |
| Up-sweep (square wave)* | **0.2–1.0** (signal duration 0.5 s) |
| White noise bands* | 0.02–20, **0.2–0.4**, **0.4–0.63**, 0.63–1.0, 1–2 |
| Warbles* | **0.8–0.2**, 0.2–0.8 (modulation bandwidth +/− 600 Hz, modulation rate 10 Hz) |
| Up-sweeps (sine wave)* | 0.2–0.6 (signal duration 0.1 s)<br>**0.2–0.6** (signal duration 0.2 s) |
| Impulsive sound | A single pulse (broadband; 5 ms in WAV file, 10 ms in the pool due to reverberations) |
| HELRAS down-sweeps | 1.3–1.44 (signal duration 1.25 s with 1 s gaps), total duration of a series of 4 repetitions: 8 s |
| Killer whale vocalizations | 0.5–8 (eight 0.5–1 s vocalizations in 20 s) |

one end of the pool (Fig. 1). The output of the sound system to the transducers was monitored with a digital oscilloscope (Tektronix 2201), a voltmeter (Agilent 34401A), and a spectrum analyzer (Velleman, PCSU1000; Fig. 2). Sounds up to 16 kHz were transmitted via a low-frequency balanced tonpilz piezoelectric acoustic transducer (Lubell, model LL 1424HP). The theoretical maximum source level that could be produced by this low-frequency transducer was 197 dB re 1 μPa @ 1 m, at 0.6 kHz. The sine wave up-sweeps between 16 and 40 kHz were transmitted with a piezoelectric high-frequency underwater transducer (Labforce, model 90.02.01). The source levels were as follows: for the 16–20 kHz sweep, 124–130 dB re 1 μPa; for the 20–25 kHz sweep, 130–136 dB re 1 μPa; for the 25–30 kHz sweep, 138–139 dB re 1 μPa; and for the 30–40 kHz sweep, 139–140 dB re 1 μPa.

Sound measurements were conducted by TNO (The Hague, the Netherlands) with a hydrophone (Bruel & Kjaer [B&K], model 8105) and a pulse acquisition system

(B&K, model 3560C) with a sample frequency of 65 kHz. The SPL in one-third-octave bands was determined by averaging over 10 s. The system was calibrated with a pistonphone (B&K, model 4223). The measurement hydrophone was at 1.5 m from the transducers at a depth of 0.5 m. Only the sounds that elicited a response (as defined in "Methodology") were recorded and analyzed.

## Methodology

The turtles were placed in the pool in two pairs of two individuals, one of each species. The first pair (male Atlantic green turtle 703938 and male hawksbill turtle 703458) was placed in the pool on 9 April 2010 to habituate to the new environment; tests were carried out on 13 and 14 April. The second pair (male Atlantic green turtle 703714 and female hawksbill turtle 705138) was placed in the pool on 16 April; tests were carried out on 20 April.

A response to a sound was defined as at least one of the two members of both turtle pairs changing its behavior during sound exposure, in comparison to the swimming behavior observed when no sound was produced. In ~80% of the cases, either both members of a pair responded, or both did not respond, and responses were also consistent between the pairs (see "Results").

During the two test days with the first turtle pair (13 and 14 April), the killer whale vocalizations were not tested. The remaining 12 sound types were tested at each frequency (79 combinations of sound type and frequency in total; Table 3), at increasing SPLs, up to the maximum level that could be produced by the system for that sound (determined by the characteristics of the transducers). On 13 and 14 April, each sound was offered at up to three SPLs (10 dB steps), and responses to the lower SPLs were rare. Therefore, in tests with the second turtle pair (20 April), each sound was offered only at the two higher SPLs.

Each synthetic sound was produced for 10 s, and the time between tests was ~8 min. Only sounds that elicited a response in the first turtle pair were also tested with the second turtle pair, alongside the killer whale vocalizations, which lasted 20 s. The turtles in each pair were exposed to each sound at a given SPL only once, except the killer whale vocalizations which were tested four times. In total, 196 exposures, each lasting 10 s (or 20 s for killer whale vocalizations and 8 s for the HELRAS signals), were conducted: 143 on 13 and 14 April with the first pair of turtles and 53 on 20 April with the second pair. Thus, the dataset was derived from baseline observations (9–20 April) and observations during approximately 33 min. of sound exposure, conducted over three test days (13, 14, and 20 April).

## Results

Observations of the turtles when sounds were not being produced, on habituation days and test days, provided a baseline: the turtles spent ~90% of their time swimming slowly, at constant speed, along the pool wall. The remaining ~10% was spent resting on the bottom of the pool, next to the wall.

Responses to sounds, when they occurred, typically consisted of increased swimming speed and/or changes in swimming pattern or direction (including swimming across the middle of the pool, swimming in small circles, and swimming away from the transducer, instead of swimming along the pool wall as usual). Startle responses were not observed: responses typically did not start until 2–5 s after a sound's onset. Once the sound stopped, the turtles' behavior returned to normal within ~1 min. The turtles were not observed to approach or investigate the transducers, and the responses of the second turtle pair, which were only exposed to the sounds which elicited a response in the first turtle pair, were consistent with those of the first pair for those sounds.

Responses were observed during 36 of the 196 exposures (Table 4). The pure tones, impulsive sound, HELRAS down-sweeps, and killer whale vocalizations elicited no response, even at the highest levels that could be produced (e.g., killer whale vocalizations: one-third-octave SPL in the pool, 173 dB re 1 μPa).

Turtles that were resting (lying still on the bottom of the pool; this occurred in 19 exposures) rarely responded to sound. Resting turtles responded only to sounds 9 and 12, by starting to swim. The turtles generally responded only to particular sound types in the 0.2–1 kHz frequency range, to three sounds above this frequency range (sounds 4, 5, and 7), and to sounds with high received SPLs (between 151 and 196 dB re 1 μPa). These SPLs were close to the maximum at which the sounds could be produced with the transducers in the shallow pool. The turtles did not respond to the sounds at SPLs that were 10 dB lower than those reported in Table 4. Of the 0.2–1 kHz up-sweeps, they responded only to those of 0.5 s duration; up-sweeps of shorter (0.3 s) and longer (1.0 s) durations did not elicit a behavioral response (all were produced at the same source level).

## Discussion and Conclusions

Only four captive adult individuals of two turtle species were available for this pilot study, so the results should be interpreted with caution. Statistical analysis could not be conducted due to the low sample sizes. However, this study makes an important contribution to the little that is known about the effects of underwater sounds on sea turtle behavior.

### Response Speed and Individual Responses

Startle responses were not observed in the turtles, even to very loud sounds; they began to respond 2–5 s after a sound began. Response time in turtles is longer than in marine mammals and fish (Boeger and Pie 2006; Robertson et al. 2013).

In most cases, the response or lack of response to a sound was consistent in both turtles of each pair, and responses were also consistent between the pairs. Due to the small sample size, it cannot be determined whether the occasional differences in responses were due to individual differences, age-related differences, or species differences. The differences appeared to be related to the behavioral state: resting

**Table 4** The characteristics of the 18 synthetic sounds which elicited a response in both turtle pairs (see Table 3 for all the sounds produced). For each sound, the table shows the broadband source level (the sound pressure level at 1 m from the transducers), the broadband sound pressure level at 1.5 m from the transducers (~ the general broadband SPL in the pool and thus the mean received SPL by the turtles), and the highest one-third-octave SPL ≤1 kHz (within the hearing range of the sea turtles; see Table 1). The spectra of these sounds are shown in Fig. 3. The exposure duration was 10 s, and the duty cycle was 100% in all cases. The actual frequency bandwidth in the pool was greater than the fundamental frequency (or frequency range) due to characteristics of the transducers and harmonics. In some cases, the harmonics contained more energy than the fundamental frequency (Fig. 3)

| Sound | Sound type | Fundamental or center frequency (kHz) | Signal duration (s) | Broadband source level dB re 1 µPa @ 1 m | General broadband SPL in pool dB re 1 µ Pa | Highest SPL ≤ 1 kHz dB re 1 µPa |
|---|---|---|---|---|---|---|
| 1 | Square wave | 0.6 | Continuous | 190 | 187 | 187 |
| 2 | Square wave | 1 | Continuous | 192 | 188 | 162 |
| 3 | Triangle wave | 0.4 | Continuous | 184 | 180 | 165 |
| 4 | Triangle wave | 8 | Continuous | 191 | 187 | 112 |
| 5 | Triangle wave | 16 | Continuous | 183 | 179 | 112 |
| 6 | Sawtooth wave | 0.6 | Continuous | 194 | 190 | 174 |
| 7 | Sawtooth wave | 16 | Continuous | 180 | 176 | 113 |
| 8 | Frequency-modulated tones | 0.2 | Continuous | 155 | 152 | 138 |
| 9 | Frequency-modulated tones | 0.4 | Continuous | 160 | 156 | 152 |
| 10 | Frequency-modulated tones | 1 | Continuous | 199 | 196 | 196 |
| 11 | Up-sweep (sine wave) | 1–2 | 0.5 repeated | 190 | 187 | 180 |
| 12 | Up-sweep (sine wave) | 0.2–0.7 | 0.5 repeated | 158 | 154 | 150 |
| 13 | Up-sweep (sine wave) | 0.7–1.0 | 0.5 repeated | 183 | 179 | 179 |
| 14 | Up-sweep (square wave) | 0.2–1.0 | 0.5 repeated | 184 | 180 | 179 |
| 15 | White noise band | 0.2–0.4 | Continuous | 164 | 161 | 159 |
| 16 | White noise band | 0.4–0.63 | Continuous | 170 | 167 | 152 |
| 17 | Warble | 0.8–0.2 | 0.1 repeated | 184 | 181 | 178 |
| 18 | Up-sweep (sine wave) | 0.2–0.6 | 0.2 repeated | 154 | 151 | 145 |

turtles were less likely to respond to sound than active turtles. However, not all sounds were tested with resting turtles.

## Responses to Sounds, Audibility, and Sweep Duration

The turtles mainly responded to sounds between 0.2 and 1 kHz (Table 4 and Fig. 3): sounds that fell within the frequency range of their hearing (Table 1; up to ~1 kHz; Piniak et al. 2016; Ridgway et al. 1969). The pool was not deep enough to allow the propagation of sounds below 0.2 kHz. In deeper pools, responses to sounds with frequencies below 0.2 kHz, which are probably audible to turtles, could be tested.

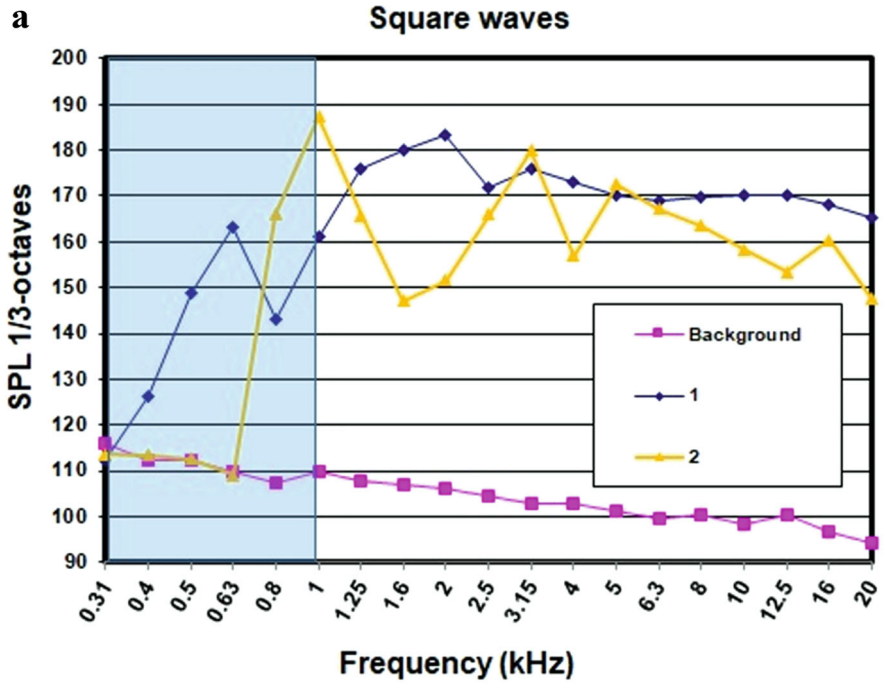
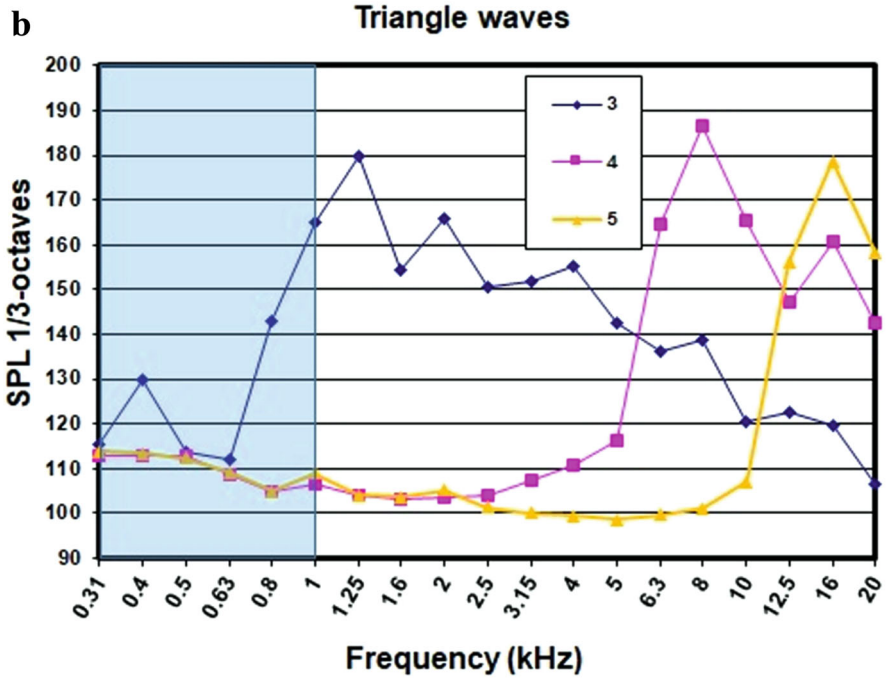



**Fig. 3** The one-third-octave band spectra of the background noise in the pool (**a**) and of the sounds to which at least one of the two members of both pairs of sea turtles responded (**a**–**g**, sounds 1–18; see Table 4). Recording distance: 1.5 m from the transducers. The actual frequency bandwidth in the pool was wider than the fundamental frequency (or frequency range) reported in Tables 3 and 4, due to characteristics of the transducers and harmonics. In some cases, the harmonics contained more energy than the fundamental frequency (for instance, with sound 9 in **d**). Table 4 shows the highest sound pressure levels ≤1 kHz (the frequency band that is shaded in blue), i.e., within the hearing range of sea turtles (see Table 1)



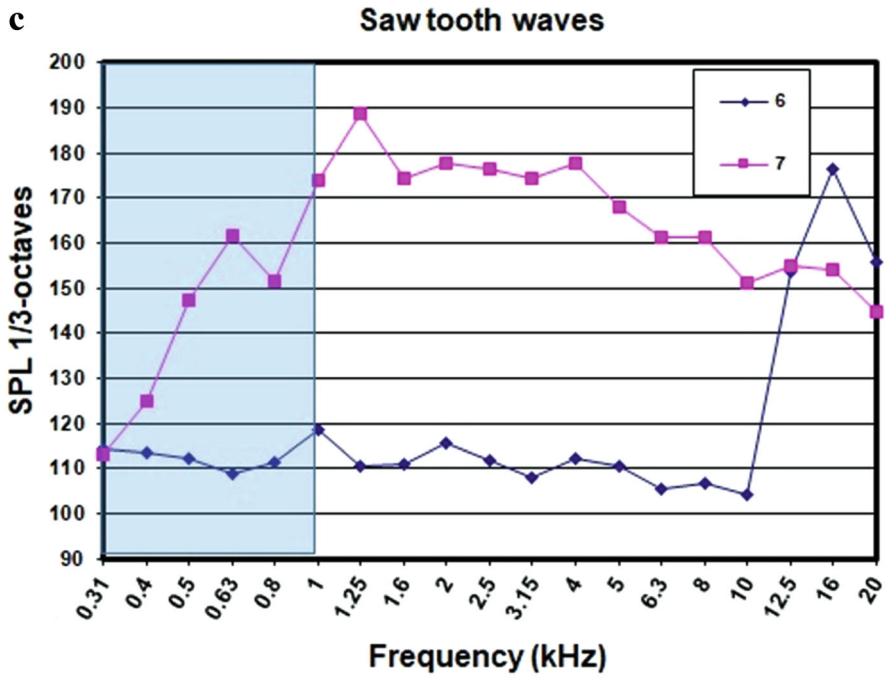




**Fig. 3** (continued)

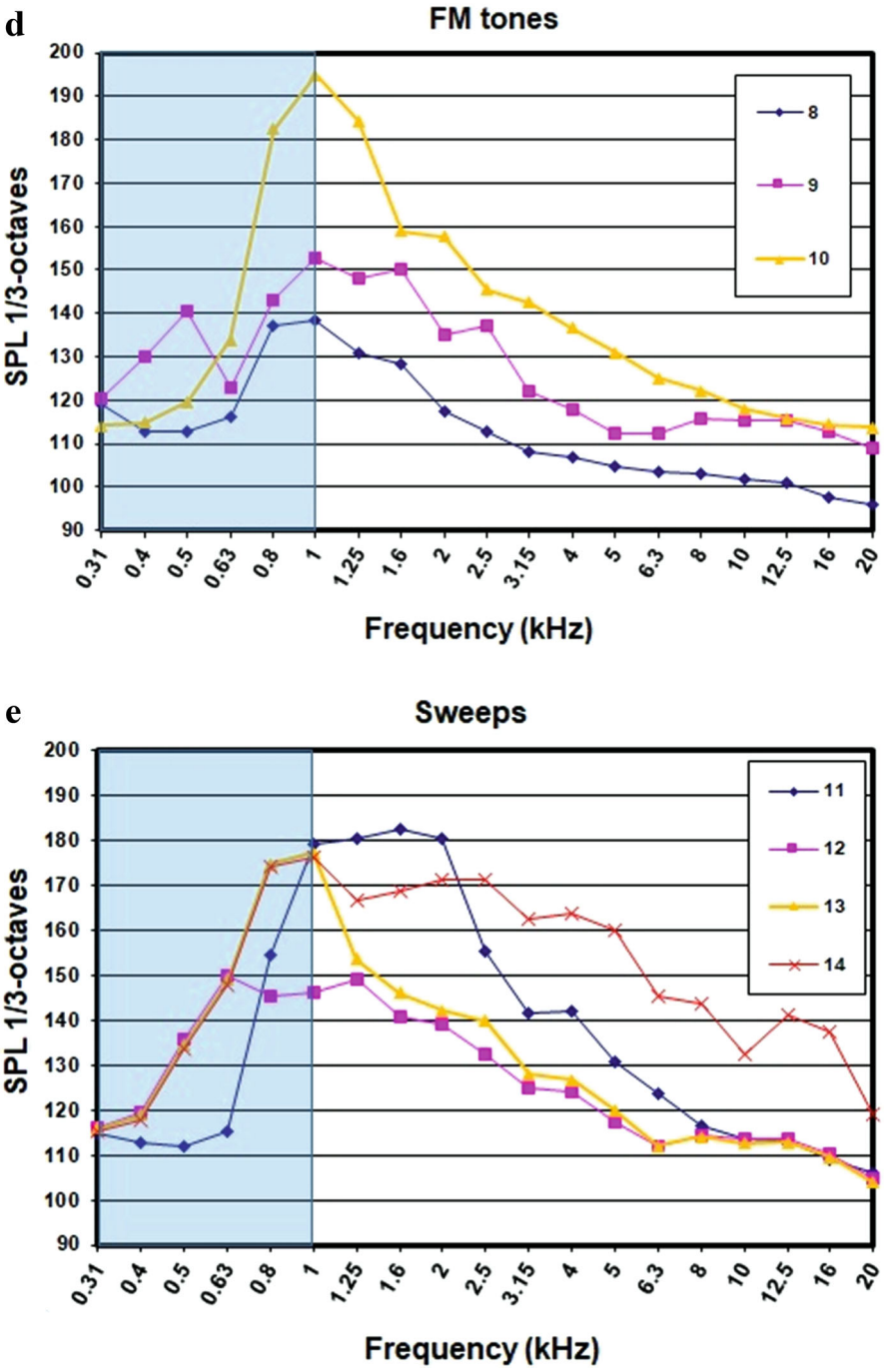
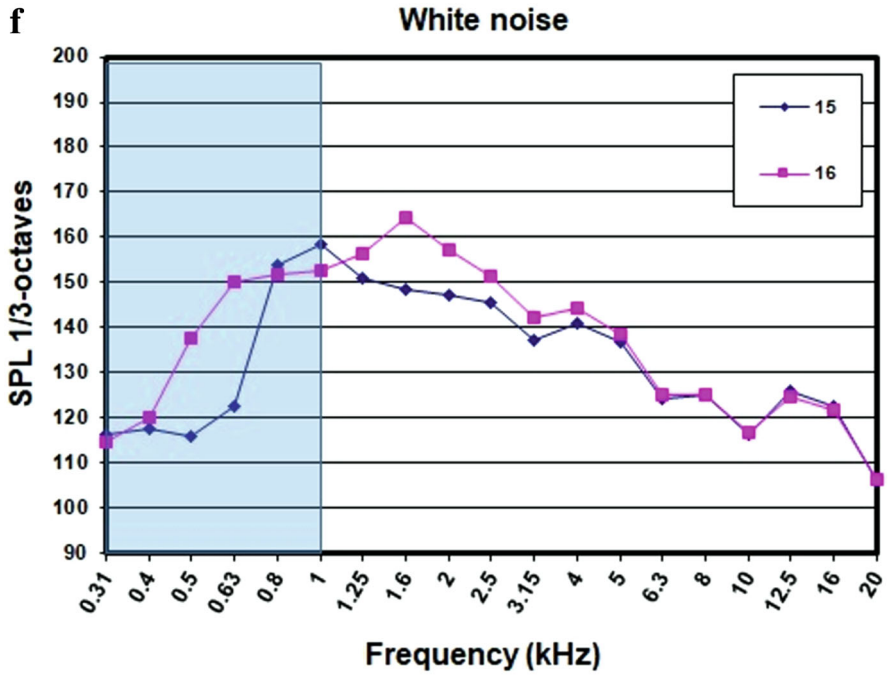



**Fig. 3** (continued)



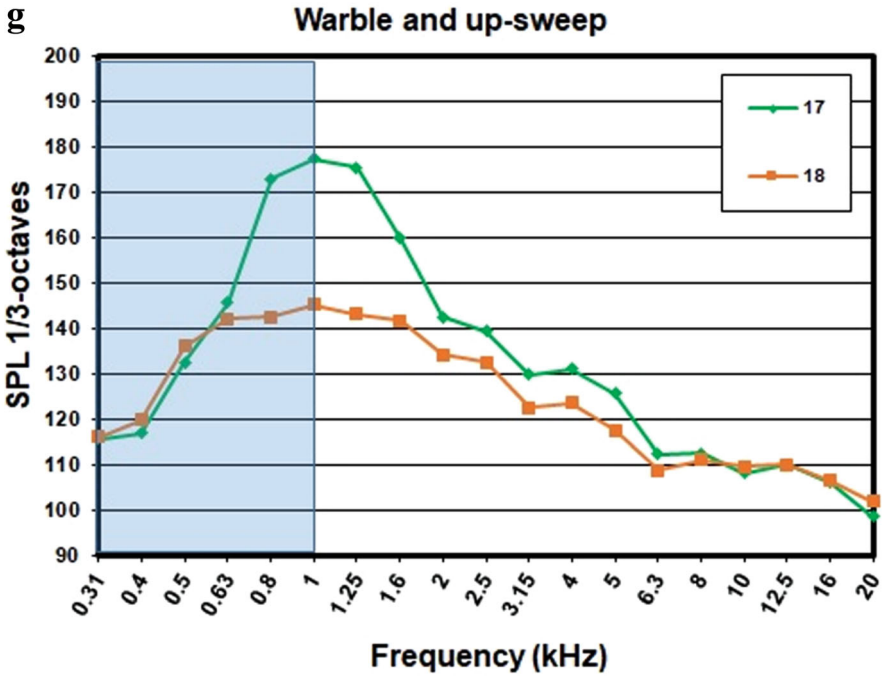




**Fig. 3** (continued)

Of the sounds with fundamental frequencies above 1 kHz, the turtles responded to the 8 and 16 kHz triangle waves and the 16 kHz sawtooth waves (Table 4). They were not expected to respond to these sounds, because the 8 kHz triangle wave (which included energy above the ambient noise at >2.5 kHz), the 16 kHz triangle wave (with energy above the ambient noise at >10 kHz), and the 16 kHz sawtooth wave (with energy above the ambient noise at >10 kHz) are all above the hearing range of the turtles (see Table 1). The energy below 1 kHz in these three relatively high-frequency test stimuli was close to the one-third-octave ambient noise level (112 dB re 1 μPa). It is possible that these high-amplitude, short-wavelength sounds caused resonance in the turtles' bodies, e.g., in the middle ear cavity (see Lenhardt 2002), or in some sections of the respiratory tract. This resonance may have been detected by somatic mechanoreceptors rather than via acoustic energy reaching the cochlea. Sound reaches turtles' ears via bone conduction in the skull, and vibrations may travel via the carapace and vertebrae to the skull (Lenhardt and Harkins 1983). Very low-frequency sounds (<0.04 kHz) can set somatosensory receptors on the skin and around internal organs into sympathetic resonance, which can result in neural activation. Sound at similar levels can also activate the semicircular canals, which can induce positional disturbance.

A strong response was observed to series of 0.2–1 kHz up-sweeps (100% duty cycle) with sweep durations of 0.5 s; a longer (1 s) and shorter (0.3 s) sweep duration elicited no response. Animals may associate up-sweeps with approaching objects, because the Doppler effect may cause approaching objects to produce apparent up-sweeps. The sweep durations (0.3, 0.5, and 1 s) were perceived differently by both operators: the series of short-duration sweeps (0.3 s) was perceived as a continuous sound, while the series of longer sweeps (1.0 s) was perceived as something approaching slowly (it therefore seemed less alarming to the human operators). The 0.5 s sweeps were perceived as most disturbing by the operators. The present study suggests that the temporal pattern plays an important role in eliciting a response in turtles. However, the low-frequency transducer had its resonance frequency at 0.6 kHz, so that the up-sweeps differed temporally not only in frequency but also in amplitude. This may have contributed to the differing effect of the sounds on the turtles. The wider the bandwidth of sweeps in the 0.1–1 kHz band (the hearing range of the turtles), the greater the amplitude modulation was.

## Directionality and Behavior

Responses consisting of diving and surfacing, as reported by Lenhardt (2002), were not observed, perhaps because the pool was rather shallow. In response to some sounds, the turtles moved away from the transducers, though only a small SPL gradient existed (within ~2 m of the transducers, depending on the frequency of the sound). The reverberant environment in the pool probably made it difficult for the turtles to determine the location of the sound source. The turtles did not approach or

show interest in the active transducers. A review of sea turtle responses to low-frequency sounds from airguns used in seismic surveys showed that behavioral responses at sea are often difficult to interpret (Nelms et al. 2016): some turtles exhibit avoidance behavior or dive, but most do not change their swimming or resting patterns abruptly. In agreement with the present pool study, startle responses to sound were not observed at sea (Nelms et al. 2016).

## Use of Underwater Sound as a Sea Turtle Deterrent

The present study shows that sea turtles respond to some complex underwater sounds with energy in the 0.2–1 kHz frequency range, and even to some sounds above their hearing range, which may be detected by sensory systems other than hearing. Recordings of the vocalizations of killer whales, potential predators of leatherback sea turtles (Pitman and Dutton 2004), are unlikely to be effective as turtle deterrents. The SPL, spectrum bandwidth, frequency modulation, and temporal pattern of other sounds determine whether a turtle responds. However, for sound to be effective as a deterrent, turtles must be able to locate the sound source and swim away from it. The nature of their responses (increased swimming speed and/or changes in swimming pattern or direction) suggests that the turtles in the present study would have responded by swimming away from the sound source if they had not been in a small, reverberant, pool. This means that sound could be an effective deterrent. When turtles are resting, it is more difficult to deter them with sound than when they are active (and potentially moving toward a dangerous area), so future research on behavioral responses to sounds should be focused on active turtles.

## Behavioral Responses of Sea Turtles to Underwater Sound

When the turtles responded to the sounds, they did so 2–5 s after the exposure began. Therefore, turtles are unlikely to respond to short-duration sounds. Responses were mainly to sounds between 0.2 and 1 kHz, with a few exceptions. Simple tonal sounds and anthropogenic sounds that are outside the 0.2–1 kHz range are unlikely to affect turtle behavior. Compared to marine mammals, the SPLs required to elicit behavioral responses in turtles are high (>151 dB re 1 μPa), which means that the radius of disturbance areas around most anthropogenic sound sources is smaller for turtles than for marine mammals. Behavioral responses depend on the SPL; on the spectrum, frequency (and possibly amplitude) modulation, and temporal pattern of the sound; and on the behavioral state of the turtle, so it is difficult to produce general acoustic dose-behavioral response functions, as is required for Environmental Impact Assessment. However, compared to marine mammals, sea turtles seem to be less responsive to sound, responding mainly to sounds with very high SPLs in a narrow low-frequency band. At sea, sea turtles may be able to lessen the effects of low-frequency underwater sound by staying near the water surface, where the SPL is often lower (Lenhardt 1994).

## Recommendations for Further Research

The present pilot study shows that several sounds can elicit responses in sea turtles. To increase understanding of how sound affects turtles and how sound could be used to deter them, research with larger sample sizes is recommended (including more turtles and other species, as well as more repetitions and more SPLs per sound type). Research could include:

- Experiments in a larger pool or bay, so that a sound gradient can be created to investigate whether or not turtles avoid sound sources.
- Experiments in a deeper pool or bay, so that lower-frequency sounds can be projected, to investigate whether turtles also show a behavioral response to sounds below 0.2 kHz, which they can probably hear.
- Investigation of whether habituation to sounds occurs and, if it does, whether habituation can be reduced by randomly varying the spectrum or amplitude of the sounds or the temporal patterns of series. Habituation to anthropogenic sound is beneficial as it can lead to reduced harm, but habituation to sounds that are intended to be deterring is detrimental to efforts to deter turtles from dangerous areas.
- Repeated exposures, both to sound types that elicit a response and to similar sounds that do not elicit a response, to determine the repeatability of the responses by the turtles. It is necessary to identify a sound palette type that results in consistent behavioral responses before an effective deterrent sound can be developed.
- Attaching bio-logging tags to wild sea turtles of all species, so that fine-scale movements in response to sound exposures can be examined (Tyson et al. 2017).

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# Acoustic Sensory Ecology of Diving Alcid Seabirds and Potential Noise Impacts

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**Abstract**

Seabirds are important components of holarctic coastal ecosystems. As humans increasingly utilize these sensitive areas, deep-diving seabirds like alcids face a unique combination of pervasive anthropogenic sound exposure in terrestrial and marine environments. However, the acoustic sensory ecology of this threatened group is largely unknown, which complicates assessment of potential anthropogenic noise impacts. Here, aerial auditory sensitivities and nesting soundscapes are compared between three deep-diving alcids: the colonial-nesting common murre (*Uria aalge*) and Atlantic puffin (*Fratercula arctica*), and the solitary-nesting marbled murrelet (*Brachyramphus marmoratus*). Auditory evoked potentials (AEPs) were used to assess aerial hearing in temporarily sedated wild individuals placed inside a field-deployed anechoic chamber. For all species, AEPs were detected across a 0.5–6 kHz frequency range, similar to other birds with available data. Lowest mean auditory thresholds were between 16- and 35-dB SPL, while hearing was generally sensitive across a range extending from 1 to 3.5 kHz. Short-term soundscape dynamics of nesting habitats were also investigated via passive acoustic recordings which show that nesting acoustic environments can vary significantly among species. Overall, frequencies of sensitive hearing in each species overlap with many anthropogenic noise sources, indicating alcid susceptibility to disturbance from a range of noise types.

**Keywords**

Hearing · Soundscape · Endangered species · Stress · Wildlife · Anthropogenic noise · Auditory brainstem response

## Introduction

Seabirds are prominent and important fauna of many holarctic habitats, acting as biovectors for nutrient transport and keystone species for coastal ecosystems (Whelan et al. 2008). Yet increased anthropogenic utilization of coastal areas has reduced seabird habitats and populations such that seabirds are considered the most imperiled group of birds, with 33% of species listed as either threatened or endangered, and approximately half of all species experiencing population declines (Croxall et al. 2012; IUCN 2014; Lascelles et al. 2016). It is therefore important for future management to understand the pressures placed on these taxa and seabird sensitivities to them (Friesen et al. 2017). Alcids are one group of sensitive seabirds with many life history and breeding traits which may result in increased susceptibility to anthropogenic disturbances. As pursuit divers, individuals spend their adult

life on the open sea and only return to land for breeding, and exhibit late sexual maturity (i.e., several years, Lee et al. 2008), high nest site fidelity (Hatch et al. 2000), monogamous breeding at localized colonies (Hudson 1985), and typically lay a single egg per annual breeding cycle (Bent 1919; Tuck 1961).

Understandably, there is growing concern about potentially negative impacts of anthropogenic noise exposure on the biology and behavior of alcids. Sound is known to be an important communication channel for most seabirds, which rely on the detection of acoustic cues to help facilitate a variety of critical behaviors. As a few examples, vocal communication can facilitate recognition among adult individuals as well as between parents and their offspring (Jouventin et al. 1999; Ligout et al. 2016; Kriesell et al. 2018), and the begging calls of chicks can reflect body condition (Gladbach et al. 2009; Klenova 2015). Noise impacts have been increasingly documented across a wide range of marine and terrestrial animal wildlife (Erbe et al. 2018; Hawkins and Popper 2018; Slabbekoorn et al. 2018), including birds (Halfwerk et al. 2018) yet there are significantly less data on seabird susceptibility to anthropogenic noise. Nesting murres exhibit behavioral changes or flushing in response to gunshots, passing aircraft, and vessels at distances ranging from 50 m out to 5 km (Rojek et al. 2007; Labansen et al. 2021). Additionally, anthropogenic noise such as off-road vehicles and pedestrians decrease adult attendance of nests and hatchling survival in American oystercatchers (*Haematopus palliatus*) and Brandt's cormorants (*Urile penicillatus*, Borneman et al. 2016, Buxton et al. 2017).

These studies suggest that some seabird species are susceptible to disturbance from even low-level noise and underscore the need for more comprehensive assessment of noise impact thresholds. Key to improvements in the understanding, prediction, and mitigation of noise effects is baseline documentation and characterization of a species' acoustic biology, such as hearing thresholds and important features of the acoustic environment within animal habitats. Together, these areas of investigation provide a vital foundation of data for the subsequent assessment of noise impacts, such as physiological and behavioral effects, damage to hearing from acoustic overexposure, and masking of important bioacoustics and communication signals (Dooling and Popper 2016).

This chapter compares data on aerial hearing and nesting soundscapes of three species of pursuit-diving alcid seabirds, the marbled murrelet (*Brachyramphus marmoratus*), the common murre (*Uria aalge*), and the Atlantic puffin (*Fratercula arctica*). The goal is to highlight the baseline characteristics of their acoustic ecology and discuss future research directions important for improved assessment of anthropogenic noise impacts.

## Field-Based Measurements of Aerial Hearing in Alcids

Establishing hearing thresholds underpin all subsequent assessment of acoustic disturbance, potential hearing loss, estimates of auditory masking, and other noise impact criteria (Southall et al. 2019), yet there currently are limited data on seabird auditory sensitivities (Crowell et al. 2015; Larsen et al. 2020; Mooney et al. 2020).

This information gap hinders implementation of targeted noise mitigation measures by making it difficult to extrapolate results across the diversity of some 350 species with different life history, ecology, and evolutionary relationships. For example, the auditory system of diving seabird species has been shaped by different evolutionary pressures than non-diving species due to the pressures of deep diving (Zeyl et al. 2022), and perhaps the significant differences in acoustic properties between air and water (Larsen and Wahlberg 2017). Moreover, diving seabirds face a unique combination of exposures to marine and terrestrial sensory cues and anthropogenic noise (Mooney et al. 2019b). Of particular concern are species in Arctic and subarctic environments where accelerated environmental warming and greater accessibility is projected to increase human encroachment into sensitive coastal ecosystems. Thus, there is a particular need to measure the hearing abilities of Arctic and sub-Arctic seabirds and document unperturbed acoustic environments within their ecosystems.

Empirical measurements of bird hearing have traditionally been collected using psychophysical tests, which provide important perceptual sensitivity information. However, such methods are time-consuming, expensive, and challenging for many species. Alternatively, electrophysiological hearing tests involve the measurement of time-averaged neurological auditory evoked potentials (AEPs) elicited by the presentation of acoustic stimuli. They are a relatively rapid way to measure a subject's overall hearing sensitivity and audible frequency range and allows for measuring untrained and even wild animals. These AEP methods have long been applied to birds in the laboratory (Brittan-Powell et al. 2002) and used for other taxa in the field (often odontocetes) for nearly two decades. However, field AEP methods have only recently been applied to birds, during which they may be safely caught, sedated, tested, and released (Mooney et al. 2019a, 2020).

Field-based AEP techniques have recently been used to collect auditory data from multiple alcid seabirds. This study describes and compares a subset of such data, including audiograms from common murres and marbled murrelets ($n = 5$ per species), and previously published audiograms of a related alcid, the Atlantic puffin ($n = 9$ individuals). All hearing tests followed consistent steps between species, which included subject capture, administration of anesthesia, auditory testing, recovery from anesthesia, and subsequent release back at the capture location. The auditory test methods used here are nearly identical to those previously detailed for the Atlantic puffin (Mooney et al. 2019a, 2020), and thus we only provide a summarized version here. Marbled murrelets were captured at night on the waters of Auke Bay, Alaska, using the spotlighting method (Whitworth et al. 1997), while common murres were captured from nesting ledges at Latrabjarg, Iceland, using a wire noose at the end of a fiberglass pole. Atlantic puffins were captured with a fishing net placed over entrances to clifftop burrows (detailed in Mooney et al. 2019a, 2020). Once captured, individuals were transported for approximately 20 min to a sheltered field-testing site on land where they received anesthesia before being placed in a portable anechoic chamber. Body temperature and respiratory rates were continuously monitored while heart rates were manually checked every 15 min. Once sedated, electrophysiological tests were done to record AEPS using two

subdermal needle electrodes placed dorsal to the auditory meatus and at the vertex of the head. A third grounding electrode was placed along the dorsum. Forty-ms epochs of evoked potential signals were recorded concurrent with production of narrowband tone pips between 0.5 and 8 kHz and at output levels controlled in 1–10 dB steps. For each stimulus frequency and sound level, between 750 and 1000 pips were projected and the concurrent epochs were averaged to extract the resulting AEP from noise. Final threshold calculations were made offline in Matlab (Mathworks, MA, USA) and Excel (Microsoft). For the common murre and marbled murrelet, a linear regression was fit to the AEP response magnitudes of each stimulus frequency as a function of stimulus level, and thresholds were estimated as the intersection of the linear regressions with the zero-AEP-magnitude level. For the Atlantic puffin, thresholds were calculated by fitting the linear regression to the magnitudes of cross-correlation functions across stimulus levels. Thresholds for each bird were plotted as a function of frequency to generate audiograms of aerial hearing sensitivity (Fig. 1).

All the studied alcids heard sounds extending from 0.5 to 6 kHz. Measures of central tendency (mean and median) between all individuals resulted in comparatively similar audiograms (Fig. 1a and b). Lowest mean thresholds were 35 and 30 dB re: 20 μPa for the common murre and marbled murrelet, respectively, and both occurred at 2 kHz. Lowest means thresholds were 16 dB re: 20 μPa for the Atlantic puffin, which occurred at 2.5 kHz (Mooney et al. 2020). Overall, these audible frequency ranges and most sensitive frequencies are similar to the majority of nonauditory specialist birds. Notably, the puffin exhibits mean thresholds that are at least 14 dB lower than the other two alcids across the entire audiogram, with the greatest differences mostly at low frequencies of 750 and 500 Hz. Yet the best way to assess a wild population's "representative" auditory sensitivity is not fully established and the desired metric may be dependent on the desired use of the data (Mooney et al. 2020). For example, when assessing potential impacts of anthropogenic noise, particularly for endangered or threatened species, a conservative estimate of auditory sensitivity may be desired, such as a composite audiogram of the lowest observed thresholds for each species (Fig. 1c).

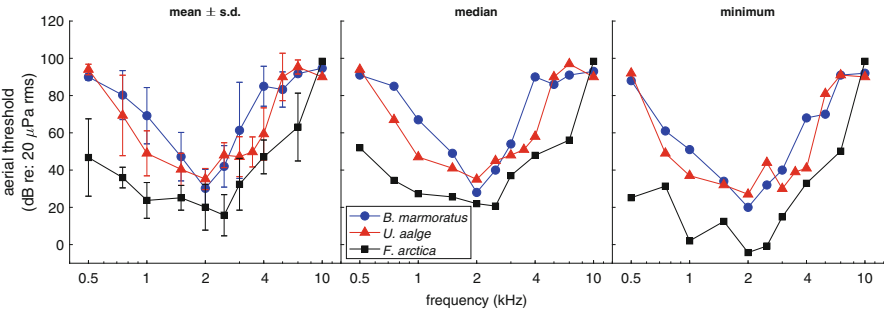



**Fig. 1** Composite in-air AEP audiograms compiled from a subset of data from the Atlantic puffin (black), common murre (red), and marbled murrelet (blue) plotted across different metrics (mean ± s.d., median, and minimum thresholds)

Regardless of which metric is used, these audiograms show the studied alcids (and likely other alcid species) possess generally sensitive aerial hearing akin to many terrestrial birds.

## Environmental Nesting Soundscapes

Habitat-associated soundscapes underlie various ecological processes in a variety of animal taxa, and acoustic cues derived from the surrounding soundscape may be important for seabirds as well. However, there are few studies on aerial soundscape dynamics of many coastal habitats, including seabird colonies. As humans increasingly utilize such areas, it is essential to characterize these soundscapes as a baseline for understanding seabird acoustic biology and thus quantifying the effects of human activity on their acoustic ecology over time. This acoustic sensory information is increasingly valuable for conservation and management efforts (Elmer et al. 2021), yet there are relatively few studies that compare auditory thresholds of wild animals to the noise levels from their particular habitat (Mooney et al. 2018). Nesting soundscapes may be particularly relevant for alcids since a variety of critical behaviors occur within the summer nesting period when they are also most exposed to aerial anthropogenic noise.

Brief snapshots of the nesting soundscape for each of the three studied alcids will be considered here. Passive acoustic recordings were made with a dual-channel Song Meter (SM4) acoustic recorder (Wildlife Acoustics Inc., Concorde, MA, USA) that was placed at known nesting locations of each species. Common murres and Atlantic puffins are colonial nesters along sea cliffs, with common murres congregating on exposed rock ledges and Atlantic puffins burrowing up to 1–2 m into the softer soil atop such cliffs. In contrast, marbled murrelets nest solitarily on large limbs of dominant trees within mature forests, but also occasionally nest on cliffs and on the ground (e.g. Simons 1980; Nelson 2020). For the marbled murrelet, the ambient soundscape was recorded approximately 500 m from a known murrelet ground nest from previous years (Willson et al. 2010, lat: 58.294586, lon: −134.453400). For the common murre, recordings were made at the top of the cliff directly above the short rock ledge where murres were captured for the auditory tests. For the Atlantic puffin, synchronized recordings were made both at the mouth of an inhabited burrow and approximately 1 m inside the burrow. Sounds were recorded either continuously (common murre and marbled murrelet) or on a 25% duty cycle every 4 min (Atlantic puffin) via omnidirectional A2 microphones. Signals were amplified by 16 dB and digitized at a sample rate of 24 kHz before being saved as dual channel .WAV files on SD cards for offline analysis. The A2 microphones were manually calibrated before and after deployments at each site with a 94 dB SPL 1 kHz tone from a calibrated piston-phone. A-weighted sound levels were measured from the recordings in 1-s intervals, and mean (± s.d.) sound levels were then calculated across 1-min bins of the data.

Figure 2 illustrates snapshot aerial soundscapes recorded at nesting sites of the common murre (5 days), marbled murrelet (2 days), and Atlantic puffin (6 days).

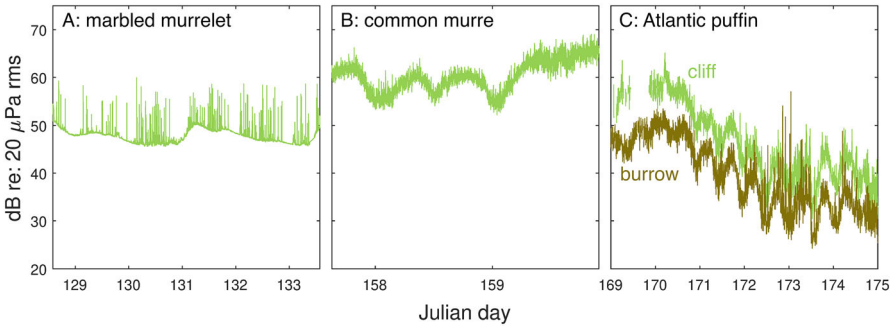



**Fig. 2** A-weighted aerial soundscape levels at nesting sites of (**a**) marbled murrelets, (**b**) common murres, and (**c**) Atlantic puffins. The puffin nesting soundscape is represented both outside and inside a single burrow

All soundscapes contained both biotic and abiotic signals. The seaside soundscapes of the common murre and Atlantic puffin nests were dominated by geophonic and biophonic sound sources (crashing waves and bird vocalizations, respectively) and exhibited the highest general noise levels overall, circa 65 dB re: 20 μPa. The puffin nesting site also exhibited the lowest sound levels circa 30 dB outside the burrow and 25 dB inside the burrow (re: 20 μPa). Baseline sound levels within the forested marbled murrelet nesting area exhibited less variable baseline sound levels between 45 and 55 dB, but this baseline was frequently punctuated by brief but intense (typically low-frequency) sources of anthropogenic sound such as flyovers by planes and helicopters and city traffic from the nearby city of Juneau (~750 m away). Notably, the temporal pattern of the marbled murrelet soundscape levels exhibited a strong diel cycle consistent with the diel nature of the anthropogenic sound encroachment, whereas the seacliff nests of the common murre and Atlantic puffin were clearly influenced by the 12.5 h tidal cycle. It is also noteworthy that time-synced sound levels differed between the inside and outside of the puffin burrow. The puffin burrow acted as a low-pass sound filter which reduced sound levels by an average of 7 (±2) dB across the recording period, with most of this reduction occurring above 1 kHz (Fig. 3).

## Future Research Directions and Anthropogenic Noise Considerations in Alcids

### The Utility of Field-Based Auditory Measurements

The auditory data highlighted in this chapter illustrates the viability of field-AEP techniques to measure hearing from sufficiently large sample sizes of wild individuals (Mooney et al. 2019a, 2020; Smith et al. 2023). With audiograms now documented for 14 of some 350 seabird species, this new method enables valuable data collection from untested species of conservation concern, such as the threatened

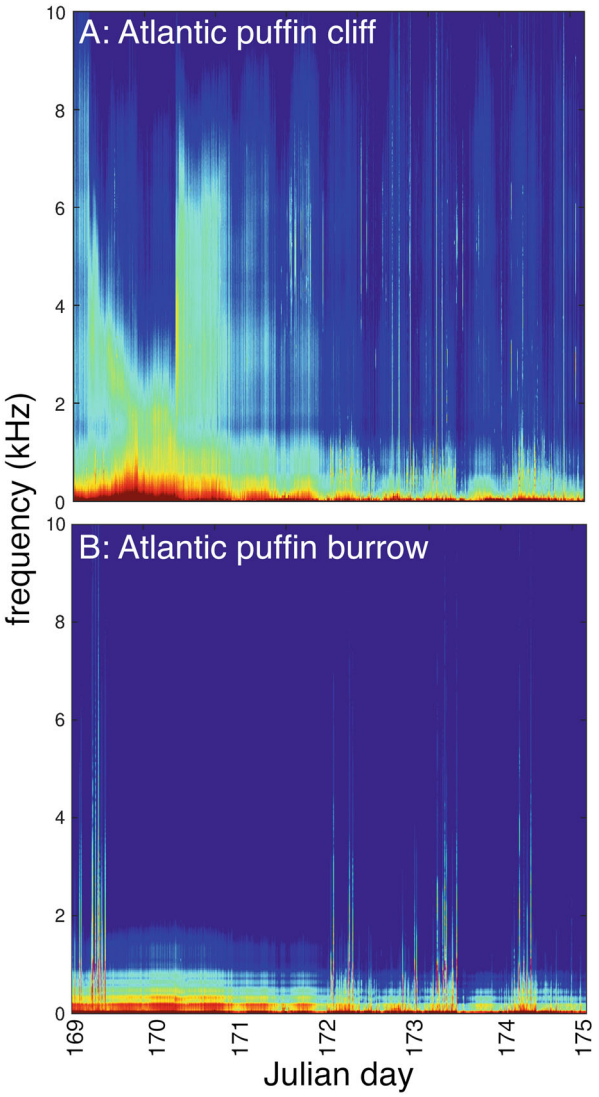



**Fig. 3** Long-term spectral average of an Atlantic puffin nesting soundscape calculated from 6 days of synchronized recordings at the mouth of a cliff burrow (**a**) and 1 m inside the burrow (**b**)

marbled murrelet (newly presented here) or related species like the thick-billed murre (*U. lomvia*) and Kittlitz's murrelet (*B. brevirostris*). In turn, it also foments potentially valuable insight generated through greater comparison. For example, recent anatomical analysis has suggested that the auditory anatomy of diving birds differs from those of fully terrestrial species, perhaps as an adaptation for baroprotection of the auditory system during diving (Zeyl et al. 2022). The audiograms of the deep-diving species studied here exhibit lowest auditory sensitivities that are similar to, or even better than, many fully terrestrial birds. This suggests that

middle-ear structural changes for baroprotection have not hindered aerial auditory reception (Smith et al. 2023) and advocates for additional comparative research on the auditory capabilities and anatomy of diving seabird species.

## Acoustic Overlap and Auditory Masking

The field-based measurement of AEP-derived audiograms shows that frequencies of sensitive hearing in these alcids spectrally overlap with a variety of common anthropogenic sounds. To varying degrees, this overlap includes sound sources in air, such as traffic noise, human speech, and airplanes flying overhead. Only a few studies have directly investigated impacts of in-air anthropogenic noise on seabirds, including the great cormorant (*Phalacrocorax carbo* (Buxton et al. 2017). Thus, with more sensitive aerial hearing than the cormorant (Larsen et al. 2020), these alcids may face similar disturbance at even lower noise level exposures. Notably, some naturally occurring sound levels within the "snapshot" soundscapes of the common murre and Atlantic puffin are akin to anthropogenic noise levels which can have deleterious effects on a variety of terrestrial bird species (circa 55–60 dBA; e.g., Injaian et al. 2018; McClure et al. 2013, 2017). Noise disturbance thresholds identified for terrestrial birds may therefore not be applicable to these species or other seabirds that similarly nest on sea cliffs or other areas exposed to consistent wave noise. This underscores the importance of incorporating species and niche diversity into avian noise-impact criteria, a need identified in other taxa (Popper et al. 2014). Behavioral sensitivity, sound source characteristics, and ambient noise conditions will almost certainly play a role as well, and the comparison of audiograms to environmental soundscape characteristics provides an initial way to consider and explore these factors.

Although acoustic communication in alcids has not been thoroughly studied, many species are known to be highly vocal. This suggests aerial acoustic cues are likely important for alcids, and the preservation of sensitive aerial hearing despite deep-diving in the three species studied here corroborates this notion. Thus, high levels of environmental background noise, including from anthrophonic sources, have the capability to mask potentially important acoustic cues. Comparison of acoustic environments and audiograms can suggest if anthropogenic sounds may fall above natural sound levels typically encountered by individuals, and thus inform what levels of acoustic masking species may naturally be equipped to deal with and provide context for possible levels of anthrophonic disturbance. For most birds studied to date, tonal signals must be around 18–38 dB above the background noise power spectral density (PSD) at the same frequency to be detected (the auditory critical ratio, e.g., Dooling 2002). Assuming similar critical ratios for alcids, Atlantic puffin individuals may experience significant levels of natural masking under the noisiest conditions of their cliffside nesting sites by virtue of their sensitive auditory thresholds (Fig. 4c and d). In contrast, common murres may experience only moderate masking of acoustic signals within their nesting soundscape (Fig. 4b), and nesting marbled murrelets likely encounter only limited masking of acoustic

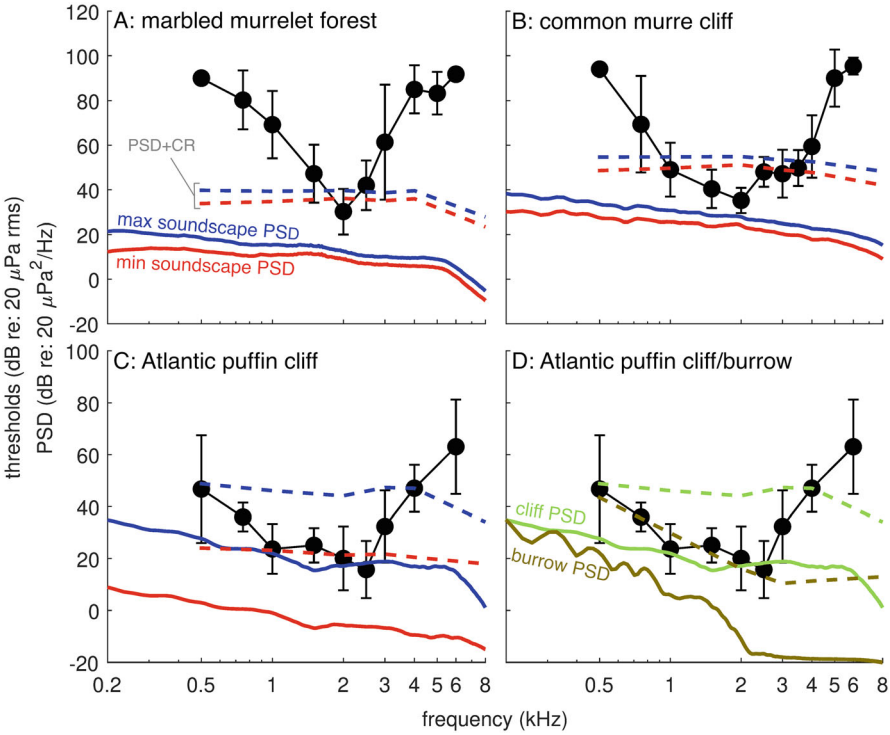



**Fig. 4** Aerial audiograms from each species (mean ± s.d., black markers) plotted with the maximum and minimum soundscape power spectral density (PSD) profiles at the species nesting site (solid red/blue lines). The dotted lines in denote estimated onset levels of auditory masking in the environment by adding mean avian critical ratios (CR) to the soundscape PSDs. In (D), the Atlantic puffin audiogram is compared to the concurrent PSDs from inside and outside the burrow

signals by the naturally occurring environmental sound levels (Fig. 4a). In general, this could suggest that anthropogenic sounds would need to be generally louder or closer to disturb the seacliff-nesting common murres and Atlantic puffins in comparison to the marbled murrelet, whose quite forest habitat and solitary-nests allow greater penetration of faraway anthropogenic noise sources (see Fig. 2a) and may result in lower thresholds for the onset of acoustic disturbance. A comparison of noise-induced disturbance relative to species and habitat type would help elucidate how sound sensitivities vary among these seabirds.

Notably, nesting soundscapes of the Atlantic puffin were also sometimes very quiet (see Fig. 2c), which might likewise increase their susceptibility to anthropogenic noise disturbance, even inside the relative safety of their burrows. Low frequencies, which dominate many anthropogenic sounds, overlap with the low-pass characteristics of the puffin burrow, and tend to attenuate less in a constant media and can propagate well through the ground. Thus, low-frequency anthropogenic noise could pose a particular problem for the burrowing Atlantic puffin when

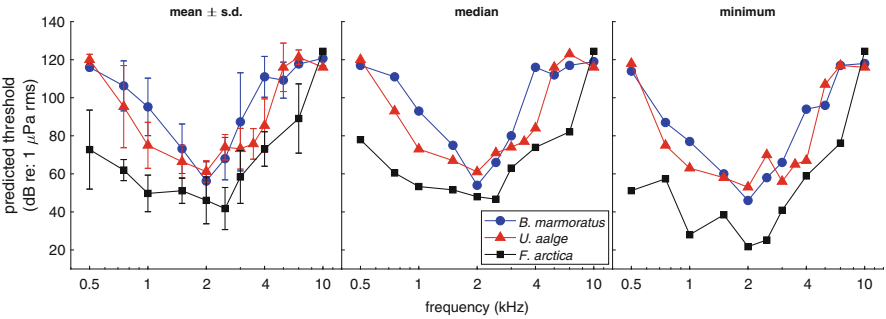



**Fig. 5** Preliminary estimates of pressure-equivalent underwater AEP audiograms compiled from the in-air audiograms presented in Fig. 1

combined with the relative sensitivity of the puffin auditory system compared to the other two alcid species presented here. Against this backdrop, additional research might investigate more specifically if the reduced noise of the burrow environment facilitates acoustic-mediated behaviors and if the relative acoustic shelter of the burrow influences their susceptibility to acoustic disturbance (Fig. 5).

## Considering Underwater Auditory Sensitivity and Use of Sound

Given these species' amphibious lifestyle, there is also a need to understand if, and how, their acoustic biology may be adapted for use underwater. Such information is a crucial for the implementation of effective underwater noise mitigation measures since diving seabirds also face potential exposure to marine anthropogenic noise (Mooney et al. 2019b). At least four species of seabird can conclusively detect underwater sound, including one alcid, the common murre (Hansen et al. 2020; Larsen et al. 2020; Therrien 2014; Sørensen et al. 2020). Of these, underwater auditory thresholds have been measured from only two: the long-tailed duck (*Clangula hyemalis,* Therrien 2014) and great cormorant (Larsen et al. 2020). Notably, the combined aerial and underwater AEP results from Larsen et al. indicated that mean hearing thresholds across all cormorant individuals were similar between the two media when compared in terms of the stimulus acoustic pressure. If this aerial-underwater threshold relationship is similar across other diving seabirds, the three alcids discussed here could be predicted to have lowest pressure-equivalent mean underwater thresholds between 42 and 61 dB re: 1 μPa at 2 kHz. While this prediction certainly represents an extrapolation, it suggests that deep-diving alcids may have sensitive hearing underwater and are therefore susceptible to impacts from underwater anthropogenic noise such as auditory injury or hearing loss. This supposition provides a hypothesis, but supporting it fully requires corroboration from robust and direct underwater threshold measurements from multiple alcid species in the future. Such data would support the needed development of appropriate underwater noise mitigation strategies.

The establishment of auditory thresholds also does not address the separate but relevant question of whether diving seabirds utilize underwater sound, even if it can be detected. Answering this separate question will provide important context for managers and regulators to assess the underwater noise impacts that are less acute than physical injury but still intrusive, such as auditory masking of potentially important acoustic signals. Thus, synergistic and interdisciplinary research projects are also needed to address what extent underwater sound is utilized by diving species.

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# Effect of Piling on Little Blue Penguins

# 87

Benjamin C. Lawrence, Leigh S. Bull, Stephen C. Arden, and Victoria E. Warren

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**Abstract**

Kororā (little blue penguin; *Eudyptula minor*) are a small penguin species native to New Zealand and often found nesting in man-made structures nearby ports and marinas. The species has a national conservation status of "At Risk – Declining" and is protected under New Zealand's Wildlife Act (1953). As such, development projects occurring in areas that kororā inhabit need to consider how this species may respond to construction noise and vibration. This chapter presents the noise assessment approach and on-site mitigation and management for the construction of a new marina on Auckland's Waiheke Island, which involved piling and other intensive construction activities close to kororā burrows. The project is still under construction, and monitoring of kororā will continue until its completion. The results of the monitoring to date show that kororā continue to inhabit and breed successfully in the breakwater on which the marina is being constructed, including during the noisiest phase of the works.

**Keywords**

Penguin · Noise effects · Behavioral response · New Zealand · Construction noise · Vibratory piling · Marina construction

## Introduction

### Effect of Noise on Birds

Birds can be adversely affected by sound, and anthropogenic noise is a key source of disturbance for species that live alongside human development. There are several ways in which birds (and animals in general) could be affected by noise, depending upon the loudness and character of the noise as well as the sensitivity of the species. At the closest point to an activity where the loudest noise levels would occur, there is a potential for noise exposure to cause physiological hearing damage in the receiving animal that can be permanent (i.e., permanent threshold shift (PTS)) or temporary (i.e., temporary threshold shift (TTS)). Further away, or when the sound is quieter, masking can occur, which is when the sound source is loud enough, and in the same time and frequency window, that it disguises other sounds that are biologically important (e.g., sounds from predators, prey, or other members of the same species). Often considered the least effect, sound has the potential to elicit a behavioral response when it can be distinguished from the ambient soundscape. Behavioral responses may be context-specific (animals may exhibit disturbance differently during their breeding season, for example), and some individual animals may be

more affected than others (e.g., mothers with young may respond differently to lone males) (Southall et al. 2021).

Behavioral responses vary greatly in severity, from large-scale obvious abandonment of a noisy habitat to fine-scale changes in foraging behavior or time spent in a behavioral state (Blickley et al. 2012; Wright et al. 2010). Behavioral responses are difficult to predict and can be challenging to quantify. While some minor behavioral responses may seem trivial (e.g., turning the head to look in the direction of a sound source), regular disturbance can lead to energetic consequences for wildlife. A constant state of alertness can lead to heart rates and stress levels that are higher than normal (DeRose-Wilson et al. 2015). The impact of stress can be acute or chronic. Chronic stress can lower immune response, making an animal more susceptible to disease or predation (Siegel 1980). Moreover, when energy is expended during a disturbance event, less energy is available for other activities, such as foraging or parental investment. Accordingly, individual disturbance events can lead to population-level consequences (Pirotta et al. 2018). If courtships and breeding are interrupted by disturbance, overall breeding productivity of a population can be affected.

## New Zealand Approach to Avoid Disturbance to Avifauna

New Zealand is currently home to 429 bird species, 179 of which have a national conservation status of "Threatened" or "At Risk" as of 2021 (Robertson et al. 2021). Many of these bird species inhabit areas that can be impacted by anthropogenic noise from human development. In general, the greatest noise risk from human development occurs during the construction phase when the highest noise emissions are produced. Construction projects must therefore be carefully managed to ensure that noise emissions do not exceed a level where there could be significant disturbance to protected species.

In New Zealand, the general approach to managing noise and vibrational disturbance to avifauna has been for consenting authorities to specify a setback distance, most often 50 m, from works. However, such an approach is not site or species specific, nor are scientific rationales provided for any specific setback distances.

## The Project

A new 180 berth marina is currently being constructed in Putiki Bay on the western side of Auckland's Waiheke Island. The marina is adjacent to the existing Kennedy Point ferry terminal and includes a fixed wharf, floating carpark, piers, pontoons, and finger jetties. These structures are all supported or secured by piles driven into the seabed and existing rock wall. A plan of the development is shown on Fig. 1.

It was discovered during the consenting and design phase of the project that kororā, little blue penguins (*Eudyptula minor*), may inhabit the rock structures around the proposed development. Kororā are protected under the New Zealand

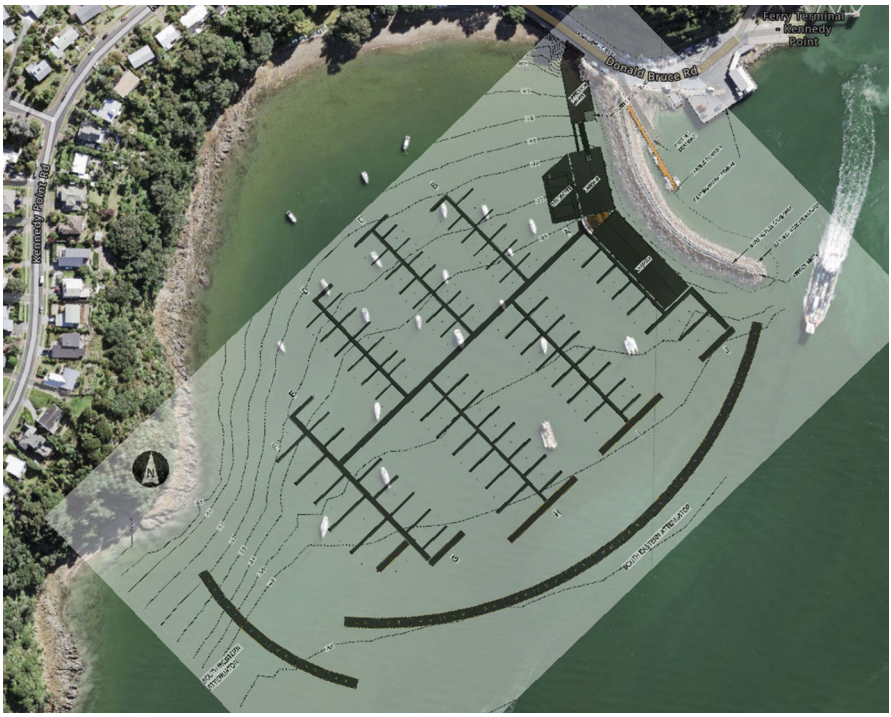



**Fig. 1** Map of the proposed marina

Wildlife Act (1953). Kororā surveys undertaken in October 2016 (DabChickNZ 2016), October 2017 (DabChickNZ 2017), and April 2021 (4Sight Consulting 2021a), using a certified conservation dog trained to detect penguins, provided data on the distribution of birds along the Kennedy Point breakwater prior to the marina construction (Fig. 2). Several detections were made close to the proposed piling works for the wharf structure.

The presence of kororā became a key factor in determining the noise management and construction methodology for the development. The management plan for the project involved setting an appropriate disturbance threshold based on available literature on kororā and similar species, and managing works on the site to keep within this limit.

## Kororā/Little Penguin

### Distribution, Habitat, and Behavior

Kororā are native to New Zealand and Australia. In New Zealand, they are widely distributed along the coastlines of the main and offshore islands (Heather and Robertson 2015; Marchant et al. 1990), with the national population estimated to

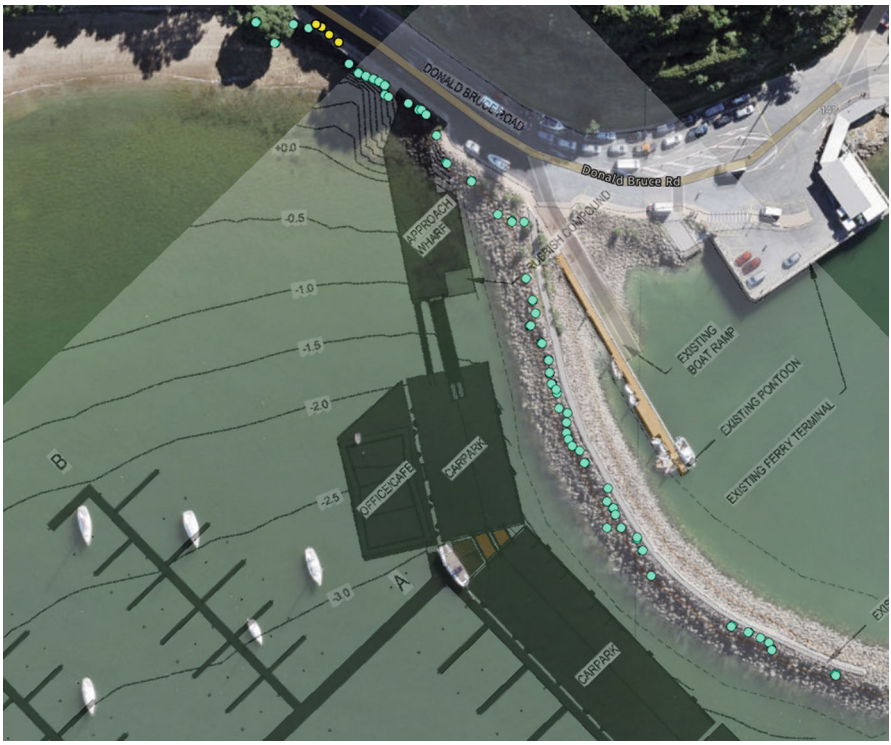



**Fig. 2** Kennedy Point Marina kororā locations (yellow circles indicate nest boxes, and blue circles indicate natural burrows)

be approximately 50,000–100,000 individuals (C. J. R. Robertson and Bell 1984; Taylor 2000). Weighing approximately 1 kg and standing 30–40 cm tall, the kororā is the smallest of all the 17 penguin species (Reilly 1994). Robertson et al. (2021) assigned an "At Risk – Declining" threat classification to kororā in New Zealand.

Kororā form loose colonies, with burrows located several meters apart (Braidwood 2009; Bull 2000a; Bullen 1997; Marker 2016). Colonies can range in size from a few to hundreds of pairs. Nests are generally situated close to the sea in burrows excavated by the birds or other species, or in caves, rock crevices, under logs, or in or under a variety of man-made structures including nest boxes, pipes, stacks of wood or timber, and buildings. Several studies have shown the ability for kororā to adapt their nesting behavior to urban environments; in response to the lack of "natural" habitats, birds successfully nest in rock crevices and human-made structures, such as breakwaters and even under rail tracks, that offer protection from the weather, tidal action, predators, and human disturbance (Bourne and Klomp 2003; Giling et al. 2008). Kororā exhibit high levels of site fidelity, generally returning to the same landing site and nest each breeding season (Bull 2000b; Pledger and Bullen 1998).

**Table 1** Indicative breeding cycle of kororā in the Hauraki Gulf

| | JAN | FEB | MAR | APR | MAY | JUN | JUL | AUG | SEP | OCT | NOV | DEC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Burrow occupation | | | | | | | | | | | | |
| Pair bond / nest building | | | | | | | | | | | | |
| Egg laying | | | | | | | | | | | | |
| Chick rearing / fledging | | | | | | | | | | | | |
| Moulting | | | | | | | | | | | | |

The yearly cycle of kororā comprises three overlapping stages: (1) occupation of burrows and pair formation; (2) breeding (eggs and chicks); and (3) molting (Table 1). Adults are present at colonies throughout the year, though numbers are lowest between completion of molt and start of breeding (Marchant et al. 1990). Molting birds remain on land for the entire molt period as they are unable to swim without getting waterlogged (Heather and Robertson 2015). During the nonbreeding and nonmolting season, birds are less frequently on land during the day, mostly returning to land at night. The breeding cycle of kororā in the Hauraki Gulf (Table 1) is based on Geurts (2006), van Rensburg (2010), McKenzie (2011), and Fitchett (2017).

## Response to Airborne Noise

As part of the effort to minimize the effects of noise on the kororā, a literature review was undertaken to ascertain their sensitivity to sound. There have been relatively few studies on the hearing abilities of penguins in general, and the research undertaken for this project was unable to uncover any specifics pertaining to kororā hearing. The applied noise threshold on site was therefore based on available literature on penguin or bird hearing in general, it being the best relevant information.

Hearing ability is determined by the frequencies that a species has the capability to hear, in addition to their sensitivity across those frequencies. The sensitivity curves of three Blackfooted penguins (*Spheniscus demersus*) from Wever et al. (1969) demonstrated that penguins are most sensitive to sounds between 600 Hz and 4000 Hz. These sensitivity curves are supported by the 1000–5500 Hz frequency range of in-air vocalizations, for kororā specifically (Nakagawa et al. 2001), as an animal's hearing is expected to be most sensitive at the frequencies produced by its own species. The frequency range of best sensitivity is relatively similar to the human range of 500–4000 Hz. However, birds are generally less sensitive than humans at the frequencies of best hearing, and their sensitivity falls off rapidly outside of this range (Halfwerk et al. 2018).

In air, continuous noise levels above 110 dBA SPL, or a single impulsive sound over 140 dB SPL, can lead to PTS in diving birds, while TTS can occur at continuous noise exposure levels above 90–95 dBA SPL (Dooling and Therrein 2012). Non-physiological effects such as masking and other behavioral responses are context specific and can occur at a range of noise levels. Penguins, including kororā, are vulnerable to direct masking of their sounds in air as they utilize auditory cues above water to identify other penguins.

Studies on behavioral response to noise for Fiordland penguins (tawaki, *Eudyptes pachyrhynchus*) (Fanning et al. 2020), Australian little blue penguins (Schaefer and Colombelli-Negrel 2021), and King penguins (*Aptenodytes patagonicus*) (Hughes et al. 2008) were reviewed for this project. The study from Hughes et al. (2008) on the King penguin was the only one that provided sound levels at which behavioral responses occurred. The study identified that penguins exhibited behavioral responses during aircraft overflights with maximum 1 s $L_{Aeq}$ levels ranging from 67 to 83 dB $L_{Aeq}$. The paper did not determine whether the reported behavioral responses were due to the sound levels produced by the aircraft or merely due to an object flying overhead. Moreover, the reported ambient noise levels were sometimes louder than the aircraft noise levels.

## A Species-Specific Approach to Minimize Disturbance from Anthropogenic Noise

The proposed marina construction activities were located within the typical required setback of 50 m from a kororā burrow and were scheduled to be carried out in August when breeding activity is generally occurring. The project's acoustic consultants, in collaboration with the project ecologist, were therefore asked to provide specific advice regarding the effect of the works on the local penguin population. Addressing noise effects on the behavior of wild species is difficult as very little is known about how free-ranging animals, and kororā in particular, respond to noise, and it is challenging to quantify behavioral responses. There is anecdotal evidence that kororā may be reasonably tolerant to noise and/or disturbance as many burrow locations are adjacent to ports, roads, and rail lines. However, this does not mean that they are unaffected by noise, and moreover, construction noise features a different character of sound to day-to-day activities.

Based on the best available evidence described above, the construction activities on site were not expected to cause PTS or TTS in the penguins, and a threshold of 80 dB $L_{Aeq(1s)}$ at the penguin burrows was recommended to minimize behavioral disturbance to the penguin population. The threshold incorporated an A-weighting (human auditory weighting) (Houser et al. 2017) to account for frequency-based hearing sensitivity of the kororā. As noted earlier, birds are generally sensitive over a narrower frequency range than humans, so the human A-weighting is considered to be a conservative assumption. Furthermore, it is convenient to measure using the A-weighting as it is widely used and is the default weighting for most sound-level meters. The 80 dB $L_{Aeq(1s)}$ threshold was accepted and certified by Auckland Council.

This chapter presents validation measurements made on site around and within the penguin burrows, which were undertaken to check the feasibility of adhering to the threshold. The findings are supported by results from ecological surveys throughout the construction works to investigate the effectiveness of the established threshold. The focus is on airborne noise received from construction activities adjacent to the identified kororā burrows in the rock wall and breakwater. Specifically, it

addresses the issue of undertaking noisy works during ecologically sensitive periods (i.e., breeding season) while the penguins are in their burrows.

## Method

### Acoustic Monitoring Methodology

A Bruel and Kjaer 2250 sound-level meter (https://www.bksv.com/en/instruments/handheld/sound-level-meters/2250-series) was used to measure noise levels from the piling on-site. The sound level meter was calibrated before and after each noise survey. The monitoring was carried out in air at 1 m above the rock face, and generally at the closest point along mean high water springs (MHWS) to the piling. Noise levels were monitored for a representative duration to capture each construction activity. Acoustic measurements were undertaken in August, October, November, and December 2021. The measurement durations are presented in detail in the results section of this chapter.

A map of the piling area and acoustic measurement locations on site is shown on Fig. 3. The acoustic measurement sites correspond to the results presented in Table 2.

The purpose of the noise monitoring was to assess the construction works against the 80 dB $L_{Aeq(1s)}$ A-weighted threshold. When an exceedance was identified, the construction methodology was reviewed to identify lower noise methods and to implement additional mitigation measures. Further monitoring was then carried out to confirm that the revised methods were sufficient to avoid exceeding the threshold.

Additional measurements were carried out on-site in April 2022 inside a representative kororā burrow to determine the noise reduction provided by the rock wall of the breakwater. One Wildlife Acoustics Song Meter Mini recorder (https://www.wildlifeacoustics.com/products/song-meter-mini) was installed inside "Burrow 13," after it was confirmed that no kororā were present inside, and a second unit was attached to a nearby tree branch approximately 1 m above the rocks. Burrow 13 is depicted in Fig. 6c. Both devices were calibrated by recording a tone from a calibrator before and after the survey, which was used to adjust the processed data to the correct level. The monitors were left on site from 1 to 6 April 2022 and recorded continuous audio between 7 am and 5 pm each day. The data was analyzed based on construction timings provided by the project team and by listening to the recordings to identify noise sources. The purpose of this monitoring was to compare the noise levels inside the burrows to the corresponding free-field levels at 1 m from the burrow entrance.

### Penguin-Monitoring Methodology

In order to determine the effectiveness of noise management efforts, as well as other disturbance reduction efforts, the kororā on-site were monitored by qualified and permitted ecologists.




**Fig. 3** Map of acoustic measurement locations (pink), and pile locations (red)

The ability to monitor kororā activity within a burrow is in part dependent on the ability to access the "chamber," which can be located at the end of a tunnel (in some cases >1 m long). Furthermore, due to the nature of the substrate in which a burrow is located, the tunnel is not always straight and therefore a direct line of sight into the chamber is not always possible. Thus, due to the difficulties in being able to observe kororā behavior directly, the objective of the kororā monitoring was to confirm the ongoing habitation of the breakwater by kororā during the construction phase of the marina.

Monthly monitoring of kororā activity has been undertaken since July 2021, except for August 2021 due to New Zealand Covid Alert Level restrictions. At the beginning (July 2021), 40 locations were monitored along the Kennedy Point breakwater. However, the number of monitoring sites increased over time due to the detection of new locations. The locations are shown on the map in Fig. 2. Burrows were numbered by the ecology team so activity within each burrow could be tracked over time. During each monitoring session, the locations of all previous kororā detections were inspected first by way of a visual check for the presence of any penguins or their sign (e.g., feathers or guano/excrement), and then using a burrowscope and/or video camera on a mobile device if possible. The following information was recorded at each monitoring location: the presence of any sign (feathers or guano), birds, or nest contents (eggs and / or chicks).

**Table 2** Noise-producing activities on-site and results from acoustic measurements, as measured from the breakwater

| Activity | Pile # and distance from measurement location | Measurement position (see Fig. 3) | Highest 1 second level (dB $L_{Aeq(1s)}$) | Cumulative duration and number of events |
|---|---|---|---|---|
| **Survey 1, 5 August 2021** | | | | |
| Vibro piling and excavator drill rig engine (pile free standing) | Test pile, 12 m | MP1 | 79 | 1 minute, 1 event |
| Vibro piling only (pile free standing) | Test pile, 12 m | MP1 | 79 | 1 minute, 1 event |
| Drilling (including mechanical component banging) | Pile 15, 12 m | MP2 | 88–95 | 11 minutes, 2 events |
| Drilling with banging avoided through careful operation | Pile 15, 12 m | MP2 | 79 | 3 minutes, 1 event |
| **Survey 2, 4 October 2021** | | | | |
| Vibro piling (pile touching pile gate) | Pile 18, 25 m | MP3 | 84–89 | 15 minutes, 6 events |
| Vibro piling (not touching gate) | Pile 18, 25 m | MP3 | 74–75 | 1 minute, 2 events |
| Vibro piling (pile touching pile gate) | Pile 16, 16 m | MP3 | 85 | 3 minutes, 1 event |
| Vibro piling (not touching gate) | Pile 16, 16 m | MP3 | 77–79 | 3 minutes, 2 events |
| Drilling with banging avoided through careful operation | Pile 18, 25 m | MP3 | 75–76 | 11 minutes, 2 events |
| **Survey 3 (17 November 2021)** | | | | |
| Vibro piling (no pile gate) | Pile 20, 21 m | MP2 | 77 | 5 minutes, 1 event |
| **Survey 4, 10 December 2021** | | | | |
| Vibro piling (gate mostly removed, slight touch during measurement) | Pile 19, 12 m | MP4 | 81 | 5 minutes, 1 event |
| Drilling | Pile 21, 20 m | MP4 | 79 | 4 minutes, 1 event |

## Results

### Noise Levels Outside the Burrow, per Activity

Noise levels were measured for the two main noise-producing construction activities on-site: vibratory piling and drilling. The measured levels are shown in Table 2, and the results for each activity are discussed in the following sections.

## Noise Measurements: Vibro Piling

A vibratory hammer suspended from a crane was used to drive the piles which were contained within pile gates. The activity consisted of 30 s to 5 min long periods of vibro driving, with quiet periods in between while the alignment of the pile was checked. Noise from the vibro hammer itself was primarily low-frequency noise but included a noticeable high-frequency rattle. The piling noise was relatively constant with levels between 74 and 79 dB $L_{Aeq(1s)}$. However, spikes in noise level of up to 89 dB $L_{Aeq(1s)}$ were caused when the steel pile came into contact with the steel pile gate as shown on Fig. 4. This issue was addressed by increasing the clearance between the gate and pile, and by careful control from the crane operator. These measures significantly reduced the problematic noise to a maximum of 81 dB $L_{Aeq(1sec)}$.

## Noise Measurements: Drilling

Drilling was carried out by a large excavator with a drill attachment and auger. The main noise from the activity was a high-frequency whine of the drill attachment turning. The low-frequency rumble from the excavator engine was audible but at a much lower level. It is common for drill rig operators to shake or counter-rotate the drill attachment to remove the spoil into the bin, but in this case a worker was using a shovel to remove spoil stuck to the auger to minimize noise emissions.

In general, the drilling was relatively quiet (75–79 dB $L_{Aeq(1s)}$ at 12 m, see Table 2). However, a mechanical component occasionally slipped during drilling resulting in loud bangs of up to 20 decibels above typical drilling (88–95 dB $L_{Aeq(1sec)}$ at 12 m, see Table 2). The banging was mitigated by training the rig operator in quiet operation techniques. Once the issue was resolved, the drilling was below the 80 dB $L_{Aeq(1sec)}$ threshold.



**Fig. 4** Photo of pile gate

## Other Noise-Producing Activities

It is important to note that other activities such as tracking of the excavator and moving/placing materials with a crane would have likely generated intermittent high noise events. This is common on most construction projects involving heavy machinery. While these intermittent events can be minimized through operator care, they are difficult to anticipate, predict, and mitigate and therefore cannot be avoided completely.

## Noise Reduction from Outside to Inside a Burrow

The data from the two Wildlife Acoustics Song Meter Mini sound level meters recorded three separate occurrences of vibro piling associated with the marina construction. The measured noise levels inside and outside the burrow are shown in Table 3 and illustrated in Fig. 5. The levels shown are an average across each vibro piling period.

The results show that the burrow provided an overall noise reduction of 8 dB, with the biggest reduction being at frequencies above 8 kHz. There was an amplification of 2–10 decibels at low frequencies (40–100 Hz). However, as identified in the literature review, kororā hearing is likely to be most sensitive in the range of 600–4000 Hz or 5000 Hz as per other penguin species, which is outside of the frequencies where the amplification occurs. Noise reduction provided by the breakwater rock wall is likely to vary depending on factors such as the position/orientation, material, and depth of the burrows but is expected to be at least 5 decibels for burrows in rock walls. Accordingly, kororā inside the burrows were not exposed to sound levels exceeding the 80 dB $L_{Aeq(1sec)}$ threshold once appropriate mitigation and methods were in place on site.

## Korora Activity

The monitoring undertaken to date has confirmed that the pattern of kororā activity recorded over a year at Kennedy Point is consistent with that of the typical annual cycle, including egg laying, incubation, chick rearing, and molting (Table 4). The birds have continued to inhabit the breakwater, and birds have been recorded present in their burrows during the day throughout the breeding and molting periods, as well as during the nonbreeding and nonmolting period (Fig. 6). These observations confirmed that the noise levels experienced by the birds during the piling activities at Kennedy Point did not stop their normal patterns of behavior.

Furthermore, comparison with kororā activity recorded in previous surveys during the same months showed that there was in fact increased kororā activity during the construction period. The numbers of kororā sign at the monitoring locations and active

**Table 3** Comparison of noise levels from vibro piling inside Burrow 13 and near entrance of Burrow 13

| Location | Overall level (dB LAeq) | 1/3 octave band levels (dB Leq) | | | | | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | 25 | 31.5 | 40 | 50 | 63 | 80 | 100 | 125 | 160 | 200 | 315 | 500 | 630 | 800 | 1k | 1.25k | 1.6k | 2k | 2.5k | 3.15k | 4k | 6.3k | 8k | 10k |
| Outside burrow | 82 | 41 | 57 | 63 | 56 | 63 | 71 | 76 | 70 | 73 | 74 | 70 | 69 | 67 | 67 | 68 | 68 | 74 | 73 | 66 | 70 | 67 | 64 | 69 | 63 |
| Inside burrow | 74 | 42 | 58 | 65 | 60 | 71 | 80 | 76 | 74 | 72 | 67 | 64 | 63 | 65 | 66 | 67 | 62 | 62 | 61 | 56 | 56 | 53 | 52 | 51 | 51 |
| Difference from outside to inside | 8 | 1 | 1 | 2 | 5 | 8 | 9 | 0 | 4 | -1 | -8 | -6 | -6 | -2 | -1 | -1 | -7 | -12 | -12 | -11 | -14 | -15 | -12 | -18 | -12 |

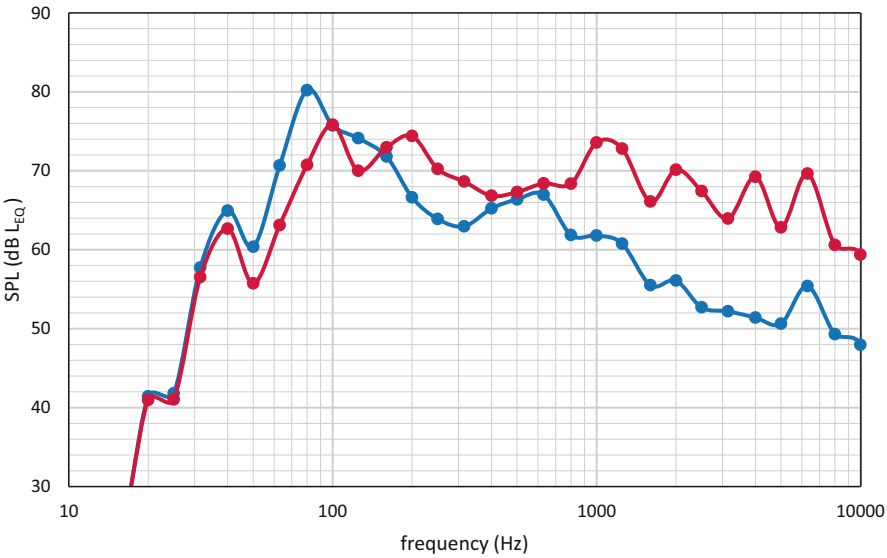



**Fig. 5** Comparison of noise levels from vibro piling inside Burrow 13 (blue) and near entrance of Burrow 13 (red)

burrows recorded over the course of the monitoring sessions to date are provided in Figs. 7 and 8, respectively. To date, there are 5 calendar months (July, September, October, November, and December) in which monitoring sessions have been conducted in both 2021 and 2022, enabling comparison. While it is important to note that there will likely be variation in the timing of annual activity due to external factors (e.g., sea-surface temperatures and food supply), the data does indicate that the numbers of kororā sign and active burrows recorded were in general higher in 2022 months than 2021. Most importantly, increased numbers of sign and active burrows were recorded during the 2022 breeding months (September through November) postbreakwater piling compared to the same months in 2021 (Figs. 7 and 8).

**Table 4** Kennedy Point kororā monitoring results

| Monitoring session | No. of burrows monitored | Korora sign | Burrow activity | Comparison with preconstruction surveys in the same months |
|---|---|---|---|---|
| July 2021 | 40 | 21 | Burrow 10 – adult incubating egg<br>Burrow 13 – adult incubating egg | July survey not previously conducted |
| September 2021 | 40 | 17 | Burrow 10 – adult with chick | September survey not previously conducted |
| October 2021 | 26 *(Full monitoring session could not be completed due to interference by members of the public)* | 11 | Burrow 5 – two adults incubating egg<br>Burrow 10 – chick<br>Burrow 13 – adult incubating egg | The October 2016 and 2017 surveys reported potentially five and nine kororā burrows, respectively, based on positive indications by the dog, although no birds were observed, nor was breeding confirmed (DabChickNZ 2016, 2017). In comparison, the October 2021 results reported 11 locations where kororā sign was observed as well as breeding activity at three burrows |
| November 2021 | 40 | 14 | Burrow 5 – adult with chick<br>Burrow 10 – adult with chick | A kororā survey previously conducted along the Kennedy Point Breakwater in November 2020 (4Sight Consulting 2020) recorded some evidence (including small patches of guano and a feather) of potential penguin use but no active burrows. In comparison, the November 2021 results confirmed the presence of two |

(continued)

**Table 4** (continued)

| Monitoring session | No. of burrows monitored | Korora sign | Burrow activity | Comparison with preconstruction surveys in the same months |
|---|---|---|---|---|
| | | | | active burrows with chicks |
| December 2021 | 40 | 22 | Burrow 5 – chick fledged<br>Burrow 10 – chick fledged | December survey not previously conducted |
| January 2022 | 40 | 17 | Six burrows with molting birds | January survey not previously conducted |
| February 2022 | 40 | 28 | Four burrows with molting birds | A survey in February 2021 (4Sight Consulting 2021b) reported 18 locations where kororā sign was observed |
| March 2022 | 41 | 6 | No birds observed<br>No active burrows | March survey not previously conducted |
| April 2022 | 40 | 25 | Six burrows contained birds<br>No active burrows | April 2021 survey using a conservation dog identified 28 locations where positive detections were made by the dog (4Sight Consulting 2021a). This is similar to the 25 locations identified during April 2022 which was based on human observations only |
| May 2022 | 42 | 16 | Burrow 1 contained a pair<br>No active burrows | May survey not previously conducted |
| June 2022 | 48 | 24 | Birds present in five burrows, including two which had pairs<br>No active burrows | June survey not previously conducted |
| July 2022 | 49 | 12 | No active burrows | Less sign observed than in July 2022 |
| August 2022 | 52 | 22 | Four burrows containing adult birds<br>Two active burrows | August survey not previously conducted |

(continued)

**Table 4** (continued)

| Monitoring session | No. of burrows monitored | Korora sign | Burrow activity | Comparison with preconstruction surveys in the same months |
|---|---|---|---|---|
| | | | (12 and 23) with adults incubating eggs<br>Two new burrow locations (50 and 51) identified on the breakwater immediately adjacent to the new piles | |
| September 2022 | 55 | 33 | Six active burrows (53, 51, 10, 13, 23, and 26) with adults incubating eggs<br>Burrows 10 and 51 immediately adjacent to new piles on breakwater | Increased number of sign and active burrows compared to September 2021 |
| October 2022 | 55 | 29 | Four burrows (10, 13, 23, and 26) were active, containing or suspected to contain chicks | Similar number of active burrows compared to October 2021. Burrows 10 and 13 were active in both years |
| November 2022 | 55 | 25 | Seven burrows (10, 18, 20, 21, 22, 23, and 26) were considered active, containing or suspected to contain chick | Increased number of sign and active burrows compared to November 2021 |
| December 2022 | 55 | 25 | Five burrows (20, 21, 23, 25, and 26) were considered active | Increased number of sign and active burrows compared to December 2021 |

## Discussion

Construction noise is a key source of anthropogenic noise for wild species that live close to human development. In New Zealand, consenting authorities commonly specify a generic distance (often 50 m) that works must be separated from the




**Fig. 6** (**a**) Adult kororā with chick in Burrow 10 (24 September 2021, breeding/molting period); (**b**) adult kororā in Burrow 10 (12 April 2022, nonbreeding/nonmolting period); and c) pink dot on rock (right) indicated location of Burrow 13 (containing bird) relative to the construction works area (12 April 2022, nonbreeding/nonmolting period). *Top two photographs taken by 4Sight Consulting Ltd*

location of protected species in order to minimize the effects of noise disturbance or hearing injury. However, these generic setback distances are not context, site, or species specific and are not based on scientific evidence.

The aim of this work was to minimize the effect of nearby construction noise on nesting kororā during an ecologically sensitive period. The study measured the sound levels on site, assessed their adherence to an ecologically informed noise threshold, and considered the results from ecological surveys throughout the construction works to investigate the effectiveness of the established threshold.

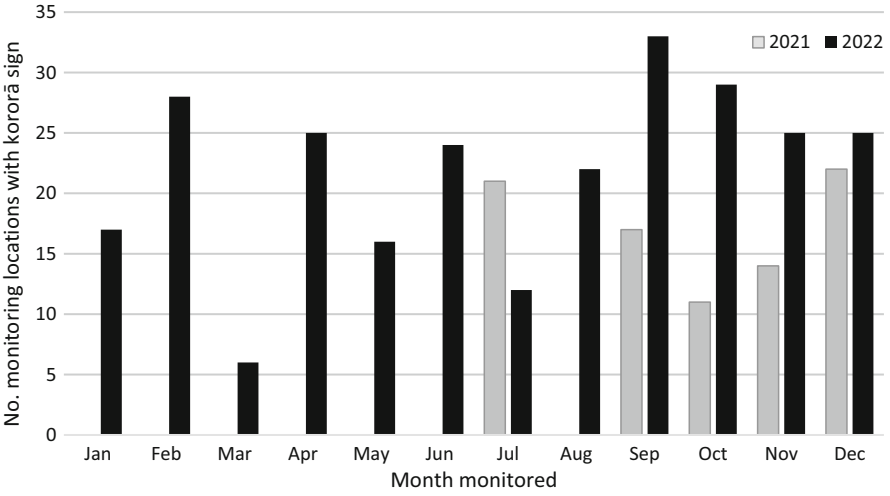



**Fig. 7** Number of monitoring locations at which kororā sign was observed during each monitoring session. Monitoring sessions began in July 2021; no monitoring was undertaken in August 2021 due to Covid-19 restrictions; and October 2021 monitoring session was incomplete due to public harassment

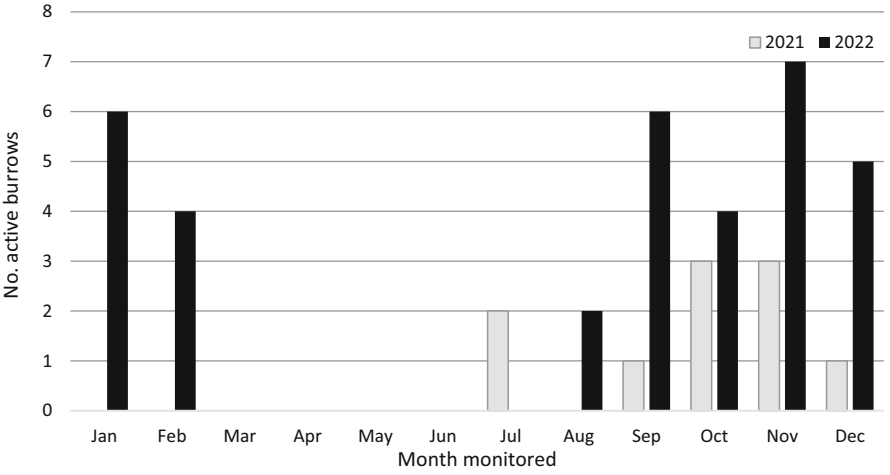



**Fig. 8** Number of active burrows recorded during each monitoring session. Monitoring sessions began in July 2021; no monitoring was undertaken in August 2021 due to Covid-19 restrictions; and October 2021 monitoring session was incomplete due to public harassment

## Feasibility of an 80 dB $L_{Aeq(1s)}$ Threshold for Construction Works

Based on verification measurements, the construction works associated with vibro piling and drilling were generally below the 80 dB $L_{Aeq(1s)}$ threshold at the location of the kororā burrows. Short periods of higher noise due to mechanical banging on

the drill rig and the pile touching the pile gate during vibro piling were resolved by providing additional training to the operators. At a representative burrow, which was known to support breeding activity, the rock wall of the breakwater was measured to provide an additional 8 dBA reduction in noise level from outside to inside, so the construction noise levels received by the kororā inside the burrows on site would have been lower than the in-air reported levels.

While the outcome of this project is satisfactory in that the noise emissions were generally able to be kept below the 80 dB $L_{Aeq(1s)}$ threshold and the kororā remained in the burrows, quieter piling methodologies, such as screw piling, are still preferable over the vibro piling and drilling methodologies and should be used where practicable. Impact piling was not used, but it is noted that this activity can produce significantly higher noise and vibration levels than the methodology used for this project and should therefore be avoided close to kororā burrows.

### Kororā Responses to Construction Activities

Kororā exhibit considerable variability in regard to reproductive success rates. A number of factors have been shown to affect kororā reproductive success, including breeding habitat (Braidwood et al. 2011; Bull 2000a; Colombelli-Négrel 2015; Perriman and Steen 2000), age and experience of birds (Nisbet and Dann 2009; Ramírez et al. 2021; Robinson et al. 2005), disturbance (Vertigan et al. 2012), sea surface temperature, and food availability (Cullen et al. 2009; Dann et al. 2000; Geurts 2006; Kowalczyk et al. 2014; Perriman et al. 2000). These factors can make it difficult to identify the specific cause of reproductive failures. For these reasons, changes to the habitation of the Kennedy Point breakwater by kororā were determined to be the only variable that could be measured in a meaningful way.

Kororā are confined to land during the day during the egg incubation phase, chick guard period, and molting period of their annual cycle; at all other times, kororā are not confined to land and can spend the day at sea. During the sensitive land-based phases, kororā were still recorded on land in their burrows while piling and construction works were occurring. Furthermore, adults in burrows (5, 10, and 13) located closest to the works did not abandon their clutch but in fact successfully fledged chicks, and comparison with kororā activity recorded in previous surveys during the same months showed that there was actually increased kororā activity during the 2021/22 construction period.

Overall, based on the kororā activity data collected, it appears that the birds did not alter their general patterns of behavior, and were able to successfully raise clutches and molt during and after the marina piling and construction works.

## Further Research

This chapter has focused on airborne noise effects on kororā in burrows. However, underwater noise and vibration also have the potential to affect kororā:

- Underwater noise can affect kororā in the same way as airborne noise (physiological effects, masking, and behavioral effects). While the vibro piling and drilling for this project were measured to have relatively low underwater noise levels, other noisier methodologies, such as impact piling, may affect kororā behavior underwater, even at relatively large distances.
- Vibration has the potential to affect kororā on land. Drilling produces negligible vibration, but vibro piling can result in relatively high vibration levels at close distances, depending on the ground conditions. No vibration measurements were carried out for this project as airborne noise was the key concern, and furthermore there is little available published literature on vibration effects on penguins to determine a suitable threshold. Nonetheless, this is an area that needs further research.

There is also limited data on the ambient noise and vibration levels that kororā are exposed to outside of construction settings. Anecdotal evidence suggests that the species is tolerant of high levels of anthropogenic noise based on the burrow locations of some free-ranging penguins (e.g., nearby busy roads, port areas, or rail lines), but ambient acoustic data has not been obtained to quantify the noise levels that these animals choose to inhabit. Teasing out the relationship between preferred foraging and nesting sites from their overlap with anthropogenic noise is likely to be extremely difficult; some individuals may persist in areas of high noise due to productive feeding opportunities or preferred nesting areas, for example. Further research into real-world received levels would help to set appropriate thresholds for minimizing disturbance in areas affected by anthropogenic noise.

## Conclusion

Although the results of this study are positive as the kororā continued to inhabit the rock wall of the marina and successfully fledged chicks, this project does not provide certainty that the 80 dB $L_{Aeq(1s)}$ threshold is appropriate for all kororā habitats. Nonetheless, it does provide evidence that this may be a suitable threshold for environments influenced by anthropogenic noise, and that intensive construction works can generally be constrained to meet the threshold. The results of this study provide a foundation for minimizing noise effects on wild species, and highlight the benefits of collaboration between technical experts and industry.

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# Section XII

# Noise Effects: Invertebrates and Plants

# Impacts of Substrate-Borne Vibrations from Pile Driving in a Benthic Marine Invertebrate

88

Youenn Jézéquel, Seth F. Cones, Frants Havmand Jensen, Hannah Brewer, John Collins, and T. Aran Mooney

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**Abstract**

Many anthropogenic activities in the ocean involve direct contact with the seabed. These benthic interactions can potentially be harmful to marine fauna. For example, pile driving is commonly used for coastal and offshore

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constructions, including wind farms that overlap with vital marine habitats. The sound from pile driving is a major pollutant of international concern, although studies on substrate-borne vibrations arising from these activities are scarce, particularly those which address impacts to key benthic invertebrates. In a field-based study, the impacts of substrate-borne vibrations from pile driving were quantified on the giant scallop (*Placopecten magellanicus*), an ecologically and commercially important bivalve. Behavioral responses of tagged scallops to pile driving were assessed at two spatial scales (< 10 and 50 m from the source). Pile driving induced valve closures in scallops at substrate-borne vibration levels similar to those measured hundreds of meters from wind turbine constructions. These responses could have long-term energetic consequences on scallops, rending them more vulnerable to predation. The results suggest offshore wind farm constructions have the potential to impact scallop populations.

**Keywords**

Climate change · Offshore wind farm · Anthropogenic sound · Pollution · Bivalve

## Introduction

Offshore wind farms (OSW) are proliferating dramatically worldwide, often overlapping with important ecological habitats and fisheries, consequently raising concerns about their potential impacts on marine species. Pile driving (PD) used to build OSW generates high-intensity impulsive sounds that have various impacts on marine fauna (Mooney et al. 2020). However, the impacts of associated substrate-borne vibrations on benthic invertebrates from PD are not known (Popper et al. 2022).

Bivalves can detect substrate-borne vibrations through their abdominal sense organs and statocysts (Roberts and Elliot 2017). A previous study showed mussels and scallops respond to low-frequency vibrations (i.e., < 1 kHz) in tanks by closing their valves (Roberts et al. 2015; Jézéquel et al. 2023), which is a common anti-predatory response with likely energetic consequences (Robson et al. 2012). However, there are needs now to scale up these experimental studies into the field to better understand the impacts from OSW constructions.

Bivalves are both a key taxonomic group for many marine ecosystems and represent a commercially important taxon (Newell 2004; Wijsman et al. 2019). Given the rapid proliferation of planned OSW projects overlapping in vital bivalve habitats and fishing zones, it is necessary to quantify the construction impacts on these species. The purpose of this field-based study was to measure the behavioral responses of the giant scallop (*Placopecten magellanicus*) to real-time PD using biologging tags.

## Materials and Methods

### Data Collection

This study was performed using 14 scallops (8.2 ± 0.5 cm) that were acclimatized in tanks at the facilities of the Woods Hole Oceanographic Institution (WHOI). Before the PD experiment started, scallops were tagged with Axy 5 XS bio-loggers (Technosmart Europe Srl, www.technosmart.eu) using the same technic described in Jézéquel et al. (2022). Basically, these tags measured the valve angle of bivalves using the strength of the magnetic field (in mV; sampling frequency = 2 Hz) between a magnetometer placed on the dorsal valve and a magnet located on the ventral valve. In addition, this tag contains an accelerometer that was used to detect the substrate-borne vibrations (sampling frequency = 25 Hz) arising from PD exposure. Substrate-borne vibrations were also monitored using a calibrated geophone (model GS-11D from Geospace®; sampling frequency = 200 Hz) placed at 8 and 50 m from the PD. Overall, these data were recorded for a period of 2 weeks of PD exposure.

### Experimental Procedure

Pile driving exposures occurred from the 14th to 29th of September 2021 in a shallow water area (41.52° N, 70.67° W; Fig. 1A). A cylindrical steel monopile (length, 10.0 m; diameter, 0.3 m; thickness, 0.02 m) was hammered into the seabed each day using an impact hammer (weight, 1500 kg; Fig. 1B) at a rate of 10 strikes per min, generating 16 kJ per strike. Tagged scallops were placed in cages located at a near (< 10 m) and far (50 m) site from the PD (Fig. 1A). Scallops were exposed daily to 2-hour-long PD events.

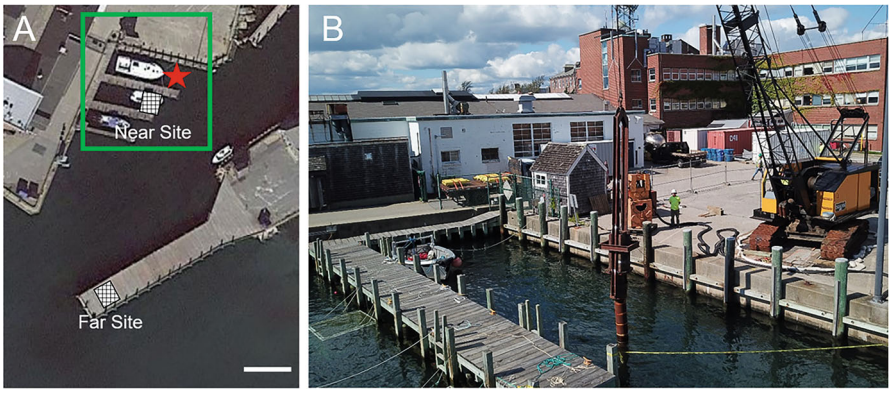



**Fig. 1** A) Top view of the experiment site at Woods Hole harbor (Massachusetts, US). Near site is located within 10 m from PD while far site is at 50 m. White bar is 10 m and red star denotes PD location. B) Picture of the PD setup at the green square shown in A

## Statistical Analyses

Linear mixed-effects analyses were conducted between scallop valve angle and different fixed effects. Individual and date were used as random effects, while fixed effects included exposure type (pre-exposure vs. exposure). Pairwise tests were also performed using post hoc Tukey tests.

## Results

Substrate-borne vibrations recorded without PD were low at both distances (Fig. 2A, D). The PD generated transient substrate-borne vibrations with calculated peak-to-peak levels ranging from 87.08 ± 2.88 to 109.95 ± 1.25 dB re 1 µm.s$^{-2}$ at 50 and 8 m, respectively (Fig. 2B, E). Single strike exposure levels were 94.39 ± 1.34 dB re (1 µm·s$^{-2}$)$^2$.s at 8 m and 72.48 ± 2.51 dB re (1 µm·s$^{-2}$)$^2$.s at 50 m. The spectra peaked at low frequencies around 10 Hz (Fig. 2C, F).

Scallops located within 10 m from the PD showed valve closures in synchronicity with PD substrate-borne vibrations (Fig. 3). This resulted in a significant diminution of 30% in valve angle during PD exposure compared to pre-exposure (Tukey test, $p < 0.001$, Fig. 4). Interestingly, no reactions were found in scallops placed at 50 m, and there were no significant differences in valve angles between pre- and PD exposure (Tukey test, $p = 0.864$, Fig. 4). These results show that scallops were not behaviorally affected by substrate-borne vibrations at 50 m from the PD.

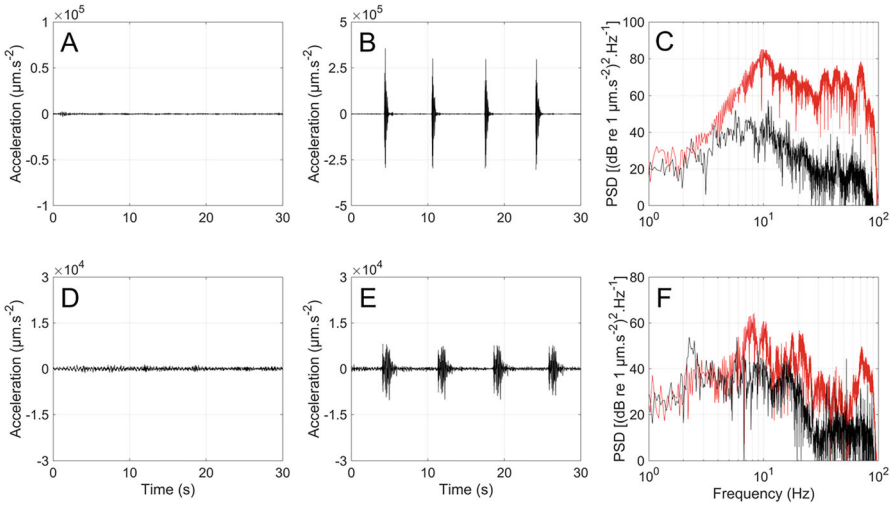



**Fig. 2** Example of recorded substrate-borne vibrations recorded by the geophone (x-axis) at 8 m (top) and 50 m (bottom) without (A and D) and during PD (B and E). Power spectral densities (PSD) of the temporal signals are presented in C and F (red: PD; black: ambient)

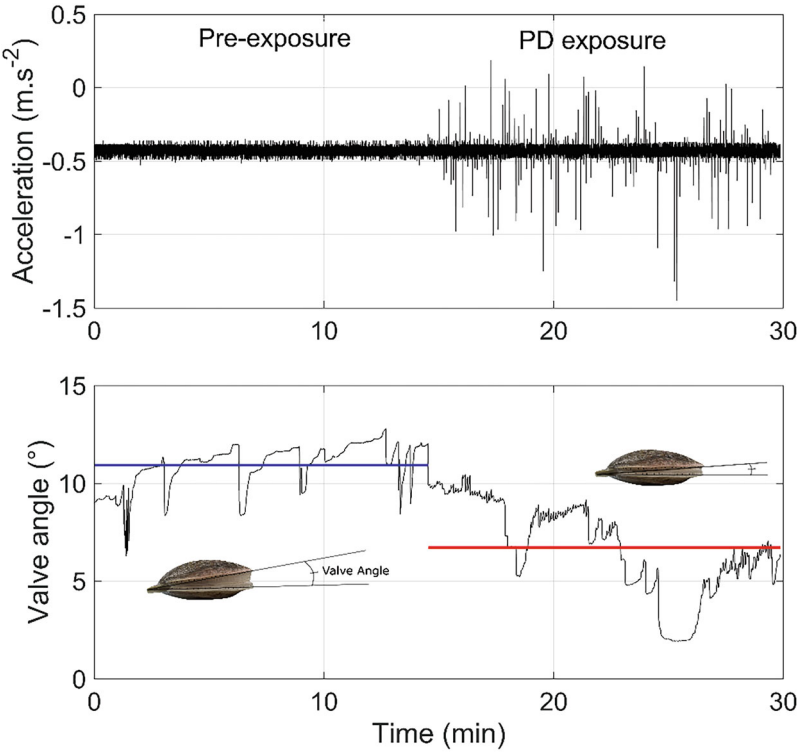



**Fig. 3** Synchronized data from one tagged scallop located in the vicinity of the PD (< 10 m), with the accelerometer (top) and magnetometer (bottom). Substrate-borne vibrations from the PD (vertical bars) were detected by the scallop, which responded by closing its valves. Horizontal bars highlight mean valve angle measured before (blue) and during (red) pile driving

## Discussion

This field-based study showed that substrate-borne vibrations arising from PD impacted scallop behaviors. While strong valve closures were found in the vicinity (< 10 m) of the PD, scallops showed no behavioral responses at 50 m despite PD substrate-borne vibrations were still detected. These data raise concerns about the potential impacts from OSW constructions on scallops.

Scallops exposed to PD responded to substrate-borne vibrations by closing their valves (Fig. 3), which is a common antipredatory response (Thomas and Gruffydd 1971). Similar results were found by Day et al. (2017) with scallops (*P. fumatus*) exposed to seismic air gun. The energy cost and ecological consequences associated with these behaviors are not known. Robson et al. (2012) showed that scallop valve closures result in large expenditures of energy, and an oxygen debt has to be paid off

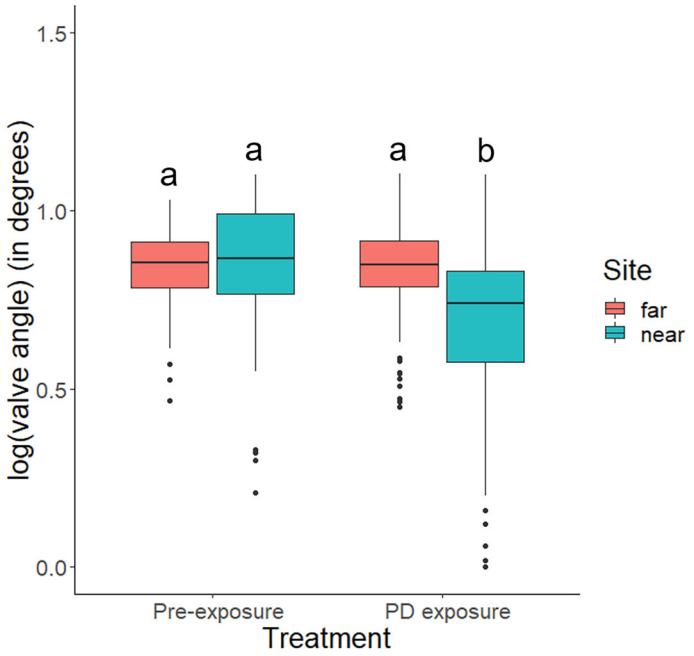



**Fig. 4** Boxplot showing the mean valve angles (log transformed) measured from tagged scallops at the far (red) and near (green) sites before and during pile driving exposure. Results of the post hoc Tukey test are shown with the letters

over long periods of time that limit further movement, including predator escapes. Hence, scallops exposed to PD could be more vulnerable to predation.

Most previous studies with marine invertebrates exposed to PD occurred in tanks at unrealistic sound exposure levels, which rend these results difficult to translate to wild populations (Popper et al. 2022). To fill this gap, scallops in this study were exposed to substrate-borne vibrations from real-time PD in an open water environment (Fig. 1). While such study remains scarce, it is to note that OSW foundations are much larger compared to the PD setup used here (Kallehave et al. 2015). Physical parameters such as bottom type also need to be considered when assessing substrate-borne vibration impacts. Taken together, the spatial range of impacts found in this study is likely to increase during OSW constructions, potentially impacting scallops at the population level.

## Conclusion

The efforts developed in this field study are an important step in understanding the impacts of substrate-borne vibrations from PD in marine benthic invertebrates. The results indicate that scallops behaviorally responded to PD with potential ecological consequences. Further studies are now required to measure the responses of wild scallop populations to OSW constructions (Popper et al. 2022).

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# Noisy Waters Hamper the Pumping Behavior of the Polychaete *Lanice conchilega*

89

Sheng V. Wang, Carl van Colen, and Jan Beermann

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**Abstract**

Anthropogenic low-frequency noise (LFN) is a continuously growing stressor in our oceans. How this noise affects most groups of marine animals is understudied especially in the case of benthic invertebrates, important ecosystem engineers that rework the sediment in which they live. Here, the effects of acute, continuous LFN on the bioirrigation behavior of *Lanice conchilega*, a tube-building worm

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abundant on the North Atlantic coasts, were investigated by tracking sediment oxygen concentrations via microsensor pre- and during exposure. Individuals exhibited lower pumping (tube ventilation) rates when exposed to LFN than when under control conditions. This may signify repercussions for both *L. conchilega* and the ecosystems it helps maintain. Thus, more research is urgently needed to predict and circumvent potentially adverse scenarios brought about by underwater anthropogenic noise.

**Keywords**

Anthropogenic noise · Low-frequency noise · Polychaeta · Bioirrigation · Bioturbation

## Introduction

Anthropogenic low-frequency noise (LFN), or sounds emitted by human activities with frequencies between 10 and 500 Hz (Carey and Evans 2011), is a rising pollutant in our oceans since recent decades. Emitted by sources such as cargo shipping, pile driving, and wind farm operation, these sounds have seeped into natural underwater soundscapes and only continue to increase in prevalence and intensity. Despite this, not much is known about how LFN affects most groups of marine animals, with its effects on benthic invertebrates being especially understudied.

Benthic invertebrates such as worms and bivalves rework the seafloor they live in or on through everyday activities such as burrowing, defecating, and ventilating. This process, known as "bioturbation," is a driver of resource exchange between different sediment layers and the sediment-water interface. Bioturbation also drives local sediment dynamics and biogeochemical cycles, deeming bioturbators crucial ecosystem engineers (Aller 1994; Aller and Cochran 2019; Kristensen et al. 2012). Almost nothing is known about how these organisms respond to LFN, and even the way most marine invertebrates perceive noise is only poorly understood.

While a few studies in very recent years have shown deleterious or no effects on invertebrates (Carter et al. 2020; de Soto et al. 2013; Fitzgibbon et al. 2017; Hudson et al. 2022; Leiva et al. 2021; Zhao et al. 2021), less than a handful of these have focused on bioturbation. A laboratory experiment by Solan et al. (2016) showed that the burrowing behaviors of a bivalve (*Ruditapes philippinarum*) and a commercially important lobster (*Nephrops norvegicus*) were reduced by anthropogenic noise. Similar results were seen in the previous work of Wang et al. (2022): the average sediment reworking rate and depth of the amphipod *Corophium volutator* were reduced in the presence of LFN. While not statistically significant, a potential anti-burrowing stress response was seen in the bivalve *Limecola balthica*, and an increase in bioirrigation (the water exchange component of bioturbation) variability was detected in the polychaete *Arenicola marina.*

To continue filling this immense knowledge gap, this study investigated how the bioirrigation behavior of the tube-building worm *Lanice conchilega* may be affected by LFN in a controlled laboratory setup. *Lanice conchilega* is an abundant polychaete on the North Atlantic coasts; its bioturbation activities facilitate biogeochemical cycles, stabilize sediment via its erect tubes built into the seafloor, and help determine ecosystem structure (De Smet et al. 2015; De Smet et al. 2016; Van Hoey et al. 2008). If, and if so, how added LFN alters the piston-pumping (tube ventilation) rate of *L. conchilega* were tested here, which might then hold potential implications for local sediment characteristics. The LFN employed in this study does not simulate that of a specific source; rather, the frequency produced falls within the range of those emitted by sources such as wind farm operation and ships. The results are intended to explore potential first cause-effect relationships to serve as a pilot for future field- or upscaled laboratory experiments.

## Methods

### Animal Collection

Individuals of *Lanice conchilega* were unearthed by shovel and collected in the Bay of Heist, Belgium (51°20′38.5″N, 3°14′04.0″E). The animals were immediately transported to the lab, planted in cores as mentioned below, and then left to acclimate for 60 h.

### Experimental Setup

Eight 200-L replicate tanks made out of polyisocyanurate plates (inner dimensions: 90 cm length, 60 cm width, 39 cm height) were placed in a temperature-controlled chamber at 13 °C. Each tank was paired with its own aerated seawater tank and pump to form a cyclical water system; filtered seawater was sourced from the facilities at Ghent University in Ghent, Belgium. Within each tank, three cylindrical plexiglass cores (5.4 cm diameter, 15.7 cm height, plugged on one end) were placed after being filled with approximately 14 cm of sieved and settled sediment from the *L. conchilega* sampling site. One individual of *L. conchilega* was planted using a glass straw as described by Ziegelmeier (1969) in a random core until all cores were occupied.

### Added Noise Treatment and Measurement

A small battery-powered submergible vibrator designed by de Jong et al. (2017), or the "noise egg," was employed to generate continuous low-frequency noise at approximately 160 Hz. The egg was attached to a PVC pipe fastened to two perpendicular rods; sandbags were fastened to the ends of the rods to weigh down the apparatus and mitigate vibration transfer from the vibrator to the tank wall. The

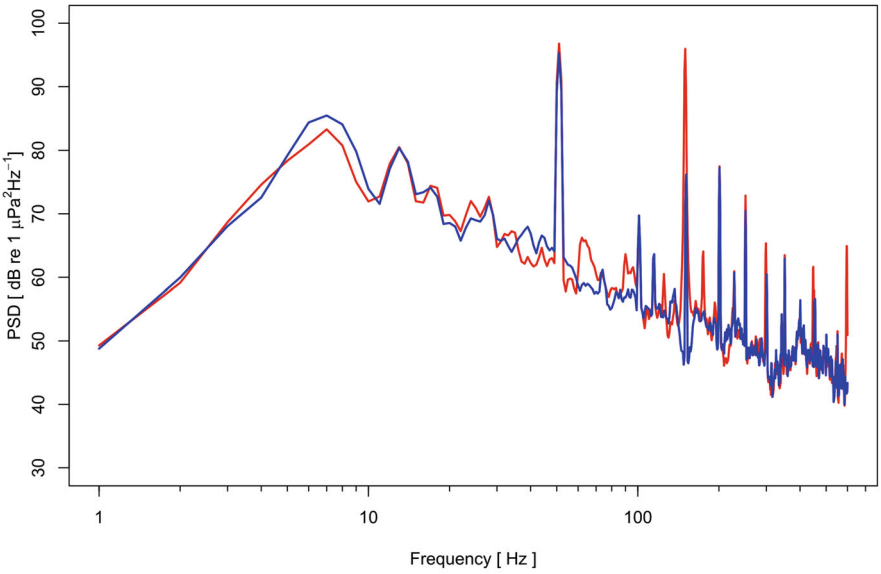



**Fig. 1** Power spectral density comparison of the added noise (red) and control (blue) conditions. The lines depict the root mean square levels. A 10 dB increase on the decibel scale signifies a tenfold increase in power. The red peak at approximately 160 Hz represents the added noise produced by the noise egg, while the red and blue peaks at approximately 50 Hz are likely ambient climate chamber noise

egg was then deployed just below the water surface, positioned between the three cores in the tank. Each replicate tank was exposed first to 1 h of ambient noise with an inactive egg and then to 1 h of added low-frequency noise with an active egg.

A calibrated hydrophone (Aquarian Scientific AS-1, −208 dBV re 1 µPa sensitivity, paired with a PA4 preamplifier and connected to a Zoom UAC-2 audio interface) was positioned 2 cm from the sediment surface of one core from each tank to monitor the frequencies produced by the noise egg. Thirty-second sound recordings of the control and noise treatments were taken in WAV format and analyzed using PAMGuide code (Merchant et al. 2015) in R (3.5.1) (R Core Team 2018) to produce power spectral density (PSD) graphs (Fig. 1) for comparison. As described in Wang et al. (2022), the following settings were used for PSD generation: Fs(Hz), 44,100; window, Hann 50%; and average, 1 s. The correction factor for calibration was calculated using manufacturer technical specifications, gain values from the preamplifier and audio interface, and the zero-to-peak voltage of the audio interface.

## Pumping Rate Measurement

Porewater oxygen concentrations (µmol/L) were measured using an oxygen microsensor (Unisense OX-25) connected to a Unisense Field Microsensor Multimeter

and recorded using the Unisense Logger software. The microsensor was calibrated using freshly oxygenated seawater as the 100% oxygen point and sodium ascorbate as the 0% oxygen point and then inserted into a motorized arm mounted on a steel frame. The microsensor was then inched approximately 1.5 mm into the sediment next to the *L. conchilega* tube until a consistent pumping signal was acquired on the logger software. For each tank, continuous oxygen concentration measurements were taken for one *L. conchilega* individual under control and added noise conditions. Pumping rates were then calculated by plotting the data in GraphPad Prism 8.0.2 and counting the number of peaks per hour; these rates were afterward statistically analyzed using the same program.

## First Results and Discussion

The added noise treatment had a significant effect on the pumping rates of *Lanice conchilega* (paired t-test, $t = 2.646$, $df = 6$, $p = 0.0382$). Individuals pumped less often when exposed to low-frequency noise (LFN) than they did when under control conditions (Fig. 2).

In five of seven replicates, the pumping pattern of the worm was visibly influenced by the LFN; apparent effects on pumping included significant changes in regularity, slight shifts in peak amplitudes, and changes in overall signature (Fig. 3).

These results, along with those from the previous study (Wang et al. 2022), provide some first indications as to how LFN might affect polychaetes. Wang et al. (2022) showed that the lugworm *Arenicola marina,* an abundant burrowing worm along North Atlantic coasts, exhibited great variability in its bioirrigation rates

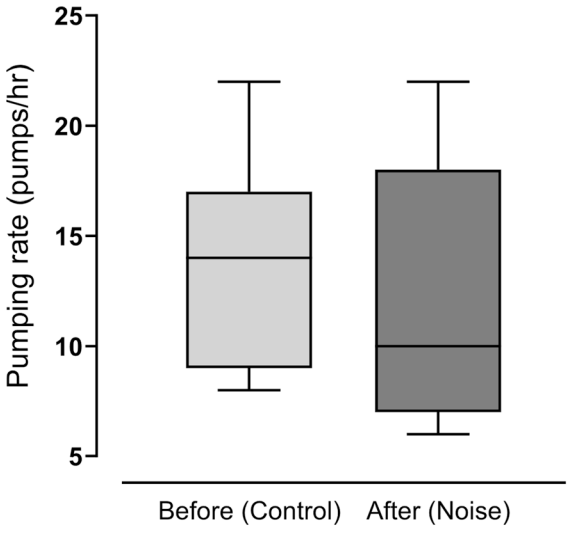



**Fig. 2** Pumping rates of *L. conchilega* before and after exposure to added low-frequency noise

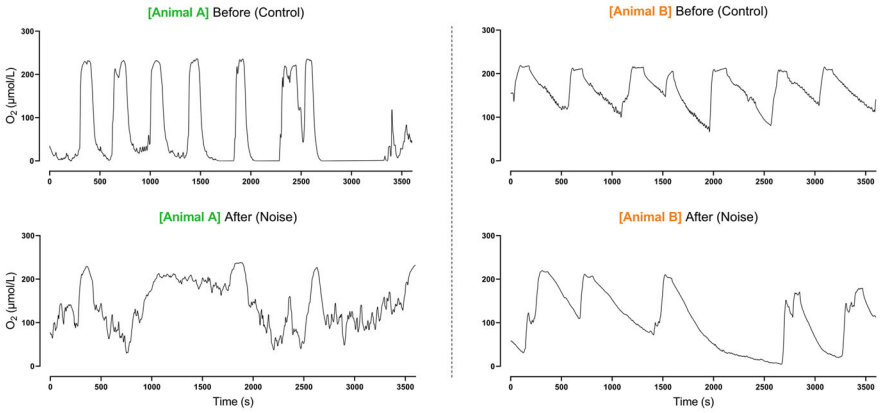



**Fig. 3** Example before-and-after pumping pattern comparisons within two replicates (**a** and **b**). The top graphs show pumping under control conditions (before), while the bottom graphs show pumping under added noise conditions (after)

when exposed to continuous LFN produced by the same source. This was done by periodically sampling sodium bromide tracer concentrations in the overlying water column, contrasting with the current study in which local sediment oxygen concentrations were measured using a microsensor. Both methods, however, ultimately quantified bioirrigation in different ways. It is then plausible that the aforementioned large variabilities might be reflections of the changed pumping characteristics seen in the current results, assuming that both *A. marina* and *L. conchilega* reacted to LFN in a similar way. Both polychaetes are infaunal facilitators of the nitrogen cycle (Braeckman et al. 2010; Volkenborn et al. 2007), and therefore, changes in their bioirrigation behaviors (rate, depth, regularity) will affect the local biogeochemistry. Yazdani Foshtomi et al. (2018) showed that *L. conchilega* in *L. conchilega* reefs, through their bioirrigation activity, had significant influence on the composition and diversity of the *nosZ* gene (a crucial gene in the microbial denitrification pathway) and, by association, the local denitrifying microbiota. *Arenicola marina*, in addition to having similar influence at depth, has also been shown to fuel microphytobenthos growth on typically low-nutrient sediment surfaces via nutrient advection when they ventilate their burrows (Chennu et al. 2015). A decrease in bioirrigation due to LFN might therefore bring about serious repercussions in terms of benthic-pelagic coupling and ecosystem functioning. Low-frequency noise may affect different life stages of the bioirrigators as well as the microbiota themselves. A more detailed understanding of how LFN affects all parts of a benthic ecosystem will be crucial in predicting and avoiding critical scenarios in our increasingly noisy oceans.

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# Effects of Repeated and Continuous Sound Exposures on Mussels (*Mytilus* spp.) in the Lab

90

Jeroen Hubert, Emily Booms, Rosalie Moens, A. Daniëlle van der Burg, Rob Witbaard, and Hans Slabbekoorn

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**Abstract**

Human activities at sea have increasingly changed the marine soundscape over the last decades, and many of these have been shown to potentially impact marine life. Bivalves and other invertebrates received limited attention in sound impact studies, despite their importance to the marine ecosystem. Furthermore, many

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studies only focus on the effects of the sound onset or on the effects of the sound exposure as a whole. However, the impact over time of an exposure or over repeated exposures may also reveal relevant information on how bivalves respond to sound and what mechanisms may explain variation in their responsiveness. This chapter analyzes the combined results of four published experiments on the effects of sound on mussels (*Mytilus* spp.) to reveal generalities in their valve gape behavior throughout the trials of all studies. Across studies, mussels responded to sound by partial valve closure and also returned to pre-exposure valve gape levels within the exposure periods. Furthermore, the closing response decayed over sequential sound exposures. These results provide insight into the initial and long-term effects of sound on mussels. Future studies should examine effect of sound in situ and link this to fitness and population consequences.

**Keywords**

Bivalves · Anthropogenic noise · Habituation · Valve gape · Pulse rate · Boat noise

## Introduction

Increasing amounts of human activity at sea have changed the acoustic scene underwater (Andrew et al. 2002; Duarte et al. 2021). Activities like shipping, seismic surveys, and pile driving produce high levels of sound and are audible to marine life (Morley et al. 2014; Richardson et al. 1995; Slabbekoorn et al. 2010). Marine animals are probably exposed to anthropogenic sound continuously due to the omnipresence of sources and the far-reaching propagation of underwater sound. Human-made sound has been shown to cause stress and hearing damage and disrupt vital behavior (Nichols et al. 2015; Popper et al. 2005; van der Knaap et al. 2021). Most studies focused on marine mammals and fishes, while invertebrates including bivalves received limited attention (Williams et al. 2015). Bivalves fulfill an important role in the ecosystem as filter feeders, reef builders, and prey species (Borthagaray and Carranza 2007; Jørgensen 1990; Kautsky 1981). Most bivalves adopt a sessile or sedentary lifestyle, making it difficult or impossible to move away from a sound source, and are therefore likely to be exposed to the same sound for long periods or repeatedly.

Bivalves have been shown to detect and respond to natural and anthropogenic sound. Planktonic larvae of bivalves have, for example, been shown to increase settlement in response to playbacks of recordings from reefs with conspecifics (Jolivet et al. 2016; Lillis et al. 2013; Wilkens et al. 2012). Anthropogenic sound has been shown to impact bivalve behavior, physiology, and survival (Day et al. 2017; Jézéquel et al. 2022; Spiga et al. 2016; Vazzana et al. 2016; Wale et al. 2019).

The mechanism of sound detection in bivalves is not yet fully understood. Bivalves lack gas-filled cavities and are therefore not expected to detect sound pressure but are rather expected to detect particle motion through mechanoreceptor cells (Ellers 1995; Zhadan 2005). Nineteen bivalve families possess an abdominal

sense organ which is shown to be involved in sound detection (Haszprunar 1985; Zhadan 2005). Bivalve lineages that lack this specific sense organ possess structurally similar organs, possibly also to detect sound (Haszprunar 1983). Hair cells and statocysts have also been suggested to be involved in sound detection (Charifi et al. 2017; Roberts et al. 2015; Zhadan 2005). However, despite the reports on impact of noise pollution on bivalves and their widespread sensitivity to sound, insights into effects from continuous or repeated sound exposures are scarce.

This chapter replots the results of four experiments from three published studies on the effects of sound on mussels (*Mytilus* spp.). The valve gape behavior of mussels from all experiments was plotted in the very same way and scale to allow better comparison of general patterns, which are addressed by integrating the results of all four experiments. This chapter also explored whether the time allowed for acclimation before sound exposure in each experiment had been long enough by examining data from nonexposed control mussels from two of the experiments.

## Methods

All valve gape data from four previous sound exposure experiments on mussels (*Mytilus* spp.) in the lab was used (Hubert et al. 2022a (two experiments); b, 2023). In these experiments, individual mussels were independently tested in separate containers, and their valve gape was measured using a valve gape monitor. Mussels' valve gape has been linked to filter-feeding rate and is therefore a relevant measure related to fitness. The valve gape monitor consists of pairs of electromagnetic coils which were glued on to opposite valves of a mussel. The active coil generated an electromagnetic field, which was measured by the responsive coil and translated to a distance between the coils. To determine the valve gape, rather than the distance between coils, the minimum distance between the coils was deducted from the distance fluctuations over time. The location of the coils on the mussel and the size of the mussels also affected the measured valve gape. For this reason, the valve gape was translated to the fraction of the maximum opening (Fig. 1). The trials typically started with 25 min of silence playback to allow the mussels to recover from handling and to acclimate to the experimental setup. This was followed by a

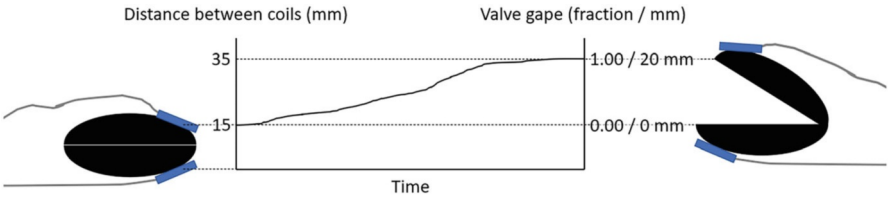



**Fig. 1** Schematic on how valve gape monitors were used to determine the valve gape of mussels. The electromagnetic coils (blue rectangles) of a pair were glued on the valves of a mussel (black half ovals). The distance between the coils was measured, and the minimum and maximum distance between the coils was used to determine the valve gape. The plot is not based on actual data

**Table 1** Description of the sound treatments, sound levels, and sample sizes (N) for all experiments discussed (I and II, Hubert et al. 2022a; III, b; IV, 2023). Sample sizes may differ slightly from the original papers as identical inclusion criteria across studies were used here

| Experiment | Sound treatments | Mean rms SPL (dB re 1 μPa) 100–600 Hz | N |
|---|---|---|---|
| I | Tones (1 min, 5 min silence):<br>150 Hz + harmonics<br>300 Hz + harmonics | 138.4<br>135.6<br>Silence: 77.6 | 100 |
| II | Pulse trains (1 s pulse, 9 s interval) or<br>Single pulses (1 s):<br>150 Hz + harmonics | 138.4<br>Silence: 77.6 | 65 |
| III | Pulse trains (5-s pulses, 5–160-s interval):<br>200 Hz + harmonics<br>350 Hz + harmonics | 141.8<br>143.2<br>Silence: 102.8 | 250 |
| IV | Looped boat sound (1 of 6 boats) or<br>Looped ambient sound (1 of 6 recordings) | 145.4<br>125.4<br>Silence: 92.4 | 130 |

sound treatment (or silent control) which varied per experiment and was designed to answer specific research questions (Table 1). The mussels were hung in the water column and did not touch the container they were in and are therefore expected to respond to in-water particle motion, rather than substrate vibration.

This chapter uses the mussels' valve gapes over time as fraction of the individual's maximum opening and plotted the median and first and third quartiles of all individuals. Mussels with a maximum valve gape of less than 0.5 mm or less than 10% of their maximum valve gape 23 min after the start of the trial were excluded because this indicated poor recovery from handling and limited or no acclimation to the experimental setup.

## Results

### Experiment I

Hubert et al. (2022a) examined whether the response of mussels to repeated sound exposures decreased over time. To test this, mussels were exposed to ten 1-min tonal sound exposures of 150 or 300 Hz (with unintentional presence of harmonics), and an 11th sound exposure with the tone this mussel had not heard before (300 or 150 Hz, respectively). Each 1-min sound exposure was followed by 5 min of silence. Mussels in the control condition were only exposed to the tenth and 11th exposure.

The figure of this chapter shows that the mussels' valves were closed at the start of the trial, probably because of the handling by the experimenter (Fig. 2). The mussels opened their valves in about 10 min to a relatively stable level of about 60–80% of their maximum observed valve gape. Upon the first sound exposure, the mussels rapidly closed their valves partially. Interestingly, the interindividual variation in valve gape clearly decreased after sound onset (based on Q1 and Q3).

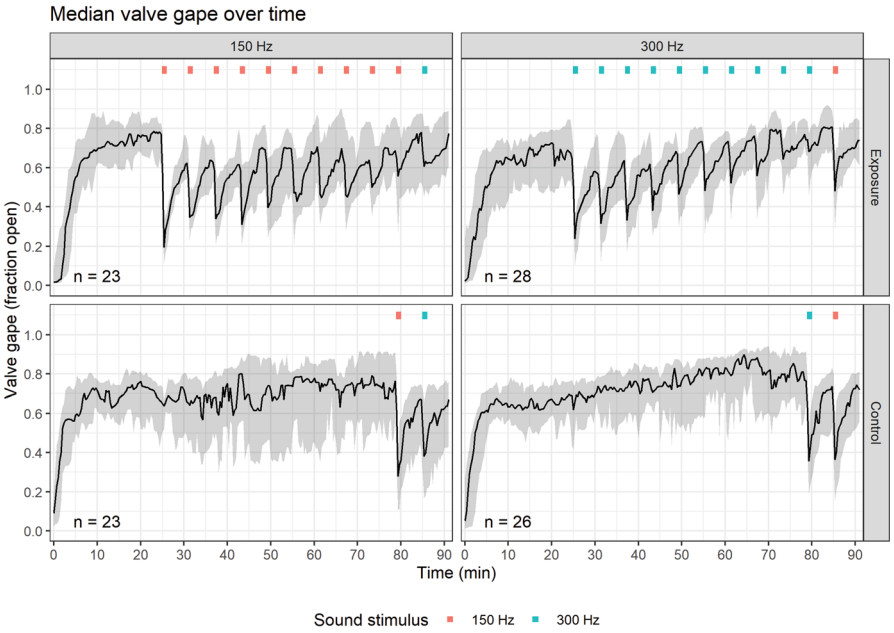



**Fig. 2** The median (solid line) and interquartile range between the first and third quartile (shaded area) of the mussels' valve gape over time. The colored blocks at the top of the panels indicate the timing and type of sound that was played back. The four panels show the four treatments in a 2 × 2 design, with the exposure treatments (top) consisting of 10 identical sound exposures followed by a single novel exposure and control treatments (bottom) consisting of the last two sounds of the exposure condition only. The treatments started with 150 Hz sound(s) (left) or 300 Hz sound(s) (right). The mussels clearly responded to the sound exposures by closing their valves, and the response magnitude decreased over sequential exposures

Moreover, during the 1-min sound exposures, the mussels started to increase their valve gape again. During recovery, the variation in valve gape among individuals increased, indicating variation in recovery speed. The response magnitude to repeated sound exposures decreased over time, and the variation in response seemed to increase.

The mussels were exposed to an 11th novel sound stimulus (with a different spectrum) to examine whether the decay in responsiveness could be attributed to habituation. Habituation is specific to a particular stimulus, while motor fatigue (muscle exhaustion) and sensory adaptation (e.g., increased sound detection threshold) are not. A stronger response to the novel stimulus therefore proves habituation; however, this was only present in one of the two exposure conditions.

## Experiment II

Hubert et al. (2022a) examined whether mussels' responses in valve gape were mostly determined by the sound onset or was also affected by the continuation of sound treatments. To test this, mussels were exposed to three 15-min pulse trains, consisting

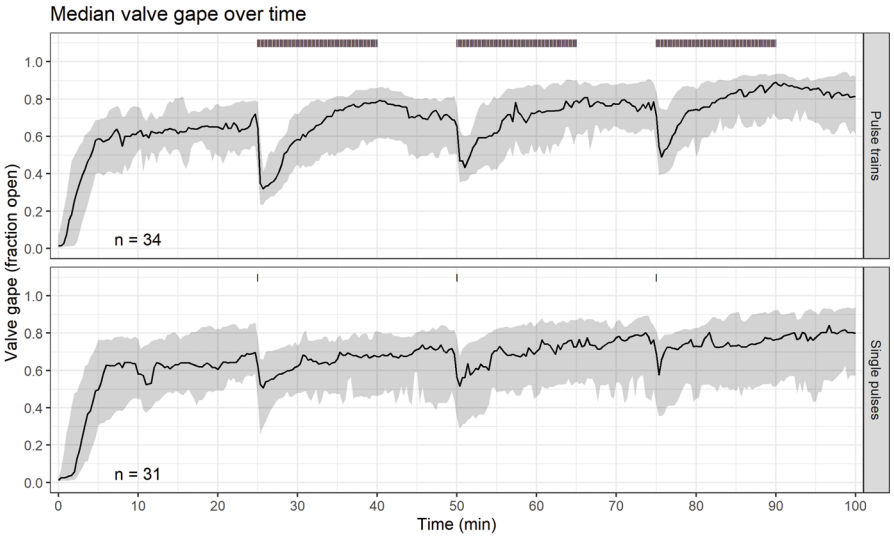



**Fig. 3** The median (solid line) and interquartile range between the first and third quartile (shaded area) of the mussels' valve gape over time. The black vertical bars at the top of the panels indicate the timing of the sound pulses that were played back. The mussels responded to both the pulse train and single pulses and returned to pre-exposure baseline levels already during the pulse train or after the single pulse

of 1-s pulses followed by 9 s of silence; each 15-min pulse train was followed by 10 min of silence. In another condition, mussels were only exposed to the first 1-s pulse for each pulse train, and the rest of the pulses were replaced with silence.

The figure of this chapter shows that the mussels in both treatments responded to the sound by decreasing their valve gape and recovered soon after, either during the pulse train or after the single pulse (Fig. 3). The response magnitude and recovery time decreased over the three sequential pulse trains or single pulses. Statistical analyses from the paper did not reveal statistically significant differences in initial response magnitude and recovery time (time to return to pre-exposure level) between pulse trains and single pulses. Based on visual observation of the valve gape data over time (Fig. 3), it seems that – if anything – the response magnitude was higher during the pulse trains, and recovery times between conditions were similar due to a steeper recovery during pulse trains. Additionally, after recovery during the pulse trains, mussels had a larger valve gape than prior to the first pulse train, perhaps to compensate lower filtering rates or higher oxygen demand during the initial response.

## Experiment III

Hubert et al. (2022b) examined whether the response of mussels to pulse trains was affected by the pulse rate. To test this, mussels were exposed to a 55-min pulse train with consistent pulse intervals ranging from 5 s to 160 s or a silent control with only a single

pulse at the end of the trial. The effect of sound on the phytoplankton clearance rates of the mussels was also explored by taking water samples at the start, after 60 min, and at the end of the trial and analyzing the water samples using a spectrophotometer.

The figure of this chapter shows that mussels responded to the start of the sound treatments by partial valve closure and that most mussels returned to pre-exposure valve gape levels during the pulse train exposure (Fig. 4). The statistical analysis from the paper revealed that recovery was fastest during faster pulse rates (Hubert et al. 2022b). The figure of this chapter reveals that, at relatively fast pulse rates ($\leq$40 s), mussels seem to respond quite strongly at the start of the pulse train, with no additional closure directly after sequential individual pulses. During the slower pulse rates ($\geq$80 s), the initial reaction seems less strong, and mussels already recovered a bit before the subsequent pulse and then respond again. For the 80-s interval, this pattern seemed to fade relatively quickly, while for the 160-s interval, responses to the individual pulses remained clearly visible at the median and Q1. At the Q3, the responses also faded fairly quickly. Around 60 min (and directly before and after the trial), an experimenter entered the experimental room to take some water samples to determine the mussels' clearance rates (cf. Hubert et al. 2022b). Here, it seems that the experimental groups that already returned to pre-exposure valve gape levels (or were not exposed in the control) responded more strongly to the disturbance from the experimenter. Lastly, the paper also analyzed the phytoplankton clearance rate and found a link with valve gape but did not find a sound-induced effect on mussels' clearance rate (cf. Hubert et al. 2022a).

## Experiment IV

Hubert et al. (2023) examined whether mussels responded to sound exposures of boat sound. To test this, mussels' valve gape behavior during one of various ambient or boat sound treatments were compared. The effects of a crab predator and potential interactions with the sound treatment were also explored by exposing the mussels to a live crab predator (either chemical cues or a free-ranging crab) or a control.

The figure of this chapter shows that the mussels strongly responded to the introduction of a live crab stimulus – or sham control – by partial valve closure. The boat sound exposure started before the mussels were fully recovered from the (sham) crab introduction. Nevertheless, and despite a 10-s linearly increasing transition (fade in) to the boat sound, the mussels responded again by closing their valves a bit more again but also recovered and caught up with the ambient control mussels (Fig. 5). The analysis from the paper showed that mussels had a lower valve gape when exposed to a live crab in the same enclosure, but not when only chemical cues were present (cf. Hubert et al. 2023).

## Pre-exposure Baseline

The control conditions of experiment I and III were used to evaluate the duration of the recovery period prior to the sound treatments in the exposure conditions.

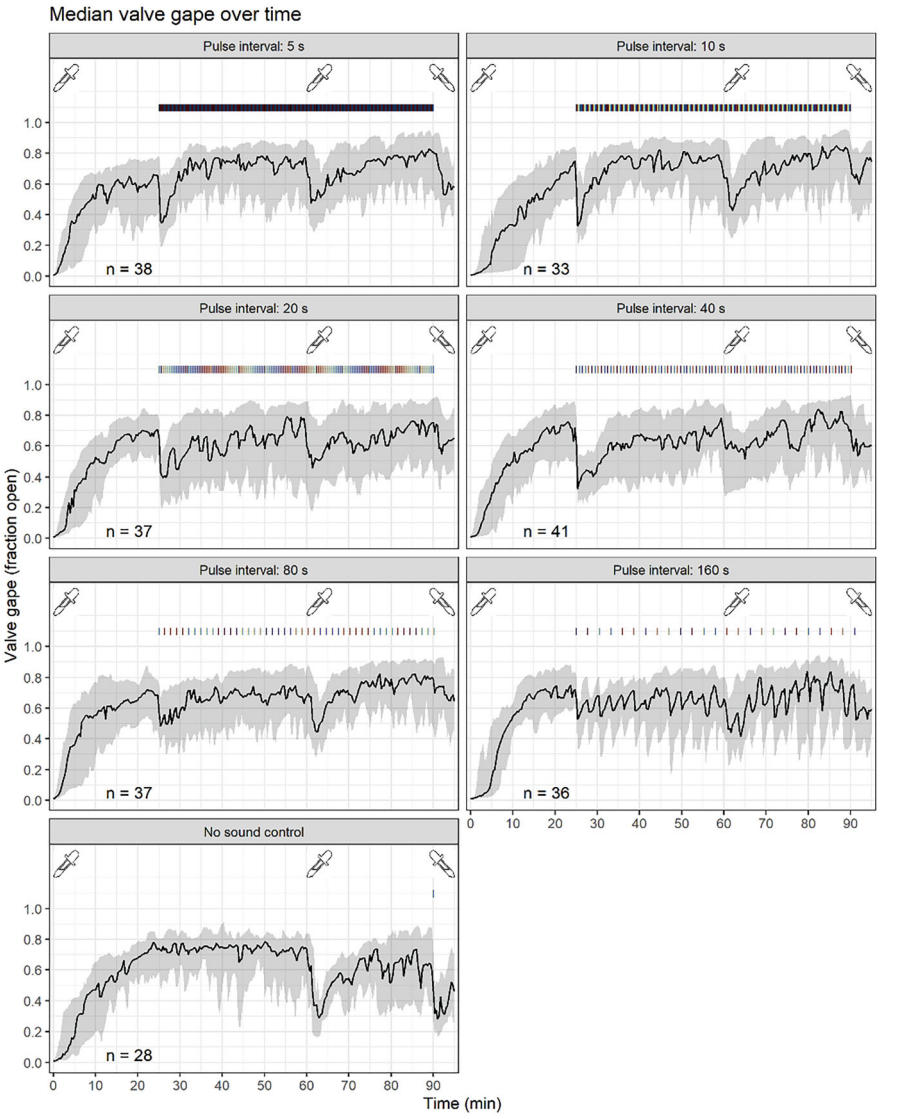



**Fig. 4** The median (solid line) and interquartile range between the first and third quartile (shaded area) of the mussels' valve gape over time. The pipets and vertical bars at the top of the panels indicate water sampling and the timing of the sound pulses that were played back. The interval between the pulses ranged from 5 to 160 s. At around 60 min, a water sample was taken to determine the mussels' clearance rates. The mussels closed their valves upon pulse train onset and mostly return to pre-exposure baseline levels during the pulse train. The recovery time was positively correlated with the pulse interval (Hubert et al. 2022b)

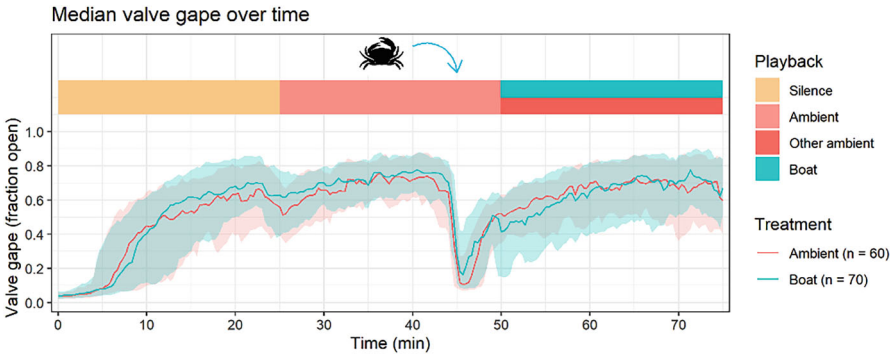



**Fig. 5** The top part of the panel shows that all mussels were first exposed to silence, followed by ambient sound (from 25 min) during which a (sham) crab introduction took place (at 45 min), followed by either another ambient sound or a boat sound playback (from 50 min). The bottom part shows the median (solid lines) and interquartile range between the first and third quartile (shaded areas) of the mussels' valve gape over time. The mussels closed their valves upon the (sham) crab introduction and again at the boat sound onset and caught up with the ambient exposed mussels after about 10 min

The sound exposure typically started after 25 min, but the control mussels in experiment I and III remained undisturbed until, respectively, 79 and 60 min after the start of the trials (Fig. 6a). A 1-min moving median was used to determine when mussels reached or exceeded the median valve gape of the last 5 undisturbed minutes. This analysis showed that a small majority reached this level before 25 min, when mussels in the exposure condition were exposed to sound (Fig. 6b).

## Discussion

Throughout all four experiments, mussels responded to the start of sound exposures by partial valve closure, and this response decayed over sequential exposures. Most of the mussels returned to pre-exposure valve gape levels during the sound exposures, and this recovery took longer during low pulse rates than during fast pulse rates. The mussels not only responded and recovered during pure tones with harmonics but also during boat playbacks with a soft start.

### Initial Valve Gape Closure

The mussels responded to sound by closing their valves. Valve closure is a general defensive reaction by bivalves and has been found in response to all kinds of changes in the environmental (Bayne et al. 1976; Carroll and Clements 2019; Curtis et al. 2000; Robson et al. 2010). Valve closure in response to tonal sound is in line with an earlier study (Roberts et al. 2015) but potentially contrasts with a study that found a larger mean valve gape in response to boat sound playbacks over a 1-h period (Wale et al. 2019),

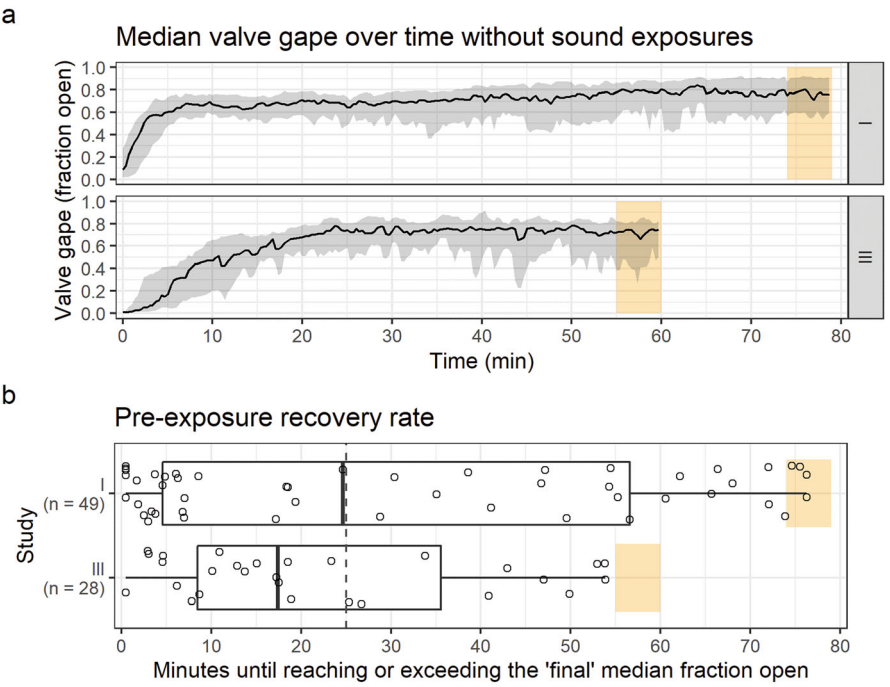



**Fig. 6** (**a**) The median (solid line) and interquartile range between the first and third quartile (shaded area) of the valve gape over time of mussels in the control conditions in experiment I and III. The yellow rectangle indicates the last undisturbed 5 min, from which the individuals' median valve gape was used as a "final" valve gape. (**b**) Time until mussels first reached or exceeded their "final" median valve gape. Most mussels reached this level before 25 min, when the sound started in exposure conditions. The open circles indicate the individuals, and the box-and-whisker plots indicate the median, first and third quartile, and minimum and maximum excluding outliers

although a larger mean valve gape does not exclude initial closure at the sound onset. Mussels' valve gape has been linked to filter-feeding rate (Hubert et al. 2022b; Jørgensen et al. 1988), but dissociation also occurs (Maire et al. 2007). Despite the link, no evidence was found for a sound-induced reduced clearance rate in experiment III, probably because sound-induced valve gape effects over the entire trial were limited (Hubert et al. 2022b). Further, the rapid closing movements in response to sound might be energetically costly (Galler et al. 2010; Robson et al. 2010). Overall, valve closure in response to sound may limit energy intake or increase energy use; future studies should examine to what extent these impacts also affect mussels in situ.

## Long-Term Effects

The mussels returned to pre-exposure baseline valve gape levels during or quickly after the exposures and responded less strong to sequential sound exposures. The decay in responsiveness to sound is in line with earlier studies on bivalves that

reported this for visual and tactile stimuli (Clements et al. 2021; Dehaudt et al. 2019; Wilkens 1986). A reduction in responsiveness can generally be explained by motor fatigue, sensory adaptation, habituation, or a combination of these (Netser et al. 2011; Rankin et al. 2009). Habituation is probably the most likely explanation for the decay in responsiveness to sound as mussels responded stronger again to a novel stimulus, at least in one condition (experiment I, ten 300 Hz exposures followed by one 150 Hz exposure). However, this result was not ambiguous across conditions, making it unable to exclude sensory adaptation (increase in sound detection threshold) (cf. Hubert et al. 2022a). The mechanisms underlying the decay in responsiveness are relevant for determining the impact of sound. If the responsiveness decays due to sensory adaptation or motor fatigue, it means that mussels may also not be sensitive or able to respond to other – potentially relevant – stimuli (Bejder et al. 2009; Domjan 2010), while habituation means that an animal is still able to detect and respond to a cue but ceases to do so as a result of nonassociative learning (Bejder et al. 2009; Rankin et al. 2009). Nevertheless, habituation of one response does not necessarily mean habituation of all responses and absence of all impacts of sound (e.g., masking of relevant cues) (Neo et al. 2018).

## Evaluation of the Pre-exposure Baseline

A small majority of the mussels in the control conditions first reached or exceeded their final median valve gape in the first 25 min of the trial, after which the sound treatment started in the exposure conditions. This measure is a proxy for recovery of handling and acclimation to the experimental setup. For this proxy, it is assumed that mussels had stable baseline valve gape at the end of the undisturbed period of these trials, as this period acted as a reference. Secondly, it was assumed that the first time reaching this valve gape level meant that the mussels had indeed reached this stable baseline level. While accepting these uncertainties, it can be concluded that a relatively large group (even thought a minority) did not reach the final valve gape before the start of the exposures at 25 min. So, for studies that require to manipulate mussels at stable valve gape levels, it is recommended to use a longer recovery and acclimation time than the experiments from this chapter.

## Future Research

Future sound impact research should be a combination of lab and field studies focusing on mechanisms that underly impact the actual effects of anthropogenic sound sources in the field and linking these effects to fitness consequences. The current lab experiments yielded insight into mechanisms underlying sound impact on mussels. This will help to understand impacts in the field. Insights into contribution of acoustic features, such as pulse rate, may help to mitigate effects. Future studies could focus on spectral and temporal components and the amplitude of boat sound to predict mitigation efficacy of boat motor design changes and slowdown measures. Lab studies are useful for such fundamental insights because they allow to

efficiently collect high-resolution data in controlled trials. However, the acoustic conditions, behavior of the mussels, and ecological conditions are likely different from the field (Slabbekoorn 2016). So, to gain insight into the actual effects of sound on behavior and physiology of mussels in situ, field experiments with actual anthropogenic sound sources should be conducted. Lastly, efforts should be made to link the found effects to changes in energy budgets of mussels and ultimately to fitness consequences (Hubert et al. 2020; Van Leeuwen et al. 2013).

## Conclusions

This chapter examined the behavioral response of mussels to short, long, and repeated sound exposures. Mussels responded to sound by partial valve closure, and they returned to pre-exposure valve gape levels during or after sound exposures, and the closing response decayed over sequential sound exposures. The mussels mostly responded to the onset of the sound treatments, but a slower recovery to pre-exposure baseline levels during pulse trains with slower pulse rates shows that mussels are still sensitive to acoustic traits beyond the onset. These new insights into the contribution of acoustic features to impact of sound on mussels should be considered in impact assessments of anthropogenic activities and are essential for designing future mitigation measures. Future research should expand the knowledge on the contribution of various acoustic features to guide mitigation measures and should examine the impact of sound on mussels in situ.

## Ethical Statement

The guidelines for the treatment of animals in behavioral research and teaching (ASAB 2018) were followed. There are no legal requirements for studies involving mussels (and crabs) in the Netherlands.

## Data Accessibility

All data are available from online repositories: experiment I and II, https://doi.org/10.5281/zenodo.4068351; experiment III, https://doi.org/10.5281/zenodo.7154170; and experiment IV, https://doi.org/10.5061/dryad.2rbnzs7sj.

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# Marine Bivalve Sound Detection and Associated Noise Impacts

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Seth F. Cones, Youenn Jézéquel, and T. Aran Mooney

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**Abstract**

Bivalves are an abundant and diverse taxon that serve many important ecological roles including ecosystem engineers and biofilters. As largely sessile animals, bivalves have evolved remarkable sensory systems to perceive their mobile and dynamic surroundings, including acoustic stimuli. Yet, the growing presence of human-generated ocean noise may have adverse effects on their sensory abilities, potentially resulting in physiological and fitness consequences. Recent studies have explored how bivalves detect sound and the impact of human-generated noise on their ecology and physiology. However, the vast variety of noise sources and characteristics, along with the limitation of experiments being primarily conducted in tanks, make it challenging to understand the overall impact of noise pollution on marine bivalves in situ. This work provides a brief review of the understanding of bivalve sound detection mechanisms, followed by a

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collation of noise impact studies to assess trends in noise impact severity relative to amplitudes of sound received by animal subjects. Data from comparable studies of similar sound types and characterizations provide evidence that acoustic impacts tend to be more severe with increasing received levels for both zero to peak particle acceleration (water and substrate) levels (dB re 1 $\mu ms^{-2}$) and root mean square sound pressure levels (dB re 1 μPa). This suggests that animals closer to a noise source face greater impact severity. Though, more research is needed to clarify these trends, and such data may aid in identifying more robust noise level thresholds among differing impact severities.

**Keywords**

Noise · Hearing · Audiogram · Particle motion · Sensory · Pollution · Vibrations

## Introduction

Bivalves (class: Bivalvia, phylum: Mollusca) are an extremely diverse and abundant taxon that serve important roles in marine ecosystems. Ecologically, bivalves are a key energy source for a myriad of species from myriad of species from rays (Ajemian and Powers 2012), marine mammals (Kvitek et al. 1988), and other mollusks (Kabat 1990). Bivalve aggregations provide complex habitats for other species (Tolley and Volety 2005), and they transport nutrients from the water column into benthic sediments (Newell 2004). Economically, global bivalve harvest nears 14 million tons and USD $16 billion (Wijsman et al. 2019). Despite their significant contributions to human and marine health, data on the auditory sensory capabilities of marine bivalves falls behind that of taxa such as fishes and marine mammals (Sole et al. 2023). This knowledge gap is concerning due to the increase and prevalence of man-made noise in coastal and offshore waters (Duarte et al. 2021).

There is mounting evidence that water and substrate-borne vibration detection is widespread within marine bivalves (see Table 1). Specific uses of sound may be vital ecologically; for example, detection of substrate-borne vibrations may aid in the perception of mobile benthic predators as seen in other taxa (Barth et al. 1988). However, sound detection capabilities can also lead to adverse impacts of anthropogenic noise (water and substrate borne). Noise impacts to mollusks can include behavioral changes (Cones et al. 2022), temporary threshold shifts in hearing sensitivity (Putland et al. 2023), and physical damage (De Soto et al. 2013). Bivalve sound detection and associated noise impacts were last reviewed by Roberts and Elliott (2017), and since then 12 new publications have become available in this emerging research area, potentially allowing for integration of these data to begin estimating thresholds of noise impacts on bivalves.

In this chapter, the potential mechanisms of sound detection capabilities in bivalves are described. Then, existing acoustic impact studies are collated. Finally, results from noise impact studies with similar methods and sound characterizations are amalgamated to describe noise impacts at various received sound levels. This

**Table 1** A summary of all studies examining the effects of noise on marine bivalves

| Study | Model species | Life stage | Tank versus field | Sound type | Sound intensity | Sound units | Frequency (Hz) | Impact | Effect category |
|---|---|---|---|---|---|---|---|---|---|
| Aguliar de Soto et al. (2013) | New Zealand scallop (*Pecten novaezelandiae*) | Larval | Tank | Seismic pulses | 164<br>165<br>3–4 | SPLrms re 1 μPa<br>SEL re 1 $\mu Pa^2s$<br>PAL $ms^{-2}$ | <1000 | Body malformations | Physical damage |
| Charifi et al. (2017) | Pacific oyster (*Magallana gigas*) | 18 months | Tank | Tones – 180 s duration | 100–160<br>0–1.2 | SPLrms re 1 μPa<br>PAL re 1 $ms^{-2}$ | <1000 | Partial valve closures | Behavior |
| Day et al. (2017) | Southern Australian scallop (*Pecten fumatus*) | Adult | Field | Impulsive – seismic airgun | 213<br>188<br>37 | SPL dB re 1 μPa<br>SEL dB re 1 $\mu Pa^2s$<br>Ground roll $ms^{-2}$ | Not specified | pH change, recessing, increased hemolymph, mortality | Behavior – physio-physical |
| Ellers (1995) | Coquina clam (*Donax variabilis*) | Not specified | Tank | Ocean wave sounds: 0.25–155 min | 120–160 | SPL dB re 1 μPa | <200 | Jumping off sediments | Behavior |
| Hubert et al. (2022a) | Blue mussel (*Mytilus edulis*) | Adult | Tank | Pulses and pulse trains | 90–113<br>138.4<br>135.6 | SPL dB re 1 μPa/Hz<br>SPLrms dB re 1 μPa<br>SPLrms dB re 1 μPa | 150 and 300<br>150<br>300 | Partial valve closures | Behavior |
| Hubert et al. (2022b) | Blue mussel (*Mytilus edulis*) | Adult | Tank | Pulses and pulse trains | 141.8<br>143.2 | SPLrms dB re 1 μPa<br>SPLrms dB re 1 μPa | 200<br>350 | Partial valve closures, no change in biofiltration | Behavior |
| Jézéquel et al. (2022b) | Giant scallop (*Placopecten magellanicus*) | Juvenile subadult or adult | Field | Impulsive – pile driving | 87<br>110 | PAL dB re 1 $\mu ms^{-2}$ | <100 | Reduced valve angle | Behavior |

(continued)

**Table 1** (continued)

| Study | Model species | Life stage | Tank versus field | Sound type | Sound intensity | Sound units | Frequency (Hz) | Impact | Effect category |
|---|---|---|---|---|---|---|---|---|---|
| Jézéquel et al. (2023) | Giant scallop (*Placopecten magellanicus*) | Juvenile subadult or adult | Tank | Tones – 0.5 s duration | 90–113<br>89–97<br>95–98 | PALrms dB re 1 $\mu ms^{-2}$<br>PALrms dB re 1 $\mu ms^{-2}$<br>PALrms dB re 1 $\mu ms^{-2}$ | 100<br>150<br>200 | Reduced valve angle | Behavior |
| Peng et al. (2016) | Razor clam (*Siliqua patula*) | Adult | Tank | Continuous | 80<br>100 | SPL dB re 1 µPa<br>SPL dB re 1 µPa | 500<br>1000 | Digging depth changed, altered O:N ratio | Behavior – physio |
| Roberts et al. (2015) | Blue mussel (*Mytilus edulis*) | Adult | Tank | 8-second vibrations | 0.06–0.55 | $ms^{-2}$ | 5–410 | valve closures | Behavior |
| Solan et al. (2016) | Japanese littleneck clam (*Ruditapes philippinarum*) | Not specified | Tank | Continuous impulsive | 135–140<br>150 | SPL dB re 1 µPa<br>SEL dB re 1 $\mu Pa^2s$ | 100–2000 | No change to tissue biochemistry, reduced movements and valve closures | Behavior |
| Spiga et al. (2016) | Blue mussels (*Mytilus edulis*) | Adult | Field | Impulsive – pile driving | 158.5<br>182 | SELss dB re 1 $\mu Pa^2s$<br>SPLzpk dB re 1 $\mu Pa^2s$ | Not specified | Change in clearance rates | Behavior – physio |
| Vazzana et al. (2016) | Mediterranean mussels (*Mytilus galloprovincialis*) | Adult | Tank | Sweeps – 30 min | 150 | SPLrms re 1 µPa | 100–60,000 | Increased glucouse | Physio |
| Wale et al. (2019) | Blue mussel (*Mytilus edulis*) | Adult | Tank | Continuous – ship | 155<br>160<br>145<br>175 | SPL dB re 1 $\mu Pa^2/Hz$<br>PAL dB re 1 $\mu ms^{-2}$<br>SPL dB re 1 $\mu Pa^2/Hz$<br>PAL dB re 1 $\mu ms^{-2}$ | 10–3000 | DNA damage, valve closures; reduced oxygen consumption | Behavior – physio |
| Wang et al. (2022) | Baltic macoma (*Limecola balthica*) | Adult | Tank | Not specified | Not specified | Not specified | Not specified | No change in digging depth | Behavior |

provides a novel, empirically based, initial model of distance and potential dose-dependent responses to noise.

## Sound Detection and Mechanisms

All bivalves tested to date have been shown to behaviorally respond to sound, with responses in the frequency bandwidth of roughly 10–1000 Hz (Roberts et al. 2015; Charifi et al. 2017; Jézéquel et al. 2023, Table 1). Bivalves do not contain gas cavities, and so are not thought to detect the sound pressure component of the sound field (Budelmann 1992). Instead, there is evidence bivalves sense particle motion (Ellers 1995) and ground vibration (Mosher 1972) through three potential auditory sensory systems: paired statocyst, cilia-lined tentacles, and the abdominal sense organ (ASO).

Many invertebrates have an internal statocyst sensory system, which is analogous to the inner ear otoliths of many vertebrates and is thought to aid in sound detection (Young 1960). Statocysts are closely associated with the pedal ganglia, often located near the muscular foot (Morton 1985). Bivalve statocysts are paired (Wiederhold et al. 1986), and anatomically they are fluid-filled spheres lined with hair cells (Budelmann 1992). The hair cells function to detect movement of the statolith or statoconia, which are a singular or multiple calcium carbonate structures that reside within the fluid-filled chamber, in response to gravity and sound (Morton 1985). This auditory sensory mechanism has been demonstrated in squid (Mooney et al. 2010).

Secondly, bivalve tentacles are ciliated, and may be able to detect water motions and pressure gradients similar to the fish lateral line system (Bleckmann and Zelick 2009). Cilia tufts that line tentacles along the peripheral mantle contain a high density of nerves in numerous bivalve clads, increasing the evidence of a sensory role (Audino et al. 2021). Lastly, the ASO is an asymmetrical organ postulated to regulate water flow into the mantle cavity (Haszprunar 1985). The ASO surface is also lined with cilia, again potentially used for detecting particle motion.

The mechanisms underlying these systems are not mutually exclusive, and a combination of some or all could aid in detection of sound, or they may be differentially used for water-borne and substrate-borne vibrations. It is currently unknown which mechanism or combination of mechanisms are ultimately responsible for sound detection in bivalves.

## Noise-Induced Impacts

Man-made noise can have varying impacts on marine bivalves, including physical damage (De Soto et al. 2013), physiological changes (Day et al. 2017), and behavioral changes (Jézéquel et al. 2022b). Reponses are often a function of received level and, in a related manner, distance from the sound source (Jézéquel et al. 2022b). Unlike marine mammals and most fishes, bivalves are potentially more vulnerable to anthropogenic noise because animals are either sessile or have very limited mobility

and therefore are unable to move away from the sound source. Thus, they must incur the received levels at a given location for the extent of the noise activity.

To summarize the current state of the literature, all bivalve sound detection or noise impact studies to our knowledge are present in Table 1. Note that this work has mostly been conducted within the last decade and most studies were conducted in enclosed tanks where the acoustic field is not representative of in situ conditions (Jézéquel et al. 2022a). Tanks distort the temporal, power, and spectral characteristics of sounds, and this requires a strong understanding of the physical impacts tanks cause and how to account for them in an experimental design and data interpretations (Jézéquel et al. 2022a). It also requires careful calibration of the acoustic field within the tank for both pressure and particle motion measurements (Jones et al. 2019).

## Aggregating Noise Impact Studies

The severity of noise impact on marine species is often a function of multiple factors such as frequency, rise time, and dose parameters including duration and received sound level, which often correlate with distance. This spatial phenomenon is frequently visually depicted as linear (see Fig. 1 Mooney et al. 2020) or two dimensional (Fig. 1, Richardson et al. 2017) where there are discrete regions around a source that transition from physical damage to audible. Here, noise impact studies were aggregated to map effects within this framework and identify potential thresholds between impact types. The goal was to create a received level-based predictive zone of impacts for bivalves. In doing so, two related challenges are addressed, which should be acknowledged as this framework is assessed.

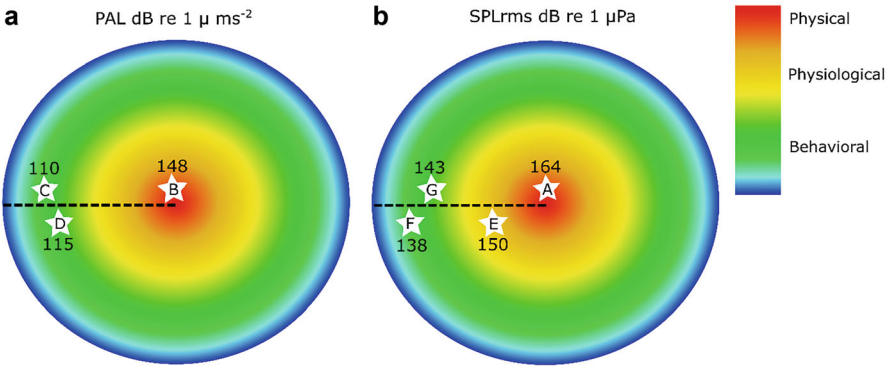



**Fig. 1** Two-dimensional noise impact maps in both particle acceleration level in zero-to-peak (**a**) and root mean square sound pressure level (**b**). The colors denote the severity of impact ranging from physical harm (red) to behavioral impacts (green). The numbered stars denote the referenced study while the adjacent number is the dB intensity reported. The location of the stars is representative of the associated impact at the reported noise level. ([A]De Soto et al. 2013, [B]Day et al. 2017, [C]Jézéquel et al. 2022b, [D]Roberts et al. 2015, [E]Spiga et al. 2016, [F]Hubert et al. 2022a, [G]Hubert et al. 2022b)

## Scant Literature on Bivalve Sound Detection and Noise Impacts

First, there were limited publications on noise impacts on marine bivalves. In total, we found 25 publications on the topic of bivalve sound detection. Of the 25 studies, 15 were experimental examinations of the effects of anthropogenic noise on bivalve behavior or physiology (Table 1). Only 3 of the 15 studies were conducted in situ using real sound sources (Spiga et al. 2016; Day et al. 2017; Jézéquel et al. 2022b). Eleven of the fifteen reported noise levels in sound pressure (SPL), while only seven of the fifteen reported either water- or substrate-borne acceleration levels (PAL) received by the animal. Given that animals detect particle acceleration and that PALs are not predictable from SPLs near the inherent (benthic) boundaries which bivalves inhabit or in tanks where many studies were conducted, this reporting limits our understanding, and predictions, of noise impacts on these taxa. Further, of the studies reporting noise in SPL, three studies reported $SPL_{rms}$ dB re 1 μPa (root mean square), two studies reported peak to peak SPL dB re 1 μPa, and two reported sound exposure levels (SEL in dB re 1 $\mu Pa^2 s$). Lastly, there were diverse sound types used in the reviewed experiments, including, pulses pulse trains, sweep tones, flat tones, and white noise.

## Lack of Audiograms

Another challenge faced when addressing sound sensitivities was the knowledge gap on hearing thresholds. Roberts et al. (2015), Charifi et al. (2017), and Jézéquel et al. (2023) constructed behavioral response curves of the blue mussel (*Mytilus edulis*), Pacific oyster (*Magallana gigas*), and the giant sea scallop (*Placopecten magallenicus*), respectively. These are key data demonstrating sound sensitivity, however, these unconditioned response curves do not necessarily provide the minimum thresholds of auditory detection. Behavioral response curves were developed by monitoring valve movements in response to sound at specific frequencies and intensities. If an animal closed its valves, it was assumed to be in response to the sound stimulus. However, it is possible that animals detect lower sound levels, but do not behaviorally respond by closing their valves. There are no current data reporting sound intensities which bivalves detect, but do not show a behavioral response. Determining if auditory thresholds are similar to sound levels inducing behavioral responses should be examined in the future. Consequently, the response map constructed here utilizes the best available data.

## Noise Impact Maps

Considering these factors, two-dimensional representations of noise impacts on bivalves were created, with the colors depicting relative noise amplitude and animal impact severity (Fig. 1). Stars represent noise impact studies with comparable units and sound types from Table 1. The letters within the stars represent studies cited

from Table 1. The location of the star describes the impact severity reported in the study and its associated number represents the measured noise intensity in decibels.

For Fig. 1a, only studies measuring either particle or ground acceleration in zero to peak and using impulsive noise sources were used (Roberts et al. [2015] was included in this analysis, which exposed animals to 8 s vibrations). For Fig. 1b, only studies measuring sound pressure in root mean square and using pulses, pulse trains, and sweeps were used. Again, as noted above, measures of pressure, especially in tanks, do not reflect the signals to which bivalves and other particle motion-sensitive animals respond. It is important to note that studies were combined across life stages, despite evidence that behavioral responses vary through ontogeny (Jézéquel et al. 2022b). The frequency of the noise stimulus was omitted from this analysis but is important to consider. The power spectra of the studies likely differed, and these relative frequency contributions may influence animal responses. However, all exposure frequencies were completely or partially in bivalve sound detection range (<1 kHz).

For both $PAL_{0\text{-peak}}$ (dB re 1 $\mu ms^{-2}$) and $SPL_{rms}$ (dB re 1 μPa), the measured noise impact tended to be more severe with increasing received sound level, although no statistical test was run due to the limited sample size. This finding intuitively suggests that, for $PAL_{0\text{-peak}}$ and $SPL_{rms}$, noise impact on bivalves is a function of distance from sound source (but see [Dahl et al. 2020]). In this context, marine bivalves closer to a noise source will be most impacted. Although there were some assumptions in this analysis, these methods can be used for other taxonomic groups to assimilate all data and identify thresholds between impact severities. Here it is demonstrated that noise sources greater than 115 $PAL_{0\text{-peak}}$ (dB re 1 $\mu ms^{-2}$) and 143 $SPL_{rms}$ (dB re 1 μPa) may lead to physiological and behavioral impairments. This information can be used to set exposure criteria guidelines for taxa (Popper et al. 2014).

## Conclusion

While there are some data in the bioacoustics literature on impacts of noise on marine bivalves, there are still several key knowledge gaps: auditory thresholds, noise exposures in varying contexts, dose-dependence impacts, and the need for studies measuring the appropriate sound metrics. The need for auditory thresholds was described in section "Lack of Audiograms."

Secondly, context such as time of day, behavior, or reproductive state may influence impact severity. For example, scallops show a clear diurnal pattern to their valve gape angle (Jézéquel et al. 2022b), and thus time of exposure would influence the extent of the response and also the vital activity occurring at that time. Context has a clear role in vertebrate noise studies and potentially for invertebrates as well (Deruiter et al. 2013).

Additionally, more noise impact studies should employ a before-and-after gradient design (Methratta 2020) when possible to address multiple noise amplitudes and concurrently monitor for a gradation of impacts. This experimental style is not only

better controlled but a more-natural reflection of likely exposure conditions (i.e., animals at multiple distances and received sound levels; Methratta 2020). The stepwise design will aid in identifying thresholds between behavioral, physiological, and physical impacts of sound, which are needed to understand how anthropogenic noise may impact the ecology of this vital taxon. Finally, considering that all bivalves live within or on top of the substrate and are thought to detect particle acceleration, future work should be directed at quantifying both water- and substrate-borne particle acceleration levels.

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# Effect of Operational Wind-Turbine Vibration on Surface-Dwelling Invertebrates

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Jordan Burgess, Stephen Thomas, Krysia Mazik, Sophie Al-Mudallal, Shiu Keung Tang, and Thomas Breithaupt

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**Abstract**

With the UK aiming to achieve 50 GW of offshore renewable energy by 2030, offshore construction will increase in tandem. The construction noise associated with wind turbine installation has gained much attention over the years, with sound emissions threatening to have an impact on the ecology of marine mammals and fish. During operation, continuous vibration is produced by blade rotation and the operating machinery near the blades (the nacelle) that travel down the external surface of the turbine tower and into the foundation and neighboring seafloor. While vibration is known to affect the behavior and physiology of marine invertebrates, the potential for these seismic interface waves, to impact the animals living on the turbine or the seafloor adjacent to the turbine, has yet to be studied. Due to the difficulty of measuring offshore, in this study, substrate-borne vibrations (seismic waves) from a turbine in operation were measured at a terrestrial wind farm at three different locations (turbine tower, concrete foundation of the tower, and adjacent soil in a 10 m distance) at different wind speeds to assess their potential effect on associated animals, particularly invertebrates. The vibration was highest on the turbine tower, with a maximum vibration recorded at 91.3 $mms^{-2}$ (114 Hz). However, most of the values fell below 53 $mms^{-2}$. The lowest recorded vibration was found within the soil, with maximum vibration not exceeding 2.1 $mms^{-2}$ (50 Hz). The recorded substrate vibration showed little overlap with the known vibration sensitivities of invertebrates at their expected locations. Mussels (*Mytilus edulis*), regularly found to be attached to the towers of offshore wind turbines, dwell in the intertidal area and are often exposed to air. This study finds that their threshold sensitivity (60 $mms^{-2}$) is generally too low to perceive the measured vibrations of the tower. Similarly, hermit crabs (*Pagurus bernhardus*) and other decapod crustaceans, which dwell close to the foundation of the wind turbines, based on their sensitivity, are unlikely to perceive the minute seismic waves that are conducted into the sediment. However, under some specific but rare conditions, vibration amplitudes were high enough to be detectable by the marine invertebrates. These measurements, while conducted in terrestrial conditions, indicate potential temporary impacts of wind farm operational noise on associated epifauna. Additional information is needed to confirm the severity of the effects on the marine environment.

**Keywords**

Substrate vibration · Turbine vibration · Anthropogenic noise · Invertebrate sensitivities · Vibrational noise · Wind turbines

## Introduction

Wind farm developments in Europe and the UK, particularly offshore, are rapidly growing to reach the 2050 net zero targets for the global energy sectors (British Energy Security Strategy 2022). The energy strategy in the UK has increased current targets from 40 GW to 50 GW by 2030, with approved projects set to increase the current capacity by threefold (British Energy Security Strategy 2022). This increase in demand has led to turbines becoming larger and more powerful (IRENA 2019), as well as to an increase in the total marine space planned for offshore developments. Concerns surrounding underwater noise have typically been related to pile driving activities, generating high amplitude-low frequency noise and vibration that propagates through the water column (water-borne particle motion) and the seabed (substrate-borne particle motion). Documented effects range from behavioral changes to increased mortality. Study taxa include marine mammals (Brandt et al. 2011; Kastelein et al. 2019), and the less studied fishes and marine invertebrates (Casper et al. 2013; Spiga et al. 2016; Jézéquel et al. 2021; Siddagangaiah et al. 2021). Operational turbine noise and substratum vibration have gained less attention in comparison, likely due to their lower intensity (Madsen et al. 2006; Tougaard et al. 2020; Stöber and Thomsen 2021).

Installing offshore turbines creates a new habitat through the introduction of hard substrata to a previously predominantly sedimentary environment (Mavraki et al. 2020). This new habitat is colonized by epifaunal marine animals, predominately filter-feeding invertebrates such as mussels and fish (Andersson and Öhman 2010; Coolen et al. 2022). After installation, the turbines are in their operational phase for approximately 25 years before decommissioning, where turbines are either fully removed or the foundations are left in place to preserve the assemblages. During these operational years, vibrations are produced from two main parts of the turbine: the blade rotation and the machinery within the nacelle, such as the gearbox and the generator (Tougaard et al. 2009; Mollasalehi et al. 2013; Escaler and Mebarki 2018). Both the blades and the machinery are supported by the tower and the foundation, which anchors the turbine into the substrate (Tougaard et al. 2009). The vibrating tower produces acoustic noise which radiates into the environment, in the form of sound pressure waves and particle motion of the medium, traveling at the speed of sound (Hawkins et al. 2021). It will also, due to the elasticity of the tower wall generate several types of substrate vibration (substrate-borne particle motion). These waves travel down the tower surface and into the sediment with different speeds depending on the type of wave, and into the substrate (Mortimer 2017). Within the sediment, waves will travel as solid shear waves and compressional (P-) waves (Hawkins et al. 2021), while on the interface between water and substratum surface, vibration travels as seismic interface waves (i.e., Rayleigh or love waves; Aicher and Tautz 1990; Hawkins et al. 2021). These anthropogenic seismic waves may influence the associated invertebrates as they are very sensitive to these vibrations (Roberts et al. 2015; Popper and Hawkins 2018; Velilla et al. 2021).

Previous studies surrounding wind turbine vibration mostly focused on the acoustic noise radiating from the tower into the surrounding fluid (Pangerc et al.

2016). Substrate vibration has also been studied regarding structural health monitoring for fault detection and general operations and maintenance (Zhao et al. 2020; Pacheco-chérrez and Probst 2022). Little attention has been given to the impact of these vibrations on colonizing epifauna (Sigray and Andersson 2011; Velilla et al. 2021). Based on sound recordings in the surrounding fluid, operational turbines are expected to produce vibrations at low intensities in a frequency range below 500 Hz (Wahlberg and Westerberg 2005; Madsen et al. 2006; Sigray and Andersson 2011; Pangerc et al. 2016).

Research surrounding the biological impacts of turbine operation-induced substrate vibration is still in its infancy. One study at a terrestrial wind farm suggested that operational vibration in the soil reduces earthworm abundance near the turbine (Velilla et al. 2021). However, only relative vibration amplitudes were recorded, not the absolute magnitude of vibration (Velilla et al. 2021). Earthworms have the ability to detect substrate-borne vibrations in the soil (Holbrook and Mortimer 2018). However, it remains unclear how sensitive earthworms need to be to detect the minute turbine vibrations transmitted into the soil. Offshore, research mostly focused on the effects of sound waves (sound pressure or associated particle motion) originating from the operating wind turbines. The operation-induced sound appears to increase chorusing among fish, although it was suggested that this could be due to an increase in fish assemblages around the turbine (Siddagangaiah 2021). Apart from a few exceptions, previous biological studies of vibration effects on invertebrates have only considered the pressure component of sound (Hawkins et al. 2021). Invertebrates do not detect sound pressure and appear only sensitive to substrate vibration and particle motion (Breithaupt 2002; Roberts et al. 2015, 2016; Day et al. 2016). Sigray and Andersson (2011) measured water-borne particle motion near an offshore wind turbine (1.5 MW) using a wireless, three-axis sensor, placed 1 m from the outer rim of the foundation. The amplitudes of water-borne particle motion were strong enough to be within the detection range of some species of fish (Sigray and Andersson 2011). Vibration amplitudes were higher during lower wind speeds (5 $ms^{-1}$ – 12.3 $mms^{-2}$ at 16 Hz) compared to high wind speeds (11 $ms^{-1}$ – 4 $mms^{-2}$ at 29 Hz) (Sigray and Andersson 2011), despite the opposite being measured in turbine health monitoring reports (Zhao et al. 2020). Both Pangerc et al. (2016) and Sigray and Andersson (2011) measured turbines far smaller (3.5 MW and 1.5 MW) than offshore turbines that are currently being installed (12–14 MW; GE 2018) and there is therefore potential for higher levels of operational vibration both on the turbine tower and in the surrounding seabed.

This study aims to understand the magnitude and frequency of operational turbine-induced substrate-vibration and whether these are detectable by invertebrates that are attached to or live near the turbines. Substrate vibrations were measured perpendicular to the surface (considering only seismic interface waves, not the compressional P-waves) on a 2.0 MW onshore turbine. Due to the difficulty in data collection at an offshore wind turbine, vibration measurements were made at a terrestrial wind farm in the UK. The origin and spread of vibration measured on the turbine tower are expected to be similar between offshore and onshore turbines. While currently built offshore wind turbines are generally larger and more powerful,

tower vibration may be dampened due to their submergence in water. Three example invertebrates were used – the blue mussel (*Mytilus edulis*), the hermit crab (*Pagurusbernhardus*), and a terrestrial crustacean, the pill woodlice (*Armadillo officinalis*) – to give an approximate, initial indication into whether substrate vibrations may affect these invertebrates.

## Methodology

### Study Site and Turbines

The studied wind farm was located onshore in the North of England, UK (E492598, N438499), consisting of six turbines of 2.0 MW capacity. The turbines had a hub height of 58.5 m, a blade diameter of 83 m, and pitch-controlled blades. All measurements were taken on a single turbine. The power generation of the turbine is displayed using power output, measured in kilowatts (KW) and wind speed ($ms^{-1}$) (Fig. 1). The power curve can be split into different zones which relate to various dynamic conditions of the turbine which will likely influence vibration. The turbine will cut in (begins rotating) at a wind speed of 3 $ms^{-1}$ (zone 2) and cut out (stops rotating) at 25 $ms^{-1}$ (zone 6). The regions between these wind speeds are considered operational, and outside the turbine would be non-operational.

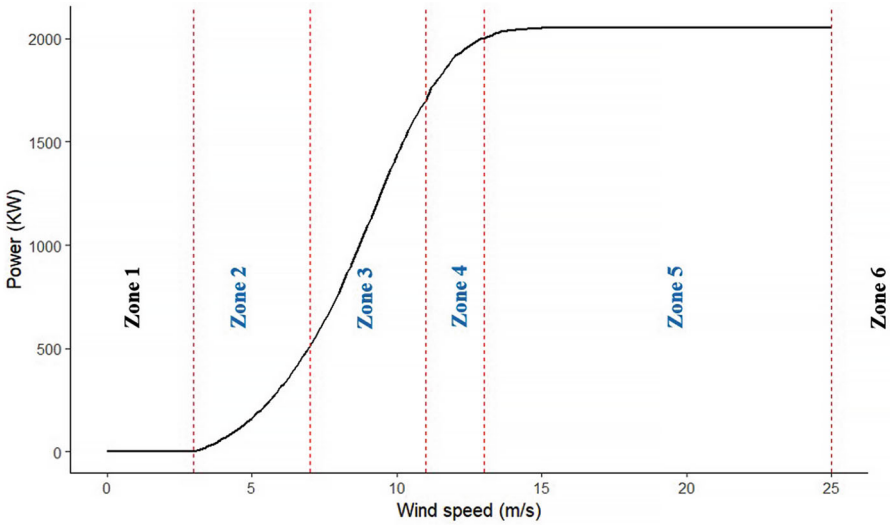



**Fig. 1** Power generation curve for 2 MW study turbine. Zones in black indicate a turbine in a non-operational state, while zones in blue relate to operational phases. Red dashed lines indicate operational zones. (Adapted from Pacheco et al. (2021))

## Study Design

Experimental data were measured and displayed as acceleration units ($mms^{-2}$). Measurements were collected across four days between August 2021 and February 2022. Measurements were taken at three different locations on and near the turbine (Fig. 2); at the turbine tower at 1 m height, on the surrounding concrete foundation, and on the soil at 10 m distance from the turbine. The accelerometer was buried in 10–15 cm depth to measure substrate vibration transmitted through the soil. Control measurements were taken on a separate day at the turbine tower during downtime periods when the turbine was switched off. During all measurements, wind speeds ranged from 3 $ms^{-1}$ up to 8.8 $ms^{-1}$, which correspond to zones 2–3 (Fig. 1). Conditions varied throughout the days, as detailed further in Table 1.

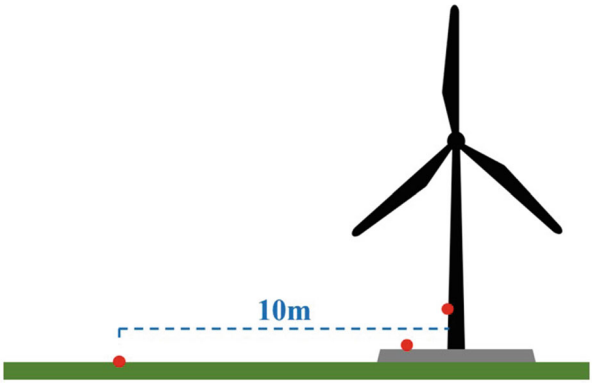



**Fig. 2** Measurement locations at the turbine sample site. Red dots indicate the placement of the accelerometer

**Table 1** Turbine variables at each visit

| | Visit 1 | Visit 2 | Visit 3 | Visit 4 |
|---|---|---|---|---|
| **Wind speed ($ms^{-1}$)** | Tower: 8–8.2<br>Concrete: 8–8.1<br>Soil: 7– 7.7 | Tower: 3.8–4.2<br>Concrete: 4.1–5.4<br>Soil: 3.4–3.9 | Tower: 3.6–5<br>Concrete: 4.1–4.3<br>Soil: 3.8–4.0 | Tower: 7.5–8 |
| **Rotation speed (RPM)** | Tower: 16.3–16.5<br>Concrete: 15.7–16.3<br>Soil: 13.3–15.1 | Tower: 1.4–1.9<br>Concrete: 1.5–1.8<br>Soil: 1.3–1.4 | Tower: 1.9–2.2.<br>Concrete: 1.4–1.7<br>Soil: 0.8–1.2 | Tower: 1.3–1.2 |
| **Tower oscillation (mG) (y)** | Tower: 2.1–3.0<br>Concrete: 4.0–4.3<br>Soil: 2.0–2.5 | Tower: 1.4–1.9<br>Concrete: 1.5–1.8<br>Soil: 1.3–1.4 | Tower: 1.9–2.2.<br>Concrete: 1.4–1.7<br>Soil: 0.8–1.2 | Tower: 1.3–1.2 |
| **Tower oscillation (mG) (x)** | Tower: 3.6–4.8<br>Concrete: 2.4–2.6<br>Soil: 4.4–6.3 | Tower: 3.5–5.9<br>Concrete: 5.2–5.6<br>Soil: 4.4–4.9 | Tower: 4.5–5.6<br>Concrete: 4.0–5.5<br>Soil: 2.4–3.3 | Tower: 0.8–1.0 |

**Table 2** High- and low-frequency settings used

| Frequency | Sample window | Time period (s) | Bin width (mHz) | No. bins | Sample rate | Range (±V) | Coupling | No. samples per time period | Sample interval |
|---|---|---|---|---|---|---|---|---|---|
| **High** | Blackman | 3.277 | 305.2 | 16,384 | 10 k S/s | 2 | AC | 23,764 | 100 μs |
| **Low** | Blackman | 32.77 | 30.52 | 16,384 | 1 S/s | 1 | DC | 32,764 | 1000 S/s |

## Measuring Equipment

Measurements were made using a calibrated Brüel & Kjær accelerometer Type 4370 (sensitivity 1.01 V/ms$^{-2}$), connected to a battery-powered Brüel & Kjær Charge Amplifier type 2635. The amplifier was further connected to a PicoScope oscilloscope (4200 series; Pico Technologies) and an HP Laptop with PicoScope 6 software (PC Oscilloscope version 6.14.44.5870). Data were collected at each location using two different frequency settings (Table 2). These frequencies were chosen based on preliminary data collected which highlighted dominant frequencies between 0 Hz and 500 Hz as well as above 500 Hz. All frequency spectra were made using a Blackman window. Spectral amplitudes were displayed as root mean square (RMS). The cable connecting the accelerometer to the amplifier was covered using tubing. The accelerometer and the cable were taped to the turbine tower or concrete to shield these cables from unwanted noise caused by wind and other disturbances. During soil measurements, the wires were also covered in tubing, and the accelerometer was covered in excess soil. Peaks recorded below 1 Hz are considered to be the result of vortex shedding behind the turbine tower rather than a result of turbine operation.

## Data Analysis

The frequency spectra of each recording were imported into MatLab, using a frequency resolution of 305.2 mHz for high frequencies (1–2 kHz) and 30.5 mHz for low frequencies (see all parameters in Table 2). At each location, recordings were replicated 32 times and averaged across each bin width using MatLab (Ver. 2021; MathWorks) to produce an average spectrum for each location. The acceleration (mms$^{-2}$) spectra were obtained using the Fast Fourier transform (FFT) built-in function in MatLab. These data were then used to calculate the overall average and standard error as a percentage (%). For invertebrate sensitivity comparison, each spectrum was scanned for peak amplitude and frequency. If frequencies (±3 Hz) appeared more than three times, these were chosen as dominant frequencies and selected for further analysis.

Supervisory control and acquisition data (SCADA) were provided by the turbine operator, including 10-minute averages of operational and environmental variables such as wind speed (ms$^{-1}$), rotor speed (rpm), gearbox and generator speed (rpm), turbine oscillation in the x- and y-axis (mG), and active power (kW). Variables are defined in Table 3. Selected data were used to understand changes in vibration associated with turbine variables (i.e., rotor speed, wind speed, and turbine oscillation).

**Table 3** Definition of variables used in SCADA data

| Variable | Definition |
|---|---|
| **Rotation speed (RPM)** | The number of rotations or spins something does in one minute, measured in revolutions per minute |
| **Turbine oscillation (x and y)** | Movement of the turbine structure in x- and y-axis measured in mG (mass and acceleration of gravity) |
| **Active power** | The power output of the wind turbine, measured in kilowatt (KW) |

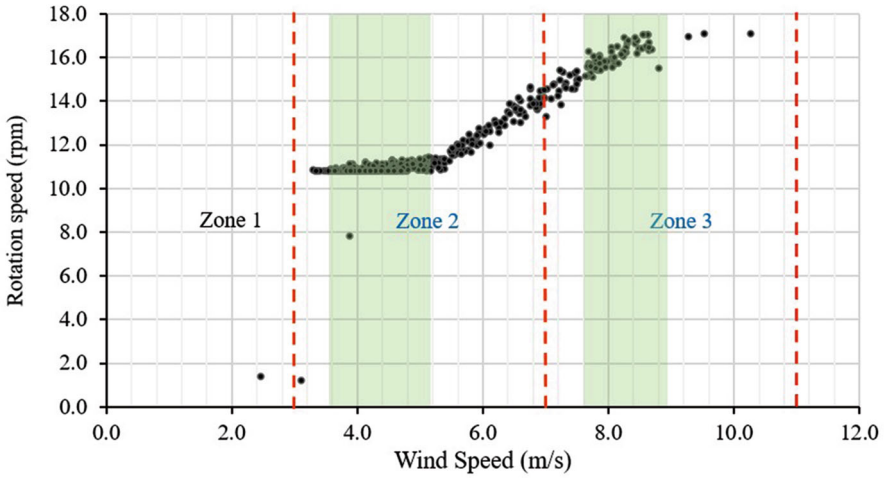



**Fig. 3** Relationship between wind speed and rotation speed of the sampled turbine. Dotted red lines indicate operational zones (see Fig. 1). Green zones indicate the wind speeds where measurements were made

The peak vibration amplitudes at selected frequencies were compared to threshold sensitivities of three invertebrates; the European hermit crab (*Pagurus bernhardus*; Roberts et al. 2016), pill woodlouse (*Armadillo officinalis*; Cividini et al. 2020), and the blue mussel (*Mytilus edulis*; Roberts et al. 2015), to understand if they would perceive the vibrations. These sensitivities indicate the amplitude at which the invertebrate started to show a response to a vibration source. For the mussel and hermit crab, responses consisted of valve closures at 60 $mms^{-2}$ (Roberts et al. 2015) and antennae flicking at 20 $mms^{-2}$, respectively (Roberts et al. 2016). The pill woodlouse displayed avoidance responses at 3.1 $mms^{-2}$, moving away from the vibration source (Cividini and Montesanto 2018).

## Results

### Turbine Vibration Characteristics

The rotation speed (rpm) of the turbine was positively correlated to wind speed (Fig. 3). Rotation remained constant at 10.8 rpm up to a wind speed of 5.8 $ms^{-1}$ (Fig. 3).

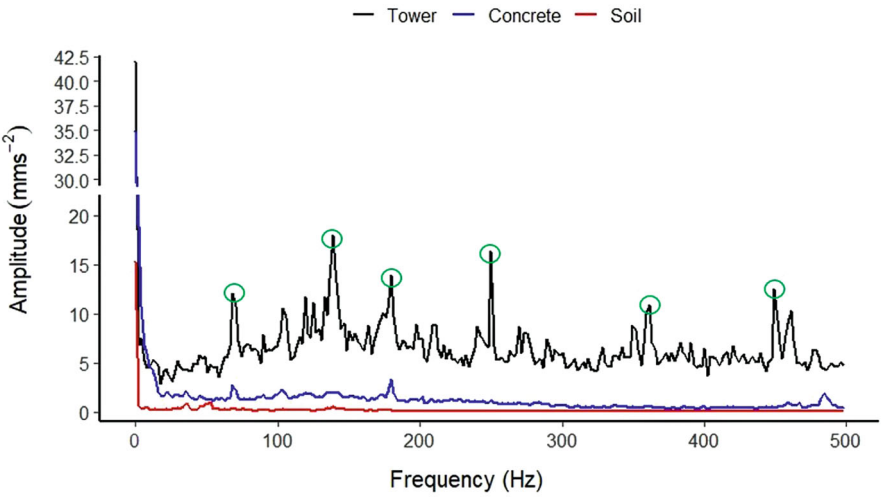



**Fig. 4** Averaged vibration amplitude at the tower (black), concrete (blue), and soil (red), as measured during visit 1. Wind speed was recorded between 7.7 and 8.2 ms$^{-1}$. Standard error ranged between 7% and 19%, 8% and 48%, and 3% and 12% on the tower, concrete, and soil, respectively

## Visit 1: Zone 3

Wind speeds on visit 1 were within zone 3 of the active power plot (Fig. 1), ranging from 7.0 to 8.2 ms$^{-1}$ (Table 1). Rotation speed (rpm) suggests that the turbine was operating at rated, or maximum power (Figs. 1 and 3) at 16.3–16.5 rpm on the tower and ranging between 13.3 and 16.3 rpm during concrete and soil measurements. Tower oscillation provided by SCADA data from the operator is shown in Table 1. Variables are different at each sample location due to wind speed changes when measurements were made at different times on that day. The average vibration measured at the tower had a peak amplitude of 18.1 mms$^{-2}$ at a frequency of 138.6 Hz (Fig. 4). Further prominent peaks are located at 68 Hz, 179 Hz, 250 Hz, 361 Hz, and 449 Hz at 12.1 mms$^{-2}$, 13.9 mms$^{-2}$, 16.4 mms$^{-2}$, 10.9 mms$^{-2}$, and 12.5 mms$^{-2}$, respectively (Fig. 4). Vibration decreased with distance from the turbine, with the dominant peak in the concrete measuring 3.3 mms$^{-2}$ at 179 Hz, while the soil measured at 1.1 mms$^{-2}$ RMS at 52 Hz, comparatively (Fig. 4). The initial peak measured between 0 and 1 Hz was 42 mms$^{-2}$, 34.8 mms$^{-2}$, and 15.3 mms$^{-2}$ at the tower, concrete, and soil, respectively.

## Visit 2: Zone 2

Throughout visit 2, wind speeds ranged from 3.8 to 5.3 ms$^{-1}$ and were within zone 2 of the active power plot (Fig. 1) and rotational speed zones (Fig. 3). Rotation speed remained constant throughout the measurements at 10.8–10.9 rpm. Tower oscillation was highest on the x-axis (back and forth motions) during all three measurements

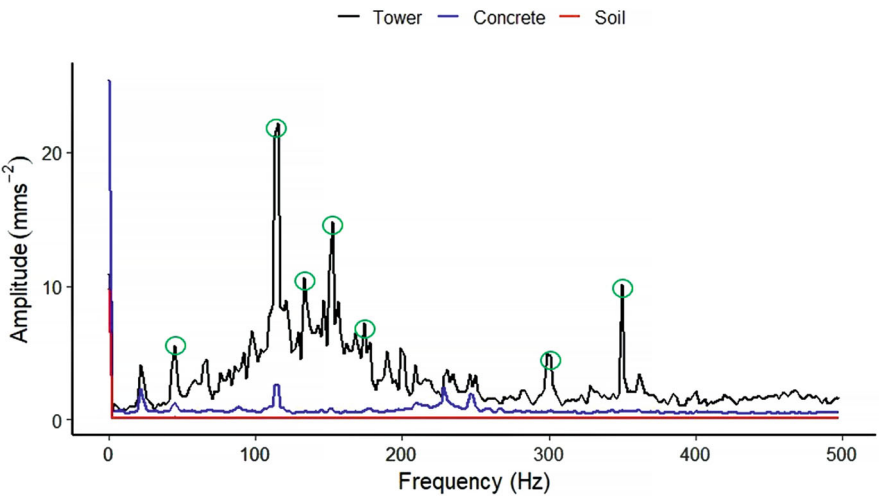



**Fig. 5** Averaged vibration amplitude at the tower (black), concrete (blue), and soil (red) as measured during visit 2. Wind speed varied between 3.4 and 5.4 ms$^{-1}$. Standard error ranged between 0.38% and 2%, 0.1% and 2%, and 3% and 16% on the tower, concrete, and soil, respectively. Peaks circled in green at 45, 114, 132, 152, 173, 298, and 349 Hz

(Table 1), exceeding the oscillations on the other visits. Variables are different at each location due to differences in wind speed as measurements were taken at different times on that day.

During tower measurements, the averaged data showed a peak frequency at 115.2 Hz with vibration of 22.2 mms$^{-2}$. Other prominent peaks were seen at 44, 132, 152, 173, 298, and 349 Hz at 5.5 mms$^{-2}$, 10.6 mms$^{-2}$, 14.7 mms$^{-2}$, 7.2 mms$^{-2}$, 4.9 mms$^{-2}$, and 10.1 mms$^{-2}$, respectively (Fig. 5). Measurements in the concrete during visit 2 showed a peak at 114 Hz at 2.6 mms$^{-2}$. For the soil, vibrations were lower with a peak of 0.08 mms$^{-2}$ at 50 Hz (Fig. 5). The initial peak recorded between 0 and 1 Hz for the tower, concrete, and soil were 10.9 mms$^{-2}$, 25.4 mms$^{-2}$, and 9.7 mms$^{-2}$ (Fig. 5).

## Visit 3: Zone 2

Wind speeds during visit 3 ranged from 3.6 to 5 ms$^{-1}$ and were within zone 2 of the active power plot (Fig. 1) and rotational speed zones (Fig. 3). SCADA data is summarized in Table 1.

Measurements on the turbine tower during visit 3 showed an average peak frequency of 115.2 Hz at 67.1 mms$^{-2}$. This was the highest peak vibration throughout all three visits. Prominent peaks on the turbine tower can be seen at 21, 42, 66, 132, 150, 193, 214, 228, and 449 Hz at 4.3 mms$^{-2}$, 6 mms$^{-2}$, 6.8 mms$^{-2}$, 21.7 mms$^{-2}$, 14.3 mms$^{-2}$, 12.4 mms$^{-2}$, 12.1 mms$^{-2}$, 15.3 mms$^{-2}$, and 9.5 mms$^{-2}$, respectively (Fig. 6). The vibration was smaller on the concrete and soil, with peaks

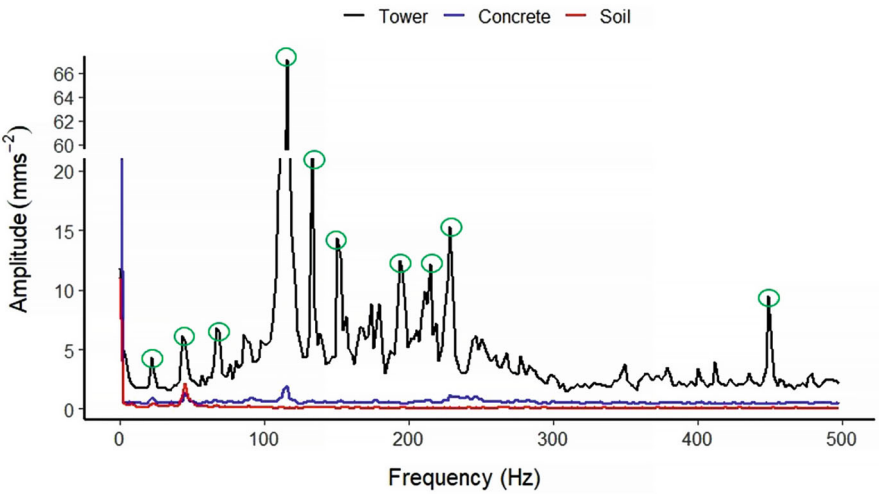



**Fig. 6** Averaged vibration amplitude at the tower (black), concrete (blue), and soil (red) as measured during visit 3. Wind speed varies from 3.6 to 5.0 ms$^{-1}$. Standard error ranged between 3% and 19%, 1% and 22%, and 1% and 23% for the tower, concrete, and soil measurements, respectively, displayed in acceleration units (mms$^{-2}$)

found on the concrete at 42, 114, and 232 Hz, measuring 0.68 mms$^{-2}$, 1.7 mms$^{-2}$, and 1.07 mms$^{-2}$, respectively (Fig. 6). The initial peak seen between 0 and 1 Hz measured at 11.7 mms$^{-2}$, 33.5 mms$^{-2}$, and 11.04 mms$^{-2}$ for the tower, concrete, and soil.

## Visit 4: Control Measurements

During the control measurement, SCADA data were still provided, and wind speeds ranged between 7.5 and 8.0 ms$^{-1}$, therefore within Zone 3 (Fig. 1). The rotation speed was at zero due to the turbine being switched off. This is summarized in Table 1.

Recordings during the downtime period show peak frequencies of 100, 200, 300, 400, and 450 Hz at 1.7 mms$^{-2}$, 1.5 mms$^{-2}$, 1.1 mms$^{-2}$, 1.2 mms$^{-2}$, and 0.5 mms$^{-2}$, respectively (Fig. 7). The initial peak between 0 and 1 Hz measured at 4.4 mms$^{-2}$.

## Comparison with Invertebrate Sensitivities

There is some overlap between the sensitivity for *A. officinalis* and the vibration amplitudes measured on the turbine tower and on the concrete base of the tower, while overlaps for *P. bernhardus* sensitivity appear only for measurements made on the turbine tower (Fig. 8). For *M. edulis* sensitivity, there appears to be no overlap

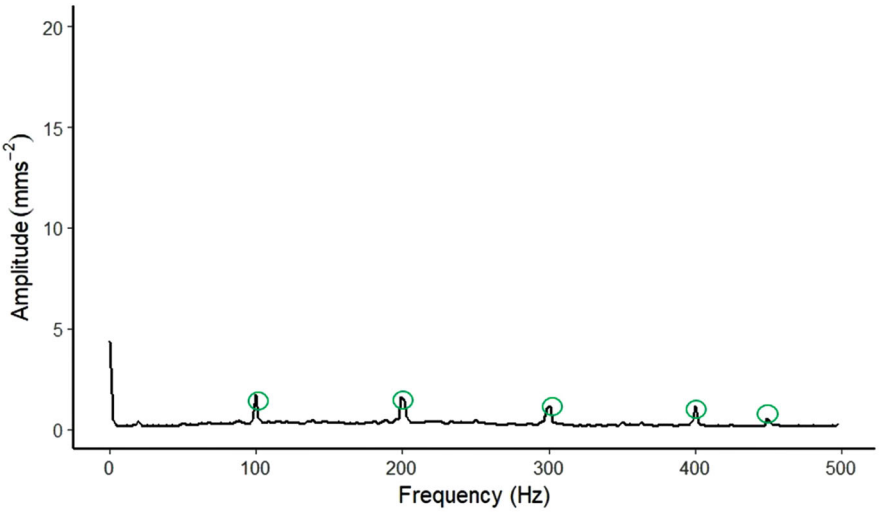



**Fig. 7** Averaged vibration amplitudes measured on the turbine tower during downtime (control measurement). Wind speed was measured at 7.8 ms$^{-1}$ and the standard error was between 3% and 41% of the mean. Green circles indicate peaks at 100, 200, 300, 400, and 450 Hz, displayed in acceleration units (mms$^{-2}$)

with the majority of the measured vibration amplitudes. Three outliers in surface vibration were recorded exceeding the threshold sensitivity of *M. edulis*. These were measured on the turbine tower at 114 Hz (78 mms$^{-2}$), 179 Hz (60 mms$^{-2}$), and 228 Hz (62 mms$^{-2}$), respectively (Fig. 8).

## Discussion

### Turbine Vibration Characteristics

Turbine vibration decreased with distance, with the highest vibration recorded at the turbine tower (maximum value 91.3 mms$^{-2}$ at 114 Hz) and the smallest measured in the soil (maximum value 2.1 mms$^{-2}$ at 50 Hz). Measurements in the soil peaked at around 50 Hz, potentially owing to ambient electrical noise picked up by the cables connecting the accelerometer to the amplifier. Overall, the highest tower vibration was recorded when wind speed was relatively low (3.6–5.0 ms$^{-1}$). This may be related to the turbine variables, such as tower oscillation, which were higher at the lower wind speeds. For example, the highest tower vibration of 67.1 mms$^{-2}$ was measured on visit 3, at a wind speed of 3.6–5 ms$^{-1}$ and a tower oscillation of 4.5–5.6 mG in the x-axis, 1.9–2.2 mG in the y-axis (Table 1). The higher x oscillation (back and forth motions) measured on the tower may be the cause of the increase in turbine vibration, due to the flexibility of the tower and the motion of the blades causing blade or tower vibrations (Staino and Basu 2015). At lower wind

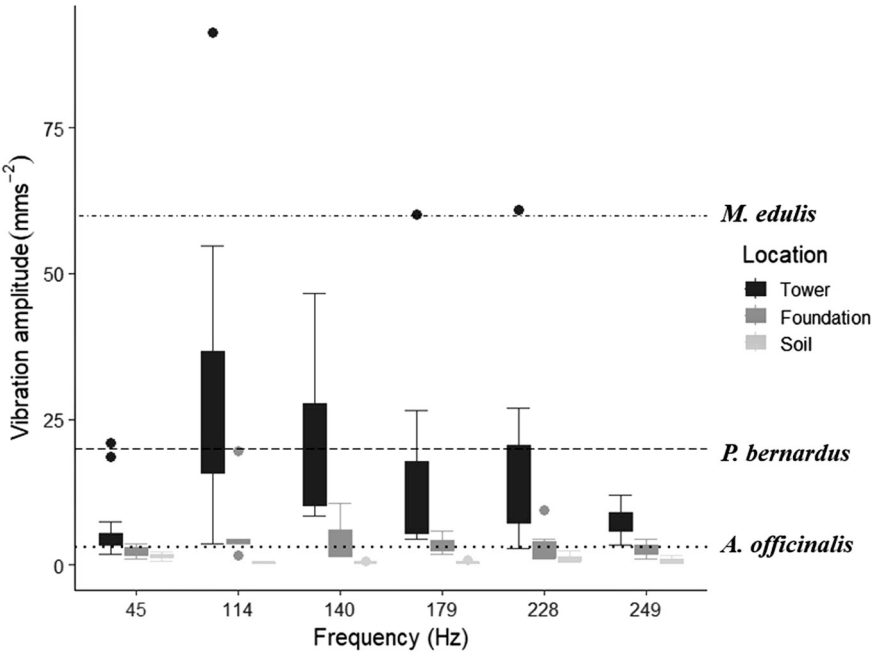



**Fig. 8** Dominant frequencies and associated vibration at the turbine tower (black), on the concrete (dark gray), and in the soil (light gray). Lines indicate sensitivities of invertebrates at 3.1 mms$^{-2}$ for the pill woodlouse *Armadillo officinalis,* 20 mms$^{-2}$ for the hermit crab *Pagurus bernhardus*, and 60 mms$^{-2}$ for the mussel *Mytilus edulis.*

speeds, the blades are fully pitched into the wind (Petrović et al. 2012). Therefore, vibrations may be higher as the turbine attempts to enhance energy capture. Tower oscillation was similarly strong during the second visit, where wind speeds ranged from 3.4 to 5 ms$^{-1}$ (Table 1) and oscillations were recorded in the ranges between 1.4 and 5.9 mG in the x- and y-axis. Peak vibration amplitudes differed with their frequency ranges on visit 1 (138 Hz) when wind speed was highest (8–8.2 ms$^{-1}$) compared to visits 2 and 3 (114 Hz), when wind speed was lower (3.4–5.4 ms$^{-1}$) (Figs. 5 and 6). Guo et al. (2017) studied the effects of gearbox fault signals, as they are particularly difficult to detect. Using their proposed methodology, it was suggested that a frequency change from 130 to 114 Hz corresponded to a faulty gearbox (Guo et al. 2017). However, there were no faults detected or noted on the days of measurements. The higher vibration found at lower wind speeds was similar to that found by Sigray and Andersson (2011) but differed from the findings of Escaler and Mebarki (2018), where a turbine of similar size showed an increase of vibration at higher wind speeds. However, Escaler and Mebarki (2018) measured the mechanical equipment within the nacelle, such as on the gearbox and generator, rather than on the tower itself. Sigray and Andersson (2011) noted that a new gearbox had been installed prior to their measurements, which may be the reason for lower vibration recordings at higher wind speeds than in other studies.

Between 0 and 1 Hz, amplitude peaks measured in this study appear to be higher when wind speed was faster because the turbine structure can create vortices on both sides of the tower when a flow passes around the structure (Viré et al. 2020). This vortex shedding creates pressure differences between wind facing and the opposite side and leads to movements of the tower (oscillations) (Viré et al. 2020). Oscillations at this frequency were higher when wind speeds were lower. The vortices may produce the same frequency as the natural frequency of the tower, leading to resonance and a larger vibration (Miceli 2020). This can be calculated using $f = St\ x\ U/D$, where St is the Strouhal number (0.2), U is the wind speed, and D is the diameter of the turbine. For example, at lower wind speeds (3.4–5.4 $ms^{-1}$) during visit 2, the vortex shedding frequency can be calculated as 0.22 Hz, within the region of the frequency peak found at the lower frequencies (Fig. 5). At the higher wind speeds of 8–8.2 $ms^{-1}$, the vortex shedding frequency can be calculated as 0.4 Hz, which can also be seen as a peak within Fig. 4. Similarly, Sigray and Andersson (2011) found that environmental variables such as waves appeared to dominate the spectrum below 4 Hz, with noisier signals occurring when waves were stronger. While there appears to be a positive correlation between wind speed and rotor speed (Fig. 3), wind speed and vibration show a negative correlation. The vibration of the turbine may be more closely related to the turbine attempting to maximize energy capture, with vibration dampening methods (such as blade pitch) coming in after energy capture is more stable at higher wind speeds. Offshore, the turbines are considerably larger, with rotor diameters reaching 222 meters (Siemens Gamesa Renewable Energy 2023), compared to 83 meters in this study.

## Sensitivity of Invertebrates to Turbine Vibration

The highest amplitude peaks were recorded at the tower, with absolute minimum and maximum values falling between 1.91 $mms^{-2}$ (50 Hz) and 91.3 $mms^{-2}$ (114 Hz). While 90% of the data fell under 54 $mms^{-2}$, below the perception threshold of the blue mussel, *M. edulis* (60 $mms^{-2}$) (Roberts et al. 2015), a few data points were within this sensitivity range. The data measured at the tower did overlap with the threshold of hermit crabs, *P. bernhardus* (20 $mms^{-2}$) (Roberts et al. 2016), and the woodlouse, *A. officinalis* (3.1 $mms^{-2}$) (Cividini and Montesanto 2018) (Fig. 8). However, tower vibration is not relevant for these species that would be found on or within the sediment surrounding the turbine rather than at the turbine tower. At the foundation, measurements were in the sensitivity threshold of *A. officinalis* (Cividini et al. 2020). It is expected that these values on the concrete would be undetectable by *M. edulis* and *P. bernhardus* due to being below threshold sensitivity values (Roberts et al. 2015, 2016). Measurements in the soil were lower than the sensitivity thresholds for all three species, and therefore it can be assumed that these would be undetectable by all three species. It is important to consider that these measurements were made on a smaller turbine than what is currently offshore, where a hub height of 83 m as in this study may have less structural loads and vibration than that of a 220 m offshore turbine. Therefore, vibrations on modern turbines may be larger and

overlap with the sensitivities of all three example invertebrates. This is supported in literature; larger turbines are often more flexible, leading to increased challenges such as greater vibrations from the blades and the tower (Staino and Basu 2015). However, submergence of the turbines in the water, due to the fluid's higher inertia, may dampen the vibration. This may explain why Sigray and Andersson (2011) recorded lower particle motion recordings (maximum value at 20 $mms^{-2}$), when measuring at a 1 m distance in the water column Their results have highlighted that turbine-induced water particle motion overlapped with the sensitivity of fish species, whose sensitivities are comparable to that of the invertebrates in this study. For example, a juvenile roach displayed startle responses at 23 $mms^{-2}$ (Karlsen et al. 2004), and a Cyprinid species showed avoidance responses to particle motions as low as 1 $mms^{-2}$ (Sonny et al. 2006).

Any overlapping of turbine vibration with invertebrate threshold sensitivities may lead to behavioral changes. The studies of both *P. bernhardus* and *M. edulis* showed changes in behavior when exposed to substrate vibrations at 20 $mms^{-2}$ and 60 $mms^{-2}$, respectively. For *P. bernhardus*, these behaviors included lifting of the shell from the substrate and antennae flicking (Roberts et al. 2015), while *M. edulis* displayed behaviors such as valve closure. Valve closures in the mussel, even if only partial, require more energy expenditure due to the use of the adductor muscles and a reduction in filter-feeding (Tang and Riisgård 2016). These partial closures are present even at levels below the threshold amplitudes measured by Roberts et al. (2015) as they are undetected in visual observations. Together, the potential underestimation of *M. edulis* sensitivity and the occasionally elevated levels of tower vibrations, indicate that blue mussels attached to offshore turbine towers may at least temporarily be exposed to conditions of high stress. Their permanence at these extreme locations indicates that they appear to be able to cope with these conditions. Spiga et al. (2016) suggested that mussels increased their filtration rate when exposed to simulated impulsive pile driving sound, while Wale et al. (2019) found that mussels decreased their filtration rate when exposed to continuous ship noise. This change in filtration rate may be due to an increase in metabolic rate (Spiga et al. 2016), which is also seen during other types of stressors such as in lead-polluted waters (Pessatti et al. 2002). Other studies suggest that mussels respond to the onset of pulse trains, but little response was seen in sequential pulse trains (Hubert et al. 2022), suggesting sensory adaptation or habituation. This variability in responses to anthropogenic sounds highlights the importance of further work on both species and sound types. While an antennae flicking response for *P. bernhardus* may not have a direct response on the individual and the local population, it may lead to the animal becoming distracted and therefore reducing fitness (distracted prey hypothesis) (Chan et al. 2010). This has been documented in terrestrial hermit crabs, whereby *Coenobita clypeatus* were exposed to boat noise (sound pressure) and simulated predators could get closer to the crab than those that were not exposed to noise (Chan et al. 2010). The terrestrial pill woodlouse (*A. officinalis*) is more sensitive, displaying a change in behavior, such as alternating turns (Cividini and Montesanto 2018), and an avoidance response to the stimuli when exposed to substrate vibrations at 3.1 $mms^{-2}$ (as recorded in the concrete measurements) (Cividini et al. 2020).

While predation change may not relate to the mussels who are sessile and attached to the turbine towers, more mobile invertebrates such as other crustaceans may also show sensitivity to turbine vibration, and therefore a potential reduction in their anti-predator response, such as that seen in the distracted prey hypothesis (Chan et al. 2010), potentially leading to population-level changes in the longer-term.

While the threshold sensitivities overlap with measurements made at the turbine; tower vibration is only relevant to *M. edulis*, given their attachments to offshore turbine towers. For *P. bernardus* and *A. officnalis*, measurements made on the concrete and the soil would be most relevant. While *A. officinalis* is not found offshore, this species highlights the variability in invertebrate sensitivities (20 times more sensitive than *M. edulis*). Even within marine species, *M. edulis* has a sensitivity three times lower than *P. bernhardus*, meaning other invertebrates offshore may be more or less sensitive to offshore vibrations. Velilla et al. (2021) suggested there was a positive correlation with turbine vibration levels and proximity to the turbine, similar to the findings in this study. They also suggested that onshore turbine vibration reduced earthworm abundance, decreasing with proximity to the turbine and increasing vibration levels. Threshold values do not currently exist for the earthworm; however, there is existing research surrounding other polychaete responses to vibration, such as that of artificial grunting using seismic vibrations below 500 Hz (Catania 2008; Mitra et al. 2009), and bird foot vibrations to hunt for polychaete prey through observations (Pienkowski 1983). Based on the low vibration measured in the soil during this study, it is unlikely that turbine vibrations are strong enough to result in population-level changes as suggested by Velilla et al. (2021). Vibrations in the soil were too low to be detected by even the most sensitive invertebrate, *A. officinalis*, studied so far (Cividini et al. 2020).

The variation in invertebrate sensitivities seen here from 3.1 to 60 $mms^{-2}$ demonstrates the complexity when considering the effects of vibration on invertebrate species. While this study suggests that there may be behavioral changes in two of the three invertebrate species, greater level changes may be seen in other species. The wide range of threshold sensitivities, between the species used in this study, highlights the difficulty in concluding how turbine vibration may impact other marine invertebrates. Offshore, other stressors will excite a turbine more than onshore, for example, waves and stronger winds. Wind speed increases with height, therefore, the larger offshore turbines will experience greater loads and vibration, likely in the lower frequencies (vortex shedding frequency) (Rueda-bayona et al. 2023). These environmental variables further add to the difficulty of measuring vibration impacts on marine species offshore. The offshore wind industry is now looking to utilize deeper waters, moving from fixed bottom structures, such as the one used in this study, to floating structures. These novel turbines will introduce new vibratory characteristics into the marine environment, mostly owing to the mooring systems (Feng et al. 2023), which also require research and understanding (Zhang and Høeg 2018). The current study highlights what kind of measurements are needed on offshore wind farms to draw any conclusions about the ecological impact on the communities associated with windfarms.

## Conclusion

The results from this study show that substrate vibration levels on the turbine, concrete, and soil from a 2 MW turbine are unlikely to influence the three invertebrates based on their sensitivity threshold values, apart from a few outliers. The outliers on the turbine tower did overlap with the sensitivity of *M. edulis,* which is particularly relevant given their attachment to the towers. Vibration in the soil at a 10 m distance was below all threshold values and it is expected that the seismic vibrations traveling through the soil are unlikely to affect species living on or within the sediment. Substrate properties at offshore turbines need further consideration, as well as more information on threshold values of other marine invertebrate species, given the high variability in sensitivity values used in this study. This study is a start to understanding how marine invertebrates may be impacted by wind turbine developments and their associated substrate vibrations. Further cumulative and longer-term understandings are required to fully assess the impacts.

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# Anthropogenic Noise Impacts on Invertebrates: Case of Freshwater Red Cherry Shrimp (*Neocaridina davidi*)

93

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**Abstract**

Sound-generating human activities have considerably increased ambient noise levels in marine and freshwater ecosystems across the globe. Anthropogenic, human-made noise is an issue of international concern and rising in ambient sound levels may have negative consequences on aquatic animals at the individual and community level. Anthropogenic noise can negatively affect the behavior of aquatic animals in several ways. While some studies have addressed the effects of noise on fish species, there are considerable gaps in our understanding of the effects of noise more widely, especially on invertebrates. Our interest lies in the effect of noise on the "lower levels of food web members," i.e., small crustaceans such as the red cherry shrimp (*Neocaridina davidi*), as model for the link between

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trophic levels. The red cherry shrimp is a gregarious species, abundant in freshwater habitats, and native to Asia. The red cherry shrimp has economic importance in the ornamental trade and ecological threat as a potential invasive species; however, its behavioral performance in response to anthropogenic noise is unknown. This chapter details the effect of sound exposure on shrimp under laboratory conditions. The findings revealed that noise impact assessments on invertebrates are critically important for understanding how anthropogenic noise alters food webs and reverberates through communities.

**Keywords**

Decapods · Noise impacts · *Neocaridina davidi* · Community · Food web

## Introduction

Sound is an important cue for many terrestrial and aquatic animals that can provide crucial information about their habitats. Underwater sound with low attenuation over large distances has quite high propagation rates and allows long-range transmission of information in the aquatic environment (Rogers and Cox 1988; Bass and Clark 2003). Sound travels almost five times faster in water than in air; therefore, it can be a useful and effective tool to transfer information in the aquatic environment (Slabbekoorn et al. 2010). Therefore, soundscapes play vital roles in the environment for transferring intricate biological signals and conveying specific information to aquatic animals for communications within and across taxa (i.e., fish, invertebrates, and marine mammals) (Ladich 2013; Putland et al. 2019; Tyack 1998). Anthropogenic activity in aquatic habitats has increased, causing environmental pollution across different modalities such as anthropogenic noise, ocean acidification, and chemical pollution, which in turn elevate ambient noise levels and enhance low-frequency sound propagation as the ocean becomes more acidic (Hildebrand 2009; Brewer and Hester 2009).

## Anthropogenic Noise Is Ubiquitous in Aquatic Habitats Worldwide

Anthropogenic activities are frequently linked to acute and chronic detrimental changes of the sensory environment in terrestrial and aquatic habitats (Swaddle et al. 2015; Wilson et al. 2020). Aquatic environments have been increasingly exploited and are exposed to anthropogenic noise-generating human activities (Slabbekoorn et al. 2010; Radford et al. 2014). Industrial developments and commercial human activities have become more frequent and have increased ambient sound levels in aquatic ecosystems, which in turn may expose aquatic organisms to a wide range of physiological and behavioral impacts (Hawkins et al. 2015; Duarte et al. 2021). Globally, anthropogenic noise sources are diverse, each with specific temporal noise features and acoustic components. Low-frequency sound pressure propagates over long distances with little

energy loss (Caruthers 1977). Although, anthropogenic noise sources have been previously examined in the scientific literature (Hawkins et al. 2008; Hawkins and Popper 2014; Popper and Hawkins 2016), there are massive information gaps in our bioacoustics knowledge, particularly on the mechanisms and impacts of anthropogenic noise on aquatic organisms across taxa (Popper and Hastings 2009; Hawkins et al. 2015). Anthropogenic noise outputs can potentially interfere with environmental information processing and resource exploitation of many aquatic organisms, therefore disrupting sensory processes and the utility of acoustic information for aquatic vertebrates and invertebrates (Popper and Hastings 2009; Slabbekoorn et al. 2010). Aquatic animals exploit sounds in the environment for many purposes. They can use sound to display courtship behavior and attract mates, detect prey, and avoid predators, all necessary adaptations that are strongly correlated with growth, reproductive success, and survival (Slabbekoorn et al. 2010). Behavioral observations can provide information on the impact of anthropogenic noise on aquatic animals. Behavioral activities such as swimming performance, movement, and spatial displacement, alongside other behavioral changes in response to noise, are research priorities for anthropogenic noise effects on fish and invertebrates (Popper et al. 2022a).

## The Importance of Aquatic Invertebrates in Food Webs

"Invertebrates" represent almost more than 95% of extant animals with enormous morpho-functional and ecological diversity within each phylum and more than 30 phyla in total (Brusca et al. 2022). They occupy all known niches in marine, freshwater, and terrestrial habitats, with a mass of ecological adaptations. However, little is known or understood about their adaptive responses to changes in ambient, anthropogenic noise (Schierwater and De Salle 2021; Brusca et al. 2022). In addition to their vast biodiversity (e.g., number of species, number of taxa), invertebrates have complex life cycles that, for many species, include agonistic interactions, courtship and mating, brooding (maternal care), larval, postlarval stages, and complex patterns of metamorphosis that could be vulnerable to anthropogenic activities such as man-made noises (Wale et al. 2021 and references therein). The large diversity of invertebrates also implies differences in the role of each species within aquatic marine, brackish, or freshwater food webs, and thus, noise impacts on some species may affect the integrity of ecological interactions within these systems (Elliott et al. 2007; Compson et al. 2019). In addition to this, many species live within a benthic habitat, either as adults and/or as larvae, and benthic invertebrates are particularly recognized to play a key role in sediment dynamics as "ecosystem engineers," but they may be particularly vulnerable to low-frequency noise (Popper et al. 2022b; Wang et al. 2022). In fact, invertebrates still only represent a small proportion of animals studied within the marine noise literature (Popper et al. 2022a; Wale et al., 2021) despite pioneer studies recognizing invertebrates as vulnerable species for anthropogenic noise (Morley et al. 2014). This highlights the necessity of further laboratory and field studies specifically involving the study of invertebrate responses to noise environments.

## Noise Impacts on Invertebrates

Empirical data, based on behavioral studies, demonstrate an expanding functional role of anthropogenic noise in the lives of aquatic animals among many taxa including fish and marine mammals (Slabbekoorn et al. 2010; Hawkins and Popper 2014). However, most recent behavioral studies explored such impacts and still speculation and anecdotal observations continue to play a dominant role (Myrberg 1997; Popper et al. 2022b). Invertebrates detect kinetic sound energy in the form of particle motion (Popper et al. 2001; Popper and Hawkins 2018; Dinh and Radford 2021), and recent evidence suggests they also respond to vibration stimuli (Roberts et al. 2016; Roberts and Howard, 2022; Roberts and Wickings 2022). Crustaceans, and invertebrates in general, are thought to benefit from mechanoreceptor organs such as the setae cells, hair-like structures that detect particle motion components of sound (Weatherby and Lenz 2000; Popper et al. 2001). Most aquatic invertebrates "hear" using three types of sensory organs including chordotonal organs, internal statocysts systems, and superficial body receptor systems (Breithaupt and Tautz 1988, 1990; Breithaupt 2002). There is also evidence that some crustacean zooplanktons in large swarms can produce sounds (Kühn et al. 2022) and these acoustic signals can be used for intra- and interspecific communication (Slabbekoorn et al. 2010; Simpson et al. 2016). However, relatively little information is known about the effects of anthropogenic noise on the behavior of aquatic invertebrates in marine habitats (Fewtrell and McCauley 2012; Olivier et al. 2022). Recent research has revealed a range of behavioral impacts of noise on marine invertebrates including molluscs (Mooney et al. 2016; Wale et al. 2019; Jones et al. 2021), crabs (Wale et al. 2013; Roberts et al. 2016), and shrimps (Spiga 2016; Ruiz-Ruiz et al. 2019). Despite the huge diversity of freshwater invertebrates, even less empirical information on this issue exists for these taxa. Azarm-Karnagh et al. (2022) demonstrated the capacity of freshwater red cherry shrimp for low-frequency sound detection (400–2000 Hz). In a laboratory-based experiment, the authors observed immediate behavioral changes both in terms of movement activities, spatial distribution, and feeding performance when exposed to acoustic stimuli using an underwater speaker (Fig. 1).

Clear, detailed accounts of experimental setups, including properties of the acoustic environment applied, are important for reproducibility in science and to facilitate enhanced interpretation of behavioral observations and research findings, particularly when contradictory sound-related behavioral responses occur. A previous study on a freshwater zooplankton, the water flea (*Daphnia magna*), used an in-air speaker to broadcast continuous broadband noise (300–1500 Hz) (Shafiei Sabet et al. 2015). Although behavioral changes in response to acoustic stimuli were observed for vertebrate animals (in this case, the zebrafish, *Danio rerio*), no sound-related behavioral changes were observed for water fleas. In another study, in-air speakers were used to ensonify a water flea tank (Shafiei Sabet et al. 2019) with continuous broadband noise (400–2000 Hz); however, no statistically significant behavioral changes were observed for the fleas. Despite this, authors found some evidence (a trend) of sound-related behavioral changes (number of hops), suggesting that water flea can detect acoustic stimuli and respond behaviorally to

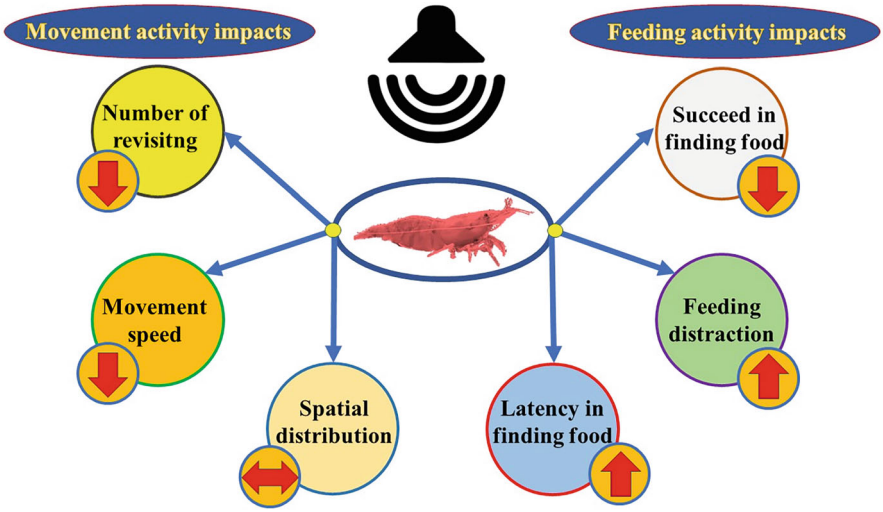



**Fig. 1** Behavioral changes of the freshwater red cherry shrimp in response to an underwater speaker. Behavioral responsiveness of shrimps when exposed to acoustic stimuli was categorized as movement activity impacts (**a**) and feeding activity impacts (**b**). For the movement activity impacts, a clear reduction occurred in the number of food source revisits, a reduction in movement speed, and avoidance of the noisiest area close to the active underwater speaker. For the feeding activity impacts, there was an increased latency to find food and feeding distraction also reduced success in locating food

it. The frequency range of auditory sensitivity for invertebrates varies (Morley et al. 2014), and no data is currently available for water flea. The main conclusion was that sound exposure levels in the experiment were not enough loud to trigger sensory mechano-receptors in the water fleas, or the frequency of sound exposure was beyond their detection range (Shafiei Sabet et al. 2015).

Recent complementary studies have further advanced knowledge on the same genus *Daphnia,* revealing that water fleas are prone to be affected by anthropogenic noise. Prosnier et al. (2022a), using an underwater speaker, found impacts of continuous broadband noise (100–20,000 Hz) on water fleas; reducing mobility and swimming activity under noisy conditions. The observed impacts on swimming speed and hopping behavior found in Shafiei Sabet et al. (2019) and Prosnier et al. (2022a) may indicate that slower individuals save energy in order to prioritize survival-relevant activities. Similarly, a reduction in swimming activity and mobility is also pronounced when water fleas are infected by parasites and potentially may be more visually profitable by predators to detect them easier (Prosnier et al. 2022b). On the other hand, increased swimming activity provides more opportunities for finding food, avoiding predators, and prioritization of their diel migration activities (Larsson and Kleiven 1996; Dodson et al. 1997; O'Keefe et al. 1998; Roozen and Lürling 2001; Chang and Hanazato 2003). A trade-off between mobility and fitness has already been reported for water fleas (Prosnier et al. 2022a). Therefore, these large variations in invertebrate behavioral responses (Wale et al. 2013; Morley et al. 2014)

support the need for further investigations and complementary studies on water fleas and other aquatic invertebrates.

In small tank environments, underwater sound fields can be complex (Campbell et al. 2019). It is important to consider that sound-related behavioral changes may be difficult to extrapolate to determine real-world impacts, both in terms of behavioral validations and acoustical complexity (see Fig. 2) from laboratory-based experiments. This may be further compounded by factors such as differences in air or underwater speakers, sound sources, sound-level variation in sampling techniques, and other method-related limitations (Fewtrell and McCauley 2012).

All fish species can detect particle motion, and some have special structures to also perceive sound pressure in their habitats. It is most likely that all sound-detecting invertebrates acquire acoustic information through the detection of particle motion and/or substrate vibration (Nedelec et al. 2016; Popper and Hawkins 2018). There is evidence that marine invertebrates, in taxa other than crustaceans, also exploit environmental soundscapes to disperse in search of suitable habitats, e.g., the Australian flat oyster (*Ostrea angasi*) (Williams et al. 2022) and coral larvae (Lecchini et al. 2018). We suggest that findings of behavioral changes in response to sound in freshwater red cherry shrimp of the present study are due to particle motion components of the acoustic stimuli used. The implication is that aquatic animals who fail to perceive and detect important acoustic information about their environment, because of distraction or the masking of relevant biological acoustic cues, may experience significant impacts to their overall performance, biological activities, and their ultimate survival (Slabbekoorn et al. 2018; Carroll et al. 2017).

## Gaps in Noise Impacts on Invertebrates: The Need to Consider Particle Motion and Vibration

All fish can detect particle motion using their inner ear and some species that have air filled cavities (e.g., swim bladders) (Slabbekoorn et al. 2010; Popper and Fay 2011). They are also able to perceive sound pressure levels. There is growing evidence in the literature that invertebrates detect sound and response to sound (Wale et al. 2013; Spiga 2016; Hill et al. 2019). In comparison to fish, invertebrates are more likely to be able to perceive low frequencies and particle motion components, and recent

| | | | | | | |
|---|---|---|---|---|---|---|
| 13.33<Y<20 | 101.22 | 99.84 | 99.28 | 101.28 | 104.67 | 109.25 |
| 6.66<Y<13.33 | 103.42 | 98.67 | 100.39 | 99.9 | 102.88 | 107.88 |
| 0<Y<6.66 | 101.12 | 101.37 | 99.67 | 101.91 | 102.06 | 102.44 |
| Y/X | 0<X<4.16 | 4.16<X<8.33 | 8.33<X<12.48 | 12.48<X<16.64 | 16.64<X<20.80 | 20.80<X<25 |



**Fig. 2** Soundscapes of the experimental setup. Rectangular tank visualization (frontal view) for frequencies between 400 Hz and 2000 Hz ensonified by an underwater speaker placed on the right side in the tank. Numbers in colored squares show sound pressure levels (SPL, dB re 1 μPa). The higher SPLs, indicated by stronger color densities, occur close to the active speaker. Measurements were conducted in the center of each square

studies have revealed that they can also detect sediment vibrations and respond accordingly to these stimuli. However, details regarding their sensitivity to particle motion or vibration, as well as the importance and consequences of their responses, are not well known (Roberts and Elliott 2017; Roberts and Wickings 2022). With progress in technology and the development of new sensory detection devices, the measurement of particle motion sound components and substrate surface vibrations is becoming more achievable. Such advances in technology and computer processing can shed further light on mechanisms that influence invertebrate behavioral change in response to acoustic stimuli (Hill et al. 2019).

Vibration sensitivity has been tested in a wide range of invertebrate species across many taxa (Budelmann 1992), as well as some vertebrates, such as bottom-dwelling fish species (e.g., flatfishes) (Berghahan et al. 1995). Despite this, research relating to the potential effects of vibrational noise is sparse and only a few recent studies have explored the effect of sediment vibration stimuli on behavioral responses of aquatic organisms, especially invertebrates. In a recent review, it was highlighted that less than 25 species, representing different taxa, have been tested across environments and these are primarily bivalves and crustacean species (Roberts and Howard 2022). The behavioral observations of red cherry shrimp suggest they avoid or escape from noisy areas in response to acoustic stimuli, and spend less time close to the active speaker with the noisiest sound environment (Azarm-Karnagh et al. 2022). They also exhibited feeding problems in the noisy area; experiencing more failures to find food sources, making fewer revisits to the food source, and displaying greater distraction in noisy conditions.

We suggest that by providing controlled settings that limit confounding factors within experimental design, laboratory-based research is still essential for exploring and answering fundamental research questions around behavioral changes and activity patterns in response to anthropogenic noise (Slabbekoorn 2016). One challenge in laboratory-based research on this topic is the ongoing debate over the validity of measuring acoustic fields within artificial environments. Fish tanks are acoustically complex as tank walls act as a secondary noise source and distort natural sound propagation patterns (Parvulescu 1967; Van den Berg and Schuijf 1985; Akamatsu et al. 2002; Gray et al. 2016; Rogers et al. 2016; Shafiei Sabet and Alizadeh Lademakhi 2022). Therefore, to ensure behavioral relevance, validity, and acoustically relevant results, it is necessary to also conduct field experiments that assess the most relevant and perceivable sound components when conducting behavioral experiments on invertebrates.

Currently, well-replicated experiments have shown two invertebrate species from different taxa, blue mussels (*Mytilus edulis*), and hermit crabs (*Pagurus bernhardus*) that are sensitive to sediment vibrations and will change their behavior in response, by, respectively, closing their shells and changing their movements after exposure to substrate-borne vibrations (Roberts et al. 2015, 2016). Vibration sources (i.e., wind turbine structures) are not only limited to the ground and substrate-borne vibrations at the sediment level of habitats (Popper et al. 2022b), but even sounds produced during seismic surveys may indirectly translate into the substrate in the aquatic environment (Hawkins et al. 2021). Auditory stimuli and acoustic perceptions play

vital roles for many aquatic species, depending on their adaptive abilities to perceive and process those acoustic signals in their daily life. Unlike sound pressure, particle motion conveys substantial information about the sound and its traveling direction (Popper et al. 2022a, b). Therefore, it has been suggested that hearing sensitivity studies for a range of animal taxa should incorporate, not only sound pressure, but also particle motion and substrate acoustical vibration (Popper et al. 2022b). Such studies do exist for marine invertebrates, but comparatively little is known about freshwater invertebrates.

We also call for the need to study more complex responses and behaviors that are directly involved for reproductive successful and survival (e.g., courtship and mating interactions, brooding and maternal care, burrowing, and other activities as ecological bioengineers). It is also necessary to prioritize the exploration of noise impacts on behavior at early life stages and during periods of large morphological and physiological changes in invertebrates (i.e., larval and postlarval swimming and feeding behavior, metamorphosis, and settlement-stage). Finally, it is important to emphasize the need for more study of invertebrate taxa and food webs as integrative units, both under environmentally controlled laboratory conditions and under real field conditions, in order to fully understand the impacts of underwater anthropogenic noise on invertebrates, at both the individual and community level (Shafiei Sabet et al. 2016).

## Conclusion

In a recent study, the anthropogenic noise effects on a freshwater red cherry shrimp in captivity. Although some behavioral changes (avoidance of noise sources and increased feeding disturbance) in response to acoustic stimulus were identified, we speculate that those changes are related to particle motion components of acoustic stimuli and potentially also vibrational noise at the substratum of the fish tank. Assessment of sound pressure, particle motion, and substrate-borne vibration on this species, simultaneously, are suggested in terms of research priority to quantifying acoustic components at the precise location of the animal. In this way, more empirical assessments of both underwater acoustic properties and behavioral response interpretations will be addressed. Careful behavioral observation with validated experimental design, controlled methodological approaches, by taking into account the complex nature of underwater acoustics, and explicit interpretation of invertebrate behavioral changes in response to anthropogenic noise need further research.

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# Sound Effect on the Maternal Care Behavior of a Crab

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**Abstract**

Maternal care in marine decapods encompasses eggs caring until larvae hatching, ensuring healthy embryonic development. The effect of different sound sources (anthropogenic and biologic) and temporal patterns (low and high rate) was studied on the maternal care of the key crab species, *Neohelice granulata*, through laboratory experiments. Three acoustic stimuli were emitted: two anthropogenic, an artificial white noise and motorboat passes obtained from the crabs' natural habitat, and one biologic, signals from a crabs' predator (fish). Behavioral variables quantified included duration of still position and maternal care behaviors. Preliminary findings showed that the high-rate anthropogenic stimuli

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increased the still position and diminished maternal care behaviors. The predatory stimulus did not affect the still position, although the high-rate temporal emission pattern reduced the maternal care behaviors. The fact that all stimuli affected the maternal care behaviors indicates a negative effect that potentially could affect offspring survival. This is important to consider given the ecosystem engineering function (e.g., burrowing) of the studied key crab species.

**Keywords**

Biologic sound · Anthropogenic sound · Maternal care behavior · Impact · Crustaceans

## Introduction

Maternal care is widespread among many animal taxa. In crustacean marine decapods, maternal behaviors involve the care of the eggs that females carry in their brood compartment until larvae is released (Diesel 1992). This behavior is related to the supply of oxygen to the embryo mass, exhibiting active brooding comportment directed toward their ventilation (Fernández and Brante 2003). Maternal care is a critical factor during embryonic development, given that oxygen limitation was demonstrated to influence survival (Baeza and Fernández 2002).

The sound production in crustaceans has been widely measured. In marine decapods, sound production was demonstrated to be involved in diverse behaviors such as defensive, mating, and agonistic interactions (e.g., Boon et al. 2009; Buscaino et al. 2011; Filiciotto et al. 2019; Flood et al. 2019). It was demonstrated that decapods appear more sensitive to low-frequency acoustic stimuli resulting from the particle motion sound component (Roberts and Elliott 2017). However, a recent study comparing the sound detection abilities among four crab species proposed that crustaceans may have alternative and complex sensory systems involved in sound perception (Radford et al. 2022).

The effect of anthropogenic sound on marine animals typically depends on noise intensity and temporal patterns of exposure (Popper et al. 2014). Particularly in crustaceans, relative few studies have been performed to assess the effect of human noise on their biology, and no previous studies have tested its effect on the maternal care behavior. In contrast, the effect of noise on parental care behavior has been demonstrated in fishes (Picciulin et al. 2010; Nedelec et al. 2017).

*Neohelice granulata* is a varunid semi-terrestrial key crab species inhabiting the intertidal zone of estuaries, salt marshes, and mangroves of the Southwestern Atlantic Ocean (Spivak et al. 2019). In the Mar Chiquita coastal lagoon (Buenos Aires Province, Argentina), a Man and the Biosphere (MAB) Reserve declared by UNESCO, *N. granulata* is a dominant species. The soundscape of the lagoon was previously characterized, describing anthropogenic (motorboat passages) and biologic (fish and crustaceans) sounds (Ceraulo et al. 2020). The sound production in *N. granulata* was reported and linked to the reproductive behavior (Filiciotto et al.

2019; Sal Moyano et al. 2019). The maternal care behaviors in *N. granulata* were previously characterized (Silva et al. 2007). In this context, this chapter aimed to analyze the effect of different sound sources, anthropogenic and biologic, considering two temporal patterns of emission, low and high rate, on the maternal care behaviors of *N. granulata*.

## Methods

Similar-sized ovigerous females ranging from 22.5 to 26.5 millimeters (mm) of carapace width in a late stage of embryonic development were collected manually from the Mar Chiquita coastal lagoon. Individuals were acclimated for 24 h in laboratory in natural seawater under a controlled photoperiod and continuous aeration.

### Acoustic Stimuli

Three acoustic stimuli were used: white noise (bandwidth range of 10 Hz–20 kHz) and two sound stimuli acquired from a previous soundscape study conducted in the natural crab's habitat – biologic sounds produced by a crab's predator fish (fish choruses), the black drum *Pogonias courbina*, and anthropogenic sounds emitted by motorboat passages (Ceraulo et al. 2020). Fish choruses have been tied to increase predation risk (e.g., Hughes et al. 2014). Spectrograms of original stimuli are shown in Fig. 1. Two patterns of emission were considered: low and high rates. In the high-rate pattern, playlists contained 1 min of the sound stimulus +1 min of silence, while in the low-rate one, playlists contained 1 min of the sound stimulus +5 min of silence. Both temporal patterns of sound emission are commonly found in the Mar Chiquita coastal lagoon during the warm season: high-rate motorboat passes on weekend days while low-rate motorboat passes during weekdays and high-rate fish choruses during peak hours (sunset: 19:00–21:00 h) while low-rate fish choruses in the rest of the day (see Ceraulo et al. 2020). Playlists for each stimulus and pattern of emission were constructed by randomly choosing different files. Ten different playlists were created for each stimuli and pattern of emission.

### Experimental System

A circular experimental PVC tank (1.2 m diameter and 1.5 m depth) filled with natural seawater was used. A subaquatic video camera connected to an NVR (network video recorder) was placed on the top and center of the tank. An underwater loudspeaker connected to a power amplifier plugged into the stereo output of a laptop was located suspended 10 cm from the tank lateral wall.

To test the experimental system, a calibrated RESON hydrophone coupled with a preamplifier connected to a digital acquisition card (Avisoft UltraSoundGate 116 h)

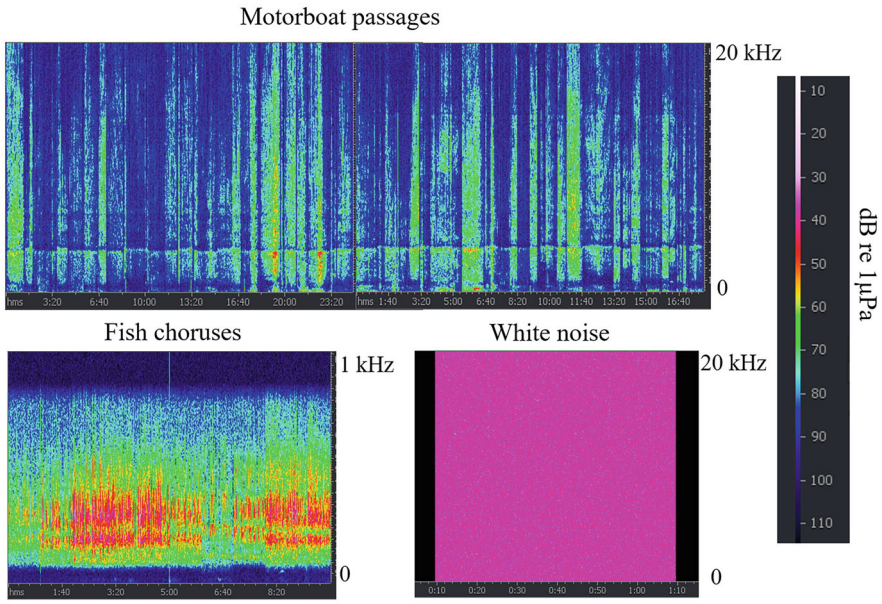



**Fig. 1** Spectrograms of the original stimuli: motorboat passages ($n = 45$ passes), fish choruses ($n = 10$ 1-min files), and the white noise. Ten different playlists were constructed for each stimulus (motorboat and fish) and pattern of emission (low and high rate)

managed by the Avisoft-RECORDER USGH software (Avisoft Bioacoustics) was used. The acoustic stimuli and tank background noise were acquired at the sampling frequency of 100 kHz with 16-bit resolution and analyzed by the Avisoft-SASLab Pro software (Avisoft Bioacoustics). The peak frequencies for emitted motorboat, white noise, and fish stimuli were 3442 Hz (amplitude 130 dB re 1 µPa), 6177 Hz (126 dB re 1 µPa), and 195 Hz (121 dB re 1 µPa), respectively. During the 30 min of sound emission, the sound pressure levels (Lp,rms dB re 1 µPa) changed differently for each stimulus (see Fig. 2). However, over the entire time interval exposition, the Lp,rms were comparable between the playlists with an average of 140 dB re 1 µPa for motorboat playlist, 138 dB re 1 µPa for white noise playlist, and 136 dB re 1 µPa for fish playlist (Fig. 2).

## Experimental Design

A female was located in the center of the PVC tank by using a net, and after a 10 min habituation period, the experiment began. The total experiment duration was 60 min, divided into two phases of 30 min each: the "before phase" without sound exposure and the "test phase" with sound exposure when the different stimuli were emitted. Ten replicates for the control and per treatment (low-rate motorboat, high-rate

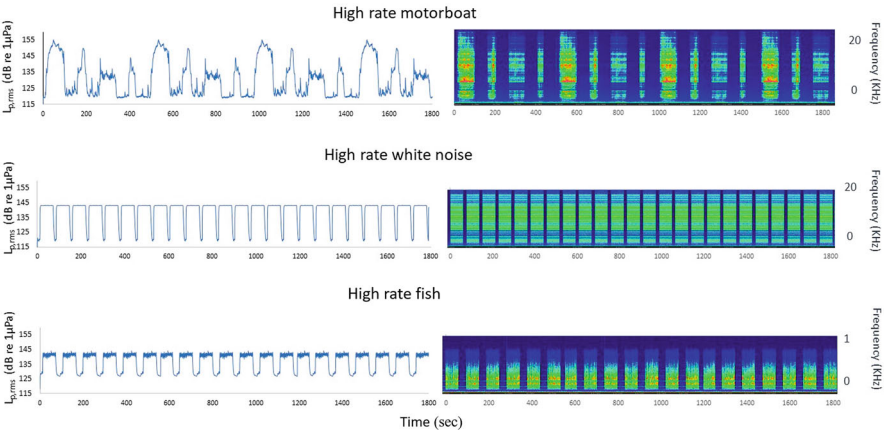



**Fig. 2** (Left) Temporal variation of the sound pressure levels (Lp,rms dB re 1 μPa) during 30 min of exposition. (Right) Spectrograms of an example of a motorboat, white noise, and fish playlist emitted during the sound exposition

motorboat, low-rate white noise, high-rate white noise, low-rate fish, high-rate fish) were performed.

Two maternal care behaviors were recognized: "abdominal flapping," characterized by the abdomen forward and backward movements, beating rhythmically the egg mass, and "chela probing," when females used the chela to pierce the egg mass (Silva et al. 2007). The time in which females were observed still (without walking or moving the pereiopods nor the chelae) for 5 s or longer was also quantified and considered as a "still position." The time duration (in s) of both behavioral variables, still position and maternal care behaviors, was quantified.

## Preliminary Findings

Statistical data analysis is currently in progress. However, preliminary findings indicate that all stimuli, anthropogenic and biologic, affected the maternal care behaviors by diminishing the duration of both flapping and probing, although only the anthropogenic stimuli increased the duration spent in the still position. The high-rate temporal emission pattern of all stimuli had a greater effect on the three behavioral variables quantified compared to the low-rate ones. Figure 3 shows the preliminary findings.

## Remarks

The fact that all stimuli, both anthropogenic and biologic, seemed to diminish the duration of the maternal care behaviors would imply a negative effect on this species that potentially could impair embryonic development. The greater effect of the

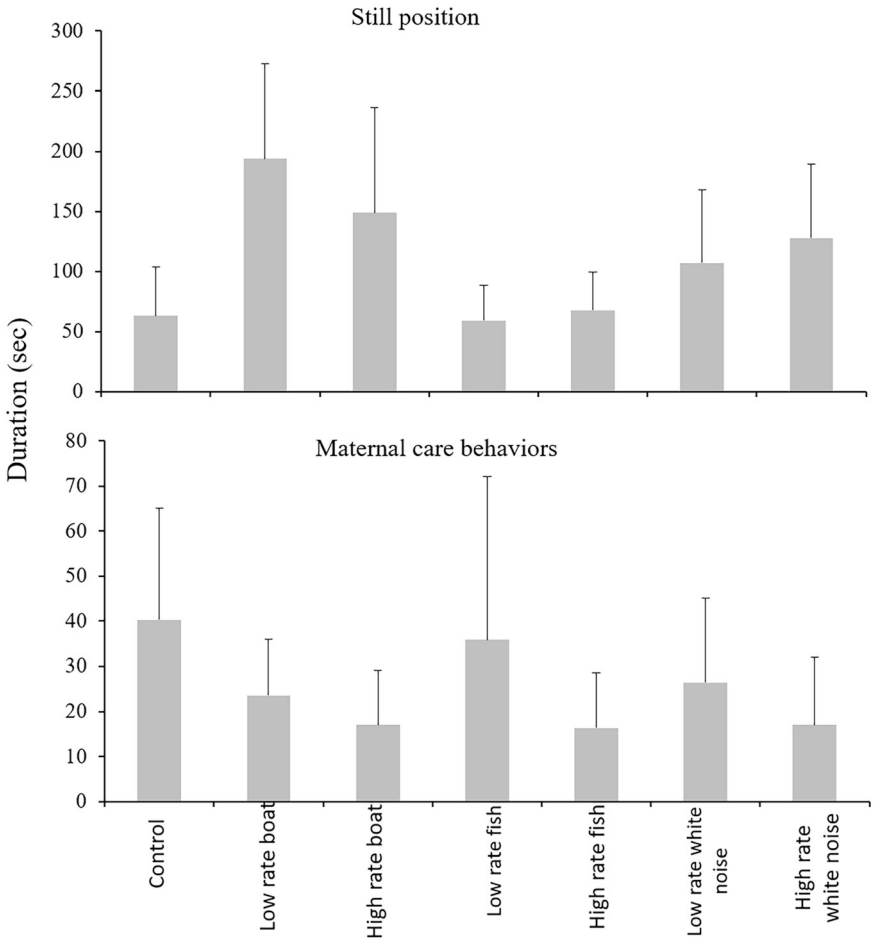



**Fig. 3** Preliminary findings showing the effect of the different stimuli emitted (low- and high-rate motorboat, low- and high-rate fish, low- and high-rate white noise) compared to the control without sound stimulus for the behavioral variables quantified: duration of still position and maternal care behaviors

high-rate temporal emission pattern compared to the low-rate stimuli may suggest that reducing the sound emission in the natural habitat could conform a mitigation strategy to diminish negative sound effects on this species.

The preliminary finding showing that only anthropogenic noise increased the duration of the still position may indicate that crabs distinguish among sound sources and, thus, their response behavior is different. In this sense, the increased duration of the still position could be related to a distraction or confusion effect given that the noise stopped the crab's locomotion. In this sense, a previous study demonstrated

that boat noise distract hermit crabs making them more vulnerable to predation (Yim-Hol Chan et al. 2010).

## Future Studies

The preliminary findings suggest a potential effect of the sound stimuli on the reduced maternal care, likely affecting eggs' oxygen supply and, thus, offspring survival. Negative effects could not cause direct egg mortality but involve a reduced larvae quality, such as a smaller size or less energy reserves, which could impair survival after hatching. In addition, the changeable and vulnerable estuary habitat of the species is characterized by high fluctuations in chemical parameters, such as dissolved oxygen. In this context, the effect of sound may imply a greater threat given that it could add negative effects, impairing embryonic development. Future long-term experiments testing the effect of sound stimuli throughout the embryonic development and assessing direct traits associated with brood survival (e.g., number of hatched larvae) would help to better quantify the effect on offspring fitness.

## Experimental Considerations of the Study

The experiment was conducted in a tank; consequently, conditions such as environmental variables were completely controlled. This is important to highlight given the high variability of the natural habitat of the crab (e.g., dissolved oxygen, salinity; see Luppi et al. 2013), the Mar Chiquita coastal lagoon estuary. However, experiments conducted in tanks can modify the transduction sound properties of the acoustic stimuli, implying differences compared to habitat sound propagation (Akamatsu et al. 2002).

Moreover, a limitation of the study includes the absence of measurements of the particle motion sound component. It was previously described that crustaceans might mainly detect the particle motion component of the sound because of the lacking of gas-filled organs inside the exoskeleton (Popper et al. 2001). In this sense, to obtain results that are more representative of what occurs in nature, future studies should be conducted in nature, or in bigger tanks, and with systems able to characterize also the particle's motion.

## Importance of the Study

This preliminary study is the first study in assessing the effect of different artificial and habitat sound sources, both anthropogenic and biologic, on the maternal care behavior of a crustacean species. In addition, *N. granulata* is a crab key species and an ecosystem engineer because of its burrowing activity that regulates the estuarine ecosystem functioning. In this sense, very few previous studies have focused on the impacts of noise on marine ecosystem services considering how it affects species that mediate ecosystem functioning (Solan et al. 2016). The present study provides important data that can be used in the development of management plans and

sustainable use in the Mar Chiquita coastal lagoon, a MAB UNESCO-declared area that provides important ecosystem services.

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# Marine Plants and Sound

95

Marta Solé and Michel André

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**Abstract**

Plants have been considered passive, static, and immutable organisms for a long time, but research shows that plants can perceive their environment and respond to stimuli by modifying their growth and development. Plants communicate with each other and with the world around them. This communication has been explained through the contact between mycorrhizae and volatile organic compounds, but recently, plant acoustic perception and communication have become a new line of research. The sounds produced by plants would contain information with ecological implications, which could be used in the management of crop fields or in the prevention of biodiversity loss. In addition, anthropogenic noise inflicts ultrastructural damage on various tissues of marine plants that could have implications at ecosystems level and in the planet's biodiversity in general. This chapter provides an overview of the plants bioacoustics and aims to underline the need to develop criteria to assess the impact of sound on marine plants. Understanding how the intricate communication networks between the roots supplies vital information for individuals in marine ecosystems, the possible capacity to

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warn their peers of dangers and the root role on the sensory and communication capacities could be new research lines in marine plant bioacoustics.

**Keywords**

Sound vibration · Plant acoustics · Sound perception · Plant communication · Acoustic impact · Statocyst · Amyloplast · Starch grain

## Introduction

Undersea natural soundscapes include sounds from wind, water currents, earthquakes, lightning, rainfall, and marine organisms (Lindseth and Lobel 2018). The relative recent introduction of anthropogenic noise (fishing, oil and gas exploration, shipping, dredging, pile driving, and drilling, sonar, scientific, and recreational activities) is having an enormous impact upon the marine environment (Popper et al. 2020). These sound sources introduce waterborne acoustic stimuli, including vibration within the seabed (e.g., direct contact with the sediment or propagation via the water column) (Roberts and Elliott 2017).

The distinction between sound and vibration in the marine environment is less clear than in terrestrial environment. Commonly, "sound" is used to describe pressure changes that may be detected by a specialized organ (Hill 2008). The particle motion component of an underwater sound may propagate not only via the water column but also through the seabed (Miller et al. 2016). Recent publications in the marine literature have used the term "substrate borne vibration" (Roberts et al. 2016) or "ground vibration" (Hawkins et al. 2021).

Biotremology, the study of the biological use of vibrational waves, is a relatively new discipline (Hill and Wessel 2016). Animals use vibrations for communication (parental care, foraging, detection of environmental cues, recognition, and predator-prey interactions) (Hill 2008; Roberts and Wickings 2022) and detect incidental vibrations for locating prey and avoiding predators, or abiotic vibratory cues such as thunder and tsunamis (1–100 Hz) as demonstrated in elephants (Roberts and Wickings 2022).

Hanlon and Budelmann (1987) define hearing as "*the ability of an animal to sense vibrations (either pressure oscillations or particle displacement) covering a wide frequency range and to integrate this information to produce an appropriate behavioural response*." In the light of this definition, they argue that cephalopods are not deaf because they can detect particle motion. The same idea can be applied to other marine invertebrates and plants. In addition, sound does not need to be "heard" to cause an effect or damage to an organism (Roberts and Elliott 2017). If acoustic or vibratory stimuli cause a behavioral change, perhaps it is more appropriate to say that these invertebrates have an "acoustic [or vibratory] sense" (Roberts and Elliott 2017).

Noise has been shown to negatively affect fishes and invertebrates, in terms of physical damage, physiology, and behavioral changes (André et al. 2011; Casper et al. 2013; Wale et al. 2013, 2019; Hawkins et al. 2014; Solé et al. 2016, 2022; Day et al. 2019). Given the prevalence of seabed vibration-producing activities, such as pile driving, many benthic species are exposed to a variety of stimuli that have the

potential to impact benthic invertebrates in their vicinity. Noise/vibration could also affect aquatic plants, particularly from sources directly contacting the sediment, because their immobility makes them highly susceptible to chronic effects. The information regarding the range of sensory capacities of aquatic plants, sound source types, and propagation conditions in the marine environment is practically inexistent. A review summarizing the current knowledge on sound perception and communication in plants and the possible implications of the introduction of anthropogenic noise in the marine environment is actually lacking. The purpose of this chapter is to provide an outline of this knowledge and to highlight the urgent need for research to assess the impact of sound on marine plants.

## The Senses of Plants

Despite their obvious immobility, plants possess the same senses as higher evolved organisms such as animals, as well as other sensory capabilities unique to plants. Generally speaking, they are able to see, smell, taste, touch, and also perceive sound (Mancuso and Viola 2015). Plants use photoreceptors, molecules that act as light receptors, which are capable of receiving and transmitting information on the direction and quality of light. These photoreceptors distinguish the quality of the wavelength of each color and use this information to regulate the processes of germination, flowering, and growth (Mancuso and Viola 2015). In addition, plants use their entire organism to "smell." From roots to leaves, plants harbor millions of cells that have receptors on their surfaces capable of capturing biogenic volatile organic compounds (VOCs). They use these compounds to gather information about the environment and to communicate with each other as well as with insects. Each compound carries specific information (imminent danger, stress, pollinating insect attraction messages, harmful insect repulsion, etc.) (Bouwmeester and Schuurink 2019). Plants also have receptors for chemicals, to search for food in the soil through the scouting action of roots, in a function analogous to our sense of taste. The roots perceive the slightest variations in the chemical gradients of the mineral salts present in the soil and are capable of detecting nitrates, phosphates, and potassium that are essential for their nutrition, developing a greater number of roots where their concentration is higher. A special case would be that of carnivorous plants that adopted an evolutionary solution to compensate the lack of nitrogen in swampy areas by modifying the shape of their leaves to turn them into traps for reptile, insects, and even small mammals (D'Amato 1998). In plants, the senses of touch and hearing are closely related. They have "mechanical-sensory channels" scattered throughout their body, but especially in the epidermal cells on their surface, which are activated when they pick up a vibration or come into contact with an object. Carnivorous plants show this "tactile" ability by triggering their traps only when the weight comes into contact with them. Other examples would be plants that react to the contact of their pollinating insects, climbing plants or roots that grow around an obstacle on their way to nutrients (Del Stabile et al. 2022). In terms of the sense of "hearing," land plants have at their disposal two sound transmission vectors, air and earth. The soil is an extraordinary conductor where the vibrations are transmitted and captured by the

mechanoreceptor channels of the plants. Like the other senses, in plants, the "ear" is a diffuse organ. Plants perceive the sound vibrations by all the cells of their body. The roots perceive a very high range of sound vibrations that can influence their directional movement (phototropism). In addition, the roots are capable of producing sounds (clicking) (Gagliano et al. 2012) probably due to the rupture of their cell wall during growth. In fact, plants emit sounds through different organs and at different stages of growth or in response to different situations. Plants have been reported to emit low frequency sounds (50–120 Hz) from the xylem (Jung et al. 2018) or ultrasonic vibrations (20–100 kHz) in case of stress (Hassanien et al. 2014; Khait et al. 2018). This ability to emit, although not voluntarily, and perceive sounds led authors to think that there could be an underground communication channel between plants (Gagliano et al. 2012; Del Stabile et al. 2022). For example, root systems behave like a swarm (Ciszak et al. 2012), which would indicate that each individual plant has some communication capacity that allows it to explore the soil effectively.

At this point, it is worth noting the fundamental role played by the plant roots due to their analogous role to the brain in the animal world. Darwin launched this "root-brain hypothesis" in his book "The power of movement in plants": "It is not an exaggeration to say that the tip of the root, thus endowed [with sensitivity] and being able to direct the movement of the adjacent regions, acts like the brain of a lower animal; the brain, located in the front part of the body, receives the impressions of the sensitive organs and directs the different movements" (Darwin 1880). The apex of the root is the zone of the plants that presents more capacities comparable to an intelligent organ: (i) perception ability of environmental stimuli – gravity, temperature, humidity, electric field, light, pressure, chemical gradients, presence of toxic substances (poisons, heavy metals), sound vibrations, and presence or absence of oxygen and carbon dioxide; (ii) decision capacity on the needed movement; and (iii) final movement implementation (Mancuso and Viola 2015). The root apparatus is a physical network in which the roots maintain an anatomical relationship with each other (Baluska et al. 2010). These roots communicate with each other through chemical signals. But these roots could be sensitive to electromagnetic fields (Masi et al. 2009) and sounds that other roots produce when growing (Gagliano et al. 2012). This enormous sensitive capacity of the roots makes them strong candidates to be authentic recipients of the consequences of noise pollution. Research on the effects of noise pollution on plants, specifically on roots, is urgent and necessary. The knowledge, practically nonexistent at the moment, of the consequences of artificial noise in the plant world would help the responsible and respectful management of biodiversity in all Earth's ecosystems.

## Communication in Plants

Like gravity, wind, tidal currents, and rain, sounds include pressure waves that have a mechanical influence on organisms (Telewski 2006). Plants respond to vibrations in a selective way (Gagliano et al. 2012; Appel and Cocroft 2014). Through their senses, plants receive information on their habitat and position themselves in their

environment. To transport information from one part of the body to another, plants use electrical, hydraulic, and chemical signals that are transmitted through their main vascular system (xylem and phloem) that connects the roots to the leaves. But plants can also communicate with each other and with their environment through chemical signals or by "touching" their roots or tops (Mancuso and Viola 2015). Sound and touch have similar physical properties (via mechano-stimulation), but plants seem to distinguish between them in sophisticated ways (Mishra et al. 2016). Trees of the same species form a community and often establish alliances with trees of other species. Trees in forests have evolved to live in cooperative and interdependent relationships, maintained by communication and a collective intelligence. Although the most visible part of the trees are their tops, the real action takes place underground (Wohlleben and Flannery 2016).

Roots are able to communicate with microorganisms, bacteria, fungus, and insects inhabiting soil. For instance, mycorrhizae are symbiotic associations between the root plants and fungi. The fungus supplies the root with minerals that are difficult to obtain, such as phosphorus, and receives in exchange sugars produced by the plant in photosynthesis, which it uses as an energy source. Fungi consume about 30% of the sugar that trees photosynthesize from sunlight. Sugar is what feeds the fungi as they search the soil for nitrogen, phosphorous, and other mineral nutrients, which are then taken up and consumed by the trees (Wohlleben and Flannery 2016). In addition, fungi supply trees with essential amino acids, protective metabolites against cryptogamic predators, other fungi, bacteria, and viruses, as well as special sugars that are stored in the soil for protection against contaminants. These soil sugars are particularly important for marine plants, as they bind metals that could become toxic in ionic form. They also contribute to the marine plant being able to resist salinity.

But not all fungi are beneficial. Plants need to be able to identify pathogenic fungi through chemical signals to avoid harmful symbiosis with them. Harmful fungi colonize tree tissues, digest them, and weaken trees. Some of these fungi feed on heartwood (inner wood), other on sapwood (outer wood), and some on roots. Fungus can destroy strong wood and create spongy or brittle wood. Examples of fungus that could cause root rot are honey fungus (*Armillaria mellea*), oak bracket (*Inonotus dryadeus*), and annosum (*Heterobasidion annosum*), but this process could rather consist of a biotic balance of the immune system, since, for example, these same species can establish symbiotic relationships with ancient trees. Process for recognizing the nature of the fungus consists of a true chemical dialogue between the root and the fungus. If it is a harmful fungus, hostilities will begin. Mycorrhizal fungi release suppressive exudates (antibiotics that inhibit infection by fungal root pathogens). Mycorrhizae also defend root systems by forming a physical barrier made of chitin to deter invasion by soil pathogens. This barrier forms a protective layer protecting root cells (St-Arnaud 2007). If the plant recognizes the fungus as beneficial, it could allow the establishment of ectomycorrhizae (the fungus generates a sleeve around the root of the plant) and even manages to penetrate its cells (endomycorrhizae). The fungal filaments could penetrate the root's cell and can lead to the creation of true hybrid organs in arbuscular endomycorrhizae where DNA sharing and gene exchange occur (Johnson and Gehring 2007).

This symbiotic relationship is essential for both tree and fungus survival. For example, tree decline processes such as that observed for cork oak (*Q. suber*) in Mediterranean forests can modify the mycorrhizal landscape and as a consequence alter the oak seedling establishment, which is a process mediated by the fungus (Ibáñez et al. 2015).

All the trees in the forests are connected to each other through underground networks of fungi. For young trees in shady areas of the forest where they cannot normally photosynthesize, the mesh is a lifesaver. They survive because large trees pump sugar into their roots through the network (Wohlleben and Flannery 2016). Trees share water and nutrients through networks and also use them to communicate. They send out distress signals about drought and disease, for example, or insect attacks, and other trees alter their behavior when they receive these messages. It seems that alarm and distress are the main topic of tree conversations. As said before, some plants emit and detect sounds (e.g., behavioral response of young roots to a continuous 220 Hz sound crackling noise, Gagliano et al. 2012). It is easy to think on mycorrhiza as a channel for efficiently transmitting these sonorous messages between trees, given their close connection with their roots. Future research could try to decipher the sonic conversations between the roots of trees to understand how the cooperative and supportive lifestyle between plant communities is essential for the conservation of biodiversity of the entire ecosystem.

Plants can communicate with animals too. In addition to chemical adaptations, plants also have adapted to different sounds in their environment. Pollination is one of the periods in which the plant depends more on communication with animals. At this time, the plants need vectors that transport the pollen from one flower to the pistil of another flower. Insects use vibrations to extract pollen from flower anthers (buzz pollination). Solanum-type flower has evolved specifically in response to sonicating bees. The bees adjust their duration of their buzzing depending on the pollen content of the flower (De Luca and Vallejo-Marìn 2013). The peak frequency of the buzzing varies between 100 and 400 Hz depending on the species of bee.

Although animals most used as vectors are insects, also birds and bats can serve this function. A special case is the carnivorous plant *Nepenthes hemsleyana*, whose pitchers have evolved to reflect the echolocation of bats. This plant had established a symbiosis with bats; they have a safe, parasite-free roosting spot, and the bats in return fertilize the plant with nitrogen-rich droppings (Schöner et al. 2015). Bats are responsible for transporting the pollen of many of the cacti of the American desert, such as the Joshua tree. Another curious case is the Cuban liana *Marcgravia evenia*, which possesses leaves in the shape of a parabolic antenna with the function of signaling the presence of flowers to the bats' sonar. The echo emitted by the leaf acts as a beacon, providing a strong, multidirectional, and recognizable signal (Simon et al. 2011). The authors of this work hypothesized that it would be necessary to analyze whether flowers pollinated by bats have developed similar acoustic signals in order to attract echolocating pollinators. Bat-pollinated flowers need to attract their pollinators in the dark and some species have developed this specialized echoic flower parts, usually concaves, that act as acoustic retroreflectors, making the flowers acoustically visible to bats. Species-specific echo sequences inform the bats about

flower morphology and show them the best way to get the nectar. This acoustic information provides essential support for efficient foraging. Bat-pollinated flowers have higher echo target strength, independently of their size, and differ in their morphology from species pollinated by insects or birds (Simon et al. 2021).

## The Plant "Ear": Sound Perception

Although various scientific papers highlight the ability of plants to perceive and react to sounds, the mechanism by which these processes occur is largely unknown. Plants have been shown to react to specific sound frequencies (Hou and Li 1997), for example, by building up defenses against the chewing of caterpillars (Appel and Cocroft 2014), the mating sounds of insect larvae, or sweetening their nectar if they sense pollinating insects (De Luca and Vallejo-Marìn 2013; Schöner et al. 2016; Veits et al. 2019). The sound waves make the leaves of the plants vibrate, accelerating the plasmatic movement of their cells (Cai et al. 2015).

Cells respond to external stress by changing a number of factors, including cell division, growth, signal transduction, gene expression, and activation of membrane ion channels (Hamant and Haswell 2017; Sparke and Wünsche 2020). Mechanosensitive ion channels have the ability to open and close as a result of mechanical forces resulting from gravity, osmotic pressure, and sound. The process of sound signaling at cellular level is unknown, but the most accepted idea is that sound vibrations activate the plasma membrane channels, evoking a membrane potential-based signaling cascade, where $Ca^{2+}$ acts as second messenger. Several studies have shown an efflux/influx of $Ca^{2+}$ after sound vibrations exposure. Also at intracellular level, exposure to sound can cause activation of stress-related genes or regulate their transcription, increase the level of soluble sugars and proteins (or change their activity), and influence the reorganization of microtubular structures as well as the amylase activity in the cap root (Sun et al. 1999; Wang et al. 2002a; Qin et al. 2003; Xiujuan et al. 2003; Hongbo et al. 2008; Mishra et al. 2016). Mechanosensory proteins present in the plasma membrane may mediate the transmission and response to mechanical stimulation of sound in plant cells, through their influence on hydrostatic pressure, plasma membrane tension, and thus traffic through it. Changes in these characteristics would cause the alteration of the resonant frequency of the cell. Plasma membrane tension modulates the processes of exocytosis and endocytosis of the cell. An increase in membrane tension stimulates exocytosis and inhibits endocytosis. Decreased membrane tension slows down exocytosis (Del Stabile et al. 2022).

More precisely, the process of sound signaling could be described as below. Sound generates mechanical tension in the membrane and cell wall, activating mechanosensitive ion channels and mechanoreceptor proteins. When ion channels are open, in response to mechanical forces, they allow ions ($Ca^{2+}$ and $K^{+}$) to cross the membrane and cause an ionic current that is translated into an electrical or chemical signal (Alexandersson et al. 2004; Ingber 2006; Mishra et al. 2016). Increase in cytoplasmic $Ca^{2+}$ activate protein kinases that stimulate H+ ATPase pump to generate

electrochemical gradients across the membrane. This process activates the increase of cytoplasmic proteins and provoke changes in the cytoskeleton. In addition, alterations in phytohormones derive on transcription changes within the nucleus and alters chromatin structure (Kim et al. 2021), and consequently the specific proteins and metabolites expression that brings to physiological changes (Mishra et al. 2016).

Different experiences have been described where this ability to perceive sound was used to influence production processes in agriculture and biotechnology. Certain frequencies, especially between 100 and 500 Hz, favor seed germination and plant and root growth, while, when subjected to high frequencies (ultrasounds, 1–3 MHz), an inhibitory effect is observed (Qin et al. 2003; Creath and Schwartz 2004; Cai et al. 2014; Hassanien et al. 2014; Mishra et al. 2016; Vicient 2017). In addition, sound can improve the immune system and the efficiency of light absorption for photosynthesis (Cai et al. 2014; Choi et al. 2017; Kratochvil and Pollirer 2017). Although sound vibrations were suggested to increase yield, quality, and protein content, organogenesis (e.g., 400 Hz and 106 dB increased the germination index, shoot height, and root activity (Wang et al. 2003a; da Silva and Dobranszki 2014; Hassanien et al. 2014) and stimulate the in vitro growth and development in rice (*Oryza sativa*) (Liu et al. 2003b), higher frequencies (70 kHz ultrasound) produced harmful effects, reinforcing the idea that the use of sound vibrations on agriculture has to be very selective and avoid ultrasounds because it causes negative botanical effects (Hassanien et al. 2014). Similarly, increased organogenesis stimulated by sound vibrations has been described in aloe (*Aloe arborescens*, Liu et al. 2003a), gerbera daisy (*Gerbera jamesonii*, Wang et al. 2003b), pumpkin (*Cucurbita pepo*, Ananthakrishnan et al. 2007), and European hazelnut (*Corylus avellana,* Safari et al. 2013), among other species. Treatment with a sound of 1000 Hz at 100 dB promotes the seed germination of *blacksamson* (*Echinacea angustifolia*) (Chuanren et al. 2004). Sound vibrations were used to increase the tomato, lettuce, and spinach number of leaves, flowers, and chlorophyll content (Hassanien et al. 2014). Treatment with sound could reduce the use of fertilizers and increase the immune response against harmful insects and diseases (Hassanien et al. 2014). In addition, these treatments can promote the acceleration of seed germination and organogenesis process by modification of phytohormonal levels (Wang et al. 2003a, 2004; Zhao et al. 2003; Appel and Cocroft 2014).

## Algae and Aquatic Plant Sound Production and Perception

Algae, both macroscopic and microscopic, are photosynthetic organisms that live in the aquatic environment (Brodie and Lewis 2007). The production and perception of sound in algae are almost unexplored processes. However, algae and aquatic plants have been shown to produce sound during photosynthesis (Kratochvil and Pollirer 2017; Freeman et al. 2018). This sound is produced by the release of oxygen bubbles by plant structures. The oxygen bubble assumes a spherical shape creating a monopolar sound source that is distributed by resonance over the seabed. This phenomenon,

where plants produce sounds from 2 to 20 kHz (Freeman et al. 2018), can be used for quantification of algae/aquatic plants in their ecosystem (Kratochvil and Pollirer 2017) and could be the basis for exact acoustic measurements, opening entirely new possibilities for plant physiology. As with terrestrial ecosystems, the identification of sound sources that contribute to underwater biological soundscapes would constitute a noninvasive method, capable of remotely quantifying their ecological status. As previously highlighted, photosynthesis in macroalgae is an acoustically active process, driven by the oxygen bubbles they produce. Soundscapes are correlated with macroalgal cover during light periods. The production rate, the size of the bubbles, and the level of sound are directly related to the dissolved oxygen concentration, which in turn depends on the amount of photosynthetically active radiation. The stress produced by climate change, overfishing, or pollution causes an increase in the macroalgae cover, which serves as an indicator of the degradation of underwater ecosystems. These signal processing methods could quickly and passively assess the relative abundance of algae and consequently the underwater ecological status of the referred habitat (Freeman et al. 2018).

The possibility of using acoustic methods to monitor the in situ response of seagrasses to environmental factors has been proposed too. Photosynthesis effects on sound propagation were assessed in grasslands of *Posidonia oceanica*. Measurements of acoustic response were conducted in situ over an extensive bed of *P. oceanica* through the monitoring of their oxygen synthesis (Hermand et al. 1998). This research showed a strong correlation between photosynthesis and the impulse response of the acoustic waveguide with a marked change of attenuation and time dispersion at the start of the photosynthesis process. During the active photosynthesis phases, sound speeds were lower (due to solved gases adhering to the leaf or running inside the leaf blades, or to gas movement in the rhizome and root systems) than in bubble-free seawater (Hermand 2004).

The plant's ability to perceive and react to sound stimuli has also been used to increase the growth and productivity of aquatic plants. *Chlorella pyrenoidosa* and *Picochlorum oklahomensis* microalgae showed a significant improvement in their growth (biomass productivity and lipid yield) after exposure to sound waves of frequency of 0.4 and 2.2 kHz, respectively (Jiang and Chen 2012; Cai et al. 2016). Furthermore, it was shown that natural sounds increase algal biomass production and that the 2.2 kHz frequency is the predominant component of most sounds found in nature (e.g., the main frequency composition is from 1.5 to 2.0 kHz in most of the wild birds' chirp, from 2.0 to 4.0 kHz in most of the crickets and cicadas' chirp, and from 1.5 to 2.0 kHz in the sound of trickling water and the sound of rain (Cai 2013). Ecological data encoding could be used to induce ecosystems to reproduce and synthesize biotechnologically relevant cellular components (such as triacylglycerol synthesis in *Chlorella pyrenoidosa* microalgae, which suggests the use of acoustic signals to improve biodiesel production (Golub and Levtun 2016)) and to improve the yield rate in different algae species (*Chlorella pyrenoidosa*, Jiang and Chen 2012; *Picochlorum oklahomensis*, Cai et al. 2016; and *Haematococcus pluvialis*, Christwardana and Hadiyanto 2017).

## Noise Effects on Plants

The ability of plants to detect vibrations may represent a very efficient way of finding information from distant sound sources, for example, for orientation towards water. Roots may detect the sounds of water moving through the soil or flowing through channels (Gagliano et al. 2017). Roots of corn seedlings can detect sound waves or vibrations and selectively use them for orientation (Gagliano et al. 2012). Gagliano et al. (2017) showed that *Pisum sativum* roots were able to locate a water source by sensing the vibrations generated by water moving inside pipes and that the presence of noise (computer-generated white noise) affected the abilities of roots to respond correctly to the surrounding soundscape (masking effect).

If investigations addressing water-seeking abilities of roots (Cassab et al. 2013) and the possible noise impact affecting them (Gagliano et al. 2017) or other noise-based affected capacities in terrestrial plants have been extremely limited, research on acoustic impact on aquatic plants has been practically nonexistent. Although considerable scientific attention has been devoted in the last decade to determining the sensitivity of aquatic animals to noise, focusing especially on cetaceans, pinnipeds, and fish (Richardson et al. 1995; Slabbekoorn et al. 2010), and recently in invertebrates (Solé et al. 2016, 2017, 2022; Day et al. 2019; Wale et al. 2019), little attention has been dedicated to assessing how much waterborne artificial noise could affect sessile organisms as plants, whose immobility makes them highly susceptible to chronic effects. Only a recent work has reported morphological and ultrastructural damage on seagrass *Posidonia oceanica* after exposure to sound, which is new to aquatic plants (Solé et al. 2021).

Columella cells, located in the inner, central region of the root cap and characterized by numerous starch-filled plastids (amyloplasts) in the distal part, are involved in gravity sensing (Yoder et al. 2001; Hashiguchi et al. 2013). The amyloplasts are double-lipid membrane cytoplasmic organelles containing DNA and store starch grains inside the membrane compartments (Pozueta-romero 1991) (Fig. 1). The sedimentation of these amyloplasts triggers gravity signal transduction, which is transmitted to the elongation zone, where it promotes differential cell growth, allowing the root to direct itself downward (Hashiguchi et al. 2013). *P. oceanica* also exhibits starch-containing amyloplasts in the cortical cells of the rhizome (Kuo 1978). Amyloplasts are responsible for gravity sensing and signal processing (Hashiguchi et al. 2013), operating like statocysts and play a fundamental role in sound and vibration reception (Gagliano et al. 2012).

Solé et al. (2021) revealed that aquatic plants present acoustic trauma when exposed to noise. Low-frequency sounds (50–400 Hz sinusoidal wave sweeps with 100% duty cycle and a 1-s period, received sound pressure level RL = 157 ± 5 dB re 1 μPa with peak levels up to sound pressure level SPL = 175 dB re 1 μPa) produced alterations on seagrass *P. oceanica* root and rhizome statocysts. Exposed plants showed a decreasing number of starch grains with time when compared with control specimens (Figs. 1 and 2). Some starch grains presented a deformed structure lacking the typical spherical shape (Figs. 1c and 2f). Other amyloplasts had deformed inner organelles and superficial holes (Fig. 2h). Exposure

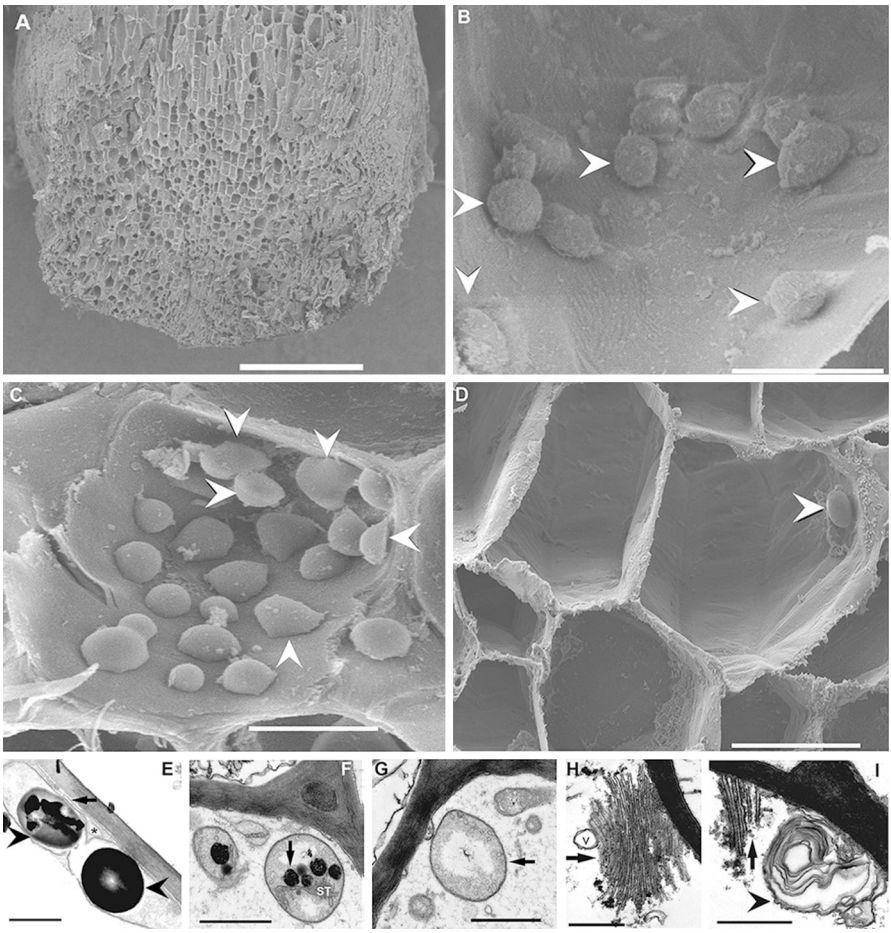



**Fig. 1** *Posidonia oceanica* root. (**a–d**) Scanning electron microscopy (SEM). (**e–i**) Transmission electron microscopy (TEM). (**a**, **b**, **e**, and **h**) Control (at 120 h after starting the experiment). (**c** and **f**) Forty-eight hours after sound exposure. (**d**, **g**, and **i**) One hundred and twenty hours after sound exposure. (**a**) Apical portion of *P. oceanica* root showing the opened root cap containing the columella cells. (**b**) Opened columella cell with normal inner starch grains (arrowheads). (**c**) Starch grains presenting deformed structure (arrowheads). (**d**) Only one remaining starch grain is visible (arrowhead). (**e**) Amyloplasts containing starch grains (arrowheads). Note the association of endoplasmic reticulum (arrow) with the amyloplasts and the presence of a vesicle (asterisk). (**f**) Absence of starch grains and presence of plastoglobuli (arrow) in amyloplasts associated with stroma thylakoids (ST). (**g**) Absence of starch grains in the amyloplast (arrow). (**h**) Healthy Golgi complex (arrow) in columella cells (v: vesicle). (**i**) Myelin-like formation (arrowhead), after sound exposure. The arrow shows the leftover of a membranous structure. Scale bar: (**a**) 0.5 mm. (**d**) 30 μm. (**b** and **c**) 10 μm. (**e–i**) 1 μm (Solé et al. 2021)

to sound caused the development of plastoglobuli (Fig. 1f) and myelin-like structures, which have their origin in endoplasmic reticulum disorganization or vacuoles rupture (Fig. 1i) induced by noise exposure. Symbiont fungi from *P. oceanica* roots

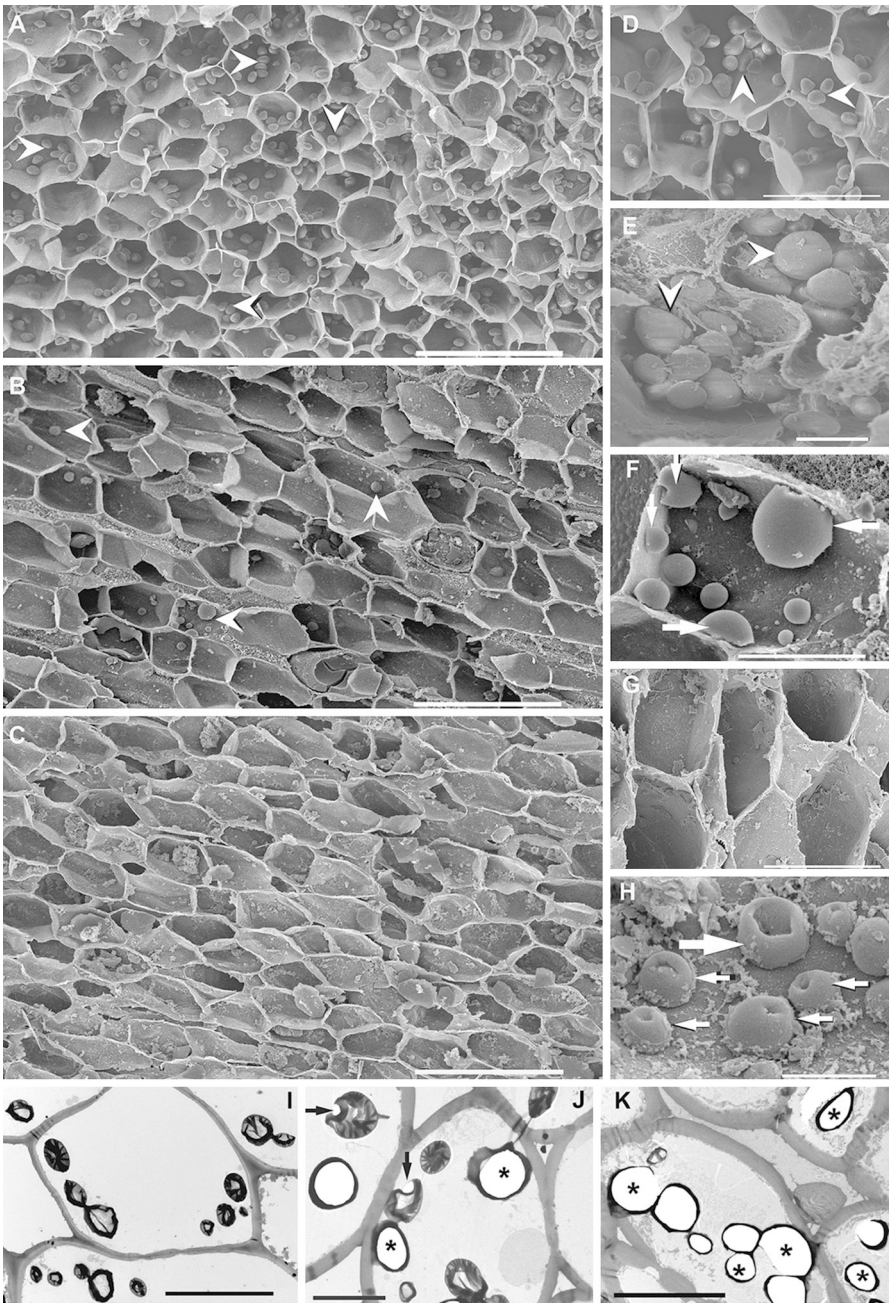



**Fig. 2** *Posidonia oceanica* rhizome. (**a**–**h**) Scanning electron microscopy (SEM). (**i** and **j**) Transmission electron microscopy (TEM). Transverse section of an apical portion of *P. oceanica* rhizome showing the opened cortical cells. (**a**, **d**, **e**, and **i**) Control at 120 h after sound exposure. (**b**, **f**, and **j**) Forty-eight hours after sound exposure. (**h**) Ninety-six hours after sound exposure. (**c**, **g**, and **k**) One

exposed to sound showed internal degradation (Fig. 3d–f) unlike controls (Fig. 3a–c). The exposed hyphae presented altered intracellular organelles, showing an empty cytoplasm 120 h after exposure to noise.

The damage was acute and immediate, significantly affecting the amyloplasts. The plant gravity perception and correct functioning mediated by these organelles could be altered. In addition, the storage capacity of the rhizome could be decreased due to the low number of amyloplasts.

Different explanation can be hypothesized to explain the loss of the starch grains after sound exposure. An acoustic trauma in the mammalian organ of Corti produce cytoarchitectural organization and actin arrangement alterations (Slepecky et al. 1982). Also in invertebrates, a downregulation of cytoskeletal proteins and microtubular proteins of the statocyst sensory epithelia has been shown after sound exposure (Solé et al. 2019). Actin microfilaments are responsible from organelles positioning in plant cells (Pozueta-romero 1991; Hashiguchi et al. 2013). The vacuolar membrane and actin cytoskeleton are involved in the amyloplast movement. The decrease in the number of starch grains after sound exposure could be explained by the distortion of the anchorage of the starch grains into the vegetal cell, resulting from changes in actin arrangements.

Sound-induced amyloplast loss could be explained by molecular mechanisms too. Double-bound organelles involved in energy storage production, e.g., mitochondria has been shown to be impacted by noise exposure in animals (Wang et al. 2002b). In *P. oceanica*, amyloplasts, which are surrounded by a double membrane, are in charge of energy storage. In mammal acoustic trauma, mitochondria trigger cell death pathways in auditory hair cells in a process mediated by calcium overload (Vicente-Torres and Schacht 2006) and by an overproduction of mitochondria ROS (reactive oxygen species) (Gonzalez-Gonzalez 2017). In a similar way, *P. oceanica* root and rhizome statocyst lesions could be originated in calcium overburden and increased oxidative stress after sound exposure. A functional interaction between cell-filling vacuoles and adjacent actin microfilaments is fundamental for controlling amyloplast status and movement in shoot statocysts of *Arabidopsis* (Hashiguchi et al. 2013). Vacuolar fragmentation, vacuolar dynamics deficiency, or endoplasmic reticulum disorganization in several types of plant cells, creating myelin-like formations (i.e., a massive accumulation of membranes that disrupt amyloplast dynamics), have been observed after drug treatments (Hashiguchi et al. 2013). The present

**Fig. 2** (continued) hundred and twenty hours after sound exposure. (**a**) High density of starch grains (arrowheads) in control cortical cells. (**b**) Small number of starch grains (arrowheads) after sound exposure. (**c**) Cortical cells totally empty of starch grains. (**d** and **e**) Details of (**a**), showing healthy round starch grains (arrowheads). (**f** and **h**) Details from (**b**). (**f**) Some deformed starch grains (arrows). (**g**) Detail from (**c**), empty columella cells. (**h**) Abnormal starch grains with empty aspect (large arrow) or holes (small arrows) at the surface. (**i**) Control cortical cells of rhizomes containing starch grains. (**j**) Starch grains partially (arrows) or totally (asterisk) lost the starch. (**k**) Empty starch grains (asterisk). Scale bar: (**a–c**) 100 μm. (**d** and **g**) 50 μm. (**f** and **i–k**) 20 μm. (**e**) 10 μm. (**h**) 5 μm (Solé et al. 2021)

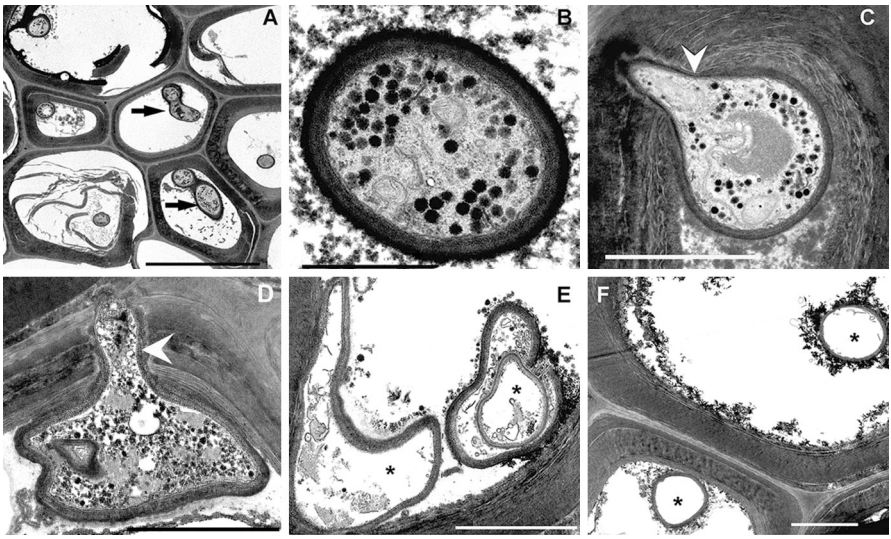
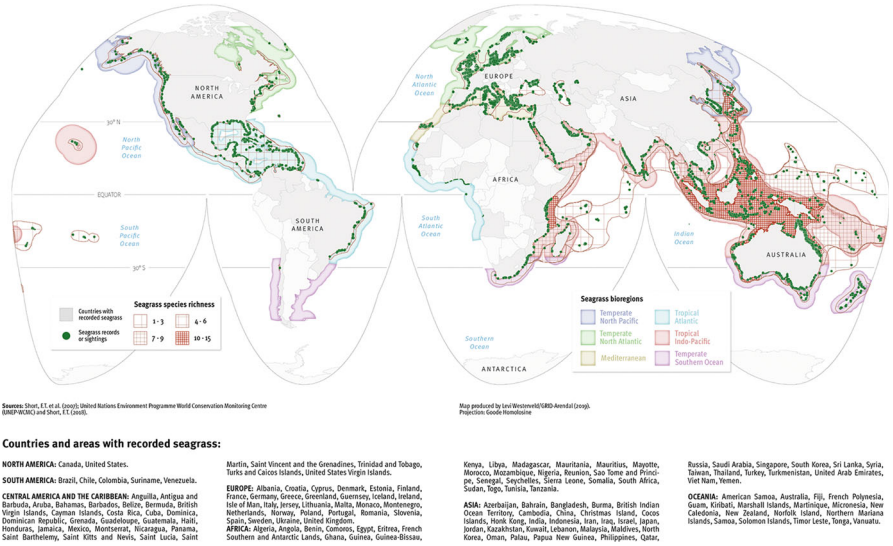



**Fig. 3** Transverse section through a *Posidonia oceanica* root showing fungal colonization. Transmission electron microscopy (TEM). (**a–c**) Control at 120 h after starting the experiment. (**d**) Forty-eight hours after sound exposure. (**e**) Ninety-six hours after sound exposure. (**f**) One hundred and twenty hours after sound exposure. (**a**) Hypodermal cells colonized by numerous intracellular hyphae (arrows). (**b** and **c**) Intracellular hyphae showing an organelle-rich cytoplasm. (**c** and **d**) An intracellular hyphae degrading the cell wall while passing from one cell to another (arrowhead). (**d**) Hypha cytoplasm presenting small vacuoles and fragmented organelles. (**e**) Intracellular hypha cytoplasm almost empty (asterisks). (**f**) Hyphae presenting a totally empty cytoplasm without organelles (asterisks). Scale bar: (**a**) 10 µm. (**b**) 1 µm. (**c–f**) 2 µm (Solé et al. 2021)

results show a similar cell organelle breakdown (Fig. 1i), which could provoke (i) a dysfunction of the normal gravitropism from amyloplast dynamics disruption and (ii) a modification of the rhizome storage function, which is fundamental for seagrass survival.

Sound exposure can be the origin of plastoglobuli detected in amyloplasts of the columella cells. Plastoglobules play a fundamental role in plastid metabolism (they are involved in their biogenesis, redox regulation, senescence, and environmental adaptation) (van Wijk and Kessler 2017). The overproduction of reactive oxygen species (ROS) would cause reactive stress, which in turn would be at the origin of processes associated with plastid biogenesis or repair mechanisms at the cellular level, resulting in the presence of plastoglobuli after sound exposure. Oxidative stress refers to elevated intracellular levels of ROS, which cause damage to lipids, proteins, and DNA.

In a similar way that dark septate endophytes colonize the roots of terrestrial plants, *P. oceanica* roots are known to be colonized by symbiotic melanized septate hyphae (Vohník et al. 2015). These symbiotic hyphae could be involved in the nutrient uptake by *P. oceanica* (Vohník et al. 2019). The degradation of symbiotic root fungi after exposure to sound could alter the nutritional capacity of seagrass




**Countries and areas with recorded seagrass:**

**NORTH AMERICA:** Canada, United States.

**SOUTH AMERICA:** Brazil, Chile, Colombia, Suriname, Venezuela.

**CENTRAL AMERICA AND THE CARIBBEAN:** Anguilla, Antigua and Barbuda, Aruba, Bahamas, Barbados, Belize, Bermuda, British Virgin Islands, Cayman Islands, Costa Rica, Cuba, Dominica, Dominican Republic, Grenada, Guadeloupe, Guatemala, Haiti, Honduras, Jamaica, Mexico, Montserrat, Nicaragua, Panama, Saint Barthelemy, Saint Kitts and Nevis, Saint Lucia, Saint Martin, Saint Vincent and the Grenadines, Trinidad and Tobago, Turks and Caicos Islands, United States Virgin Islands.

**EUROPE:** Albania, Croatia, Cyprus, Denmark, Estonia, Finland, France, Germany, Greece, Greenland, Guernsey, Iceland, Ireland, Isle of Man, Italy, Jersey, Lithuania, Malta, Monaco, Montenegro, Netherlands, Norway, Poland, Portugal, Romania, Slovenia, Spain, Sweden, Ukraine, United Kingdom.

**AFRICA:** Algeria, Angola, Benin, Comoros, Egypt, Eritrea, French Southern and Antarctic Lands, Ghana, Guinea, Guinea-Bissau, Kenya, Libya, Madagascar, Mauritania, Mauritius, Mayotte, Morocco, Mozambique, Nigeria, Reunion, Sao Tome and Principe, Senegal, Seychelles, Sierra Leone, Somalia, South Africa, Sudan, Togo, Tunisia, Tanzania.

**ASIA:** Azerbaijan, Bahrain, Bangladesh, Burma, British Indian Ocean Territory, Cambodia, China, Christmas Island, Cocos Islands, Honk Kong, India, Indonesia, Iran, Iraq, Israel, Japan, Jordan, Kazakhstan, Kuwait, Lebanon, Malaysia, Maldives, North Korea, Oman, Palau, Papua New Guinea, Philippines, Qatar, Russia, Saudi Arabia, Singapore, South Korea, Sri Lanka, Syria, Taiwan, Thailand, Turkey, Turkmenistan, United Arab Emirates, Viet Nam, Yemen.

**OCEANIA:** American Samoa, Australia, Fiji, French Polynesia, Guam, Kiribati, Marshall Islands, Martinique, Micronesia, New Caledonia, New Zealand, Norfolk Island, Northern Mariana Islands, Samoa, Solomon Islands, Timor Leste, Tonga, Vanuatu.

**Fig. 4** Global map of seagrass distribution, species richness, and bioregions (Ashkar 2020)

meadows in addition to impairing their ability to root due to the alteration of the plants' perception of gravity.

Noise exposure not only affected *Posidonia* health status and the plant's capacity to root itself, but also affected the plant's correct nutritional processes by decreasing the number of rhizome starch grains that store energy, as well as by degrading the specific fungal symbionts of *P. oceanica* roots and consequently altering the seagrass metabolism. Seagrass meadows considered an oceanic equivalent to primary forests rank among the most valuable biomes for the planet (Fig. 4) (den Hartog 1970; Boudouresque et al. 2006). This work revealed the sensitivity of aquatic plants to sound. Man-made noise could alter marine biodiversity and contribute to the fragility of seagrass meadows. More research is needed to determine if artificial noise affects seagrass in the juvenile stages, when the plant needs to use its ability to sense gravity in order to find soil and roots.

## Conclusion

This chapter provided current knowledge about the sensory capabilities of plants and their communication abilities above and below ground, as well as advances in plant acoustics, including responses to sound at behavioral, physiological, and molecular levels. In addition, it presented recent findings on acoustic impact on plants, specifically on marine species. Particularly, it was shown that anthropogenic noise can inflict ultrastructural damage on various *Posidonia oceanica* tissues, which could have implications at the whole marine ecosystem level.

More research is needed on acoustic emission and perception abilities in plants, including:

1) A better understanding of the molecular and cellular mechanisms of sound perception and signal transduction, specifically for aquatic plants.
2) Understanding the possible underground sound communication capacities in plants; understanding the intricate communication networks established between plant community individual roots as well as with other trophic levels.
3) Elucidating if marine communities, such as seagrass, possesses some communication system, chemical, or acoustic, to warn of the presence of damaging elements at physiological, ultrastructural, or behavioral level.
4) Assessing if the underground sophisticated root mesh is capable of transmitting other danger announcements related to stress factors caused by climate change, pollution, and overfishing.
5) Consideration must be given to the essential role of the root as "a brain" or point of sensory information input and organization, as well as a command center for decision-making.
6) Knowledge of damaging sound levels to develop criteria for evaluating the impact of sound on marine plants.

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# Effects of Anthropogenic Noise on Marine Zooplankton

96

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**Abstract**

Anthropogenic underwater noise is increasing in the oceans, making the establishment of sound exposure criteria for marine taxa imperative for protecting and managing a good environmental status of the seas. Zooplankton play a key role in the marine carbon cycle and act as linkage between phytoplankton and higher trophic levels. Still, studies of the effects of underwater noise on zooplankton are

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limited. This chapter reviewed 21 studies that found positive, negative, and no responses of marine holo- and meroplankton to continuous and impulsive underwater noise from experiments conducted in the laboratory, field, or both. The majority of the examined effects were related to development (28%), mortality (25%), and orientation and settlement (19%). Meroplankton (68%) were the most investigated zooplankton group. These studies indicated that both lethal and sublethal effects are stage- and species-specific. Variations were found in the implemented sound exposures, and a standardized manner in reporting sound levels is needed. Further studies combining laboratory and field experiments are required to establish sound exposure criteria for marine ecosystems, which can be used as international directives for underwater noise.

**Keywords**

Zooplankton · Noise pollution · Anthropogenic noise · Mortality · Behavior · Development · Impulsive noise · Continuous noise

## Introduction

Anthropogenic underwater noise is an increasing source of environmental stress (Fritschi et al. 2011; Duarte et al. 2021). The goal of directives, such as the EU Marine Strategy Framework Directive (Directive 2008/56/EC 2008), is to monitor underwater noise and establish threshold values that do not adversely affect the marine environment. Typically, noise is divided into continuous and impulsive underwater noise, for which no threshold value has yet been integrated (Dekeling et al. 2020; Vasilakopoulos et al. 2022). Throughout the chapter, the term "sound exposure criteria" is used instead of "threshold values," which is defined as "sound levels, based on acoustic response thresholds, above which sound levels may have adverse effects on specified animals" (Hawkins et al. 2020).

Continuous underwater noise can be described as a constant or slowly fluctuating sound pressure over a long time interval, which neither is instantaneous nor involves (typically) pulsed characteristics. Continuous underwater noise is a broad term for several different sound sources, such as boat traffic or turbine sound from offshore wind parks (Kinneging and Tougaard 2021), that increases ambient sound levels, particularly at low frequencies. The sound levels are determined by the number and size of the boats and turbines, whereas boat noise levels are further controlled by the speed of the vessel (Kaplan and Solomon 2016; Tougaard et al. 2020). In contrast, impulsive underwater noise is defined as an instantaneous change in sound pressure, often of high amplitude, e.g., from air guns used in seismic surveys and from pile driving. During seismic surveys, air guns emit impulsive sound primarily at low frequencies by rapidly releasing compressed air (Hovem and Tronstad 2012). Pile driving emits intense broadband impulsive sound that can propagate far through the water and ocean floor (Stenton et al. 2022). Much of what is known about the effects

of anthropogenic underwater noise, such as altered or disrupted communication, orientation, feeding, and antipredator behavior, comes from studies investigating marine mammals and fish (Popper and Hawkins 2016, 2018; Duarte et al. 2021). Less is known about marine invertebrates (Hawkins and Popper 2017; Murchy et al. 2019), such as zooplankton.

Zooplankton are an essential link between primary producers and higher trophic levels; hence their fitness and mortality directly affect the path and storage of organic carbon in the food web (Steinberg and Landry 2017). They comprise a large group of animals that drift in the pelagic zone, either entirely (holoplankton) or partly (meroplankton), throughout their life cycle. Zooplankton differ in morphological and functional traits, e.g., size (Lindeque et al. 2013; Deagle et al. 2018; Evans et al. 2020), energy content (Chen et al. 2019), and behavior (Kiørboe et al. 2010; Almeda et al. 2017). Therefore, various groups or species may be affected by external stressors, such as underwater sound, in different ways and degrees.

By interpreting the results and methods of previous studies investigating the effects of anthropogenic noise on zooplankton, this chapter summarizes the previous findings and highlights the knowledge gaps for future studies.

## Methods

All studies included in this chapter were systematically selected by first creating an overview of marine zooplankton groups (Table 1). Note that ichthyoplankton, meroplankton consisting of fish eggs and larvae, were not included. This group contributes a relatively small proportion of plankton, and it is challenging to implement a systematic definition for different species at the transition stage of nekton. The second criterion used in the literature search was sound source. For continuous underwater noise, all low-frequency sound sources (independent of the sound exposure duration used in the studies) that were not impulsive by definition (see Introduction) were included in this chapter. Noise that did not originate from a real-life anthropogenic sound source found in the ocean was categorized as artificial low-frequency noise. For impulsive underwater noise, only two sound sources have been investigated for their effects on zooplankton: seismic air guns and pile driving.

Potential studies were searched using the phrase *noise effects on zooplankton*. The following terms were used for sources of noise: *underwater noise*, *low-frequency noise*, *continuous noise*, *boat/vessel/traffic noise*, *turbine noise*, *wind farm noise*, *air guns*, *seismic exploration*, *seismic surveys*, and *pile driving*. The groups listed in Table 1 were used to search for zooplankton species and (sub) phyla.

Three main types of information were extracted from the collected studies: (1) sound exposure, in which the sound sources and sound characteristics of various studies were presented, (2) study animals and developmental stages, and (3) study design and outcomes. For the latter, the outcomes of each study were categorized into lethal (mortality) and sublethal effects. The sublethal effects were further

**Table 1** Overview of marine zooplankton groups (Animalia)

| Kingdom | Holo-meroplankton | (Sub) phylum | Class or order |
|---|---|---|---|
| Animalia | **Holoplankton** | Cnidaria | Hydrozoa (also part of meroplankton) |
| | | | Myxozoa |
| | | | Cubozoa |
| | | | **Scyphozoa*** |
| | | Ctenophora | Ctenophora |
| | | Rotifera | Rotifera |
| | | Platyhelminthes | Platyhelminthes |
| | | Nematomorpha | Nectonema |
| | | Mollusca | Heteropoda |
| | | | Pteropoda |
| | | Crustacea | Cladocera |
| | | | Ostracoda |
| | | | Isopoda |
| | | | **Copepoda*** |
| | | | Mysidacea |
| | | | Amphipoda |
| | | | Euphausiacea |
| | | Chaetognatha | Chaetognatha |
| | | Chordata | Appendicularia |
| | | | Pyrosoma |
| | | | Doliolida |
| | | | Salpida |
| | **Meroplankton** | Mollusca | **Bivalvia*** |
| | | | **Gastropoda*** |
| | | | Nudibranchia |
| | | | Cephalopoda |
| | | Cnidaria | **Anthozoa*** |
| | | Chordata | **Ascidians*** |
| | | Crustacea | **Thecostraca*** |
| | | | **Malacostraca*** |
| | | Annelida | Polychaeta |
| | | Nemertea | Nemertinea |
| | | Bryozoa | **Gymnolaemata*** |
| | | Echinozoa | **Echinoidea*** |

The zooplankton groups reported in the chapter are marked in bold and with an asterisk (*). The table is adapted from the *Census of Marine Zooplankton*. Retrieved 10.11.2022: http://www.cmarz.org/species_pages/phyla/phyla.htm

divided into physiology (including oxidative stress and gene expression), morphology (including injuries to cells), development, orientation or settlement, and behavior (including swimming, feeding, and antipredator behavior). The responses for each outcome were then categorized as either negative (e.g., increased mortality),

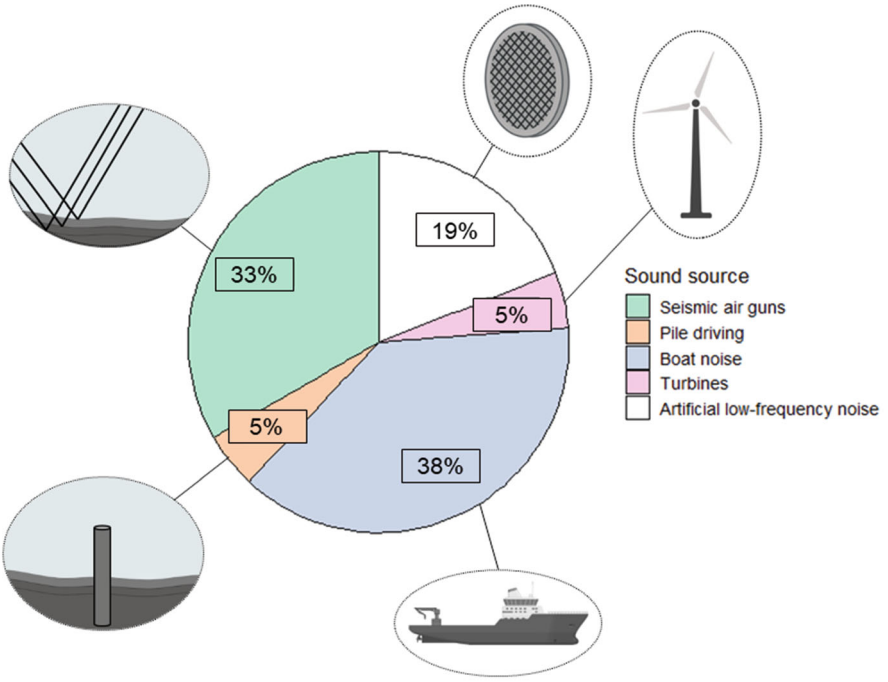



**Fig. 1** Schematic overview of the proportion of studies covering the five main sources of sound exposure ($n_{all\ studies} = 21$): seismic air guns, pile driving, boat noise, turbines, and artificial low-frequency noise. The legend is read counterclockwise through the pie chart. (Icons by biorender.com)

other (e.g., different patterns in the settlement but not clearly negative), or none (no response observed). Note that potential positive effects were categorized as "other," as it is with the present information not possible to determine whether, for example, increased settlement positively affects an individual or populations. All study animals were individually considered as either meroplankton or holoplankton. In addition, if a study presented several degrees of effects, for example, negative effects on mortality, but no effect on settlement, both outcomes were included. Therefore, the number (n) of each parameter differed among the graphical overviews (Figs. 1, 2 and 3).

## Sound Exposure

Of all the studies ($n = 21$), 13 (62%) investigated the effects of continuous underwater noise, and 8 investigated (38%) the effects of impulsive underwater noise on zooplankton (Fig. 1, Table 2).

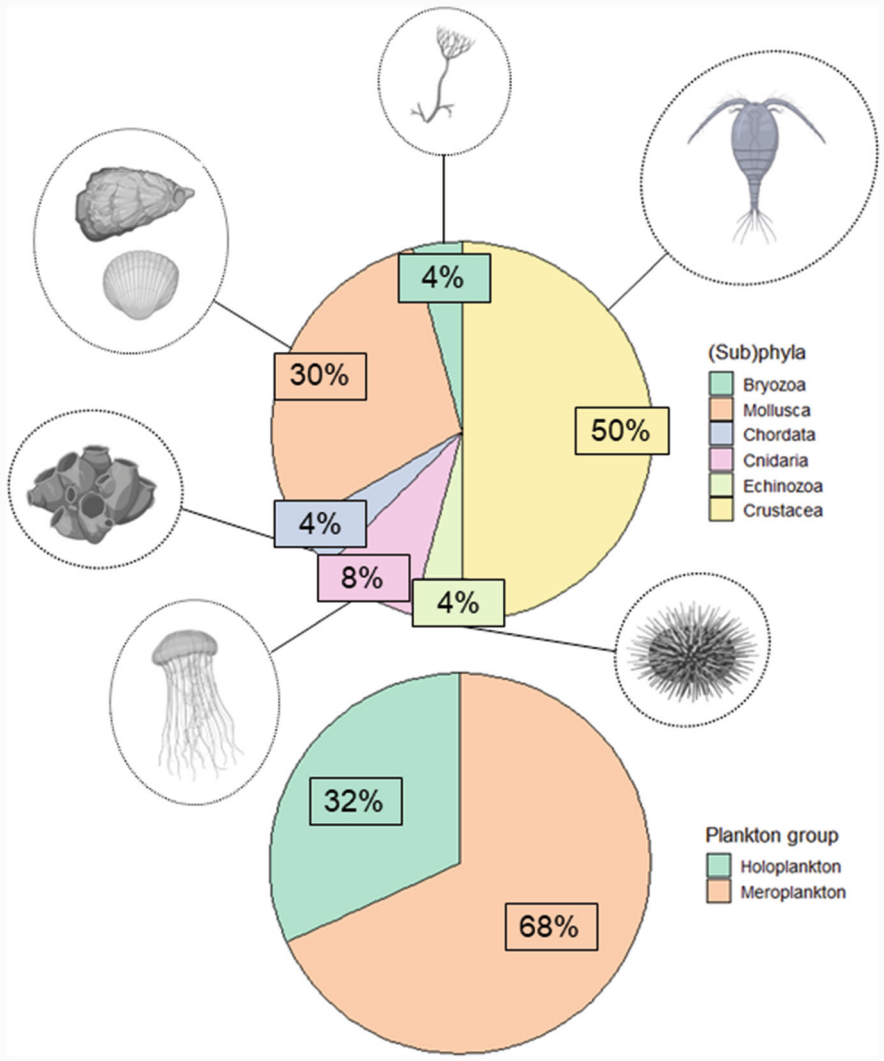



**Fig. 2** Schematic overview of the proportion of different zooplankton groups. Upper panel, groups of subphylum or phylum covered in the studies ($n_{\text{total reported (sub)phylum}} = 24$): Bryozoa, Mollusca, Chordata, Cnidaria, Echinozoa, and Crustacea*; and lower panel, plankton group ($n_{\text{total reported plankton groups}} = 22$): holoplankton and meroplankton. Note that one study could include several sub(phyla) or plankton groups. The legends are read counterclockwise through the pie chart. *In McCauley et al. (2017), in situ zooplankton samples were investigated. However, as the majority consisted of copepods, the study animals are categorized as Crustacea. (Icons by biorender.com)

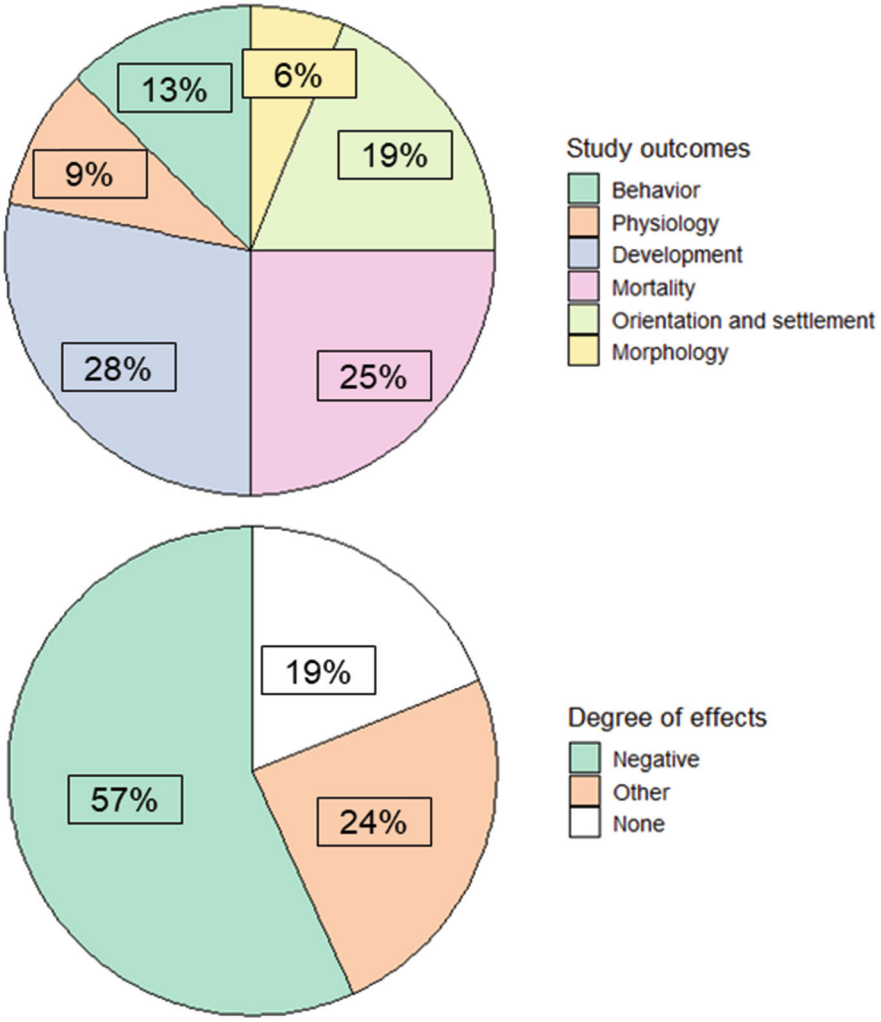



**Fig. 3** Schematic overview of the proportion of different outcomes and the degree of effects observed in the studies. Upper panel, categories of study outcomes covered in the studies ($n_{\text{total reported study outcomes}} = 32$): behavior, physiology, development, mortality, orientation and settlement, and morphology; and lower panel, the degree of effect ($n_{\text{total reported effects}} = 37$): negative, other, and none. Note that one study could include study outcomes or report different effects. The legends are read counterclockwise through the pie chart

**Table 2** Overview of the sound exposures from reviewed studies. Information is given on the sound sources, sound levels, and exposure durations

| Study | Sound source | Sound levels | Duration |
|---|---|---|---|
| Tremblay et al. (2019) | Artificial: Noise Egg (method, de Jong et al. 2017) | < 5000 Hz up to 20 dB (PSD) ↑ than ambient | 9 days |
| Solé et al. (2021) | Artificial: Playbacks of discrete frequencies of 100–1000 Hz and discrete and combined frequencies of 350 and 500 Hz | 160–195 dB (SEL) re 1 $\mu Pa^2$ | Few hours up to 2 weeks |
| Nedelec et al. (2014) | Boat noise: playback | 110 dB (PSD) re 1 $\mu Pa^2$ $Hz^{-1}$; 80 dB re 1$(\mu m\ s^{-2})^2$ $Hz^{-1}$ at around 100 Hz | 12-h cycle for 5 days |
| | | 20 dB ↑ than ambient | |
| Branscomb and Rittschof (1984) | Artificial: Discrete frequency playback at 15, 25, 30, 37, and 45 Hz | No information | 20 h |
| McDonald et al. (2014) | Boat noise: playback | 30–100 Hz 110–120 dB (RMS) re 1 µPa $Hz^{-1}$; 128–141 dB re 1 µPa | Loop of 2-min sequences: up to 26 h |
| Stocks et al. (2012) | Boat noise: playback | 100–300 Hz −30 dB re 1 $\mu Pa^2$ $Hz^{-1}$; 30–50 dB ↑ than control | 5 min |
| Pine et al. (2012) | Wind turbine: playback | 145 dB re 1 µPa; 20 dB ↑ than mudflat sounds | Every 6 h checked, up more than 200 h |
| | Tidal turbine: playback | | |
| Jolivet et al. (2016) | Boat noise: playback | 127 ± 3 dB re 1 µPa 100–1000 Hz; 20 dB ↑ than ambient | 67 h |
| Wilkens et al. (2012) | Boat noise: playback | 126 and 100 dB (RMS) re 1 µPa; 60 dB and 40 dB ↑ than ambient | 8–16 h |
| Sal Moyano et al. (2021) | Boat noise: playback | 129.5 dB (SPL RMS) re 1 µPa; 9 dB ↑ than ambient | 5 min |
| Solé et al. (2016) | Artificial: playback of 50–400 Hz sinusoidal wave sweeps | 157 ± 5 dB (not clear) re 1 µPa; peak 175 dB (SPL) re 1 µPa | 2 h |
| Lecchini et al. (2018) | Boat noise: playback | At 400 Hz and 900 Hz 90 dB (SPL) re 1 µPa; >10 dB ↑ than controls | 4-h playback track |
| Kühn et al. (2023) | Boat noise: playback | Peak at 130 Hz 150 dB (PSD) re 1 $\mu Pa^2$ $Hz^{-1}$; 174 dB PSD (RMS) re 1 $\mu Pa^2$ $Hz^{-1}$ one boat passing | 24-h loop of a 20-min recording |
| McCauley et al. (2017) | Seismic air guns: 1 air gun (150 $inches^3$) | SEL 156 (509–658 m) – 153 (1.1–1.2 km) dB re 1 $\mu Pa^2 s$ | App. 110 shots |
| | (13.8 MPa) | | |

(continued)

**Table 2** (continued)

| Study | Sound source | Sound levels | Duration |
|---|---|---|---|
| Fields et al. (2019) | Seismic air guns: 2 air guns*260 inches$^3$ (13–14.4 MPa) | SEL 221 dB re 1 μPa$^2$s (0 m), 183 dB re 1 μPa$^2$s (25 m) | Not given |
| Vereide et al. (2023) | Seismic air guns: 2 air guns*40 inches$^3$ (11 MPa) | SEL 180 (50 m), 166 (1100 m) dB re 1 μPa$^2$s | 2.5 h |
| Parry et al. (2002) | Seismic air guns: Air gun array (24) (total volume 3542 inches$^3$) | Max. source strength 211 dB/rel to 1 μPa at 1 m at frequency 50 Hz | Not given (but 2 km transect) |
| de Soto et al. (2013) | Seismic air guns: playbacks of an air gun array (6920 inches$^3$) | SEL from 161 to 165 dB re 1 μPa$^2$s | 24–90 h, 3 s intervals |
| Pearson et al. (1994) | Seismic air guns: Air gun array (7), total volume 840 inches$^3$ | Max. (1 m) 230 dB re 1 μPa | 10 s intervals |
| Day et al. (2016) | Seismic air guns: 1 air gun (3 setups): 45 inches$^3$ (2000 psi), 150 inches$^3$ (1300 psi), 150 inches$^3$ (2000 psi) | Max. SEL 186, 189, 190 dB re 1 μPa$^2$·s (3 setups) | 17.2–24.3 min, 11.6 s intervals |
| Stenton et al. (2022) | Pile driving: playback | 170 $dB_{pk\text{-}pk}$ re 1 μPa; 82.7 ± 3.4 – 84.6 ± 2.6 $dB_{pk}$ re 1 μm s$^{-2}$ | 5 days |

## Continuous Underwater Noise

The effects of boat noise on zooplankton were examined using playback sound exposure in a laboratory tank or field (Table 3), whereas only McDonald et al. (2014) used real-life observations to confirm the results of the playback exposure. Furthermore, all studies, except for Nedelec et al. (2014), conducted playback experiments in laboratory tank setups without additionally measuring particle motion. Note that Nedelec et al. (2014) conducted playback experiments in the field. These studies also differed in their control exposures, using silent or aquarium ambient sound controls (Stocks et al. 2012; Wilkens et al. 2012; McDonald et al. 2014; Jolivet et al. 2016; Kühn et al. 2023) or playback recordings from natural ambient soundscapes (Nedelec et al. 2014; Lecchini et al. 2018; Sal Moyano et al. 2021). The duration of the sound exposure ranged from 5 min to several days (Table 2). Lastly, the studies reporting spectral information on the original recordings measured similar sound pressure and particle motion levels for the playback exposures. However, sound levels varied between the studies due to different boat noise recording approaches and playback setups (Table 2).

Only one study examined the effect of turbine noise (Pine et al. 2012). Laboratory experiments were conducted to test the effects of tidal turbine noise (digital analog based on published spectra) and coastal wind turbine noise (playback) (Table 2). The maximum output sound exposure levels used in their experiments were 30 dB and 9 dB less than the published in situ sound levels from tidal and wind turbines, respectively.

Four studies used different approaches to generate artificial low-frequency noise: broadband noise, discrete frequencies, or specific combinations of frequencies (Table 2). Overall, the exposure sound levels, especially at low frequencies, were well above the ambient sound levels in studies that provided information on the measured sound spectra.

### Impulsive Underwater Noise

In seven studies that investigated the effects of air guns used in seismic surveys on zooplankton, variations were found in the number of air guns used, distance to the air guns, and exposure duration (Table 2). Therefore, the measured sound levels showed considerable differences between studies (Table 2). Furthermore, the studies differed in terms of exposing the animals to one or a series of shots (Table 2). One study used playback exposure from air guns (de Soto et al. 2013), while the rest conducted field experiments. Only one of these studies provided data on particle motion (de Soto et al. 2013).

Stenton et al. (2022) conducted two playback pile driving exposure experiments and reported the sound pressure and particle motion levels (Table 2). The measured playback sound levels resulted in a smaller proportion of sound at lower frequencies but increased at higher frequencies compared to the in situ pile driving recordings. The control playback consisted of ambient recordings made in the field, without anthropogenic sounds.

## Study Animals and Developmental Stages

Of all the included studies, 7 investigated the effects on holoplankton, and 15 investigated the effects on meroplankton. Furthermore, 50% of all study species were crustaceans (Fig. 2 and Table 3). Additionally, the reported effects on zooplankton varied between negative (57%), other (24%), and none (19%) (Fig. 3 and Table 3).

### Continuous Underwater Noise

Of the studies investigating the effects of continuous noise on zooplankton, 4 of the 21 species were holoplankton and the rest were meroplankton (Table 3).

#### Zooplankton Groups

Studies that focused on holoplankton ($n = 4$) examined the potential effects of noise on copepods and jellyfish. The physiology, morphology, and behavior of all copepods were negatively affected by noise (Table 3). Jellyfish were negatively affected by noise in their morphological structure (Table 3). Nine studies examined meroplankton (Table 3). In crustaceans, continuous noise had either negative or no effect on settlement or development (Table 3). Likewise, Cnidaria and Bryozoa were

**Table 3** Overview of the study animals and outcomes from reviewed studies

| Study | Lab/ field | Holo/mero (plankton) | (Sub) phylum | Species | Stage | Outcomes | Effect |
|---|---|---|---|---|---|---|---|
| Tremblay et al. (2019) | L | Holo | Crustacea | *Acartia tonsa* | Preadult | Physiology | Increased stress |
| | | | | | | Behavior | None |
| Solé et al. (2021) | L, F | Holo | Crustacea | *Lepeophtheirus salmonis* | (pre-)Adult | Morphology | Injuries |
| Nedelec et al. (2014) | F | Mero | Mollusca | *Stylocheilus striatus* | Larvae | Mortality | Increased |
| | | | | | Egg, embryo | Development | Inhibited |
| Branscomb and Rittschof (1984) | L | Mero | Crustacea | *Balanus amphitrite* | Larvae | Orientation/settlement | Decreased |
| | | | | | | Development | Decreased |
| McDonald et al. (2014) | L, F | Mero | Chordata | *Ciona intestinalis* | Larvae | Orientation/settlement | Increased |
| | | | | | | Development | Increased |
| | | | | | | Mortality | Decreased |
| Stocks et al. (2012) | L | Mero | Mollusca | *Bembicium nanum* | Larvae | Behavior | Increased |
| | | | Bryozoa | *Bugula neritina* | | | Decreased |
| | | | Echinozoa | *Heliocidaris erythrogramma* | | | None |
| | | | Mollusca | *Crassostrea gigas* | | | Increased |
| Pine et al. (2012) | L | Mero | Crustacea | *Austrohelice crassa* | Larvae | Development | Delayed |
| | | | | *Hemigrapsus crenulatus* | | | Delayed |
| Jolivet et al. (2016) | L | Mero | Mollusca | *Mytilus edulis* | Larvae | Orientation/settlement | Increased |
| Wilkens et al. (2012) | L | Mero | Mollusca | *Perna canaliculus* | Larvae | Orientation/settlement | Increased |
| Sal Moyano et al. (2021) | L | Mero | Crustacea | *Cyrtograpsus angulatus* | Larvae | Orientation/settlement | None |
| | | | | *Cyrtograpsus altimanus* | | | |
| | | | | *Neohelice granulata* | | | |
| | | | | *Leptuca uruguayensis* | | | |

(continued)

**Table 3** (continued)

| Study | Lab/ field | Holo/mero (plankton) | (Sub) phylum | Species | Stage | Outcomes | Effect |
|---|---|---|---|---|---|---|---|
| Solé et al. (2016) | L | Holo | Cnidaria | *Cotylorhiza tuberculata* | Adult | Morphology | Injuries |
| | | | | *Rhizostoma pulmo* | | | |
| Lecchini et al. (2018) | L | Mero | Cnidaria | *Pocillopora damicornis* | Larvae | Orientation/settlement | Disruption settlement |
| | | | | *Acropora cytherea* | | | |
| Kühn et al. (2023) | L | Holo | Crustacea | *Acartia tonsa* | (pre-)Adult | Behavior | Decreased |
| McCauley et al. (2017) | F | Holo | Crustacea | Various, including Copepoda | Adult, larvae | Mortality | Increased |
| | | Mero | | | | | |
| Fields et al. (2019) | F | Holo | Crustacea | *Calanus finmarchicus* | Adult | Mortality | Increased |
| | | | | | | Behavior | None |
| | | | | | | Physiology | None |
| Vereide et al. (2023) | F | Holo | Crustacea | *Acartia tonsa* | Larvae | Mortality | Increased |
| | | | | | | Development | Decreased |
| Parry et al. (2002) | F | Mero | Mollusca | Not given | Larvae | Mortality | None |
| de Soto et al. (2013) | L | Mero | Mollusca | *Pecten novaezelandiae* | Larvae | Development | Decreased |
| Pearson et al. (1994) | F | Mero | Crustacea | *Cancer magister* | Larvae | Mortality | None |
| | | | | | | Development | None |
| Day et al. (2016)[a] | F | Mero | Crustacea | *Jasus edwardsii* | Larvae | Development | None |
| Stenton et al. (2022) | L | Mero | Crustacea | *Nephrops norvegicus* | Larvae | Mortality | Increased |
| | | | | | | Development | Delayed |
| | | | | | | Physiology | Increased stress |

Information is given on whether the experiments were conducted in the laboratory or the field, plankton group (holoplankton or meroplankton), (sub)phyla, study species, development stages, study outcomes, and the degree of effects

[a]The adult females were exposed, but the effects were measured in hatched larvae

negatively affected by noise in settlement and swimming behavior, whereas Echinozoa species showed no behavioral response (Table 3). Other types of effects not categorized as negative were only found in studies that looked into sublethal outcomes in bivalves, chordates, and gastropods (Table 3). However, Nedelec et al. (2014) also found adverse effects of sound on the mortality and development of gastropods.

### Development Stages

For the holoplankton groups, copepodite and adult copepod stages and the medusa stage in jellyfish (Scyphozoa) were used in the experiments (Table 3). Most meroplankton studies looked into the developmental stages before metamorphosis and larvae settlement. For example, two studies investigated the effects of continuous noise on Malacostraca species (Pine et al. 2012; Sal Moyano et al. 2021) and looked into the last larval stage before settlement, megalopa, considered as an intermediate stage between planktonic and benthic. Nedelec et al. (2014) investigated the mortality of veliger stages after hatching and the development of eggs or embryos of sea hares when exposed to boat noise. Lecchini et al. (2018) looked into the coral larval stage, planulae.

## Impulsive Underwater Noise

Of the studies examining the effects of impulsive underwater noise, three out of eight looked at holoplankton, whereas more than half looked at the effects on meroplankton (Table 3).

### Zooplankton Groups

Studies on holoplankton have focused primarily on copepods. Two studies solely investigated copepods, reporting adverse but limited effects on mortality after air gun exposure (Table 3). McCauley et al. (2017) looked at the effects of air guns on copepods but as a proportion of a natural zooplankton community from in situ samples. The authors reported severe effects on mortality after exposure.

All included meroplankton species (Table 3) were crustaceans or bivalves, but none of them investigated the same species. Furthermore, only two studies have reported adverse effects, one from air gun noise on scallop larvae and the second from pile driving on lobster larvae. Interestingly, both studies reporting negative effects exposed the animals to sound in the laboratory, whereas the other six exposed the animals to impulsive underwater sounds in the field.

### Developmental Stages

Of the two studies examining the effects on copepods, one focused on adult stages and the other on naupliar stages (Table 3). In contrast, McCauley et al. (2017) tested the effects on in situ zooplankton samples and reported several species and stages in the same study. Regarding meroplankton, only the younger stages prior to the settlement of bivalves and crustaceans have been investigated.

## Study Designs and Outcomes

The majority of studies examined sublethal effects: behavior (13%), physiology (9%), development (28%), orientation and settlement (19%), and morphology (6%) (Fig. 3). A number of the studies investigated several outcomes, e.g., combining development and settlement (Fig. 3, Table 3).

### Continuous Underwater Noise

The exposure duration ranged from instant (5 min) to long time intervals (up to 8 days) when investigating sublethal outcomes. For meroplankton species, the focus was on orientation and settlement, development, and changes in swimming behavior (Tables 2 and 3). Two studies (McDonald et al. 2014; Nedelec et al. 2014) found additional lethal effects of long-term sound exposure (Table 3). For holoplankton, feeding behavior and physiological and morphological aspects, such as oxidative stress and cell injury, were investigated (Table 3). All outcomes were measured either during or directly after the end of the experimental sound exposure.

### Impulsive Underwater Noise

Studies have covered different outcomes, from mortality to sublethal effects, such as behavior, gene expression, and development (Table 3). In addition, studies have focused on both short- and long-term effects, from immediate and long-term mortality to direct and long-term effects on physiology and development. For example, McCauley et al. (2017) examined the effects on mortality directly after air gun exposure, whereas Fields et al. (2019) investigated mortality additionally 1 week after exposure. Similarly, Vereide et al. (2023) measured mortality every day for 6 days after air gun exposure in copepod nauplii.

## Discussion

Similarities, variations, and inconsistencies were found among existing studies on the effects of underwater noise on zooplankton, which are discussed in the following section. Additionally, general predictions are made regarding the potential effects of noise pollution on individual zooplankton and their subsequent effects on marine ecosystems.

### Sound Exposures

All the presented studies on continuous underwater noise were conducted in a laboratory or field setup, where no studies used real-life in situ sound sources (except

for the additional field observations by McDonald et al. 2014). In general, responses in zooplankton were found during sound exposure from 10 dB above ambient sound levels, which is concerning because continuous noise through shipping led to more than 30 dB elevated sound levels, especially in coastal areas (Kinneging and Tougaard 2021; Farcas et al. 2020). In addition, Pine et al. (2012) observed adverse responses to sound levels well below the in situ measurements of offshore turbines. Thus, the reported effects of boat noise, wind, and tidal turbines can be observed in high traffic, fairways, and operational areas (Kinneging and Tougaard 2021). Furthermore, the use of artificial low-frequency noise (Table 2) may provide information on the general frequency and sound-level characteristics that affect marine organisms.

The comparability among studies is complicated for impulsive underwater noise owing to the different setups and difficulty of simulating impulsive sound in experimental tanks. Several studies have reported varying effects despite implementing similar sound levels but often with different exposure durations and distances from the study animals (Tables 2 and 3). The sound pressure level from air guns depends on the number of air guns (often consisting of 18–48 air guns distributed in subarrays), operating pressure, and total gun volume (NDP 2021). The volume of air guns used in real seismic surveys ranges from 20 to 800 inches$^3$, which is within the range of air guns used in the included studies. On the other hand, yielding source levels have been calculated up to 260 dB rms re 1 μPa at 1 m output pressure, which is substantially higher than those reported in the included studies. In addition, the peak spectral levels for industry arrays are often in the 5–300 Hz range (Hildebrand 2009). Another important aspect of transferring reported results to a real-life setting is the natural movement and location of the zooplankton. Zooplankton move along with water currents and undergo diel vertical migration, which may affect the timing and duration of impulsive noise exposure. Therefore, although a real-life survey may cover 1000–3000 km$^2$ (Hovem and Tronstad 2012) and continuously shoot over weeks or months (Weilgart 2013), the animals will not be constantly exposed throughout that period. The duration of exposure in the reviewed studies lasted for a maximum of 3–4 days, which may even be considered too long to be transferred into a real-life setting, considering advection and migration. To transfer the results from field or laboratory studies to a natural environment, other potential factors that may alter the strength of noise exposure must be considered.

## General Anthropogenic Underwater Noise

Anthropogenic noise from, e.g., boat traffic and seismic surveys potentially overlap as low frequencies can travel several thousand kilometers in deep waters. Boat noise is predominant in coastal areas, in shallow waters where low-frequency noise propagation is limited (Kinneging and Tougaard 2021; Farcas et al. 2020). An overall increase in commercial shipping can be expected in the future, in combination with larger ships and increased shipping distances across the oceans (Kaplan and Solomon 2016). As wind turbines are a constant noise source, the increased construction of wind parks (often several 100 turbines) will lead to locally increased ambient sound levels (Tougaard et al. 2020). A model by Siebert et al. (2014)

indicated that noise from air guns would be impulsive to intermittent up to 1000 km but may change beyond continuous underwater noise. The model shows that ambient sound levels can increase by 45 dB between 500 and 2000 km from the source, particularly at low frequencies. Hence, the long-distance effects of air gun shots on zooplankton could be comparable to those presented here for continuous underwater noise (see Sound Exposure). Furthermore, seismic surveys may occur for several months over large distances; therefore, the spectrum of impacts on zooplankton based on near-field experiments may underestimate the full range of effects.

An overall challenge when investigating the literature on continuous and impulsive anthropogenic underwater noise is that many studies present the received sound and exposure levels in different and limited formats. Reporting the information in a standard manner would facilitate the interpretation of the relationship between sound levels and the degree of effects in the study animals, which is urgently needed for the integration of sound exposure criteria.

Future studies would benefit from reporting (1) sound levels in standard formats, such as PSD for continuous underwater noise for comparisons among calibrated hydrophones and SEL for impulsive underwater noise. The amplitude of the PSD is normalized over the spectral resolution, which makes it independent of hydrophone sampling rates. Further, studies should report (2) distances, (3) durations, (4) ambient or controlled sound levels, and (5), if possible, particle motion. Particle motion can injure animals, mask sound cues, induce stress, or alter the behavior of marine organisms (Nedelec et al. 2021). In addition, particle motion levels in noise-related laboratory tank experiments are higher than those in in situ recordings or playback experiments in the field (see data by Nedelec et al. 2014, 2015). Hence, the true exposure perceived by zooplankton cannot be predicted from the measured sound pressure levels.

## Study Animals and Developmental Stages

The degree to which marine animals are affected by anthropogenic noise is often species- and stage-specific. Furthermore, impacts may be linked to their perception of sound, physiology, and type of sound source used in the exposure.

### Perception of Underwater Sound

Despite the increasing use and knowledge of anthropogenic sound in the oceans, understanding how invertebrates affect and detect sound is limited. In particular, sound perception in zooplankton needs to be fully understood (Hawkins and Popper 2017). In contrast to marine mammals and certain fish species, zooplankton are incapable of "hearing" and may be affected in other ways by underwater sound, e.g., through mechanoreception or cell injuries.

Crustaceans, such as copepods, use mechanoreceptive setae to detect predators, prey, and potential mates. Mechanoreception requires relative physical movement, which can be produced by the hydroacoustic motion of a receptor-activating structure involving a stretch-sensitive ion-channel molecule (Yen et al. 1992; Lenz and

Hartline 2014). By detecting hydrodynamic stimuli, several copepod species, such as *Lepeophtheirus salmonis* (Heuch and Karlsen 1997) or *Calanus finmarchicus* (Weatherby and Lenz 2000), detect and avoid prey and obstacles. Thus, anthropogenic noise can disrupt the fitness-relevant processes in crustaceans. For example, anthropogenic noise may mask the acoustical cues from a soundscape that could alter the settlement choice for suitable habitats (Branscomb and Rittschof 1984) or potentially mask the hydromechanical cues from a prey (Kühn et al. 2023). Noise can also distract the animals and alter physiological processes, such as increased stress (Tremblay et al. 2019, Stenton et al. 2022), or lead to cell injury (Solé et al. 2021). For example, fusion of sensory cells was found in the parasitic copepod species *L. salmonis* (Solé et al. 2021). Solé et al. (2021) hypothesized that such changes in an animal's sensory organs may impair the detection of a potential host.

Bivalves may also detect and react to sound (Stocks et al. 2012; Wilkens et al. 2012; Jolivet et al. 2016). However, the underlying mechanisms are not yet well understood. For example, Eggleston et al. (2015) found an increased settlement rate in oyster larvae (*Crassostrea virginica*) during sound exposure compared to silent conditions, whereas clam (*Mercenaria mercenaria*) larvae showed no differences. Because bivalves lack gas-filled spaces, they are most likely unaffected by pressure changes (de Soto et al. 2013). However, studies have suggested specialized organs and statocysts as mechanisms that make them perceptible to sound particle motion (Hubert et al. 2022).

Other mollusks, such as gastropods, may also be capable of detecting sound via their statocysts. For example, Nedelec et al. (2014) hypothesized three mechanisms by which noise affects the development and survival of sea hares. First, the development might have been affected by tissue damage owing to strong molecular vibrations from boat noise, where a potentially similar mechanism was found by Solé et al. (2016, 2021). Second, the survival of sea hare veligers may have decreased due to barotrauma. Finally, they suggested that sea hares might suffer from stress when exposed to noise.

Jellyfish possess hair cell structures that respond to mechanical displacements (mechanoreceptors) from, e.g., vibrations and sound pressure waves (Solé et al. 2016). How noise causes cell damage in the two Cnidaria species is unknown. Solé et al. (2016) hypothesized that noise propagation vibrations could directly trigger the mechanical displacement of external sensory cells. Like jellyfish, other Cnidaria species, such as coral planulae, can detect sounds through cilia. Therefore, coral planulae can distinguish between soundscape habitats and boat noise, which may lead to the avoidance or masking of the signal needed from a suitable habitat (Lecchini et al. 2018).

In summary, the effects of anthropogenic noise may be either detection-dependent (mechanoreception) or detection-independent. First, if zooplankton detect noise with their sensory organs, it could increase stress, distract the animal, or mask acoustic signals from their surroundings. Second, strong vibrations from noise can cause morphological damage independent of their detection. However, knowledge still needs to be completed, and for many invertebrates, it remains to be seen if and how they detect sound (Stocks et al. 2012).

### Differences in Morphological Characteristics and Developmental Stages

The morphological differences between zooplankton groups may indicate that certain species are affected by one component of underwater sound, such as particle motion (Nedelec et al. 2016; Popper and Hawkins 2018), whereas others may be more affected by other components. For example, adult copepods, with a robust external layer and internal lipid sack of different densities than the surrounding water, may be affected by pressure changes from air guns. Similarly, de Soto et al. (2013) suggested that damage observed in scallop larvae after air gun exposure, because of their small size and absence of strong tissue density gradients, is more related to particle motion rather than pressure changes. These morphological differences may also explain the variation in the results of studies examining different zooplankton groups. For continuous underwater noise, contradictory results, namely, no response versus adverse effects, have been found in behavioral experiments in copepods (Tremblay et al. 2019, Kühn et al. 2023), whereas physiology and morphology were negatively affected in all investigated holoplankton groups (Solé et al. 2016, 2021; Tremblay et al. 2019). Contradicting results were also found in studies investigating the effects of impulsive noise (Table 3), although only two studies found negative effects on meroplankton, in contrast to the holoplankton studies that observed adverse effects in all studies (Table 3). Overall, the responses to the sound exposures were highly species-specific for holo- and meroplankton, but crustaceans seemed to be most often negatively affected. The sensitivity of crustaceans to sound may be due to their highly sensitive mechanoreception, as described above, and the reaction thresholds are stage-dependent (Fields and Yen 1997). In addition, only one study focused solely on the juvenile stages of holoplankton (Table 3). Regarding the importance of nauplii and their role in ecosystems, more knowledge is needed regarding earlier plankton stages. In general, the majority of studies focused on meroplankton in order to study benthic organisms (Table 3). Regarding the importance of zooplankton as a functional group, more studies focusing on holoplankton and their different developmental stages are needed.

## Study Designs and Outcomes

Continuous underwater noise can alter the sublethal processes and mortality after only a few hours of exposure (Tables 2 and 3). Note that the reported swimming activity and orientation effects were seen after 5 min of sound exposure (Stocks et al. 2012, Sal Moyano et al. 2021). Hence, continuous underwater noise alters behavioral responses to short-term exposure; however, there is no information about long-term behavioral responses in relation to physiological responses. More than half of the impulsive underwater noise studies either focused primarily on or included mortality but also on development and physiology. In general, there is an increasing trend in the combination of areas of biology and outcomes (Wale et al. 2021). This was also reflected in Wale et al. (2021), who examined the trends in methods and future directions of studies investigating the effects of anthropogenic noise on

marine invertebrates and reported that 39% of the studies in the same period covered a combination of biological outcomes. Combining investigations into physiological, morphological, and behavioral outcomes in noise research will facilitate unraveling the underlying mechanisms of noise impacts in different species.

Another important aspect to investigate is the effect of the experimental arena by comparing anthropogenic underwater noise studies done in the laboratory with studies done in the field. Several studies (e.g., Day et al. 2016) have previously highlighted this issue, which is highly relevant when translating possible effects into real-life effects. First, the studies included in this chapter presented and discussed contrasting laboratory and field results. For example, de Soto et al. (2013) conducted experiments in small tanks in the laboratory using air gun playbacks and reported negative effects on development in lobster larvae. However, Day et al. (2016) suggest that the long wavelengths produced by accurate sound sources, such as an air gun, cannot be emulated in a small tank. Also, the sound reflected from the surfaces in small tanks might result in an unrealistic picture and relationship between the sound pressure and particle motion of the sound (Parvulescu 1964, 1967; Rogers et al. 2015, Nedelec et al. 2016). Second, laboratory and field studies provide different possibilities regarding study design and outcomes. For example, when using a laboratory approach, the mechanisms behind potential damage to zooplankton can be easier isolated than out in the field. Furthermore, one can potentially exclude other impacts by keeping the study animals in a controlled environment. For example, when conducting field experiments, one needs to consider background mortality from, e.g., boat propellers or other physical disturbances, which are easier to control in the laboratory. In addition, a controlled environment is relevant when looking into long-term vs. immediate effects, as long-term effects may be challenging to detect in the field in a natural environment – but are highly relevant. For example, short-time experiments may miss the impact of potential habituation, as found in blue mussels' reduced responsiveness over several sound exposures (Hubert et al. 2022), and possible population effects.

## Potential Population and Ecosystem Effects

Based on the results of this chapter, any response of zooplankton to underwater noise can affect ecosystem functioning. For example, when zooplankton species are affected by noise, any positive or negative outcomes on mortality, physiology, or fitness-relevant behavior could lead to increased or decreased species abundance. In turn, these changes in abundance could lead to modifications in grazing and predation pressure on phytoplankton and other zooplankton groups (Lynam et al. 2017). As noise can alter fitness-relevant behavior at the individual level, it may also affect the strength and direction of intra- and interspecies interactions (Lagardère 1982; Schwarz and Greer 1984; McCauley et al. 2003). Therefore, a potential outcome in a system with low resilience may shift the phytoplankton and zooplankton abundance and community structure. Furthermore, higher trophic levels may be affected by an increase or decrease in energy transfer from lower levels. On the other hand,

zooplankton, especially copepods, are important for deep-sea carbon storage by fast-sinking fecal pellets (Shatova et al. 2012; Turner 2015). Hence, a change in abundance may increase or decrease the rate of this process.

## Conclusions

Underwater noise affects zooplankton, which is either detection-independent or detection-dependent. For the latter, interpreting the noise exposure in terms of particle motion is important, and a standardized manner of reporting sound levels is needed. However, several studies have reported no effects of underwater noise, indicating that the effects vary among sound sources, strength, and zooplankton groups. In order to understand how noise affects zooplankton and potentially establish sound exposure criteria, the interplay of the effects of noise on physiology, morphology, and behavior must be investigated simultaneously in different developmental stages. Information on the differences between short- and long-term exposure effects and potential habituation to various sound sources is largely missing. Further studies combining laboratory and field experiments are needed to predict future outcomes in the real world and, hence, the effect of noise on ecosystem functioning.

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# No Evidence for an Effect of Chronic Boat Noise on the Fitness of Reared Water Fleas

97

Loïc Prosnier, Emilie Rojas, and Vincent Médoc

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**Abstract**

Ecotoxicological studies mainly focus on chemical pollution, however, since past decades, there has been a growing interest for acoustic pollution. Previous studies on underwater acoustic pollution showed that noise affects vertebrates' behavior, like fish and marine mammals. However, little is known about

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invertebrates and there is a huge lack concerning zooplankton species which are widely used as bioindicators in chemical pollution. Consequently, it could be useful to assess the impact of noise in terms of fitness (survival and fecundity). Here, isolated water fleas, *Daphnia magna*, were reared from birth to death in the presence or absence of motorboat noises. Effects on lifespan and clonal offspring production (e.g., clutch size, number of offspring produced along life, etc.) were measured and chronic exposure to boat noise did not affect *Daphnia*'s fitness. The spectral and temporal features of the sounds could explain the results. This study highlights the importance to integrate noise pollution into ecotoxicological research.

**Keywords**

*Daphnia magna* · Acoustic pollution · Boat noise · Fitness · Survival · Reproduction

## Introduction

Freshwater ecosystems suffer seriously from all types of anthropogenic pollution (e.g., chemicals, light, radioactivity, nanopollution, and sounds) (see for instance Longcore and Rich 2004; André et al. 2011; Song et al. 2020; Jan et al. 2022) but the most documented to date remain chemical pollutions (e.g., industrial effluents, urban waste, pesticides, and drugs) (Truhaut 1977; Villeneuve and Garcia-Reyero 2011). Toxicological studies have documented, in a comprehensive and accurate way, the effects of different types of pollutants (e.g., ion, heavy metals, and drugs), exposure durations (acute or chronic), intensities (e.g., median lethal dose $LD_{50}$), and the interactions between them and with environmental parameters (temperature, acidity, humidity, etc.). Those results have contributed to the general knowledge allowing the evaluation of other types of pollution.

This study focuses on acoustic – or noise – pollution described as a pervasive and omnipresent pollutant found in all ecosystems (terrestrial, marine, and freshwater) (Shannon et al. 2016; Popper and Hawkins 2019; Kunc and Schmidt 2019), and which represents a growing research topic (Williams et al. 2015; Slabbekoorn 2019). Specifically, this study focuses on the effect of ship noise, an important noise disruption threatening aquatic systems (Rountree et al. 2020; Duarte et al. 2021). Moreover, studies on noise pollution have mainly focused substantially on behavior (Richardson et al. 1985; Duarte et al. 2021). Thus, contrary to many ecotoxicological studies since Truhaut (1977), it remains a gap in understanding how noise pollution affects individual fitness, i.e., survival and fecundity (Francis and Barber 2013; Read et al. 2014).

Although the effects of noise on large invertebrates, such as decapods or bivalves, have recently received substantial interest (see the reviews of Popper et al. 2001; Solé et al. 2023), research largely neglected zooplanktonic invertebrates

(Hawkins et al. 2015; Prosnier 2022; ▶ Chap. 96, "Effects of Anthropogenic Noise on Marine Zooplankton"), despite their ecological importance and general use as bioindicators in ecotoxicology (Parmar et al. 2016). Zooplanktons do not possess hearing structures, however, they present mechanoreceptors that allow them to detect particle motion, the other component of a sound with pressure (Gassie et al. 1993; Buskey et al. 2002). For instance, Gassie et al. (1993) found that marine copepods (*Acartia fossae*) can detect water vibrations, and Buskey et al. (2002) showed that vibrations can lead to an individual acceleration of *Acartia* spp. Marine zooplanktons (e.g., copepods) exposed to acute air guns show a reduction of their survival (McCauley et al. 2017; Fields et al. 2019; ▶ Chap. 96, "Effects of Anthropogenic Noise on Marine Zooplankton"). Copepods also show reduced foraging rate during boat noise exposure (▶ Chap. 96, "Effects of Anthropogenic Noise on Marine Zooplankton"). *Chaoborus flavicans* larvae, an important zooplankton predator, show more body rotations, interpreted as an antipredatory behavior, when exposed to boat noise for the first time (Rojas et al. 2021). These works highlight that noise could affect the fitness and behavior of zooplankton species. However, long-term exposure has not been investigated, yet except *D. magna* exposed to chronic broadband noise that showed a higher fitness and a lower speed (Prosnier et al. 2022b).

A particular acoustic parameter of noise is the temporal pattern that it can exhibit (Francis and Barber 2013) with chronic exposure at the one hand and acute exposure at the other (Duarte et al. 2021). Chronic exposure means a continuous or intermittent, regular or random sound (e.g., turbine or boat noise) whereas acute exposure represents a punctual sound (e.g., air gun) (Nichols et al. 2015; McCauley et al. 2017). The spectral (i.e., sound level of each frequency) and temporal patterns of noise are known to affect the behavior and physiology of organisms in different ways. For instance, in underwater chronic noise studies, fish are more affected by a random noise than by a continuous or regular one (Nichols et al. 2015). These results are interpreted as an ability for vertebrates to habituate to some predictable long-term noise exposition (Rojas et al. 2021). Consequently, it asks if some results with unrealistic noise, as continuous broadband noise (Prosnier et al. 2022b), could be extrapolated to real situation where organisms are exposed to boat noise, thus exposed to intermittent random noise (i.e., unpredictable) with high spectral variability.

The aim of this study was to investigate the effect of chronic exposure to motorboat noise (intermittent and random noise with spectral variability) on the fitness of the water flea *Daphnia magna*, a common zooplankton species widely used in ecotoxicology (Ebert 2022). Previous studies found no change in their mobility when exposed to acute noise (Sabet et al. 2015, 2019), whereas prior experiment with chronic exposure to broadband noise (i.e., a continuous noise) showed alterations in both fitness and behavior (Prosnier et al. 2022b). Consequently, motorboat noise should negatively affect *D. magna* fitness, by reducing survival, fecundity, or both, according to McCauley et al. (2017) and Solé et al. (2023). However, note that in numerous cases no effect was observed (Solé et al. 2023), and a positive effect was found for *D. magna* chronically exposed to broadband noise (Prosnier et al. 2022b).

## Material and Methods

### Collection and Maintenance of Organisms

*Daphnia magna* had been purchased from Aqualiment (Grand Est, France) and stored in a 20-L rearing aquarium, filled with aged tap water (physiochemical composition is available on Zenodo repository (Prosnier et al. 2022a)), for 1 month. They were reared at 18 °C under a 12:12 light:dark cycle. *Daphnia magna* were fed, every 2 days, with 0.05 g of algae (i.e., 736 mJ) with a mix of 80% of *Arthrospira platensis* and 20% of *Aphanizomenon flos-aquae* (Algo'nergy® Spiruline + Klamath).

### Fecundity and Mortality

Reproductive success and survival were measured during an experiment (similar as done in Prosnier et al. 2022b). Gravid *D. magna* were collected from the rearing aquarium and isolated in 50-mL, 0.3-mm jars containing Volvic® water. Newborns (< 24 h) were transferred individually into 150 mL (5.6 × 8.6 cm) glass microcosm, closed with a 0.3 mm mesh tissue allowing water flows and noise transmission (Fig. 1).

Eighteen glass microcosms were disposed at 20 cm of an underwater loudspeaker in four 150 L rectangular tanks (75 × 60 × 35 cm), filled with 90 L of aged tap water, at 20–22 °C and under a 12:12 light:dark cycle. Silence was broadcasted in the two control tanks and a daily boat noise playlist (see below) used as treatment was broadcasted in the two other tanks. For each *D. magna* mother, half of the newborns were exposed to the control treatment, and the other half to the noise treatment, therefore individuals were clones in the two acoustic conditions. Every day, survival and newborns production were controlled – if *D. magna* spawned, offspring were counted and taken off. Every 2 days, individuals were fed with 2 mL of algae (1 g/L), and water was changed once a week. During the first 8 days of the experiment (i.e., before the first hatching), dead *D. magna* were replaced by new newborns (isolated mothers were maintained in 50 mL jars during this initial period to be able to initiate new replicates with newborns) to increase the number of replicates. Experiments were performed with a total of 115 juveniles (58 in control and 57 exposed to noise) coming from 25 mothers; almost half of the juveniles in each condition (26 in control and 23 in noise) reached maturity. The experiment lasted 46 days, from the birth of the first individual to the death of the last one (the oldest *D. magna* survived 39 days).

Based on daily survival and daily clutch, populational data was analyzed using the Euler-Lotka equation ($\sum f_x m_x e^{-rx} = 1$), with $f_x$ the fecundity at age $x$, $m_x$ the survival at age $x$, and $r$ the intrinsic rate of increase. This equation allows to calculate the reproductive output $R_0$ ($R_0 = \sum f_x m_x$), the generation time $G_t$ $\left(G_t = \frac{\sum x f_x m_x}{\sum f_x m_x}\right)$, and the intrinsic rate of increase $r$ $\left(r = \frac{\log R_0}{G_T}\right)$ (Leung et al. 2007; Starke et al. 2021).

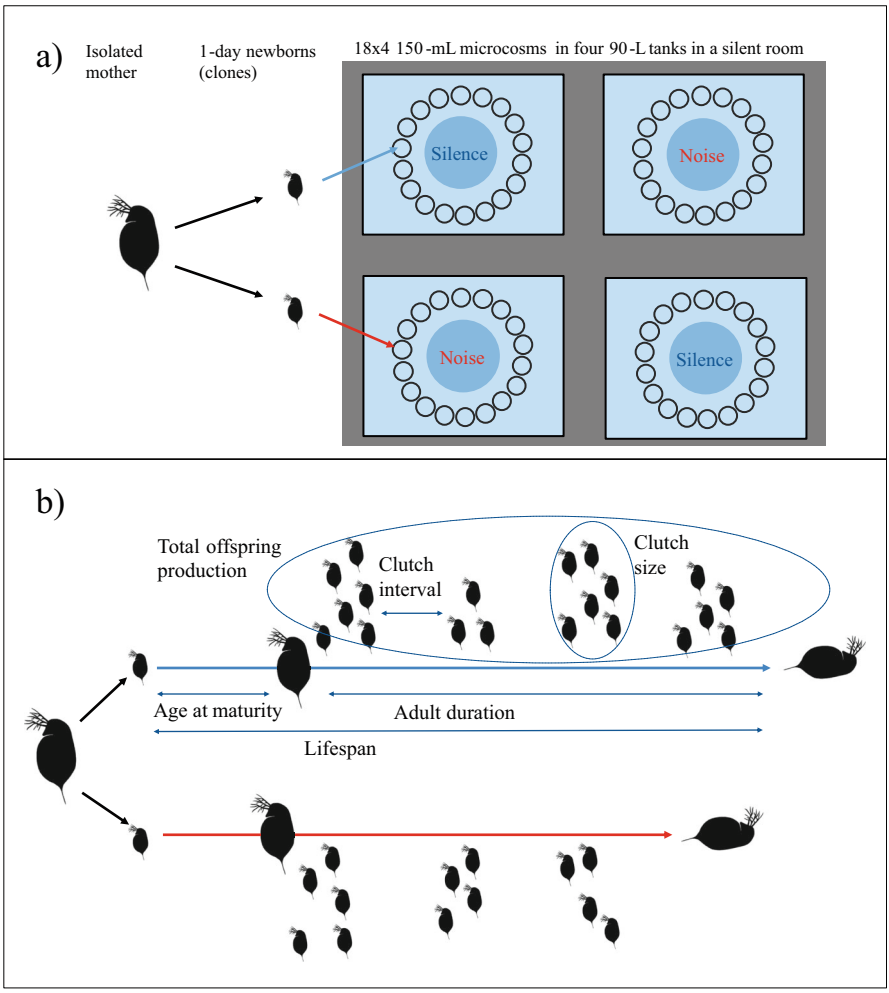



**Fig. 1** Setup. (**a**) Experimental design. In the four tanks blue circles are loudspeakers, dark small circles are microcosms closed with net. (**b**) Summary of all measures from birth to death of clonal individuals in the two treatments

## Acoustic Treatments

*Daphnia magna* were exposed to two acoustic treatments (see Rojas et al. 2021 for more details): a repeated 1 h playlist without sound for the control (i.e., only the ambient noise), or boat noise for the treatment with a playlist including 15 recordings of motorboat sounds previously made in the Grangent lake (45°45′07.54″N, 4°25′56.47″E, Loire, France). Their intensity was modulated from 0 to −25 dB Re 1 μPa by 5 dB to create 75 sounds from 103–150 dB RMS Re 1 μPa – a naturally occurring range of noise levels found in lakes (V. Médoc, pers. obs.) – that were

duplicated to obtain the total of 150 boat noises used in the experiment. Boat noise playlist was broadcasted from 9 a.m. to 6 p.m. (Fig. 2a). Both playlists (stereo WAV files) were generated using the software Adobe Audition 2020 (13.0.0.519, Adobe Systems Inc., Mountain View, CA, USA) and were played back using a Zoom® H4n recorder connected to an amplifier (DynaVox® CS-PA 1MK), and an underwater loudspeaker UW30 (Electro Voice®).

To check spectra and noise levels in both control and noise microcosms, recordings were made (Fig. 2b-e) with a Zoom® H4n coupled to a hydrophone (Aquarian Audio H2A-HLR Hydrophone, frequency response from 10–100 kHz) previously calibrated with a hydrophone (8104, Brüel & Kjær, Naerum, Denmark; sensitivity

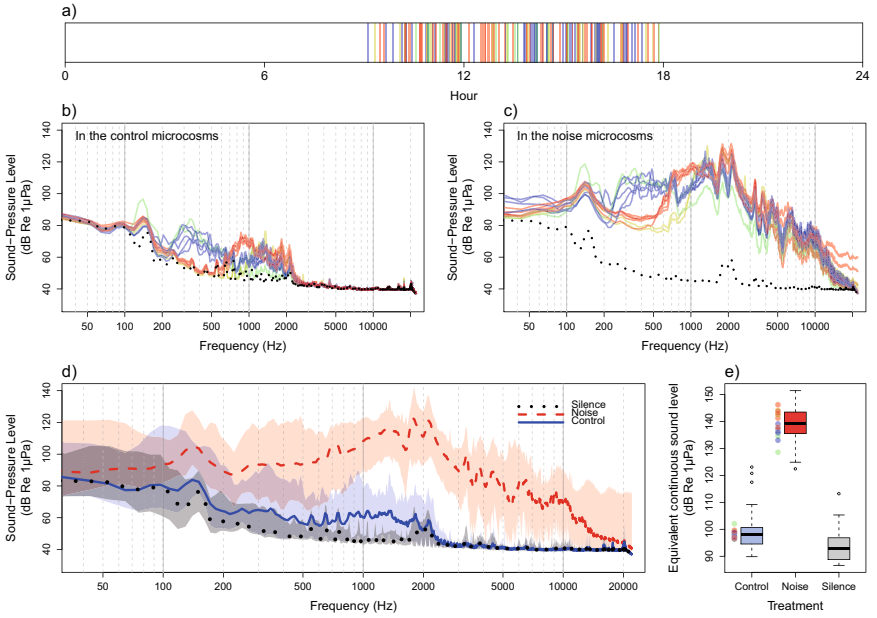



**Fig. 2** Acoustic treatments. (**a**) 24 h temporal sequence of the broadcasted motorboat noises, from 9 a.m. to 6 p.m. Each vertical line represents a boat. (**b**) Soundscapes of the control microcosms recorded in half microcosms with the records of the 15 boat sounds (solid lines), broadcasted at their maximal intensity in the noise tanks, and silence (dotted line). (**c**) Soundscapes in the noise microcosms recorded in half of the microcosms during the broadcast of the15 boat sounds at their maximal intensity (solid lines) and silence (dotted line). (**d**) Spectra in half of the microcosms. Thick lines are means for control (full blue line) and noise treatment (dashed red line) during the 15 boat broadcasts at their maximal intensity, and during the silence period (dotted black line). Shaded areas delimit the min and max sound pressure level. (**e**) Sound levels in half of the microcosms. Central bars represent the median, boxes the interquartile range, and dots the outliers (> 1.5 times the interquartile range). Colored dots are the sound levels in control and noise microcosms during the 15 boat broadcasts at their maximal intensity. The four colors (red, yellow, green, and blue) correspond to four noise structure spectra that were visually determined (for instance red and yellow have low energy between 200 and 700 or 1000 Hz compared to green and blue boat noises)

−205 dB re. 1 V µPa − 1; frequency response from 0.1 Hz to 180 kHz) connected to a sound level meter (Bruël & Kjaer 2238 Mediator, Naerum, Denmark). Emitted boat noises were firstly corrected to make their spectra closer to those of the initial 15 boat noise spectrums (Fig. 2). Boat noises were rerecorded only in half of the microcosms (in each tank) given that they were qualitatively and quantitatively similar due to the symmetry of the setup and after controlling with a broadband noise (Prosnier et al. 2022b). Note that the playback of boat noise was perceived in the control microcosms with intensities around 100 dB RMS Re 1 µPa, which was almost comparable to the sound level during the silence period (Fig. 2e), allowing to neglect noise transmission between the treatments. Particle motion cannot be measured due to the absence of adequate equipment, despite its importance for non-hearing species (Nedelec et al. 2016). However, Olivier et al. (2023) showed that results can still be qualitatively relevant when based solely on sound pressure level.

## Statistical Analyses

Statistical analyses were performed using R (version 4.2.2) with a significant threshold at 5%. Data allowed to analyze separately the effects on mortality (death age and adult survival) and fecundity (age at maturity, clutch frequency, mean clutch size, and daily clutch size). The combination of mortality and fecundity was used as a proxy of fitness and quantified through total offspring production. Data was also described at the population scale using the Euler-Lotka equation (but without statistical analysis due to absence of populational replicates). A survival analysis (log-rank test) was performed to compare survival (death age and adult duration) and age at maturity (first clutch age) between the control and noise treatments. For the fecundity parameters, only individuals that clutched at least one time (i.e., that reached maturity) were considered in the analyses. Clutch frequency (i.e., mean time between two clutches) and mean clutch size were analyzed using linear mixed-effects model, with tanks as random effect, thanks to the normal distribution of the data checked with a Shapiro test. The effect of both noise and age on daily clutch size was analyzed by a type II analysis of variance, completed with a pairwise Wilcoxon test between the treatments within each age. To test noise effect on the total number of clutches and offspring along life a generalized linear mixed-effects model was used, with tanks as random effect and a log function as the link function for Poisson distribution.

## Results

Chronic boat noise did not affect (Table 1) the survival of *D. magna* ($p$-value = 0.51, Fig. 3a) with a median survival of 4 days for the control and 5 days for the noise treatment. It did not affect fecundity parameters, with clutch interval around 2.5 days ($p$-value = 0.24, Fig. 3b), mean clutch size around 10 offspring ($p$-value = 0.74, Fig. 3c), and age at maturity around 8 days ($p$-value = 0.65). Daily clutch size was not influenced by noise (noise: $p$-value = 0.89, noise x age: $p$-value = 0.35,

**Table 1** Statistical results of chronic boat noises effects on fecundity and mortality of *Daphnia magna* (Fig. 3)

| | Mortality | | Reproduction | | | Fitness | |
|---|---|---|---|---|---|---|---|
| | Survival | Adult duration | Age at maturity | Clutch interval | Mean clutch size | Number of clutches | Number of offspring |
| df | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| $\chi^2$/W | 0.4 | 1.6 | 0.2 | 1.4029 | −0.54276 | 3.8197 | 1.8263 |
| *p*-value | 0.51 | 0.21 | 0.65 | 0.236 | 0.737 | **0.031** | 0.099 |
| $R^2$ | 0.0041 | 0.03 | 0.0046 | 0.03 | 0.007 | 0.096 | 0.327 |

pairwise: *p*-values $> 0.38$; Fig. 4), but changed with age (*p*-value $= 0.003$) with larger clutches at intermediate ages. However, taking into account the whole lifespan, there was an effect on the total number of clutches (*p*-value $= 0.003$) with five clutches for the control and six clutches for the noise treatment, and a tendency for higher total offspring production under boat noise exposure (*p*-value $=$ 0.099, Fig. 3d, e), with on average 60–70 newborns.

The populational analysis done with the Lotka-Euler equation confirmed the tendancy on total offspring production with a reproductive output ($R_0$) higher for the noise treatment with 63 offspring compared to the 54 offspring in control. Generation time ($G_T$) was longer in the noise treatment, with 16.6 days, compared to the 15.2 days in the control. The combination of both led to an intrinsic rate of increase of 0.25 $day^{-1}$ in the noise treatment compared to 0.26 $day^{-1}$ in the control.

## Discussion

This study investigated the effect of exposure to chronic boat noise on the fitness of the water flea *Daphnia magna*. Contrary to what expected, no effect on the survival and fecundity parameters was observed, which differed from results obtained with exposure to another type of chronic noise (Prosnier et al. 2022b). Such difference in the results might result from variations in the temporal and spectral features of the noise. Moreover, as one of the first experiments on chronic noise, it raises questions on the interactions with other pollutants, and on the effects on complex communities.

Chronic boat noise had no effect on *Daphnia magna*'s fecundity and survival. These results are opposite to those on acute exposition of copepods to air gun: After passing a boat with air gun, McCauley et al. (2017) observed a high mortality for numerous marine zooplankton species, as copepods; and Fields et al. (2019) showed that air gun exposure leads to an increased mortality in *Calanus finmarchicus* within an hour. In addition, results are opposite to those of Prosnier et al. (2022b), where an exposition to a continuous broadband noise leads to a surprising increase of *D. magna*'s fitness due to higher survival and clutch size. Here, although boat noise had no statistical effect on the survival and fecundity parameters, there was a tendency for a higher total offspring production with noise, which would be consistent with the results of Prosnier et al. (2022b). However, at a population scale, the

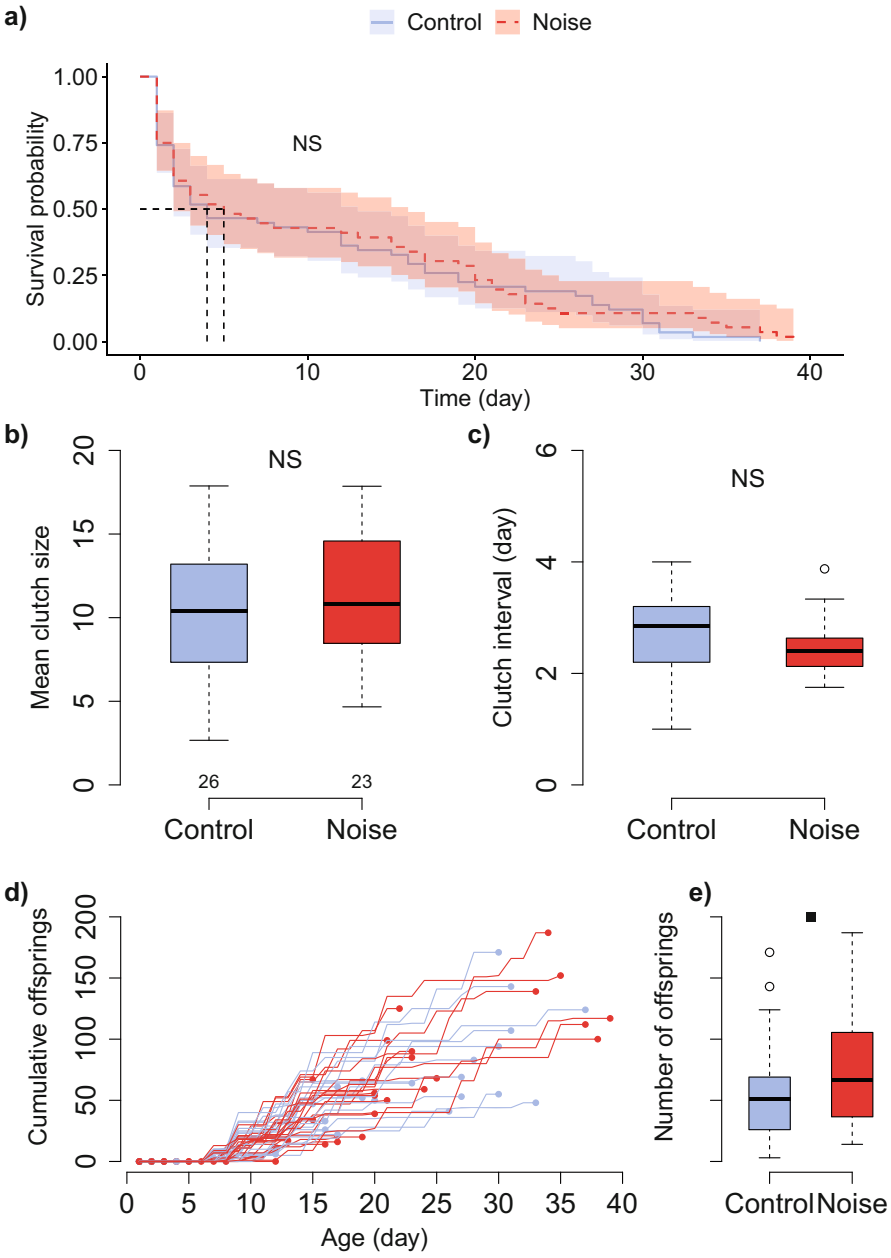



**Fig. 3** Effects of noise treatments on *Daphnia magna* survival and fecundity. (**a**) Survival of *D. magna*; (**b**) mean clutch size; (**c**) clutch frequency; (**d**) cumulative offspring production along life; and (**e**) total number of offspring during lifetime. Numbers in **b**) are the numbers of *D. magna* for the two treatments for **b**, **c**, and **e**. **a**) Representation according to the Kaplan-Meier method; **b-e**) central bars represent the median, boxes the interquartile range, and dots the outliers (> 1.5 times the interquartile range). Statistical analysis: dot *p-value* < 0.1, * *p-value* < 0.05; ** *p-value* < 0.01; NS *p-value* > 0.1. See Table 1 for statistical values

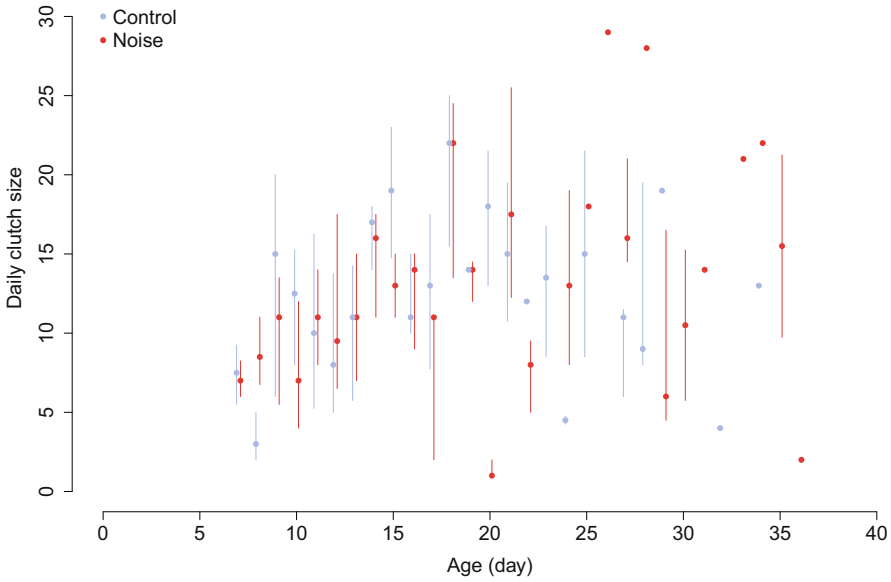



**Fig. 4** Effects of noise treatments on *Daphnia magna* daily clutch size. Dots represent the median of the daily clutch size and lines are interquartile ranges. See Fig. 3a, d as complement for the number of individuals at each age. Note that there could be only one clutch for an age (i.e., no interquartile lines) or clutches only for one treatment (i.e., only one point for an age)

Euler-Lotka equation suggests a tendency for a lower growth when exposed to boat noise due to a shorter generational time. These differences, or at least these lower intensities of effects, suggested that the effect of chronic noise pollution, on zooplankton species, could depend on the temporal and spectral structure of noise. Indeed, in Prosnier et al. (2022b), the broadband noise broadcasted was continuous, at high level (130 dB RMS re 1 µPa) for all frequencies (from 0.1–20 kHz), whereas, in this study, irregular sounds (a total of 2 h of noise per day) were used at various intensity levels (from 108–136 dB RMS re 1 µPa) and various frequencies (some boat noises with low intensity between 200 and 800 Hz). It is already known that in hearing vertebrates (i.e., with dedicated organs to detect as inner ear sound pressure variation) animals respond differently to chronic noise pollution if there are, or not, temporal variations (continuous, regular, and irregular) or spectral variations (i.e., variation in the frequencies), thus predictable or not (Francis and Barber 2013). Nichols et al. (2015) showed that fish were more stressed (higher cortisol concentration) with a higher noise level (from 126–141 dB RMS re 1 µPa), and with an intermittent random noise (i.e., unpredictable) compared to a continuous noise. However, the review of de Jong et al. (2020) on noise effects on fecundity revealed that continuous noise with spectral variations, such as boat noise, was more prone to impact physiological markers (cortisol level, ventilation rates, and metabolic rate) and behavior (startle and freeze responses, and horizontal and vertical avoidances). Another study on zebrafish larvae showed a higher negative effect of a continuous

white noise on survival and a higher cortisol concentration (Lara and Vasconcelos 2021). Thus, it seems that for hearing vertebrates, depending on the developmental stage and the considered characteristic, both temporal and spectral characteristics need to be considered. For zooplankton, little is known about the importance of noise patterns. Here, the comparison with Lara and Vasconcelos (2021) and Prosnier et al. (2022b) suggests that, contrary to what is reported with fishes, negative effects seem higher with a continuous noise.

Despite there was no effect on fecundity, it would be necessary to focus on the offspring coming from mothers exposed to noise. Here, there was no qualitative effect reported during the current experiments, i.e., all the offspring produced seemed viable and mobile, and Prosnier et al. (2022b) observed no effect on size. This would be consistent with the study of Day et al. (2016) where the exposure to air gun did not affect the embryonic development of the spiny lobster *Jasus edwardsii* (Decapodae). However, air gun exposition reduces growth and development stage of *Acartia tonsa* nauplii (▶ Chap. 96, "Effects of Anthropogenic Noise on Marine Zooplankton"). This aspect seems important as it is known that mother and early stress could affect daphnia's development (Mittmann et al. 2014; Mushegian et al. 2016) and that stress effects could differ across generations (Campos et al. 2016). Consequently, impact studies on noise should focus on embryonic development and perform multigenerational experiments to determine long-term effects of chronic noise, possibly due to embryonic misdevelopment (Mushegian et al. 2016), maternal effects (Radersma et al. 2018), and acclimatation or adaptation (Ringot et al. 2018; Abdullahi et al. 2022).

An interesting question is to consider the effect of noise as part of cocktails of pollutants. It is now a common question in ecotoxicology to ask how stressors (e.g., pollution, temperature, and food quality) act synergistically (Altshuler et al. 2011). For instance, Starke et al. (2021) showed that food quality impacts more *D. pulicaria* at some higher temperature due to the increased metabolism. Prosnier et al. (2015) studied the antagonistic effect of copper and nutrient enrichment on the *Daphnia* – algae interaction with modeling methods. Regarding noise, Stenton et al. (2022) showed both synergetic and antagonistic effects of noise and cadmium exposure on Norway lobster larvae (*Nephrops norvegicus*), depending of the considered parameters (survival, development, oxidative indicators). Another, McMahon et al. (2017) studied interactive effects of both light and noise pollutions on a frog-parasite interaction. Moreover, it could be also useful to better understand experimental results. Many controlled setups (light, temperature, and nutrient controlled) produce uncontrolled noises (L. Prosnier, pers. obs.), but their potential interactions with the stressors studied are unknown. For instance, here, there was a very high mortality in juvenile compared to similar studies (Parisot et al. 2015; Prosnier et al. 2023b) that suggests suboptimal condition for *D. magna* (i.e., other stressor than the noise) that potentially affects outcomes through the masking of effects, for example. However, on the other side, suboptimal conditions could make individuals more prone to be affected by an additional acoustic stress. Note that, despite the same suboptimal conditions, Prosnier et al. (2022b) obtained a difference between the control and noise treatments. The recent *Larvosonic* system, developed by Olivier et al. (2023),

could be useful to study the impact of noise on zooplankton with a better control of the environmental conditions.

This study is a first step in our understanding of the importance of noise patterns for invertebrates (i.e., predictable versus unpredictable noise) in comparison with vertebrates. It is mandatory to better understand how noises could affect complex communities with both vertebrates and invertebrates (Francis et al. 2009; Slabbekoorn and Halfwerk 2009; Slabbekoorn 2019). For instance, Rojas et al. (2023) experimented boat noise effects on a freshwater community. In this experiment, (1) unpredictable noise should affect more fishes, at top trophic levels, than invertebrates and (2) bottom-up effects could also be expected due to various reactions of zooplanktonic species. About effects on invertebrates, the diversity of responses by various zooplanktonic species is still lacking (► Chap. 98, "Response of Freshwater Zooplankton Communities to Chronic Anthropogenic Noise") – for instance if zooplanktonic predators are affected in the same way than their zooplanktonic prey. Moreover, indirect effect could be expected in a community (Read et al. 2014). For instance, noise do not affect frog abundance but reduce their parasite's one (McMahon et al. 2017). This interest for invertebrates, in particular arthropods, raises also for terrestrial communities (Morley et al. 2014). Thus, a more general overview of the impacts of anthropogenic noises on all invertebrates should be beneficial to prevent them (Francis and Barber 2013).

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# Response of Freshwater Zooplankton Communities to Chronic Anthropogenic Noise

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**Abstract**

Man-made sounds are now recognized as a pervasive pollutant, and impacts on wildlife have been researched for many years. However, less knowledge is available on certain species, particularly small freshwater invertebrates, which are abundant, highly diversified, and occupy key positions in food webs. Also, it is not clear whether the responses to noise observed at the level of individuals have consequences on communities. A mesocosm investigation was performed to assess the response of a freshwater planktonic community to chronic motorboat noise. Noise was expected to disturb trophic links within the community and particularly the consumption of cladocerans by dipteran larvae. To test this hypothesis, the functional response of *Chaoborus* larvae feeding on *Daphnia* was derived, and their behavior during the foraging process was recorded in microcosms (aquariums). Although noise did not induce obvious alteration in the community composition, a significant increase in the abundance of cladocerans was found that was not supported by the results of the microcosm investigation, showing no difference in *Chaoborus* functional response or behavior between the noisy and noiseless conditions. The results of this chapter suggest that the composition of freshwater zooplankton and particularly cladocerans is likely to be altered by chronic noise, with further investigations needed to understand the mechanisms. They also illustrate how scaling up the effects of noise from individual responses to community remains difficult.

**Keywords**

Freshwater zooplankton · Motorboat noise · Functional response · Trophic links · Daphnids · *Chaoborus* larvae

## Introduction

Threats to freshwaters include habitat degradation, flow modification, over-exploitation, invasive species, and disease (Dudgeon et al. 2006; Williams-Subiza and Epele 2021), and result in a decline in biodiversity at rates that exceed what is reported in most terrestrial and marine habitats (McRae et al. 2017). Anthropic pressures on freshwaters are not expected to ease given the growing human needs and also because people seek to reconnect with nature, a need reinforced by the recent crises like the Covid-19 pandemic. Managers of freshwater socio-ecological systems worry about the rise of recreational motorized activities and their associated noise emissions (Reid et al. 2019) that can disturb the various populations of users as well as wildlife.

Noise pollution has recently been categorized as an emergent threat to freshwaters (Reid et al. 2019), with motorized boats as the most widespread source of noise. Impacts of noise on fishes are well documented with physiological stress responses and alterations in communication, reproduction, mobility, foraging, and predator

avoidance (reviewed by Mickle and Higgs 2018; Popper 2003; Slabbekoorn et al. 2010). Although invertebrates are highly diverse, widespread, and possess statocysts or external sensory hairs that allow them to perceive sounds through particle motion (Popper and Hawkins 2018), interest in their response to noise pollution came later compared to vertebrate species, and 77% of the impact studies on invertebrates are less than 10 years (Wale et al. 2021). While cephalopods, large crustaceans (crabs, lobsters, and shrimps), and bivalves are among the most common model species studied (Fernández Robledo et al. 2019; Smith et al. 2018), little is known about small zooplankton despite its pivotal role in the functioning of aquatic food webs, maintaining energy flow between primary producers and higher trophic levels (Turner 2004; Vargas et al. 2010).

Available evidence on zooplankton shows a diversity of effects. While very loud emissions from seismic surveys have been found to cause mortality in both larval and adult stages of marine zooplankton (McCauley et al. 2017), vessel noise can act as a positive cue for larval settlement in the blue mussel *Mytilus edulis* (Jolivet et al. 2016). Exposure to low (30 Hz) and high (20 KHz) frequencies seems to promote grazing in the marine copepod *Acartia tonsa* (Yiwei and Berggren 2018). The water flea *Daphnia magna* (Cladocera) shows no alteration in mobility when exposed to either continuous or intermittent 300–1500 Hz band-pass filtered white noise (Sabet et al. 2016). More recently, it was found that larvae of the phantom midge *Chaoborus* (Diptera) made more body rotations in response to motorboat noise (Rojas et al. 2021).

In addition to the imbalance between vertebrates and invertebrates in the very rich literature on the impacts of noise pollution, there is also a discrepancy between the biological integration levels with a lack of research on ecosystems compared to behavioral and physiological outcomes (Sordello et al. 2020). Although a few empirical evidences from terrestrial systems illustrate how noise-induced changes in behavior can propagate through nested ecological interactions (Francis and Barber 2013; Phillips et al. 2021), scaling up the effects of noise from individuals to populations and communities without any experimental validation might overestimate impacts.

In this chapter, the effect of chronic motorboat noise on the dynamics of a freshwater zooplankton community was investigated, bringing together cladocerans, copepods, ostracods, and dipterans. Noise was expected to alter the structure of the community through changes in abundance and/or changes in the activity of the predators. To test this hypothesis, the response of the community was evaluated in mesocosms over 6 weeks, and, as a second part, it was assessed how the *per capita* predation rate of *Chaoborus* larvae (dipterans) varied with cladoceran density (the functional response) under control and noisy conditions. *Chaoborus* larvae are a relevant dominant predator of large filter-feeder zooplankton (cladoceran species) known to be a main structuring force within the community (Castilho-Noll and Arcifa 2007; Vanni and Findlay 1990). *Chaoborus* larvae have been found to make more body rotations in response to motorboat noise (Rojas et al. 2021), which could be associated with reduced foraging. Noise might therefore alter community dynamics through the modulation of the trophic pressure by *Chaoborus* larvae.

## Materials and Methods

### Mesocosm Experimental Design

The mesocosm experiment lasted 6 weeks from September to October 2021 and was carried out on the PLANAQUA platform of the CEREEP-Ecotron Ile-de-France research station (48° 16′10.92 N, 2° 43′50.879 E, Seine et Marne, France). Two acoustic conditions (with or without boat noise, see section "Playback Tracks") were applied in 16 outdoor plastic enclosures (diameter: 1.40 m, depth: 1 m, volume: 1 $m^3$, n = 8 replicates *per* condition) positioned in two lines and distributed in a systematic way to balance the effect of spatial distribution between the two conditions. All mesocosms included a 15-cm layer of Loire sand and were filled 2 months before the experiment with water from the littoral zone of one of the two storage lakes from the PLANAQUA platform, to reach a 70-cm water column. An underwater loudspeaker (Electrovoice UW30, 0.1–10 kHz) was fixed 10 cm below the water surface in the middle of each mesocosm. It was connected to an amplifier (Dynavox CS-PA 1MK), itself connected to an audio player (Handy's H4n zoom), both placed inside a waterproof electric box next to the mesocosm. One week before starting the experiment, temperature loggers attached to a ballast were positioned in the sunniest part of each mesocosm. The water temperature was 24 °C at the beginning of the experiment and decreased with some small fluctuations over time to reach 18 °C at the end of the experiment.

### Zooplankton Dynamics

At day 0, +10, +26, and +42, 8 L of water were sampled with a 2-L sampling bottle at four different positions and depths in each mesocosm. Water was filtered with a 50-µm mesh size nylon filter to collect zooplankton species, which were immediately fixed in 15 mL of 90% ethanol. Species identification and classification on day 0 and day +42 were performed by the engineering office © 2021 SAGE Environment (Annecy, France). To save costs, accurate classification was done for all the mesocosms at day +42, while the data *per* noise condition at day 0 was pooled. At day +10 and day +26, only the numbers of cladocerans and *Chaoborus flavicans* larvae were quantified, as they were the most structuring trophic links expected in the communities. At the end of the experiment, a multiparameter probe (YSI ExO-2) was used to assess the main physicochemical parameters (temperature, pH, conductivity, turbidity, and chlorophyll).

### Functional Response and Behavior of *Chaoborus* Larvae

The functional response (FR) of *Chaoborus* larvae feeding on five densities (3, 6, 12, 24, and 48) of *Daphnia* sp. coming from the storage lake has been derived with four replicates *per* density and *per* noise condition. To account for potential

habituation to the noise condition and better explain what happened in the mesocosms, larvae from the control and noisy mesocosms were collected and exposed to the same noise condition (see section "Playback Tracks") during the FR tests. The experiment took place in two 90-L rectangular tank (75 × 60 × 20 cm, one *per* noise condition) filled with filtered (50-μm mesh size) water from the storage lake. A UW30 underwater loudspeaker was positioned in the center of each tank 20 cm above the bottom. A single larva was presented to the water fleas at one of the five densities (3–48) for 8 h in a 150-mL glass beaker (height = 7.2 cm, diameter = 7 cm) covered with a 0.03 mm mesh allowing water flow only. A total of 20 beakers *per* mesocosm were used and placed at 10 cm all around the loudspeaker. At the end of the experiment, each larva was removed and put into 90 °C alcohol to be measured under a binocular loop and using a rule. The number of remaining prey was counted to determine the number of prey eaten.

The behavior of the *Chaoborus* larvae was studied in a 50-L aquarium (length×width×height: 60 × 25 × 35 cm) filled with filtered water from the mesocosms and equipped with an UW30 underwater loudspeaker in the center and 20 cm above the bottom. A 150-mL glass beaker containing 20 larvae and covered with a mesh allowing water flow only was positioned inside the aquarium at 10 cm of the loudspeaker. The number of body rotations performed by each larva was counted over a 20-min period of ambient noise (recorded in one of the mesocosm) or ambient noise supplemented with motorboat noise (see section "Playback Tracks").

## Playback Tracks

An Aquarian Audio H2A-XLR hydrophone (frequency response from 10 Hz to 100 kHz) connected to a ZOOM H4next Handy recorder was used for all the recordings and a UW30 underwater loudspeaker (Electrovoice) connected to a Dynavox CS-PA 1MK amplifier itself connected to a ZOOM H4next Handy player for all the playbacks.

Natural background noise did not differ between the mesocosms and was around 90 dB re 1 μPa. In the control mesocosms, a 1-h audio track of silence was looped continuously. To make the audio tracks of the noisy mesocosms, 25 sounds from commercial vessels and recreational boats were recorded from the river Seine after the lock of Champagne sur Seine (48°22′1.348 N, 2°29′37.401 E) at 1 m depth. The 25 original sounds were duplicated, changing a bit the intensity between the two replicates, and the resulting 50 sounds were distributed over 14 consecutive 1-h audio tracks of silence so as to mimic the mean daily activity of the Champagne sur Seine lock (Table 1). The boat sound audio tracks were broadcasted from 6 a.m. to 8 p.m. and silence the rest of the time. The intensity of each boat sound was modified with the Audacity 2.2.1 software to obtain realistic signal-to-noise ratios (SNR) ranging from 25 to 30 dB (Fig. 1a), calculated after re-recordings in the mesocosms and using the *SNR* function of *Seewave* R package (Sueur et al. 2008) with:

**Table 1** Composition of the 14-h playback track broadcasting silence supplemented with motorboat sounds during the mesocosm experiment

| Hour | Number of boat sounds | Boat ID | Duration in min | Start position in the 1-h track |
|---|---|---|---|---|
| 6 h | 3 | 17 | 07′00 | 5′00″ |
| | | 31 | 05′35 | 47′00″ |
| | | 45 | 06′10 | 51′00″ |
| 7 h | 1 | 29 | 06′00 | 23′50″ |
| 8 h | 2 | 44 | 04′20 | 13′50″ |
| | | 20 | 01′25 | 40′30″ |
| 9 h | 4 | 8 | 01′40 | 0′00″ |
| | | 46 | 07′15 | 14′30″ |
| | | 47 | 03′02 | 32′25″ |
| | | 2 | 04′37 | 36′15″ |
| 10 h | 6 | 11 | 06′05 | 2′55″ |
| | | 10 | 02′20 | 13′35″ |
| | | 42 | 02′30 | 16′25″ |
| | | 50 | 03′30 | 19′30″ |
| | | 41 | 02′00 | 18′40″ |
| | | 33 | 02′35 | 54′10″ |
| 11 h | 4 | 48 | 03′02 | 16′45″ |
| | | 42 | 02′30 | 33′15″ |
| | | 30 | 05′00 | 35′50″ |
| | | 30 | 05′00 | 53′05″ |
| 12 h | 2 | 2 | 04′37 | 8′25″ |
| | | 27 | 07′00 | 18′45″ |
| 13 h | 4 | 6 | 05′00 | 01′45″ |
| | | 38 | 04′00 | 17′40″ |
| | | 50 | 03′30 | 22′00″ |
| | | 46 | 07′15 | 40′50″ |
| 14 h | 6 | 40 | 04′15 | 01′45″ |
| | | 17 | 02′00 | 05′30″ |
| | | 24 | 03′02 | 07′45″ |
| | | 20 | 01′25 | 29′15″ |
| | | 25 | 06′30 | 35′10″ |
| | | 10 | 02′20 | 42′05″ |
| 15 h | 8 | 42 | 02′30 | 00′35″ |
| | | 15 | 01′30 | 07′00″ |
| | | 50 | 03′30 | 03′35″ |
| | | 2 | 04′37 | 14′21″ |
| | | 9 | 02′35 | 21′54″ |
| | | 12 | 08′00 | 26′45″ |
| | | 12 | 08′00 | 44′16″ |
| | | 18 | 05′05 | 54′55″ |
| 16 h | 4 | 20 | 01′25 | 00′00″ |
| | | 44 | 04′00 | 06′05″ |
| | | 23 | 07′15 | 20′52″ |
| | | 35 | 06′05 | 32′19″ |
| 17 h | 2 | 45 | 06′10 | 39′17″ |
| | | 46 | 07′15 | 46′22″ |
| 18 h | 1 | 13 | 03′30 | 27′07″ |

(continued)

**Table 1** (continued)

| Hour | Number of boat sounds | Boat ID | Duration in min | Start position in the 1-h track |
|---|---|---|---|---|
| 19 h | 3 | 31 | 05′35 | 24′03″ |
| | | 30 | 05′00 | 30′43″ |
| | | 42 | 02′30 | 49′29″ |

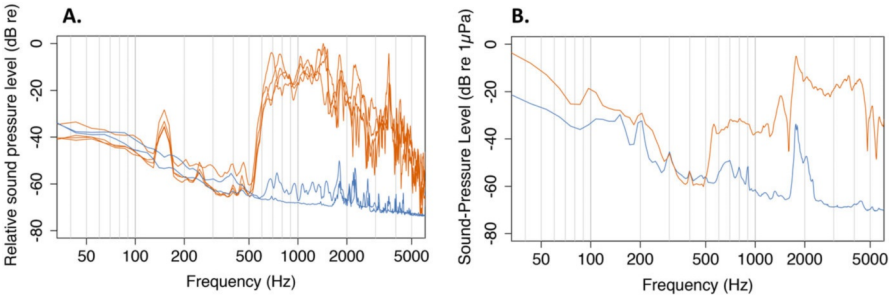



**Fig. 1** Sound spectra of the two noise conditions (ambient noise in blue and boat noise in red) in: (**a**) the mesocosms used for the community investigation (each orange line corresponds to a recording of the same boat noise made in four noisy mesocosms and each blue line corresponds to a recording of 3 min of ambient noise made in two control mesocosms) and (**b**) the aquariums (50-L rectangular tanks) used for functional response derivation

$$\mathrm{SNR} = 20 \log_{10}(\mathrm{RMS}_{\text{boat sound}}/\mathrm{RMS}_{\text{ambient noise}})$$

where RMS corresponds to the root-mean-square sound pressure level.

For the control condition of the FR experiment, an 8-h (playlist 1) audio track of natural background noise previously recorded in one of the mesocosm and whose level was adjusted to match that in the mesocosm around 90 dB re 1 µPa (Fig. 1a) was broadcasted. For the boat noise treatment, an 8-h recording (playlist 2) from a noisy mesocosm (Fig. 1a) was broadcasted using the same process than for control condition.

Concerning the behavior of the *Chaoborus* larvae, a 20-min period of playlist 1 was randomly selected to be used for the control condition and a 20-min sequence of playlist 2 corresponding to the period with the largest number of boat sounds was used for the noisy condition (Fig. 1b).

## Data Analyses

The R software version 4.0.3 (R Core Team 2018) was used for all the statistics with a significance level of 5%. A chi-square test of independence was performed to assess the homogeneity of taxa at day +0 between both noise treatments. A generalized linear mixed models with a quasi-Poisson distribution (GLMMTMB) was performed to explain the dynamic of cladocerans as a function of three fixed factors and their interactions: the noise condition (ambient or motorboat noise), the

abundance of *Chaoborus flavicans* larvae, the sampling date, and considering the tank identity as random factor to account for repetitive measures. A quasi-Poisson (or quasi-likelihood) distribution was used because it is recommended to consider the overdispersion (variance exceeding the mean) often found in count data (Ver Hoef and Boveng 2007). A Wilcoxon test was used to test for significance the difference in physiochemical parameters between the two noise conditions at day +42.

For the FR experiment, a one-way ANOVA test was performed to detect heterogeneity in the size of *Chaoborus* larvae between the two noise conditions as the data met the normality and homoscedasticity assumptions. FR analysis was done with the *Frair* R package (Pritchard et al. 2017). The three main categorical FR types (linear type I, Rogers' type II, and Hassel's type III) were modeled by maximum likelihood estimation (Bolker 2008) with the *frair_fit* function, and the fits were compared using the second-order Akaike information criterion (AIC). This allowed to exclude the types II and III whose AIC values were always the highest ($\Delta AIC > 2$ with the type I). A type I FR is characterized by a linear increase of consumption rate as a function of prey density (Holling 1959). Both FRs being of type I, the delta method implemented by the *frair_compare* function was used to perform pairwise FR comparison from parameter estimates with the null hypothesis that the difference in attack rates (D*a*) between the two FRs does not differ from zero (Pritchard et al. 2017). In addition, the overlaps of the 95% confidence intervals (BCa CIs) which correct for bias and skewness in the distribution of bootstrap estimates (*a* and *h* parameters) were inspected using the *frair_boots* function (bootstrapping method, $n = 2000$) (Pritchard et al. 2017).

Concerning the behavior of the *Chaoborus* larvae, the total number of body rotations was compared between the two noise conditions using a generalized linear mixed model with a negative binomial distribution with the noise condition (ambient or motorboat noise), the sampling date as predictors, and the tank where they came as a random factor to account for repetitive measures.

## Results

At day +0, communities between ambient noise and ambient + motorboat noise did not differ in taxa density (chi-square test = 63.333, df = 56, p-value = 0.2336). At day +42, no significant differences in the physicochemical parameters between the two noise conditions (Tables 2 and 3) were found.

The zooplankton communities of the noisy and noiseless mesocosms included cladocerans (*Daphnia* sp., *Bosmina* sp., *Chydorus* sp., and *Ceriodaphnia* sp.), copepods (especially Calanoïda and Cyclopoïda), ostracods, and dipterans (especially *Chaoborus* larvae). Cladocerans species and more particularly *Daphnia* sp. were the most abundant at the beginning and at the end of the experiment in the control mesocosms (57.31% and on average 52.11%, respectively), whereas in

**Table 2** Physicochemical parameters (mean ± SD) in the control (ambient noise) and noisy mesocosms (boat noise) at the end of the experiment, with the results of the Wilcoxon tests performed to test the difference between the two noise treatments

| Parameters | Ambient noise (mean ± SD) | Boat noise (mean ± SD) | W | p-value |
|---|---|---|---|---|
| **pH** | 7.60 ± 0.59 | 7.70 ± 0.37 | 266.5 | 0.665 |
| **Temperature °C** | 15.66 ± 0.08 | 15.66 ± 0.13 | 257.5 | 0.536 |
| **[Chlorophyll] µg/L** | 1.49 ± 0.87 | 1.71 ± 1.68 | 331 | 0.38 |
| **ODO mg/L** | 9.64 ± 0.32 | 9.62 ± 0.33 | 277.5 | 0.836 |
| **Conductivity uS/cm** | 165.32 ± 8.11 | 167.66 ± 5.74 | 212.5 | 0.121 |

the noisy mesocosms, copepods and cladocerans were more abundant at day +0 (48.25% and 39.84% respectively) but cladocerans increased to reach 71.24% of the whole community at the end of the experiment (Table 4, Fig. 2).

The abundance of cladocerans was significantly increased by noise (p-value = 0.044, Table 5, Fig. 3a) but was not influenced by the abundance of *Chaoborus*, the date, and the interactions between predictors. The abundance of *Chaoborus* larvae did not differ between the two noise conditions (GLMMs, Estimate = 0.026, Std. Error = 0.501, z-value = 0.050, p-value = 0.960, Fig. 3b).

Concerning the FR experiment, there was no difference in the size of the *Chaoborus* larvae between the two noise conditions (one-way ANOVA: $F_{1,18}$ = 0.269, p-value = 0.61). Irrespectively of the noise condition, the FR was a type I (linear increase of *per capita* consumption rate in function of prey density). No significant difference in attack rate (fixing the slope) was found between the two noise conditions (Estimate = 0.087, Std.Error = 0.108, z-value = 0.817, p-value = 0.413) supported by a strong overlap of the 95% BCa CIs, suggesting similar FRs (Fig. 4a).

No significant difference in the number of body rotations was found between the two noise conditions (noise: Estimates = 0.2957, Std.Error = 0.4268, z-value = 0.693, p-value = 0.488, Fig. 4b). However, interindividual variability in both FR and behavior was greater with boat noise than for controls.

## Discussion

In this chapter, a mesocosm investigation was conducted to assess the effect of chronic motorboat noise on the dynamics of freshwater zooplankton. Predation tests were also performed in microcosms through the functional response (FR) derivation to test the prediction that in case of a noise-induced alteration in community dynamics, this would be linked with a change in the foraging behavior of invertebrate predators, focusing on *Chaoborus flavicans* as the main predator within the zooplanktonic community.

No marked effect of chronic motorboat noise on the zooplankton community was found except for water fleas (*Daphnia* sp.), which represented the most abundant

**Table 3** Physicochemical parameters obtained at Day+0 and at Day+42 for all outdoor mesocosm

| Tank | Noise condition | Temperature (°C) | pH | Conductivity (μs/cm) | Green algae (μg/L) | Cyanobacteria (μg/L) | Diatoms (μg/L) | Chlorophyll concentration (μg/L) | Turbidity (FNU) |
|---|---|---|---|---|---|---|---|---|---|
| 1 | Ambient | 15.57 | 9.2 | 159.33 | 6.14 | 2.02 | 0.48 | 0.85 | 1.44 |
| 2 | Boat | 15.79 | 9.71 | 180.33 | 6.81 | 0 | 0.33 | 0.75 | 0.97 |
| 3 | Ambient | 15.68 | 9.76 | 172.9 | 7.55 | 0.92 | 0.3 | 0.88 | 1.15 |
| 4 | Boat | 15.73 | 9.55 | 165.63 | 7.58 | 1.65 | 0.87 | 0.45 | 1.1 |
| 5 | Ambient | 15.69 | 9.56 | 165.87 | 7.69 | 0.22 | 0.34 | 1.17 | 1.36 |
| 6 | Boat | 15.62 | 9.22 | 171.4 | 7.73 | 0.62 | 0.18 | 0.87 | 1.32 |
| 7 | Ambient | 15.71 | 9.37 | 181.07 | 7.76 | 0.25 | 0.24 | 1.05 | 1.18 |
| 8 | Boat | 15.59 | 9.96 | 166.23 | 7.95 | 1.08 | 0.43 | 0.96 | 0.99 |
| 9 | Boat | 15.39 | 9.86 | 161.8 | 7.91 | 6.21 | 0 | 5 | 6.04 |
| 10 | Ambient | 15.72 | 9.37 | 154.33 | 7.82 | 3.88 | 0.37 | 3.65 | 3.65 |
| 11 | Boat | 15.77 | 9.43 | 161.7 | 7.8 | 1.13 | 0.41 | 0.98 | 1.14 |
| 12 | Ambient | 15.73 | 9.92 | 161 | 7.92 | 0.76 | 0.16 | 1.36 | 1.26 |
| 13 | Boat | 15.65 | 9.32 | 167.5 | 7.9 | 0 | 0.21 | 0.79 | 0.86 |
| 14 | Ambient | 15.63 | 9.72 | 160.97 | 7.89 | 0.07 | 0.14 | 0.97 | 0.93 |
| 15 | Boat | 15.7 | 9.93 | 166.7 | 7.93 | 0.55 | 0.19 | 0.83 | 1.29 |
| 16 | Ambient | 15.53 | 10.21 | 167.17 | 8.05 | 0.26 | 0.07 | 0.66 | 0.95 |

**Table 4** Count of taxa at Day+0 and at Day+42 for all outdoor mesocosms (n = 16) in both noise conditions (ambient noise, n = 8; boat noise, n = 8)

| Order | taxa | Day+0 ambient noise (total, n = 8) | Day+0 boat noise (total, n = 8) | Day+42 ambient noise (total, n = 8) | Day+42 boat noise (total, n = 8) | Day+42 ambient noise (mean ± SD per mesocosm) | Day+42 boat noise (mean per mesocosm) |
|---|---|---|---|---|---|---|---|
| **Copepods** | **Cyclopoïds** | 79 | 73 | 177 | 131 | 22.13 ± 28.2 | 16.38 ± 14.86 |
| **Copepods** | **Calanoïds** | 158 | 162 | 162 | 183 | 20.25 ± 23.06 | 22.88 ± 24.47 |
| **Cladocerans** | ***Bosmina* sp.** | 0 | 0 | 25 | 0 | 3.13 ± 8.84 | 0 |
| **Cladocerans** | ***Ceriodaphnia* sp.** | 2 | 0 | 25 | 2 | 3.13 ± 8.84 | 0.25 ± 0.71 |
| **Cladocerans** | ***Chydorus* sp.** | 10 | 5 | 33 | 53 | 4.13 ± 6.33 | 6.63 ± 9.65 |
| **Cladocerans** | ***Daphnia* sp.** | 439 | 189 | 567 | 1226 | 70.88 ± 44.32 | 153.25 ± 113.55 |
| **Crustacean** | **Ostracods** | 76 | 25 | 74 | 106 | 9.25 ± 12.08 | 13.25 ± 9.84 |
| **Insect** | **Ephemeroptera** | 11 | 0 | 4 | 9 | 0.5 ± 1.07 | 1.13 ± 2.10 |
| **Insect** | ***Chaoborus flavicans*** | 7 | 16 | 11 | 7 | 1.38 ± 1.77 | 0.88 ± 0.83 |
| **Insect** | **Hydracarina** | 2 | 1 | 6 | 2 | 0.75 ± 0.89 | 0.25 ± 0.71 |

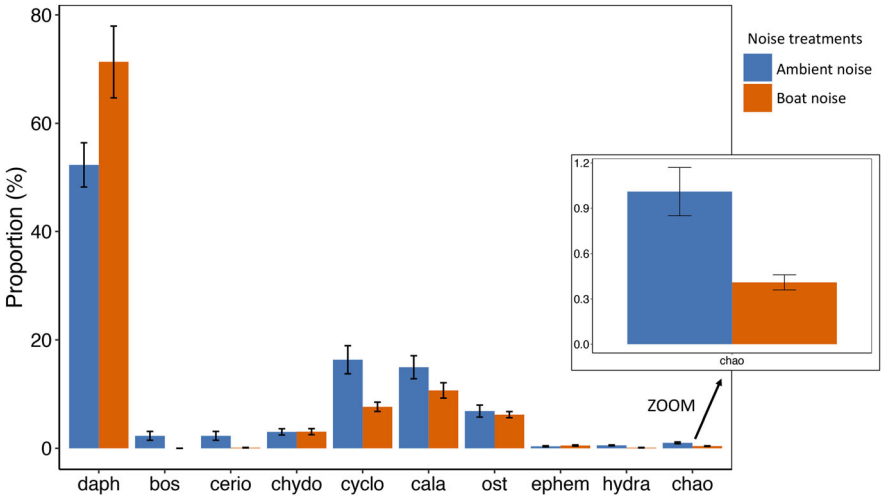



**Fig. 2** Frequency distribution (mean ± SD) of the zooplankton taxa founds in the control (blue) and noisy (orange) mesocosms at the end of the experiment with a focus on *Chaoborus flavicans* larvae, the main predator (daph: *Daphnia* sp., bos: *Bosmina* sp., cerio: *Ceriodaphnia* sp., chido: *Chidorus* sp., cyclo: Cyclopoids, cala: Calanoïds, ost: Ostracods, ephem: Ephemeroptera, hydra: Hydracarina and chao: *Chaoborus* larvae (Diptera)

**Table 5** Model-averaging coefficient estimates for the predictors included in the model used to explain the variation in cladoceran abundance. Predictors correspond to the noise condition broadcasted in mesocosm (silence or chronic motorboat noise), the presence of *Chaoborus* larvae, the date, and the interactions between all predictors. Significant p-values are in bold

| | Estimate | Std. error | Z value | P-value |
|---|---|---|---|---|
| Intercept | 4.35259 | 0.19096 | 22.793 | **<0.001** |
| Noise | 0.49052 | 0.24385 | 2.012 | **0.0443** |
| Chaoborus | −0.06774 | 0.15784 | −0.429 | 0.6678 |
| Date | 0.06991 | 0.13999 | 0.499 | 0.6175 |
| Noise*_Chaoborus_ larvae | 0.20685 | 0.19719 | 1.049 | 0.2942 |
| Noise*date | 0.19450 | 0.21705 | 0.896 | 0.3702 |
| _Chaoborus_ larvae*date | −0.18475 | 0.17620 | −1.049 | 0.2944 |
| Noise*_Chaoborus_ larvae*date | 0.41613 | 0.41402 | 1.005 | 0.3148 |

taxon and were significantly more numerous in the noisy mesocosms. This apparent positive effect could be indirect, considering that noise has no or a very limited direct negative effect on water fleas but negatively influences their natural enemies. Although no investigation on the response of water fleas to noise was made, the absence of direct effect is partially supported by the little literature available. Sabet et al. (2016) did not find any alteration in mobility in *Daphnia magna* exposed to either continuous or intermittent 300–1500 Hz band-pass filtered white noise, a result that was also obtained working on motorboat noise (Rojas et al. unpublished

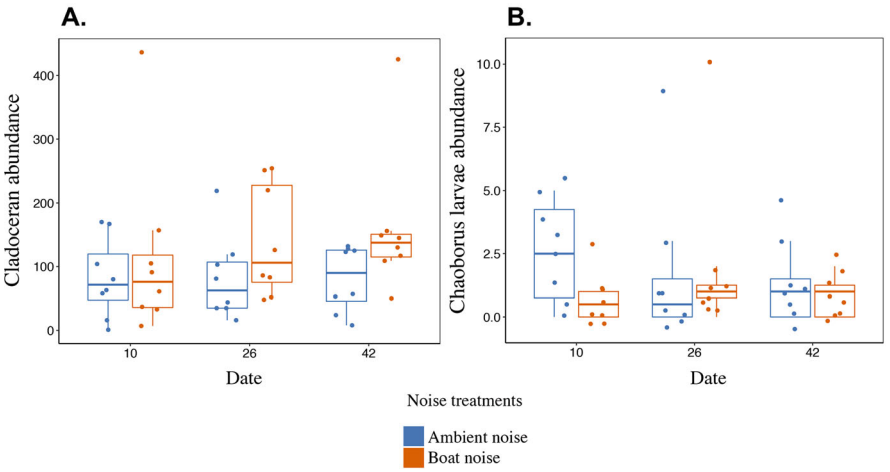



**Fig. 3** Variation in the abundances (median and interquartile range) of Cladocerans (**a**) and *Chaoborus* larvae (**b**) in the control (ambient noise, blue) and noisy mesocosms (ambient noise supplemented with boat noise, orange) at days 10, 26, and 42

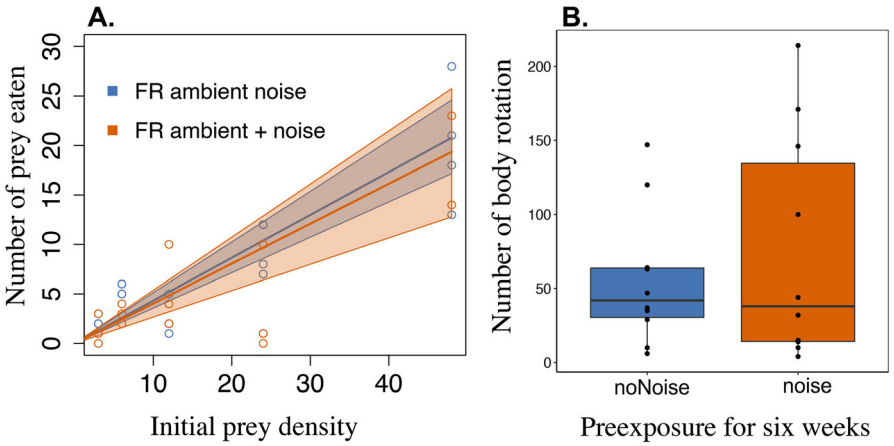



**Fig. 4** (**a**) Number of water fleas eaten by single *Chaoborus* larvae as a function of initial water flea density (functional response) under ambient noise (blue) or chronic motorboat noise (red). Dots indicate the raw data and shaded areas the confidence intervals. (**b**) Number of body rotations (median and interquartile ranges) made by the *Chaoborus* larvae exposed to recordings of ambient noise (blue) or chronic motorboat noise (red)

data). More recently, Yağcılar and Yardımcı (2021) found that exposure to 432 Hz and 440 Hz frequency sounds resulted in lower egg numbers and heartbeats in *D. magna*. However, the use of pure tones that do not refer to any kind of noise pollution in nature as well as the absence of information on sound levels make these

results difficult to compare with the results of this chapter and also difficult to extrapolate to natural populations.

Concerning water fleas' natural enemies, it was reasonable to expect from the FR results that water fleas had experienced predation by *Chaoborus* larvae in the mesocosms. However, contrary to this assumption, noise did not alter the FR of *Chaoborus* larvae nor their behavior assessed through the number of body rotations. The main difference between this chapter and the study by Rojas et al. (2021), where *Chaoborus* larvae displayed more body rotations with motorboat noise, is that this study accounted for repeated exposure (i.e., chronic noise), what Rojas et al. (2021) did not. So, it might be that *Chaoborus* larvae show more body rotations when exposed to noise for the first time and then resume normal behavior with repeated exposure, a phenomenon also referred to as "habituation" that has not been addressed in the present chapter as the response of "naïve" larvae was not tested. Habituation to noise has been reported in many species including fishes (Johansson et al. 2016; Kusku 2020; Rojas et al. 2021) and aquatic invertebrates (Hubert et al. 2022), and could result from sensory or motor fatigue, or associative learning between the repetition of a given stimulus and the absence of any threat.

Similar FRs irrespectively of the noise condition does not support the hypothesis that the water fleas of the noisy mesocosms benefited from a noise-induced reduction in *Chaoborus* predation. Surprisingly, the FR of *Chaoborus* larvae was of type I (linear increase of *per capita* predation rate with increasing prey density), while they were found to display a type-II FR (decelerating rise to an asymptote) in previous studies (Cuthbert et al. 2019; Krylov 1992; Spitze 1992; with *Daphnia pulex*, *D. longispina*, and *Culex pipiens* as prey, respectively). Regarding the *Chaoborus* and *Daphnia* populations used in this chapter, the highest prey density used ($n = 48$) was not enough to reach saturation, and it is not possible to exclude an effect of noise at higher prey densities. Another reason why it was difficult to use the behavior of the *Chaoborus* larvae to explain the increase in cladocerans in the noisy mesocosms could be that predation tests in small and highly controlled experiment units are not representative of the foraging patterns occurring in more complex systems (i.e., the mesocosms used in this chapter). For instance, many zooplanktonic species including *Chaoborus* larvae and *Daphnia* show vertical migrations (Dawidowicz et al. 1990; Haupt et al. 2009). Noise might disturb trophic links within zooplankton through alterations in the species-specific spatial patterns. In other words, the tests performed in aquariums might have underestimated the negative effect of noise on *Chaoborus* predation.

To understand how noise influenced the zooplanktonic communities of the mesocosms, a focus was made on the trophic link between *Chaoborus* larvae and *Daphnia*, and no work was made on the other ecological interactions, in particular competition. Cladocerans are known to compete with rotifers and copepods for common food resources (Gilbert 1988; Lehtiniemi and Gorokhova 2008), copepods being the second planktonic group (after cladocerans) in terms of abundance in the mesocosms. The response of freshwater copepods to noise remains unknown, but a negative effect could make the competition even more asymmetric in favor of cladocerans. The three groups are also engaged in apparent competition by sharing

*Chaoborus* larvae as predator (Elser et al. 1987; Swüste et al. 1973). An interesting perspective would be to assess their respective contribution to *Chaoborus*' diet under chronic noise.

To conclude, this chapter suggests that chronic motorboat noise is likely to disturb the composition and dynamics of freshwater zooplankton without providing evidence for any alteration in the trophic link between *Daphnia* sp. and *Chaoborus* larvae. The effects of chronic motorboat on freshwater zooplankton probably involve the modulation of ecological interactions, but this remains to be further investigated. This chapter also illustrates how scaling up individual responses obtained in highly controlled conditions to the level of communities remains tricky. Additional research on the long-term effect of noise on freshwater zooplankton, as well as on fish-dominated planktonic communities, is needed.

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# Section XIII

# Management of Noise

# Near Real-Time Underwater Sound Modeling of Dredging Noise to Meet Regulatory Noise Thresholds

99

Thomas Folegot, Erwan Hemon, Georg Nehls, Mara Schmiing, Stefan Bräger, Michael A. Bellmann, Stephan Gerlach, Rainer Matuschek, and Judith Flamme

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**Abstract**

The Fehmarnbelt Fixed Link consists of an immersed tunnel connecting Denmark and Germany in the western Baltic Sea. Marine sediments are excavated to create the tunnel trench. The importance of the Fehmarnbelt for the harbor porpoise is recognized and managed in German waters through a Natura 2000 area. The authorities defined two indicators to safeguard its function. The first threshold demands that not more than 20% of the cross-section between Lolland and Fehmarn shall be exposed to construction Sound Pressure Level exceeding 144 dB ref. 1 μPa. The second threshold states that not more than 1% of the Natura 2000 area shall be exposed to construction Sound Pressure Level exceeding 140 dB ref. 1 μPa from June to September.

Measurement of dredging sound does not allow to assess the compliance with these thresholds since public shipping sound regularly exceeds them already. Therefore, a hybrid approach combining real-time modeling and continuous measurements has been developed, using measurements for continuous calibration of the model.

Results of the first and half construction year revealed an average deviation between the model and measurements less than 1 dB. Compliance with the thresholds is assessed with confidence and reliability despite the interference of sounds introduced by other maritime activities.

**Keywords**

Sound map · Noise map · Modeling · Calibration · Threshold · Dredging · Harbor porpoises · Compliance

## Introduction

Anthropogenic underwater noise is recognized as a threat to marine wildlife, and there are growing needs to monitor, mitigate, and regulate it. Underwater noise may be harmful to cetaceans through interfering with their acoustic communication, orientation, and prey capturing abilities and – at high doses – might even lead to auditory damage. While much knowledge has accumulated about the responses of marine mammals to impulsive sounds from offshore pile driving or seismic surveys (Richardson et al. 1995; Southall et al. 2007; National Marine Fisheries Service 2018), much less is known about their reaction to continuous sound from shipping or dredging (McQueen et al. 2020). Although ships are not the loudest sources of underwater noise, their sheer abundance makes shipping sound a very important

source of chronical anthropogenic noise in the oceans (Hildebrand 2009; OSPAR Commission 2009). Ambient noise levels have been rising in some regions along with an increase in shipping activity (Anderson et al. 2008; Andrew et al. 2011; Miksis-Olds et al. 2013; Kaplan and Solomon 2016). In the context of overall rising underwater noise levels, also dredging noise receives growing attention and the Central Dredging Association also addressed the issue (CEDA 2011). CEDA acknowledged that dredging involves a variety of activities, which produce underwater sounds but also concluded that appropriate management practices lack standardized monitoring protocols. Dredging sound is diverse but usually only of moderate intensity except for Trailing Suction Hopper Dredgers, which emit relatively high underwater noise levels (Robinson et al. 2011). As dredging operations may include a variety of vessels, an assessment of underwater sound cannot simply focus on the dredgers but must assess the whole operation.

The construction of the Fehmarnbelt Fixed Link as an immersed tunnel involves extensive dredging operations for an extensive period of time and space. The Fehmarnbelt Fixed Link will consist of a 17.6 km long, immersed tunnel connecting the Danish Island of Lolland and the German island of Fehmarn in the western Baltic Sea (Fig. 1). For the first time in Germany, the relevant authorities have imposed restrictions to underwater sound from dredging and associated activities to preserve the habitat of the population of 2000–4000 harbor porpoises staging and nursing in the Fehmarnbelt (FEMM 2013). These restrictions induce an obligation to monitor, manage, and limit the overall sound levels according to requirements defined in the plan approval decision. The first restriction demands that not more than 20% of the cross-section between Lolland and Fehmarn shall be exposed to construction Sound Pressure Level exceeding 144 dB ref. 1 μPa. The second restriction states that not more than 1% of the Natura 2000 area shall be exposed to construction Sound Pressure Level exceeding 140 dB ref. 1 μPa from June to September.

In this study, we outline a hybrid approach that combines continuous sound modeling and in situ underwater sound measurements as the most suitable and robust method for assessing the compliance with regulatory metrics. The continuous assessment of the compliance with two sound thresholds defined by the regulatory authorities is described and the uncertainties assessed.

## The Sound Exposure Issue Related to the Tunnel Construction

### Activities Involved in the Tunnel Construction

For the construction of the tunnel, marine sediments need to be excavated to create a tunnel trench about 90 m width and 16 m depth for the subsequent immersion of the tunnel segments. The construction includes extensive dredging operations. The dredging will result in around 19 million cubic meters of sand, stone, and soil.

Different types of dredging vessels conduct the dredging operation at sea (Fig. 2). In the coastal areas, backhoe dredgers are deployed. Further offshore, in deeper

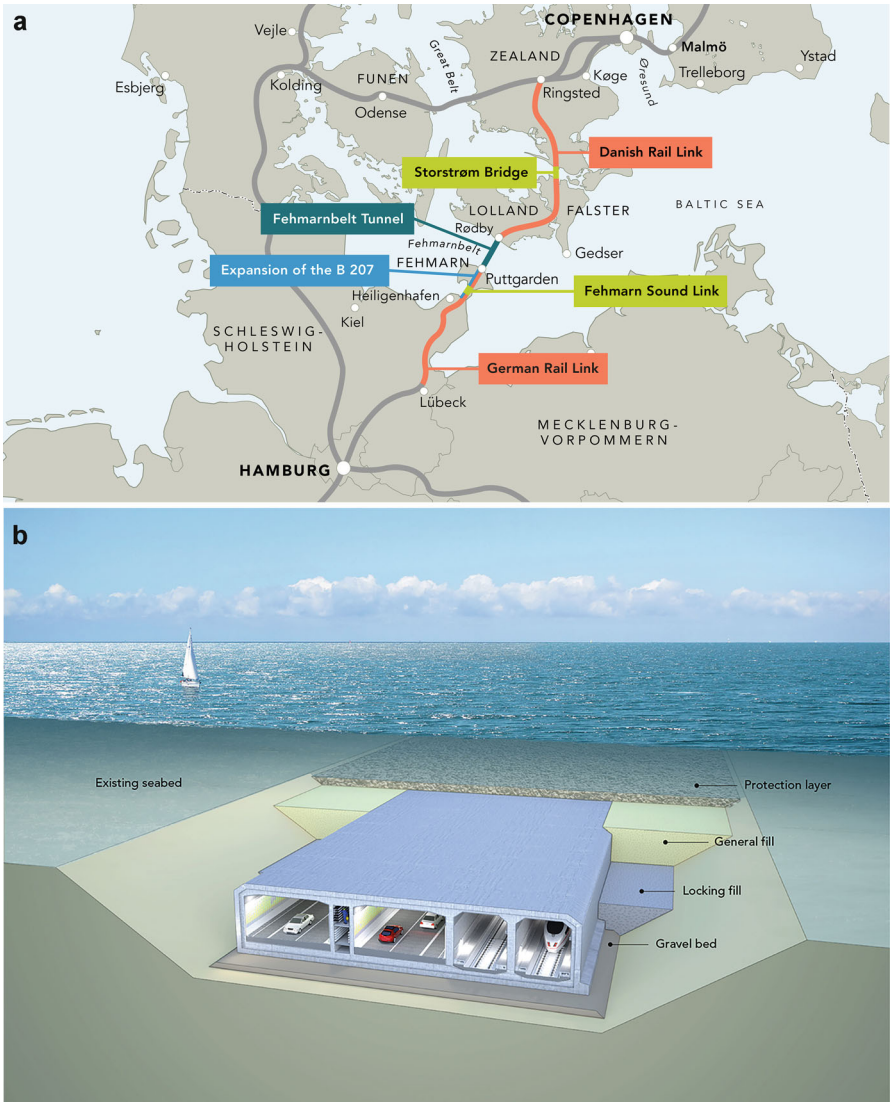



**Fig. 1** The Fehmarnbelt tunnel project. (Copyright: Femern A/S)

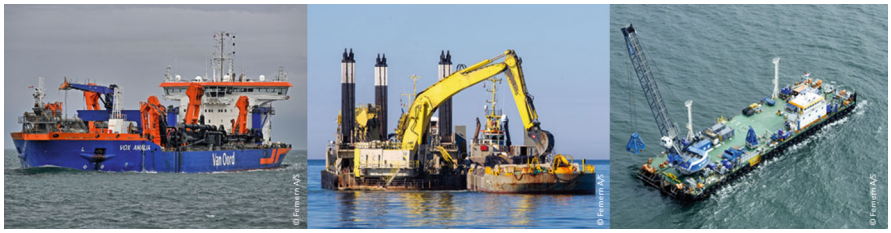


**Fig. 2** Example of different dredgers involved in the tunnel construction. (Copyright: Femern A/S)

waters so-called grab dredgers and trailing suction hopper dredgers are used. In total, more than 100 vessels have been involved 24 h per day during the first and half year of construction, including Trailing Suction Hopper Dredgers, Backhoe Dredgers, Grab Dredgers, Tugs, Barges, Survey vessels, etc.

## Sound Exposure for Harbor Porpoises

The Fehmarnbelt is a natural habitat of the harbor porpoise (*Phocoena phocoena*), a small odontocete whale which is the only cetacean species regularly reproducing in the Baltic Sea. Harbor porpoises of the Fehmarnbelt belong to the population of the western Baltic Sea which comprises about 20,000–40,000 individuals. The wider Fehmarnbelt area has a function as a staging and nursing area for 2000–4000 individuals with highest numbers during spring and early summer (FEMM 2013). During spring and autumn, porpoises are migrating through the Fehmarnbelt which is the main connection between the Danish Belt Sea and the Baltic Proper. In the center of the Fehmarnbelt, a marine protected area as part of the European Natura 2000 network has been established in 2007 to protect harbor porpoises.

As harbor porpoises show aversive responses to underwater sound, it has been extensively discussed with German authorities and NGOs whether the construction of the tunnel may negatively affect the function of the Fehmarnbelt for this species. The environmental impact assessment concluded that porpoises are less sensitive to continuous noise and that the construction of the tunnel will not impair harbor porpoise population in the Fehmarnbelt (Femern A/S and LBV SH 2013).

## Sound Indicators and Thresholds to Prevent Impacts

For the first time in Germany, the relevant authorities have imposed restrictions to underwater noise from dredging and shipping as well as a demand to monitor and manage noise according to the following requirements:

1. Not more than 20% of the cross-section between Lolland and Fehmarn shall be exposed to construction Sound Pressure Level exceeding 144 dB ref. 1 μPa to avoid creating a barrier effect that may impede porpoise migration (indicator 1).
2. Not more than 1% of the protected Natura 2000 area shall be exposed to construction Sound Pressure Level exceeding 140 dB ref. 1 μPa during the reproductive period of harbor porpoises from June to September to minimize the impact on the protected area and on reproducing porpoises (indicator 2).
3. The compliance to these thresholds shall be monitored continuously as part of the environmental construction monitoring.

## The Hybrid Model-Measurement Approach

### The Maritime Context of the Fehmarnbelt

The Fehmarnbelt is a strait connecting the Bay of Kiel and the Bay of Mecklenburg in the western part of the Baltic Sea between the German island of Fehmarn and the Danish Island of Lolland (Fig. 3). Ferries connect Puttgarden and Rødby on the two islands. The strait features a 10 nautical miles wide area with depths of 20–30 meters. Currents in the strait are weak and mostly dependent on wind.

The Fehmarnbelt is one of the busiest waterways in Europe. Figure 4 is representing the cumulative time when the ambient sound exceeded 140 dB ref. 1μPa (dominance map) before the project activities in the marine environment started. This dominance map shows that the sound of the shipping route connecting the North Sea and the Baltic Sea and the ferry line between Germany and Denmark cause levels of ambient sound pressure levels that may overcast construction sounds above 140 dB ref. 1μPa up to about 10% of the time (Fig. 4).

### The Model-Measurement Hybrid Approach

Since the commercial shipping and the ferry vessels generate sounds that already regularly exceed 140 dB ref. 1μPa and 144 dB ref.1 μ Pa, monitoring construction sounds in the Fehmarnbelt, the monitoring of construction sound and compliance with the thresholds, i.e., the two noise indicators, cannot be achieved by sound

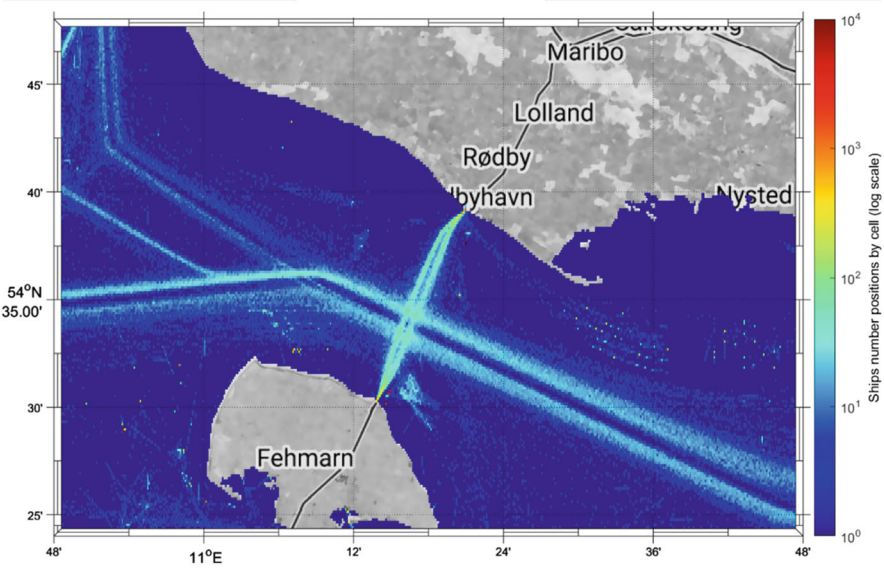



**Fig. 3** Maritime traffic density map in the Fehmarnbelt before the tunnel construction started in 2020

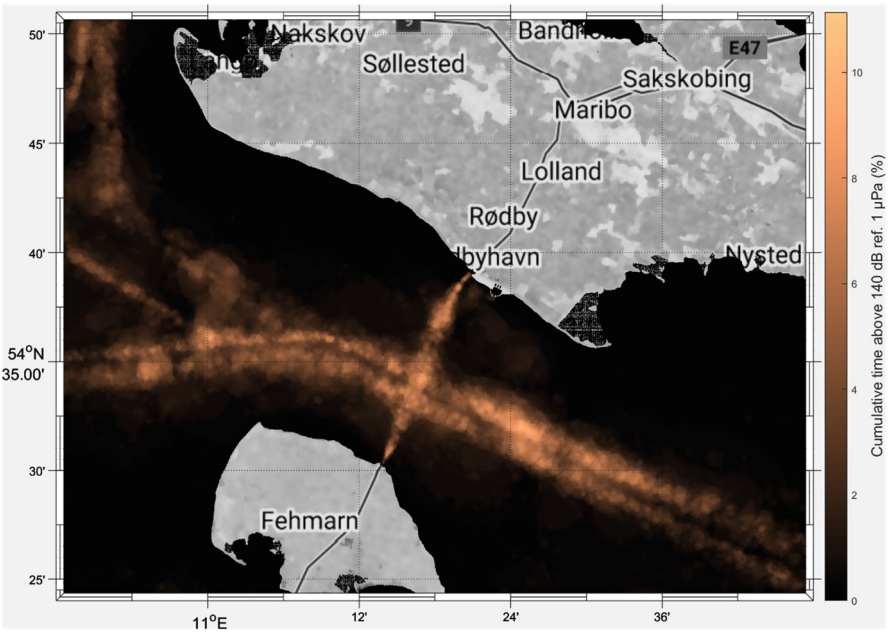



**Fig. 4** Proportion of time when the shipping sound exceeding 140 dB ref. 1 µPa before any tunnel trench dredging started in 2020

measurements alone. Indeed, direct measurement of the sound in the Fehmarnbelt would include sounds from all sources, commercial shipping, ferry vessels, etc., mixed with the sounds of the tunnel construction activities. The analysis of measured only values it would not be conclusive whether any excess of the thresholds is originated from the construction activities or from the other vessels. By nature, the sound measurement records all sounds, and it is not possible to separate contributions from the construction with sounds from the overall traffic. In addition, the spatial nature of the indicators, calculated along the cross-section and into the Natura 2000 area, cannot be assessed with only hydrophones.

A method to handle this intrinsic characteristic of the measured data for model calibration has been developed and is detailed hereafter. The construction sound monitoring follows a hybrid approach which enables the cumulative sound from construction activities and the ambient sound from the usual maritime traffic being assessed separately. The hybrid nature of the approach is combining near real-time modeling, continuous sound measurements along the construction sites, and dedicated measurements of underwater radiated noise (URN) of most of the vessels involved in the project (Fig. 5).

## Sound Measurement Strategy

A dedicated protocol has been defined to perform the sound measurements in order to ensure that the data would be suitable for the implementation of the hybrid approach.

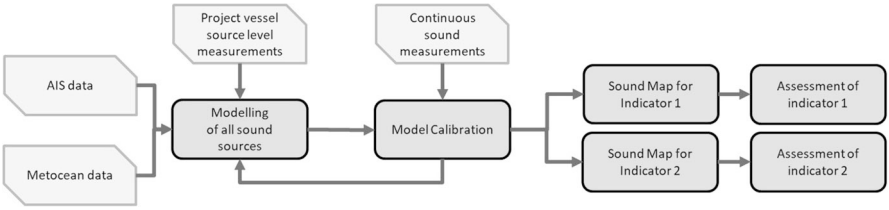



**Fig. 5** Overview of the hybrid methodology to model, calibrate, and assess both regulatory indicators

Three categories of measurement have been defined: category A consists of the measurement of the source levels (underwater radiated noise – URN) of the vessels involved in the construction. Category B consists of measurements at the vicinity of the construction site. Category C consists of long-term measurement at a distance from the construction areas.

Category A measurements are dedicated to the source-level measurements of various construction vessels involved in the tunnel construction. Levels are measured individually or representatively per vessel type. The processing of the URN takes into account the environmental conditions of the measurement site (such as sediment grain size) and the configuration of the measurement (such as the range between the hydrophone and the vessel). Source levels are estimated for different representative activities such as dredging and disposal by backhoe dredger (BHD), grab dredger (GD), and trailing suction hopper dredger (TSHD), steaming, anchoring, etc., including also tugs and service vessels operating under various activities. The source levels obtained, which includes the measurement of the source directivity, are integrated in the sound model.

Underwater sound is also continuously measured along the offshore working sites and used to validate and continuously recalibrate the model. Four mobile underwater sound measuring devices (Fig. 6) per offshore working site are deployed within a few hundred meters of the construction-related work. These mobile measuring devices are maintained weekly, i.e., the position is adjusted according to the construction progress and the measuring data are retrieved for subsequent data analysis.

In addition to the mobile underwater noise measuring devices in the near field of the offshore working areas, two permanent measuring stations are deployed at some distance to the construction areas (Fig. 7). These monitoring positions are maintained bimonthly.

For the mobile and the fixed measuring stations, standardized underwater noise measuring recorders are used that meet all requirements of the measuring regulations for underwater sound measurements (BSH 2011) (Verfuß et al. 2015) and are also calibrated regularly. The measurements are performed continuously with a sampling rate of at least 48 kHz so that the frequency range between 20 Hz and 20 kHz is covered. Measured sound pressure levels are processed into 20-s windows data in 1/3-octaves.

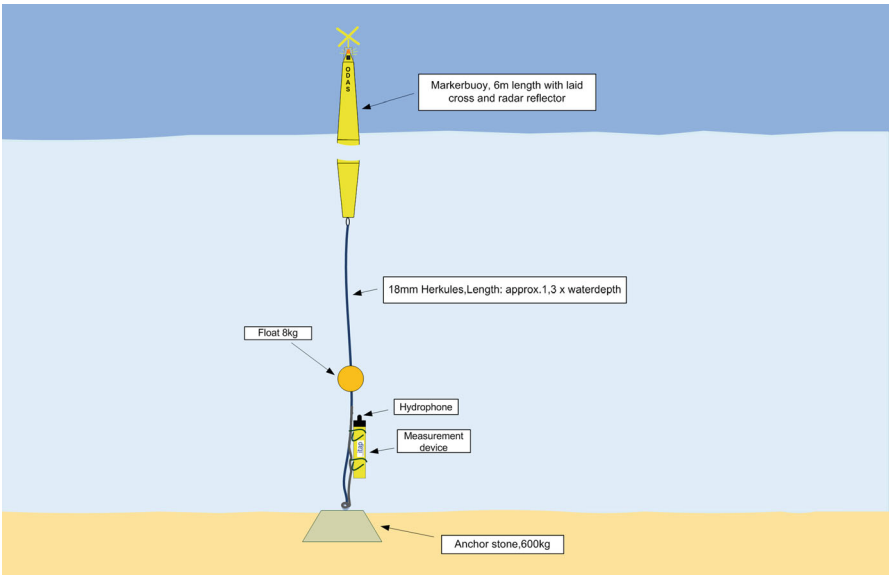



**Fig. 6** Sketch of the acoustic moorings

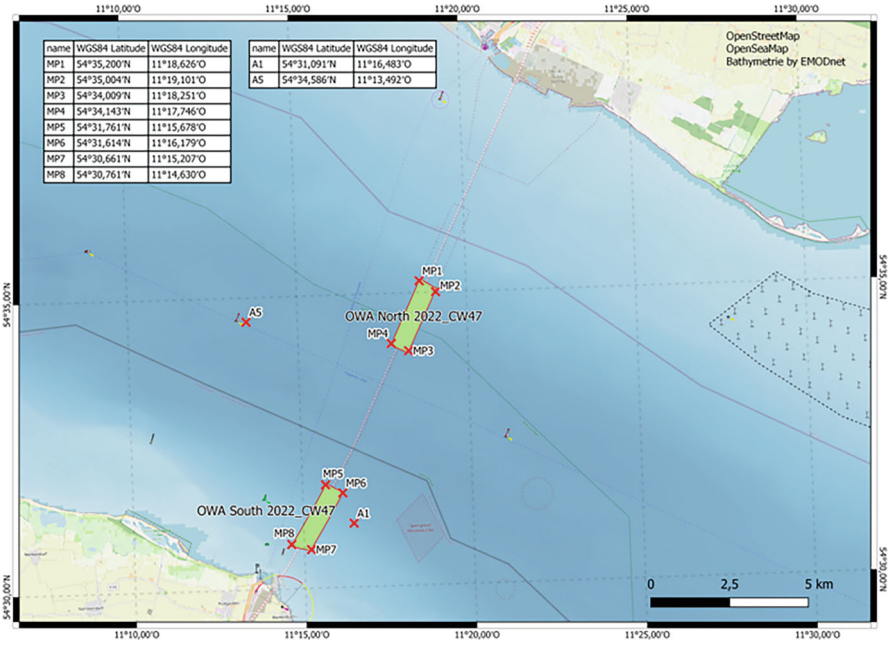



**Fig. 7** Example of the design of the sound measurements done in the construction sites and the long-term measurements at a distance of the construction activities

Noise measurement devices measure the entire total noise including natural noise from waves, splashes, and rain as well as other shipping sounds than that originating from construction activities. Sound measurements thus do not allow a direct assessment of construction sound. Therefore, the recorded underwater sound measurements.

## Sound Modeling

Sounds are modeled with the web-based Quonops© Online Services modeling platform (Folegot 2009, 2011). Although Quonops© supports other modeling techniques, in the framework of this project, Quonops© is solving the sound propagation Helmholtz equation by using the parabolic equation approximation parabolic modeling (Collins 1994; Collins et al. 1996). Broadband modeling is done in 1/3 octave bands from 16 Hz to 4 kHz for each sound source independently. Automated Identification System (AIS) data provides the near real-time position of all vessels continuously. Project vessel and related activity sounds are derived from the category A measurements, while the source levels from the non-project vessels are estimated from vessel size and sailing speed based on the RANDI 3 model (Wagstaff 1973; Ross 1976; Breeding et al. 1996).

The modeling of sound propagation is done for each sound source by a series of cylindrical vertical planes (Nx2D technique) with azimuths around the sound source spaced by 3°.

The oceanographic context is considered in the modeling: the sound speed profiles in water are derived from water temperature, salinity, and pressure (or depth) provided by operational oceanographic services and updated every hour. The main effect of these non-homogeneities in sound speed distributions is to create propagation channels. These complex phenomena are, however, predictable by numerical simulation.

The sediment properties and the changes of the bathymetry of the marine environment are also taken into consideration in the resolution of the Helmholtz Equation by considering the propagation in the marine substrate and the reflections from the water-sediment interface. These characteristics of the marine environment induce absorption, reflection, and diffusion of the sound through the soil and in the water column. The descriptive data of the bottom and bathymetry are taken from available local data.

The model calculates the sound originating from natural sources derived from the wind field data (Ainslie 2010) and all non-construction vessels – which in combination equates the so-called background noise – and construction vessels separately.

The model predicts the project vessel-based cumulative sound by integrating the individual source levels measured for the vessels involved in the construction activities and their near real-time positions provided by the AIS stream. This provides modeling of the background sound (i.e., the sum of all non-project-related sounds) and project-related sounds in near real time, separately as well as combined to form the total sound field (Fig. 8).

This approach allows for calibration of the model from cumulative sound levels recorded from non-project and project sounds, on the one hand, and the assessment of both conditions to verify compliance every 15 min, 24/7, on the other hand.

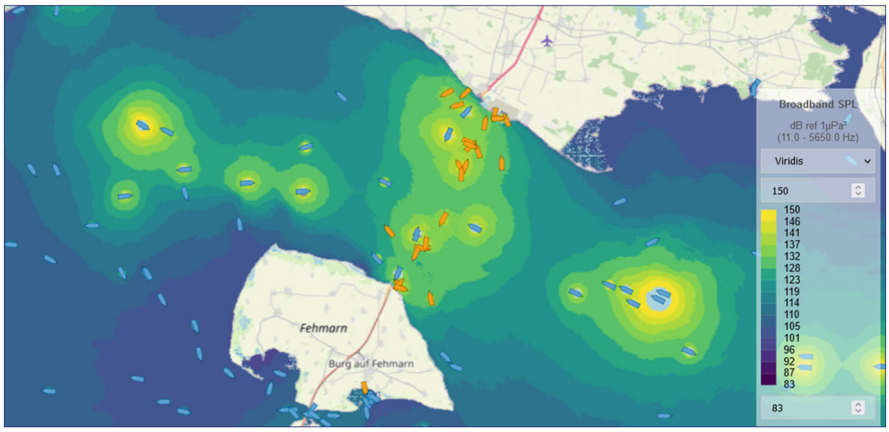



**Fig. 8** Instantaneous total sound map. Project and non-project vessels are displayed in orange and blue, respectively

## Model Calibration

The hybrid approach combines near real-time modeling provided by the Quonops© Online Services platform and the sound measurements. The modeling results are validated by recurring category B and C underwater noise measurements during the construction activities, allowing the model to be recalibrated periodically with the most recent underwater noise measurements. This procedure ensures that the model provides the best estimate of the overall sound at each measurement position.

The comparison of measured and modeled data is based on the total sound composed of construction sounds and background sounds. However, neither a detailed analysis of different situations for different time periods of the recordings nor a complex individual consideration of individual activities is possible.

The calibration methodology has been initially developed during the BIAS project (Folegot et al. 2016). The Cumulative Density Function (CDF) is used to overcome the difficulty that the measurement and the modeling are not performed at similar time resolutions by defining a common robust metric. The CDF describes the statistical content of the sound for the period considered similarly for both the measurement and the modeling, regardless of the technical characteristics of the measurement or the modeling. The CDF expresses the relationship between the sound level and the percentile, e.g., the proportion of time a given level occurs.

The calibration process consists of finding the best set of model parameters that minimize the difference between the CDF provided by the measurement and the CDF provided by the model at the position of the measurement and for the same period. The calibration is performed by minimizing the differences between both measured and modeled CDF by considering the following uncertainties:

- Wind: the wind fields are converted into sound of natural origin. The measurement provides a ground truth of natural noise and help to consider the local conditions that influence the natural noise generated by wind.

- Sediment: it is acknowledged that uncertainties in the description of the soil parameters need to be addressed. Therefore, we use a calibration process that will enable to model to decide on a set of geo-acoustic parameters that make the model fit best the natural noise component of the measurements.
- Non-project source levels: it is acknowledged that the diversity of commercial vessel designs, propellers, sizes, and hulls introduce uncertainties in the estimation of the source levels of the vessels. The RANDI 3 source-level model provides an a priori assessment. The calibration process accounts for this uncertainty and uses the measured ambient noise levels to adjust the source levels of the non-project vessels.

The assessment of the quality of the calibration is quantified by the calculation of the root mean square of the absolute difference between both measured and modeled CDF between the 5th and 95th percentiles (Fig. 9, left) and is updated on a weekly basis for all available measurement positions. From the start of the dredging in July 2021 to April 2022, more than 300 calibrations have been performed and the model-measurement mismatch has never exceeded 2 dB and is usually around 0.7 dB (Fig. 9, right).

## Model Validation

To validate the model, the modeling of the total sound (project and non-project activities in the Fehmarnbelt) is regularly compared with the acoustic measurement. An illustration of the comparison is provided in Fig. 10. The figure reveals a good agreement between the model and the sound measurements. It is important to note that there is a discrepancy between the continuous measurements of the total sound that are processed in 20-s windows and averaged on 15-min windows (black lines). The model on the other side provides a snapshot every 15 min (blue dots for the total sound, orange dots for the sound not related to the project activities). The gray area in the background explicit the variability of the measurement between the 1st and 99th

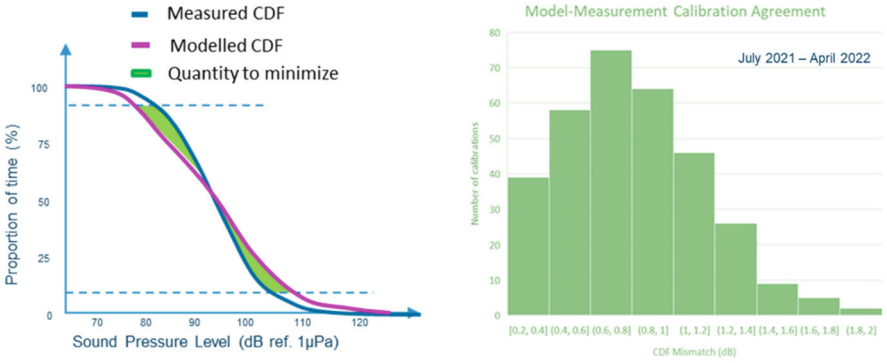



**Fig. 9** Left: The calibration of the sound modeling is based on the minimization of the difference between the measured and modeled Cumulative Density Functions (CDFs). Right: Distribution of the model-measurement statistical mismatch from July 2021 to April 2022

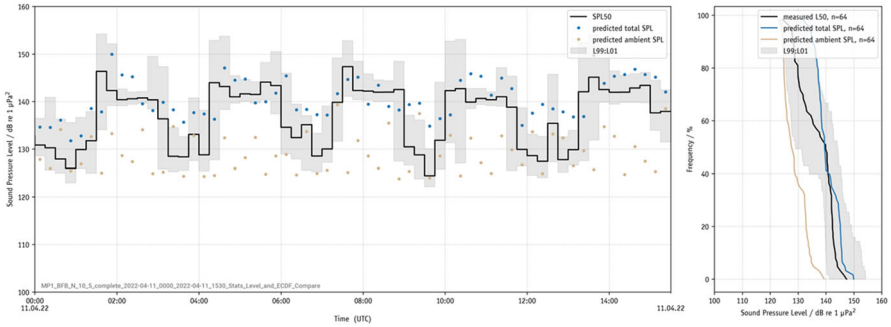



**Fig. 10** Course of the modeled and measured sound levels in April 11, 2022, at the measuring station MP1. The sound pressure level data are processed in 20-s windows and then averaged in 15-min windows (black line). The blue and brown dots represent the total and background sound, respectively, as predicted by the model. The gray background represents the 1–99% excess level of the measured values for the 15-min averaging intervals. The figure on the right is the Cumulative Function representation of the data

percentiles. There is a good agreement between the total sound levels modeled (in blue) that are close to the higher percentiles of the measurements. On the other hand, the modeled background levels not related to the construction activities are minimizing the lowest percentile measured. This is an expected result which highlights that the total sound is a mixture of construction and non-project sounds that cannot be reported by the measurement.

## Compliance with the Noise Indicators

### Indicator 1: Prevent a Barrier Effect

Indicator 1 attempts to represent the blocking effect potentially induced by the noise of the project on the passage of underwater fauna in the Fehmarnbelt channel. It is assumed that this effect becomes significant when more than 20% of the cross-section between Lolland and Fehmarn is exposed to sound levels above 144 dB ref. 1 µPa at all times.

To assess indicator 1, only the project's stationary activities near the site of the future tunnel are taken into account: this concerns project vessels sailing at speeds less than 3 knots (limit for considering vessels engaged in noisy dredging, towing, unloading, etc.) and which are located less than 300 meters from the cross-section (imaginary line of 17.6 km between Germany and Denmark, drawn at the location of the future tunnel) (Fig. 11).

The cumulative noise of these activities is calculated by the Quonops model every 15 min (Fig. 12), and their sound footprint beyond 144 dB ref. 1µPa is projected on the cross-section: the value of the indicator is given by the proportion of cross-

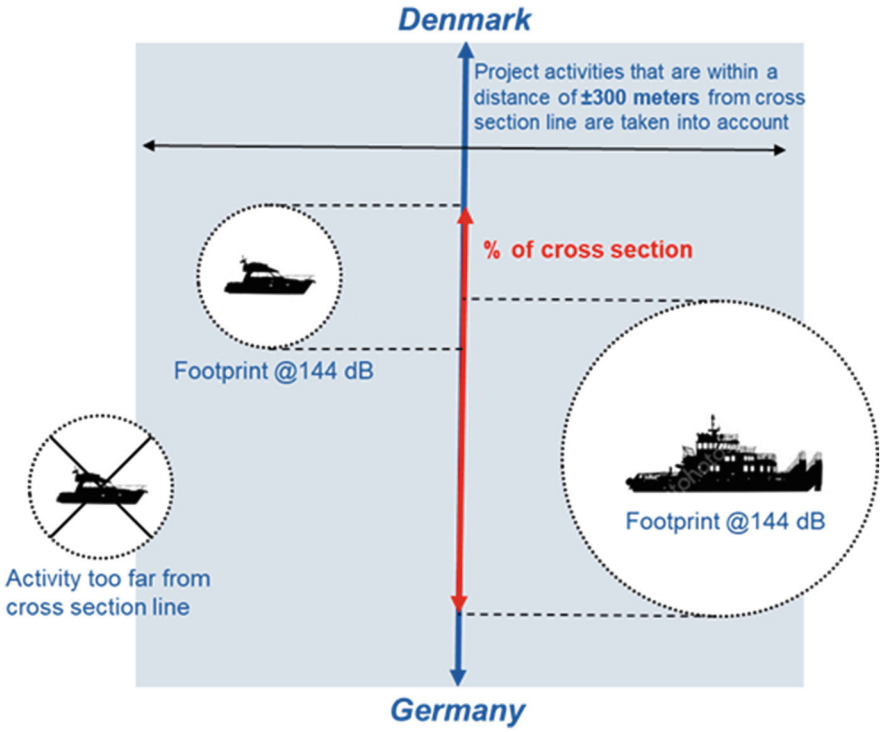



**Fig. 11** Overview of the methodology to assess indicator 1

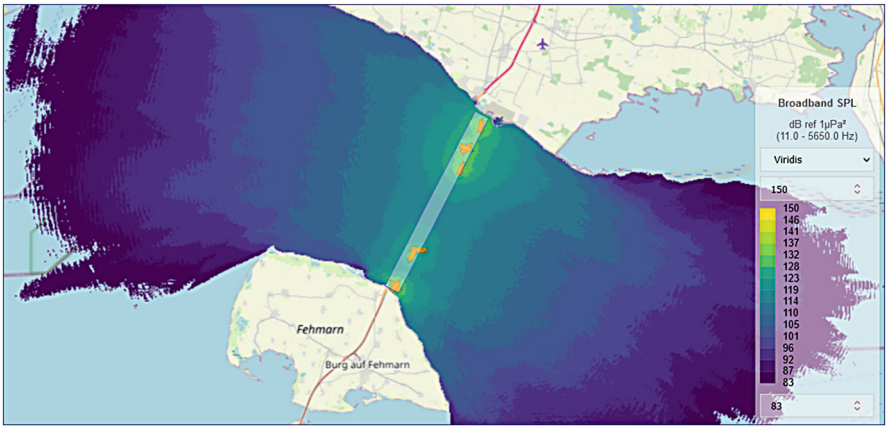



**Fig. 12** Example of sound map used for the calculation of indicator 1

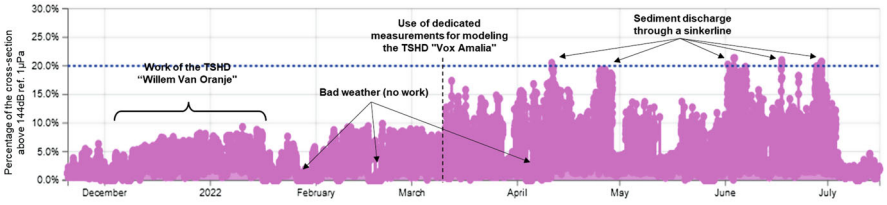



**Fig. 13** Percentage of the cross-section of the Fehmarnbelt above 144 dB ref. 1µPa (indicator 1)

section subject to a sound level greater than 144 dB ref. 1µPa compared to the total length of the cross-section.

The project shall not exceed the 20% limit imposed on indicator 1: this means that the noise generated at the construction site beyond 144 dB ref. 1µPa must not occupy more than 3.6 km (cumulative length) of the width of the channel between Germany and Denmark.

Figure 13 shows the evolution of the values for indicator 1 from December 2021 to July 2022. The fluctuations give a history of the project: highest values testify the local dominance of sound from project activities along the cross-section. More specifically, the Trailing Suction Hopper Dredgers such as "Willem Van Oranje" and "Vox Amalia" are among the loudest project vessels and contribute significantly to the values of indicator 1. The lowest values of indicator 1 can be attributed to periods of bad weather which caused the work to stop and in fact the emission of noise by the project in the channel Fehmarnbelt.

Following the use of dedicated measurements of the TSHD "Vox Amalia" in the modeling, the values of indicator 1 increased significantly from the end of March 2022. In April and June, the values briefly exceeded the 20% threshold, during the discharge of sediments by the TSHD into a sinkerline, the blocking effect of which is taken into account by the model. In such situations an investigation is performed, and a solution shall be found to reduce the noise level.

## Indicator 2: Limit Sound in the Marine Protected Area

For indicator 2 (1% of the Natura 2000 site), all construction vessels, irrespective of their speed and position, are considered by determining the construction-related noise levels. The project site intersects a Natura 2000 marine area where porpoises are protected during their summer breeding period (from the beginning of June to the end of September). Indicator 2 attempts to represent the proportion of this protected area exposed to loud project noise set to 140 dB ref. 1µPa by the authorities.

The cumulative sound of these activities (Fig. 14) is calculated by the Quonops model, and the value of indicator 2 is defined by the proportion of the marine protected area exposed to sound levels higher than 140 dB ref. 1µPa, compared to its total surface (280 $km^2$) (Fig. 15).

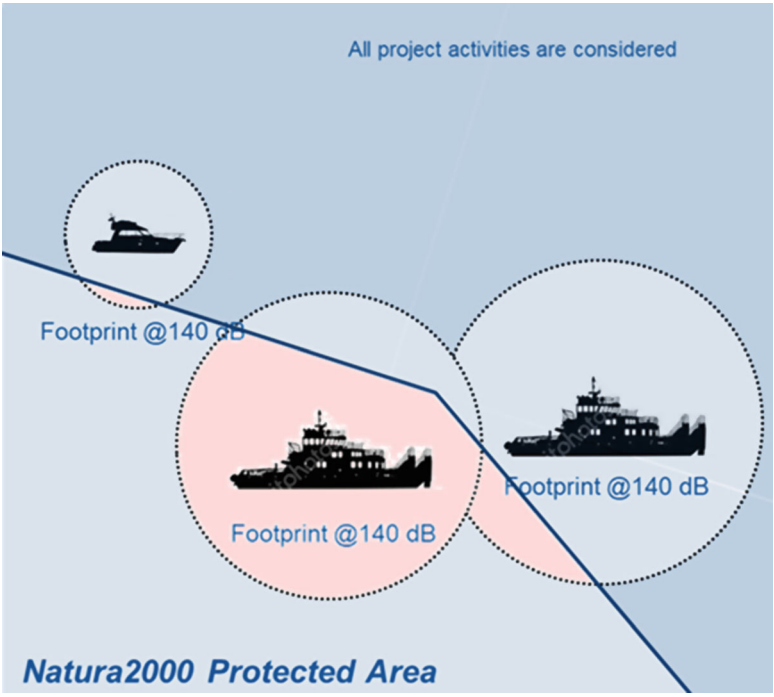



**Fig. 14** Example of sound maps used for the calculation of indicator 2

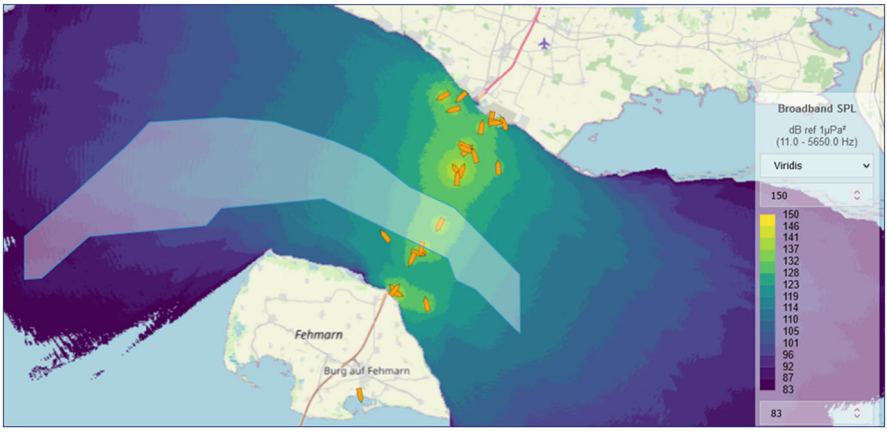



**Fig. 15** Overview of the methodology to assess indicator 2

The project shall not exceed the 1% limit imposed on indicator 2 from June to September: this means that the noise generated by the construction site beyond 140 dB ref. 1µPa must not occupy more than 2.8 $km^2$ of the surface of the Natura

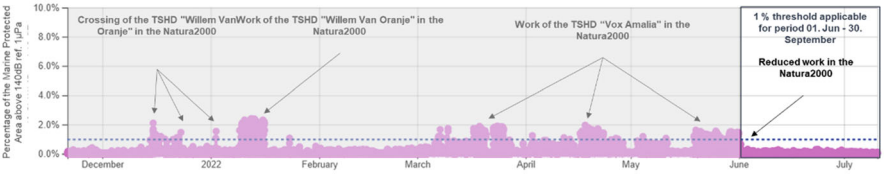



**Fig. 16** Percentage of the Natura 2000 area above 140 dB ref. 1µPa (indicator 2)

2000 marine protected area during this time of the year. Outside this period, the limit is raised to 10%.

Figure 16 shows the evolution of indicator 2 between December 2021 and July 2022. Similar to indicator 1, the fluctuations of indicator 2 can be attributed to particular events in the construction project: some high and isolated values correspond to occasional passages of project vessels crossing the Natura 2000 protected area, while the lower values correspond to localized works in the area or in the vicinity of the area. Outside the porpoise reproductive season in summer, the threshold for indicator 2 may not exceed 10%. However, from June to September, the threshold is 1%: the active management of the project has led to a reduction of vessel activities from the first of June 2022 to comply with the threshold.

## Conclusion

The regulatory thresholds defined to protect the marine environment from the sound generated by the construction project of the tunnel across the Fehmarnbelt aim at preventing a barrier effect and maintaining a quality of the habitat in the Natura 2000 area "Fehmarnbelt."

The busy maritime context of the Fehmarnbelt generates a large amount of ambient sound that may overcast construction sounds more than 10% of the time. Therefore, a hybrid approach that combines in situ measurement with modeling was therefore mandatory to enable the assessment of the sound originated from the construction independently from the sounds of all other human activities. The method developed has led to an operational solution, providing near real-time monitoring of the regulatory thresholds continuously every 15 min.

The results of the first year and half year of real-time modeling and measuring underwater noise of the dredging work revealed a good fit of modeled underwater noise with the measurements. Therefore, project-related sound across the Fehmarnbelt can be predicted confidently and with sufficient accuracy to assess compliance with the regulatory thresholds for underwater sound despite the interference with the sounds introduced by the heavy commercial traffic.

Although the real-time assessment is fully automatized, it proved to be essential to have precise knowledge of the construction activities and the specifications of the construction vessels and their source levels. Though source levels can be calculated

from available sources, it is recommended to measure source levels of specific activities at the construction sites to reduce the uncertainties of the model as much as possible.

Underwater noise modeling was accompanied by extensive noise measurements which allowed routine calibration of the model. The stability of the model has been achieved thanks to the numerous measurement stations.

The monitoring of the sound indicators is actively used by the project management to plan and organize the operations of the project. The compliance achieved since the start of the project has proven to be successful to protect the marine environment and the approach can be recommended to other projects where the occurrence of protected species may require managing underwater sound.

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# EU Marine Strategy Framework Directive-Compatible Approaches for Impulsive Noise Assessment

100

C. Juretzek, A. Müller, Dominique Clorennec, Erwan Hemon, B. Niesterok, Peter Sigray, J. F. Borsani, Thomas Folegot, and M. Boethling

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**Abstract**

The Marine Strategy Framework Directive (MSFD) (Directive 2008/56/EC) promotes sustainable use of the seas by requiring member states to achieve and maintain Good Environmental Status (GES), including underwater noise as one of of the eleven specific criteria for evaluating GES. In 2022, the EU established quantitative threshold values for addressing the consequences of impulsive underwater noise input into the marine environment. In this chapter, a data-driven stepwise approach is introduced for the implementation of an MSFD compatible assessment methodology. Corresponding standardized practices are presented. For the data-driven evaluation of regional requirements and of the applicability of harmonized methodologies within EU regions, exemplary regional case studies were performed for the Baltic Sea, North Sea, and Mediterranean Sea. Further, the estimation of effect ranges and corresponding risks of adverse effects is important for an impulsive noise assessment. Regional differences in the marine environment were addressed. Main challenges of impulsive noise propagation modelling were identified based on both numerical and analytical/empirical propagation modelling. Finally, recommendations for the impulsive noise registries are presented.

**Keywords**

Impulsive underwater noise · Regional assessment · Noise registries

## Introduction

The EU Marine Strategy Framework Directive (MSFD) (Directive 2008/56/EC) provides an important framework for a sustainable use of the seas within the European Union, which explicitly includes the effects of underwater noise in an assessment of the state of the marine environment. The MSFD was adopted to protect the marine ecosystem and biodiversity and obliges EU member states to achieve and maintain *Good Environmental Status* (GES). Descriptor 11, underwater noise, is one of the eleven specific descriptors for Good Environmental Status. Descriptor 11 (D11) comprises two criteria, with the first criterion (C1) relating to *Anthropogenic impulsive sound in water* (Commission Decision 2010/477/EU). In 2022, the EU introduced MSFD-related quantitative threshold values for underwater noise, including impulsive noise (D11C1), for the first time. The focus of these thresholds is on the protection of habitats of noise-sensitive marine species.

*Commission Decision (EU) 2010/477/EU* provides relevant criteria and methodological standards for the Descriptors of GES. For the assessment of GES in relation to D11, the temporal and spatial distribution of impulsive underwater noise (loud, low and mid frequency impulsive sounds) was identified as a primary criterion (D11C1). It is specified as *proportion of days and their distribution within a calendar year over areas of a determined surface, as well as their spatial distribution, in which anthropogenic sound sources exceed levels that are likely to entail significant impact on marine animals (Com. Dec. 2010/477/EU).* Commission

Decision (EU) 2017/848 detailes further criteria elements for the 11 qualitative descriptors and specifies methodological standards for the scale of the assessment, for standardized methods of monitoring and assessment and for units to evaluate the criterion. However, the criteria definitions and methodological standards must be further developed and specified with regard to the practical implementation and assessment of GES within the different EU regions and subregions.

Since the introduction and propagation of anthropogenic underwater noise is of transboundary nature, the assessment is relevant to all marine regions and sub-regions under the MSFD. Given the diversity of ecosystem characteristics and the various species affected by underwater noise in different regions, the assessment methods under Criterion D11 must take account of the regionally diverse conditions. Nevertheless, to ensure consistency and coherence of assessments in the marine regions and subregions, Commission Decision (EU) 2017/848 requires the definition of methodological standards and harmonization of the application of criteria for assessing the extent to which GES is achieved by Member States.

Since the adoption of the MSFD, a number of different assessment approaches for impulse noise have been proposed or used by different regions and Member States individually (Dekeling et al. 2020). This chapter describes results from the EU project HARMONIZE (see project report for details) (Müller et al. 2021), which was funded by the General Directorate (DG) Environment of the European Commission under the DG ENV/MSFD 2020 Call to support the preparation of the next 6-year cycle of implementation of the MSFD with respect to the level of anthropogenic impulsive underwater noise introduced into the marine environment. Against this background, HARMONIZE conducted the first cross-regional attempt to examine the applicability of assessment approaches for D11C1 to the diverse marine areas, to identify immediately applicable methods for assessing impulsive noise based on regional impulsive noise registries and to analyze the different assessment requirements in different EU regions and sub-regions. Thus, the project supported the definition of thresholds and evaluation of GES regarding underwater noise to fulfil upcoming MSFD reporting obligations of EU Member States.

HARMONIZE used a data-driven stepwise approach to recommend harmonized methodologies and standardized practices to be integrated in an D11 assessment that is applicable in practice, scientifically justified, comparable, and replicable across regions, sub-regions, and Member States. Key activities of this stepwise approach were the sourcing of assessment data, the identification of best available practices, and regional and sub-regional requirements as an important basis for harmonization. Another key activity was the investigation of regional case studies. These tested the practicality of the harmonized, data-driven assessment methods against regional requirements.

## Handling of Impulse Noise in the EU Employing Noise Registries

As part of the implementation of the MSFD, regional noise registries have been established in cooperation between EU Member States and regional marine conventions since 2015. They provide the database for the MSFD assessment of Good Environmental Status with respect to impulsive underwater noise. EU Member

States have to report impulsive noise events annually to these regional noise registries. D11C1 requires the consideration of three main aspects: the spatial distribution, the temporal extend, and the levels of impulsive underwater noise. The regional noise registries were developed to be able to include these aspects accordingly, but they use simplified concepts (i.e., source strength classes and anthropogenic activity categories). The most common reported time resolution of impulsive noise events is on a daily basis, but an even lower reporting resolution is also accepted by the registries. The spatial information on events spans from the exact source location of an activity, to within standardized geographic polygons of different sizes. Finally, noise registries classify impulsive noise event according to five noise level categories.

The evaluation of noise registry data provided the basis for determining the practicable level of accuracy that can be achieved, which largely depends on both the availability and resolution of data collected in the regional noise registries. For this analysis, three publicly available regional databases were used that currently include datasets on impulsive underwater noise: the ICES noise registry for OSPAR and HELCOM regions (International Council for the Exploration of the Sea (ICES) n.d.), the QuietMed noise registry for the Mediterranean Sea region (Enguix et al. 2019), and the EMODnet database (EMODnet n.d.) with subsets of data from different regions (Fig. 1).

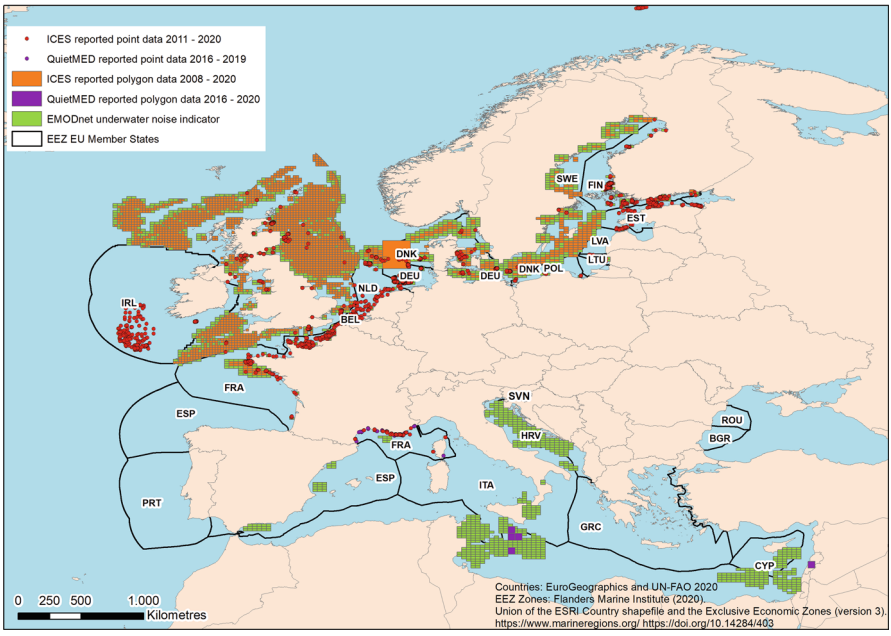



**Fig. 1** Visualization of publicly available data reported to regional noise registries

The analysis of available data in the noise registries has shown that a major insufficiency with respect to a quantitative assessment of impulsive noise arises from the incompleteness or non-reporting of impulsive noise data. Likewise, the heterogeneous level of details associated with reported impulsive noise data and corresponding parameters, i.e., differences in temporal and spatial information detail of events as well as the representation of the source strength, have a significant impact on the accuracy and reliability of a quantitative assessment. An essential prerequisite for the regional assessments, using the database provided by the registries, is data harmonization and quality assurance. Further, the temporal and spatial coherence and overlap between physical data on impulsive noise emissions on one hand and biological data on the other hand were identified as fundamental condition for the success of any assessment of the environmental status regarding D11C1.

## To Assess Impulsive Noise According to the MSFD

### Assessment Approach

Noise registries were designed based on MSFD-related guidance by the EU and were intended as a fundamental tool for a joint monitoring on regional or sub-regional level (Dekeling et al. 2014a, b, c). Over the past years, several decentralized assessment methodologies were developed by regions and member states. The introduction of quantitative EU threshold values motivated a harmonization of these approaches, taking the specifics of the different marine regions into account.

Four different assessment methodologies were included into the harmonization process for D11C1. These methodologies were reviewed with respect to various relevant aspects (e.g., required data, required resources, expert knowledge, and applicability to EU regions and sub-regions). They were further classified into several characteristic categories, as shown in Table 1 (adapted from Table 1 in Müller et al. (2021)).

An analysis of these assessment methodologies demonstrated that different components of the approaches were comparable to each other and included elements that could be adjusted to design a harmonized assessment framework for immediate application within the next MSDF cycle. Importantly, all assessment methodologies followed a common overall structure and were based on the common monitoring programm for D11C1, i.e., the regional noise registries. However, it became apparent that the four methodologies required a different accuracy and extend of input data.

For the purpose of evaluating the possibility to integrate the methodologies into a harmonized methodology, four *Harmonization Criteria* were identified, with corresponding minimum requirements (Müller et al. 2021):

1. *Suitability*: available data and information allow for an evaluation of the environmental status, uncertainties are well delimited, and results can be practically translated into regulation if needed.

**Table 1** Comparison of different assessment methodologies

| Main aspects | A: Mixed approach (Dekeling et al. 2020; Merchant et al. 2017) | B: Population-based approach (Dekeling et al. 2020; Heinis et al. 2015) | C: Habitat-based approach (Dekeling et al. 2020; Müller et al. 2020) | D: Habitat-based approach (Dekeling et al. 2020) |
|---|---|---|---|---|
| Baseline | Disturbance | Disturbance | TTS/disturbance | Disturbance |
| Source data | Noise registry | Noise registry | Noise registry | Noise registry |
| Effect ranges | Observations for species (based on specific adverse effect e.g. disturbance related to a specific activity) | Propagation model | Empirical propagation model/ measurements | Propagation algorithms/ predefined buffers |
| Assessment areal focus | MA, e.g., OSPAR II, MPAs | MA | MA (EEZ) and MPA | MA and PUHA |
| Adverse effect | Population (density) | Population (decline in density) | Habitat affected (quality loss) | Habitat affected (quality loss) |
| Assessment metric/ parameter to evaluate/ interpret | Exposure curve (index) | Population dynamics (model-based) | Adaptable percentage of area over time (based on sensitive periods) | Percentage of area and time exposed |
| Options for threshold for GES | Change in the form of the exposure curve (or change in index) | Change of population based on a model result, perhaps x % reduction in population | Percentage of area (x % EEZ or MPA) exposed over time | E.g., x % area exposed (MA and MPA) and y % time exposed |

*MA* management area, *MPA* marine protection areas, *PUHA* potentially usable habitat area

2. *Applicability*: assessment methods are well defined; uncertainties can be estimated by each Member State and the method can be applied for setting regulatory measures and evaluating the sufficiency of measures.
3. *Reproducibility*: methods applied are transferable to sub-regions and regions, results are non-ambiguous and allow for comparison of GES at union level.
4. *Feasibility*: assessment effort is such that each Member State is able to implement the methodology with own resources and capabilities.

The four assessment approaches included in Table 1 were analyzed against the four *Harmonization Criteria* established for the purpose of harmonization. Emphasis was placed on the applicability by each Member State at national level taking into account the available capacities. Furthermore, emphasis was placed on the unambiguity of the assessment results in light of the available input data. Minimum

requirements to be fulfilled for the implementation of a methodolody were stipulated for each of the four *Harmonization Criteria* as follows (Müller et al. 2021):

1. Minimum requirements for criterion *Suitability*: Data availability concerning species and habitats.
2. Minimum requirements for criterion *Applicability*: (2a) Methods available are widely accepted and well tested in case studies for as many EU regions as possible and are transferable between regions, (2b) uncertainties are defined and delimited, and (2c) monitoring may be used and integrated into the assessment to evaluate the effect or sufficiency of measures, which aim at improving the status of the marine envionment.
3. Minimum requirements for criterion *Reproducibility*: Assessment methodology and monitoring actions allow to compare results and efficiency of measures among regions and member states.
4. Minimum requirements for criterion *Feasibility*: Each member state should be able to participate with financial capacity and with resources currently available, without depending on expertise that exists only in some Member States or specialized institutes.

As basis for the recommendation of a robust assessment and for the estimate of realistic uncertainties and options for their reduction, these minimum requirements were considered as worst-case scenario for a regional assessment. Overall, the availability of input data required for a habitat-based assessment of the temporal and spatial exposure of regions and sub-regions within marine waters of most EU Member States was rated as high. One reason for this high rating is probably the fact that many Member States are involved in transnational long-term surveys on species and habitatsand have good access to environmental data within their national waters such as bathymetry, currents, and temperature profiles. Contrasting this, data required for predictive models in the context of population-dynamic modelling was rated as scarce for most EU Member States. A likely reason for this is the fact that the acquisition of physiological, demographic, or telemetry data at the necessary temporal and spatial resolution requires extensive monitoring efforts leading to increased financial ressources.

The criteria showed that a practicable and robust assessment approach must be based on a comparatively large, empirical database that isavailable to the Member States.

Another important aspect identified was the possibility to identify areas that are most severely affected by impulsive noise and to be able to formulate measures in response to this information. Thresholds compatible with the concept of habitat-conservation approaches were rated as capable of providing transparent results to be used for an effective handling of measures by the regulatory authorities in the Member States.

In the light of the structure and completness of available input data for the current MSDF cycles, an in-depth analysis of individual and heterogeneous datasets (as common practise in, e.g., certain Environmental Impact Assessments) was

rated to be not in line with current assessment priorities and the regional focus of the MSFD. Hence, a quantitative assessment of marine habitats and marine regions exposed to impulsive noise on the basis of available data in noise registries was evaluated to be directly applicable and feasible for all EU regions and sub-regions. Further, an assessment of the environmental status regarding D11C1 on different time scales, i.e., ranging from days to years, was identified as realistic ambition level for a harmonized assessment approach in all marine areas within EU MSFD responsibility.

The general procedure of an immediately applicable assessment approach is described by the following steps (Müller et al. 2021):

- Definition of the Management Area (MA), e.g., Marine Reporting Units (MRU).
- Determination of the assessment area/habitat of the indicator species.
- Evaluation of completeness and quality of the data.
- Determination of Level of Onset of Biological adverse Effect (LOBE).
- Selection of a propagation model to estimate sound pressure fields, effect ranges, or scientifically justified effect ranges (predefined ranges).
- Determination of the temporal and spatial noise pressure.
- Calculation of exposure using assessment/habitat area and noise pressure and determining the exposed habitat (% area and time exposed).
- Assessment of the environmental status based on threshold values for tolerable temporal and spatial exposure.

## Decision Scheme for Noise Propagation Modelling

The general objective of the noise propagation modelling in the context of the D11C1 assessment is to quantify the spatial exposure of noise-sensitive species in marine habitats due to one or more impulsive sound events. Sound propagation from a sound source to a receiving point is largely determined by various parameters including the source location and duration, the source depth, the type of source, the source spectra and distribution of energy, the bathymetry and seabed, the oceanographic conditions (such as temperature and salinity), currents, tides, the weather conditions such as wind and waves and the receiver location.

An important prerequisite for harmonization was that the determined exposed areas (in 2D) or the volume (in 3D) are comparable, i.e., lie in the same result range, regardless of the sound propagation modelling approach used. To achieve such comparable results, also the input data have to be well defined and accurate. For the selection of appropriate propagation models in a regional assessment, a decision scheme was developed as illustrated in Fig. 2. The decision scheme follows four interlinked steps (Müller et al. 2021), which are summarized in the following.

*Scope of application*: Depending on indicator species and marine region, the approach for the noise propagation modelling needs to consider that:

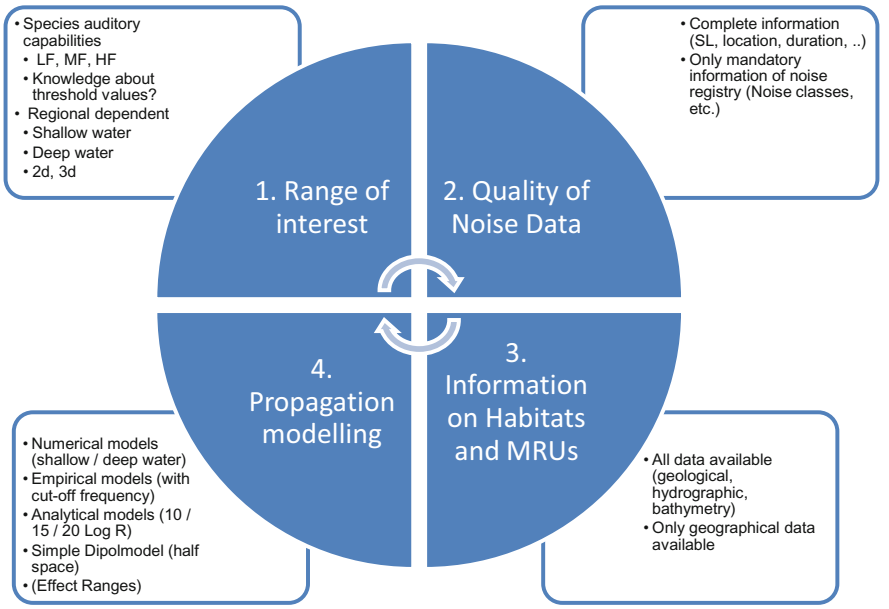



**Fig. 2** Decision scheme for the selection of appropriate propagation models as proposed in Müller et al. (2021)

- Each habitat or MRU has different physical and biological characteristics;
- Each indicator species or groups of species react differently depending on auditory capabilities (e.g., low frequency, mid-frequency, or high frequency), sound source frequency, and sound levels;
- Sound propagation in shallow waters, e.g., in the North Sea, differs from propagation in deep waters, e.g., Mediterranean Sea.

*Quality of noise data in registries*: Ideally, a complete dataset of the source event should be available for determining the sound levels using propagation modelling (source type, source level, location, duration). In addition to the mandatory data reported to noise registries, additional data, e.g., frequency distributions, specified source level, and measured levels, would significantly improve the quality of the noise propagation modelling. Irrespectively, the quantity and quality of available noise data should be evaluated carefully prior to an assessment.

*Selection of Habitat/Area of Assessment*: Habitat or area relevant for assessment should be defined with respect to environmental (bathymetry, topography, currents) and biological (species spectrum, occurrence) characteristics. For regulatory purposes, the selection of administrative sectors could be a suitable choice, which allows Member States to regulate and monitor activities and to define measures to achieve GES, if required.

*Information on geographical, geological, and hydrographic data for habitats or MRUs*: Hydrographic and geological information is important for the quality of the

noise propagation modelling. Sound propagation conditions depend on several parameters such as soil properties, stratification of the water column, and sound velocity profiles, which may change seasonally. There may be situations where required input data may or may not be available, which must be included in the decision-making process when selecting the appropriate modelling approach.

These interlinked steps should be considered for the selection of a noise propagation modelling approach. Based on the results of each step, the most appropriate choice may correspond to advanced numerical simulations, an empirical model, an analytical model, or even an effect range.

The decision-making process is based on a cycle scheme, since input data and selected procedures at all four levels influence the assessment of the environmental status and thus the assessment of GES. Additionally, uncertainties within each step influence the quality of the final assessment. On one hand, this can be seen as a disadvantage, but on the other hand it allows the process to be optimized.

Based on current experience with regional impulsive noise assessments, given the number of events, the accuracy of reported data and the current knowledge of habitats, populations, and species, simple propagation models are in most cases sufficient for the purposes of regional assessments. However, if more detailed input information is available, more complex propagation models can be used if the appropriate resources are available. Due to its EU-wide applicability for all marine regions, empirical approaches are universal and easily accessible in terms of implementation. However, acoustic wave fields are complex and for determining uncertainties in the assessments, a comparison between different propagation models is crucial.

## Numerical Modelling of Impulsive Noise

Both structure and information subject to annual reporting by Member States to the regional impulsive noise registries were defined by the EU-TG Noise during the beginnings of the MSFD. The requirements for reporting were kept low to encourage an active reporting on impulsive noise by EU Member States. The aforementioned requirements include obligatory parameter such as information on the date and location of impulsive noise introduced in the marine environment and other voluntary parameters. The description of the sources and sound propagation plays a key role in converting impulsive noise source information reported in the registry to effective representations of noise levels expected to occur in the marine environment.

Numerical simulations were performed to support the assessment strategies regarding the evaluation of effect ranges. The sound fields were estimated by numerical simulation involving the use of “range-dependent” models taking the previously mentioned parameters (bathymetry, sound speed profile, meteo-oceanic data, sound sources, sedimentology, etc.) into account. For frequencies above 1 kHz BELLHOP ray tracing was used (Jensen et al. 2011). The RAMSurf model using parabolic equations was used for frequencies below 1 kHz (Collins and Kuperman 1994). However, the choice of the model also depends on the modeling context, e.g.,

a ray-tracing method is not appropriate for the description of narrow waveguides (Küsel and Siderius 2019). Modeling was performed in two-dimensional planes (distance x depth), considering a single sound source described as a monopole. The data used as input for the numerical modelling was the data available in regional noise registries, hosted at ICES. To consider important regional differences, the North Sea, the Baltic Sea, and the Ligurian Sea in the Mediterranean Sea were selected as areas for case studies. For each of the case study regions, a sound-intensive activity was selected that represents a realistic impulsive noise event reported in the region. In addition to the regional perspective of each case study, a specific technical focus was given to challenges of quantitative assessments for each of the case studies.

Test case study 1 addressed seismic airgun arrays in the Mediterranean Sea. Airgun blasting is an activity commonly used by the oil and gas industries to locate reservoirs deep beneath the seafloor, involving high levels of impulsive sound. The context of the Mediterranean Sea requires modeling in very deep waters. An objective of this test case was to assess the sensitivity of acoustic propagation with respect to regional variability in bathymetry and bottom type, as well as seasonal variability in sound propagation conditions in water as detailed in Table 2.

The seasonal variation of oceanographic conditions had a strong impact on sound propagation with up to 20 dB difference in broadband level in the surface layer between the summer and winter season based on results from the modeling of scenarios 01A and 01B. The effect of sound speed variability between summer and winter concerned mainly frequencies above 1000 Hz in the upper water layer (up to 30 dB difference at 1000 Hz). This is related to the proximity of the source to the surface (with a source depth of 6 m), as well as by the fact that only the surface layer is affected by seasonal variations in sound speed. The depth-dependent gradient of sound speed in water depends on the temperature, pressure, and salinity of the water.It causes an oriantation of the wavefront by refracting the sound rays toward the surface or toward the seabed. Hence, sound propagation becomes affected by the state of the sea at the surface, or by the geo-acoustic properties of the seabed and its varying reflectivity. This phenomenon adds uncertainty to the estimation of sound levels at the receiving point, and requires that the environment is accurately described. Figure 3 shows an example of seasonal differences in the sound propagation at 500 Hz in the deep water context of the case study region in the Mediterranean Sea. Based on the modelling scenarios 01C and 01D, representing two different propagation directions from the same sound source, the bathymetry and variability of the seafloor properties have a strong impact on sound propagation. One one hand, the propagation of sound may be limited or interrupted in shallow water, e.g., when the sound field reaches the coast. On the other hand, the interaction of sound with the seabed increases for decreasing water depth and the sound propagation becomes affected by the geo-acoustic properties of the bottom. This caused an increase in sound levels of up to 5 dB based on the scenario results.

In order to study the influence of frequency discretization on the modeling results, the scenario 01C for a distance of up to 30 km was repeated by multiplying the frequencies in the calculation of the frequency range between 32 and 1000 Hz. To

**Table 2** Test case 1: SeismicScenarios description

| Ref. | Area | Source | Source depth rel. to surface | Bathymetry | Sediment | Source latitude | Source longitude | Water depth (m) | Azimut (°) | Season | Acoustic frequencies (central frequency of the octave band) (Hz) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 01A | Mediterranean | Seismic | 6 | Deep water | Local | 43.0 | 8.0 | 2650 | 90 | Summer (September) | 31-63-125-250-500-1000 |
| 01B | Mediterranean | Seismic | 6 | Deep water | Local | 43.0 | 8.0 | 2650 | 90 | Winter (March) | 31-63-125-250-500-1000 |
| 01C | Mediterranean | Seismic | 6 | Shallow water | Local | 43.1 | 6.5 | 450 | 90 | Summer (September) | 31-63-125-250-500-1000 |
| 01D | Mediterranean | Seismic | 6 | Shallow water | Local | 43.1 | 6.5 | 450 | 0 | Summer (September) | 31-63-125-250-500-1000 |

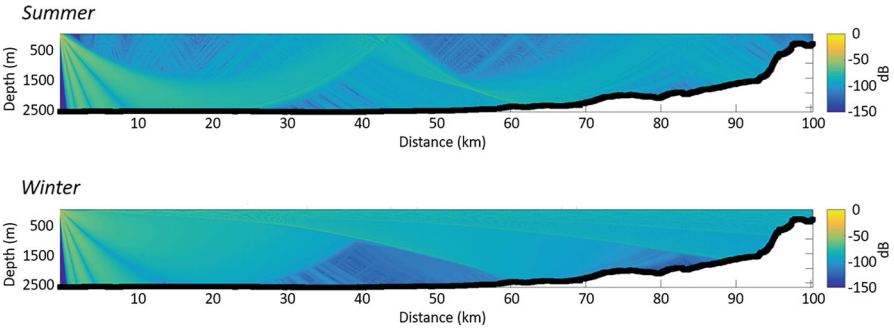



**Fig. 3** Seasonal propagation difference in a deep-water context at 500 Hz

**Table 3** Test case 1: Seismic—Results of frequency discretization (mean error and standard deviation of the transmission loss)

| Mean ΔTL (dB) by frequency (Hz) | Layer | 32 Hz | 63 Hz | 125 Hz | 250 Hz | 500 Hz | 1000 Hz |
|---|---|---|---|---|---|---|---|
| 1/3 oct. band discretization | Surface [0–100 m] | $-0.1 \pm 1.7$ | $0.1 \pm 1.2$ | $0.1 \pm 0.8$ | $0 \pm 0.7$ | $0.8 \pm 0.9$ | $0.2 \pm 0.7$ |
| | Mix [100–1500 m] | $0 \pm 0.5$ | $-0.1 \pm 0.3$ | $-0.3 \pm 0.3$ | $-0.3 \pm 0.2$ | $-0.1 \pm 0.2$ | $-0.3 \pm 0.3$ |
| | Deep [1500–2244 m] | $-0.4 \pm 0.8$ | $-0.3 \pm 1.8$ | $-0.9 \pm 0.8$ | $-0.3 \pm 0.9$ | $0.4 \pm 0.6$ | $-0.7 \pm 0.5$ |
| 1/1 oct. band discretization | Surface [0–100 m] | $1.1 \pm 3.1$ | $-0.5 \pm 2.3$ | $0 \pm 1.5$ | $-0.5 \pm 1.1$ | $2 \pm 2.3$ | $-0.2 \pm 1.4$ |
| | Mix [100–1500 m] | $-0.4 \pm 0.7$ | $-0.6 \pm 0.5$ | $-0.8 \pm 0.6$ | $-1.4 \pm 0.5$ | $0 \pm 0.4$ | $-0.6 \pm 0.6$ |
| | Deep [1500–2244 m] | $1.3 \pm 1.5$ | $-1.8 \pm 2.2$ | $-1.5 \pm 1.6$ | $0 \pm 2$ | $3.7 \pm 1.5$ | $0.6 \pm 1$ |

estimate the influence of the frequencies discretization on the accuracy in the modelling, three frequency schemes were tested. First, one central frequency was modelled per octave band (i.e., 6 calculated frequencies). Secondly, the calculation was repeated by modeling one frequency per third octave band (i.e., 18 frequencies calculated). Thirdly, one frequency per tenth octave band was modelled (i.e., 60 frequencies calculated). The results from these three schemes were then averaged to yield a single value of transmission loss per octave band to be able to compare the results. The comparison was then carried out by taking the tenth octave band calculation as a best possible case, assuming that the calculation with the finest discretization results corresponds tothe best representation of the real conditions. Table 3 shows the difference of the median values at the 50% percentile of the transmission losses (averaged error ± standard deviation over the range) calculated for each octave band of the one octave band calculation and the third octave band calculation.

The choice of the frequency discretization during the calculation of the propagation had a significant impact on the precision of the calculation. While the average deviations observed were close to zero, differences of up to 20 dB appeared locally. It is possible that these deviations affect the surface layers and the deep layers more frequently. The choice of an appropriate frequency discretization for sound propagation calculations hence corresponds to a compromise between the desired accuracy of the results and the time allocated to the calculation.

Test case 2 addressed underwater explosions in the Baltic Sea. Underwater explosions occur in different contexts (e.g., civil engineering such as rock excavation or fragmentation before drilling, military activities such as detonations of unexploded ordnance or ship shock trials) and are among the noisiest man-made activities in the seas. The objectives of this test case were to analyze the effect of sediment variability, the effect of bathymetry, the influence of source profiles, and the frequency dependence of underwater propagation as described in Table 4.

The joint influence of sediment variability and bathymetry has a strong impact on sound propagation, as demonstrated by the choice of two different bottom types (rock or sand) and two different bathymetric profiles for the modeling of scenarios 02A-B-C-D. Figure 4 shows the influence of the bottom properties on the acoustic propagation in a shallow-water context at 63 Hz. Overall, the rocky-type sediment was found to be more favorable for sound propagation, especially at low frequencies. The difference in transmission loss with respect to the sandy-type sediment reaches 80 dB for the lowest frequency of 32 Hz. This causes a difference on the broadband level received at 100 km from the sound source of up to 35 dB. These results highlight two strong influenceing factors on the propagation of sound: the cutoff frequency onone hand, which has the effect of blocking the propagation of energy below the cut-off frequency. The value of which depends on the water level and the type of sediment. The second phenomenon is the interaction of the acoustic wave field with the seabed. The reflection coefficient determines the proportion of energy that is reflected back into the water column and thus not lost in the sediment. At long distance, the repeated interactions of sound and the seabed make the effect of the reflection coefficient significant. Further, comparison of two different modeled source profiles has highlighted the importance of including the frequency distribution of the acoustic energy emitted by the source in the assessment.

Test case 3 addressed pile driving activities in North Sea, which involves repeatedly pounding long steel pipes into the ocean floor to support other structures, such as bridges, piers, and wind turbines (Table 5). A large hammer is used to strike the top of the pile, generating vibrations in the air, water, and sediment. The objectives of this test case were to study the effects of different technical noise abatement measures and their effectiveness depending on the source profile used. Details on technical noise mitigation solutions and examples of their associated mitigation effectiveness as observed at specific sea sites as also used in this case study are described in a cross-project experience report for offshore windfarm construction projects in Germany (Bellmann et al. 2020).

The considered source spectrum has a large effect on the modelled distribution of the underwater noise generated at the pile. Mainly because piling is performed in

**Table 4** Test case 2: Underwater explosions—Scenarios description

| Ref. | Area | Source | Source depth rel. to bottom | Bathymetry | Sediment | Source latitude | Source longitude | Water depth (m) | Azimut (°) | Season | Acoustic frequencies (central frequency of one-third octave band) (Hz) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 02A | Baltic-sea | Explosive confined | −2 | Shallow water | Rock | 58.6947 | 17.3147 | 10 | 130 | Summer | 31–63–125–250 |
| 02B | Baltic-sea | Explosive confined | −2 | Shallow water | Sand | 58.6947 | 17.3147 | 10 | 130 | Summer | 31–63–125–250 |
| 02C | Baltic-sea | Explosive confined | 20 | Shallow water | Rock | 58.8324 | 18.6669 | 90 | 130 | Summer | 31–63–125–250 |
| 02D | Baltic-sea | Explosive confined | 20 | Shallow water | Sand | 58.8324 | 18.6669 | 90 | 130 | Summer | 31–63–125–250 |

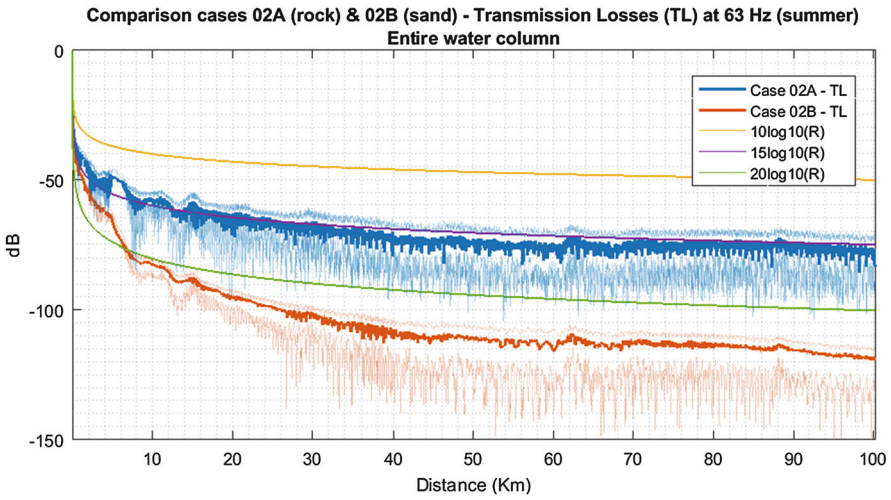



**Fig. 4** Influence of the bottom properties on the acoustic propagation in a shallow-water context at 63 Hz

shallow water, which exhibits a strong influence by the cutoff frequency. It has an even larger effect on the modelled efficieny of mitigation systems. The efficiency of the mitigation strongly depends on the source profile, since the attenuation is frequency dependent, with maximum reduction at mid frequencies (300–4000 Hz). Using a different source profile derived from Bellmann et al. (2020) altered the results. Further issues that add to the uncertainty of results result from the interaction with the seabed, which is site specific and influences the contribution of the sea-bottom thus altering the performance of the mitigation system. Using unrealistic source spectra potentially introduce deviations of about 10 dB in the estimate of the efficiency of the mitigation system.

## Conclusions

As result from the data harmonization, regional case studies, and quality assessment, it was demonstrated that a quantitative assessment of exposed marine habitats by impulsive sound based on available data in noise registries would be directly applicable for all EU regions and sub-regions. Further, a time-dependent assessment of the environmental status regarding D11C1 for different time-scales was identified as a realistic goal for a harmonized assessment approach in all marine areas within EU MSFD responsibility.

A key characteristic of the habitat-based approach identified as basis for harmonization was its potential to assess and limit impacts due to underwater noise by setting spatial and temporal thresholds for noise in certain habitats or parts of habitats (e.g., national reporting units, subregional units). Furthermore, the approach can be applied in line with the precautionary principle. For preventing impacts on marine

**Table 5** Test case 3: Piling—Scenarios description

| Ref. | Area | Source | Source depth rel. to bottom | Bathymetry | Sediment | Source latitude | Source longitude | Water depth (m) | Azimut (°) | Season | Acoustic frequencies (one-third octave band) (Hz) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 03A | North sea | Piling without mitigation | 2 | Shallow water | Sand | 55.66 | 4.2372 | 35 | 90 | Summer | 10 Hz −10 kHz (31 freq) |
| 03B | North sea | Piling with BBC | 2 | Shallow water | Sand | 55.66 | 4.2372 | 35 | 90 | Summer | 10 Hz −10 kHz (31 freq) |
| 03C | North sea | Piling with DBBC | 2 | Shallow water | Sand | 55.66 | 4.2372 | 35 | 90 | Summer | 10 Hz −10 kHz (31 freq) |
| 03D | North sea | Piling with IHC-NMS | 2 | Shallow water | Sand | 55.66 | 4.2372 | 35 | 90 | Summer | 10 Hz −10 kHz (31 freq) |

life due to impulsive noise the approach enables the protection of habitats by safeguarding the availability of a minimum proportion of undisturbed habitat at all times. Hence, the approach may ensure that relevant populations sensitive to underwater noise are able to use their habitat for vital functions despite being subjected to noise.

Priority targets for a successful implementation of such a harmonized assessment in the next MSFD cycles were identified as follows:

- Implement a robust area-based assessment methodology. This conclusion is supported by the fact that uncertainties are introduced by the regional registries that do not support a high-resolution assessment.
- Focus has to be directed to achieve data completeness in the noise registries. An assessment based on data, which islacking the majority of noise events would introduce uncertainties that are not acceptable.
- Focus on the improvement of the quality, reliability, and completeness of biological input data.
- Identify and agree on the biological component addressed in the assessment, such as at least one relevant species per region.
- Establish biological thresholds for species, species groups, or habitats of interest for the assessment in order to quantify the area affected by sound.
- Acquire information on the occurrence of species and their spatial and temporal distribution.
- Analyze impulsive noise characteristics, including spatial and temporal variations and differences between noise generating activities.
- Document and acquire additional information on impulsive noise occurances, such as results from measurements or results from the application of mitigation measures (e.g., technical noise abatement, spatial/temporal restrictions, and deterrence protocols).

Future objectives to be addressed after Member States and regions have gained experience with the evaluation of the current environmental status are how to use the assessment to determine the good environmental status (GES). In addition, a future task will be to identify targets and measures to be applied if the assessment shows that GES has not been achieved.

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# Acoustic and Carbon Dioxide Deterrents for Invasive Bigheaded Carps (*Hypophthalmichthys molitrix* and *H. nobilis*)

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**Abstract**

Invasive bigheaded carps (*Hypophthalmichthys molitrix* and *H. nobilis*) are two aquatic invasive species that are expanding their range throughout the Mississippi River Basin. These carps display negative phonotaxis to broadband sound, and thus, acoustic deterrents have shown promise for managing their populations. Numerous dams block direct upstream access for the carp; however, fish can bypass these physical obstructions by swimming through navigational locks. Nonphysical acoustic deterrents have the advantage of not effecting navigation, are relatively inexpensive, and could be tuned above the hearing range of non-ostariophysian fishes. Fish also react negatively to high levels of carbon dioxide, and it has been proposed to combine these two stimuli into one deterrent system to reduce potential habituation to long-term acoustic playbacks and increase barrier efficacy. Here, the possible synergies of combining these two stimuli into integrated deterrent systems are evaluated.

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## Introduction

A species introduced to a new range by humans is considered invasive when it reaches densities that can negatively affect native species and the environment. Historically, invasive species were considered second only to habitat loss endangering native species in the United States (Wilcove et al. 1998). Over a third of the species federally listed as threatened or endangered are imperiled primarily due to predation by, or competition with, non-native species (Pimentel et al. 2005). Indeed, non-native species are the second most common cause of extinction, after biological resource use, for all species worldwide since 1500 CE (Bellard et al. 2016). Invasive species are a growing problem caused by human introductions and will require active management to mitigate or prevent damages.

Two aquatic invasive species of recent concern in North America are the bighead carp (*Hypophthalmichthys nobilis*) and silver carp (*H. molitrix*). These carp species were first introduced to the United States from China in the early 1970s as a non-chemical method for plankton control in public water treatment facilities (Kelly et al. 2006). By 1975, silver carp had escaped and established wild populations in Arkansas waterways (Kolar et al. 2007). Their spread into open water systems was accelerated by extensive flooding of private sector aquaculture operations in the 1990s (Kelly et al. 2006). Common names for non-native species can have unintended social consequences (Larson 2005), such as fostering xenophobia. For this reason, although the common name of "Asian carp" for these species persists colloquially, hereafter the two species of silver and bighead carp will be collectively termed bigheaded carps.

Bigheaded carps are currently expanding across the United States (Noatch and Suski 2012). There is concern that these invasive carps will travel from the tributaries of the Mississippi River Basin where they have currently established populations and ultimately colonize the Laurentian Great Lakes (Kolar et al. 2007). After establishment, extirpation is nearly impossible, so prevention is the most efficient and cost-effective method to control highly invasive species (Lodge et al. 2006). Research on new barrier technology aims to prevent further range expansion into yet uninvaded ecosystems (Putland and Mensinger 2019; Vander Zanden and Olden 2008; Vetter et al. 2017a). The general goal of these selective barriers is to prevent the passage of undesirable species while simultaneously allowing the natural migration of native fish (Rahel and McLaughlin 2018).

Bigheaded carps have caused ecological damage where they have established large breeding populations, including inducing declines in native fish condition (Irons et al. 2007), and alteration of zooplankton communities (Sass et al. 2014). Currently, bigheaded carps account for 70% of fish biomass in the lower Illinois River (Garvey et al. 2012). There are concerted efforts underway to prevent

upstream passage into Lake Michigan through the Chicago Area Waterway System (Kolar et al. 2007). An electrical barrier in a lock chamber near Chicago prevents fish passage; however, it has numerous disadvantages: high maintenance expenses including energy costs, continual power usage with susceptibility to outages, potential lethality to humans, and reduced efficacy against small fish (Davis et al. 2017; Noatch and Suski 2012). Additionally, no single behavioral barrier is likely to stop all fish, and redundancy in deterrent systems would be necessary to achieve 100% efficacy (Davis et al. 2017).

To reduce economic costs such as impeding or stopping commercial vessel traffic, there is interest in developing nonphysical carp barriers, meaning they do not disrupt the flow of water or boat traffic (Noatch and Suski 2012). Besides electricity, broadband sound has been effective in deterring carp swimming (Vetter et al. 2015; Vetter et al. 2017b) and even prevented schools from passing through a narrow channel (Murchy et al. 2017). However, there is concern that habituation would occur in naturalistic environments and over longer exposure periods.

In addition to acoustic deterrents, fish can sense carbon dioxide ($CO_2$) through gill receptors (Perry and Gilmour 2002) and exhibit negative chemotaxis when $CO_2$ concentrations exceed certain levels. Bigheaded carps are known to exhibit various aversion behaviors to carbon dioxide including negative chemotaxis (Cupp et al. 2017; Cupp et al. 2018b; Donaldson et al. 2016; Tix et al. 2018). Future studies will need to demonstrate whether changes in carp movement responding to nonphysical deterrents over time are due to habituation, fatigue, or a combination of factors.

Dams and adjacent lock chambers offer pre-existing infrastructural pinch points to reduce the movement of bigheaded carps. A vessel traveling upstream through a lock and dam structure enters the lock through a gate, which then closes behind the vessel. Fish may also enter the lock chamber simultaneously with the vessel. The water level in the lock is adjusted to meet the level of the river upstream, raising the vessel. Then, the upstream gate opens so the vessel and any fish that remain in the lock chamber can pass upstream. Nonphysical deterrents may be able to encourage fish to move out of lock chamber before the downstream gate closes or prevent fish from passing through the downstream gate in the first place (Fig. 1).

In addition to moving through the lock chamber during vessel passage, fish may be able to pass upstream through other parts of a dam structure. During periods of unusually high water in the upstream reservoir or river segment, it may be necessary for infrastructural security or to reduce upstream flooding to open additional spillways that are typically closed during normal river conditions. Two design elements for dam spillways found on the Illinois River are wickets and gates (Coulter et al. 2018). To allow increased water flow past a dam, gates can be raised, and wickets lowered. In gated dam designs, a gate is raised above the water level to allow increased flow. In contrast, slabs called wickets are hinged to a foundation on the river bottom and can be laid flat onto the riverbed. In wicket-style dam designs, wickets are lowered to allow increased water discharge downstream. The design differences of each dam type may influence the ability of fish to pass through the structure.

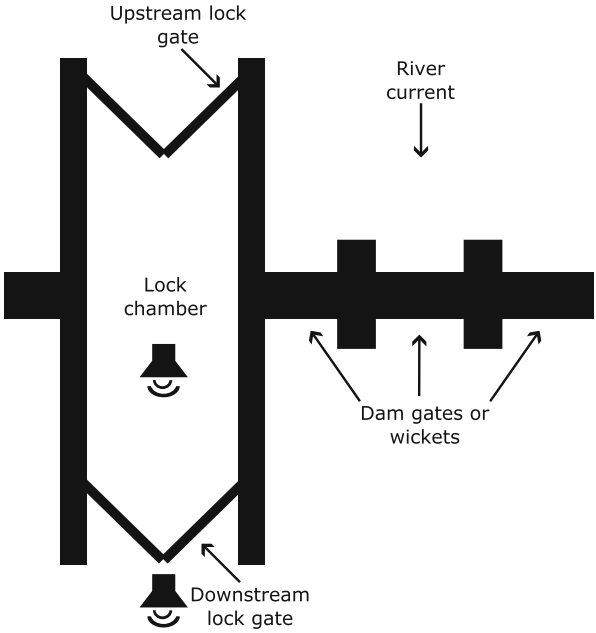



**Fig. 1 Lock and dam structure**. A lock chamber allows vessel and fish passage through the dam. Acoustic deterrents could be placed in the lock to encourage fish to exit before the downstream gate closes or outside the lock to prevent entry. Potential locations for acoustic deterrent placement are indicated by speaker icons

Characterizing current movement trends through physical lock and dam barriers will inform best practices for nonphysical barrier placement. Bigheaded carps were found to travel more frequently through wicket-style dams in the lower Illinois River than through gated-style dams found upriver (Coulter et al. 2018). Some dam designs present higher risk for carp transmission than others and may be better candidates for nonphysical deterrents. In gated-style dams on the Illinois River, bigheaded carps were found to travel through the dam gate more often than through the lock along with transiting vessels (Lubejko et al. 2017). Therefore, it may be useful to place behavioral deterrent systems near dam gates or wickets as well as near lock chambers (Fig. 2).

The majority of carp crossings may occur during open-river conditions when the dam gates are either completely open or overtopped, compared to normal river conditions (Lubejko et al. 2017). Overtopping presents a serious threat to physical barrier efficacy due to the uncontrolled river state during floods. For example, the Starved Rock Lock and Dam (Illinois River, USA) has been overtopped six times between 1933 and 2017 (Lubejko et al. 2017), and this condition is expected to occur more often with increasing precipitation in the Midwest due to climate change (Dai et al. 2016). In addition to strategic deployment near locks or dam gates selected for high risk of carp passage, secondary deployment above dam structures for use during overtopping events may be another potential application of behavioral barriers. In this way, nonphysical acoustic and/or $CO_2$ barriers could potentially extend the effective dam head height for invasive carp during flood conditions. However, future

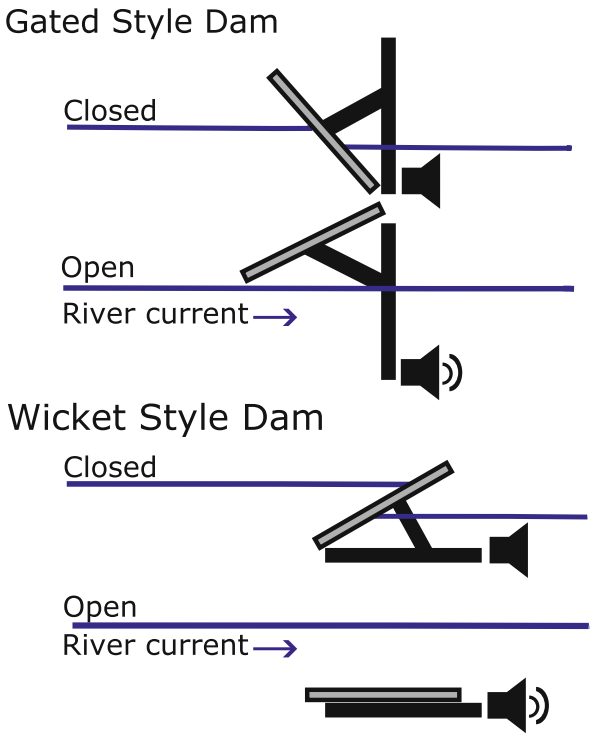



**Fig. 2 Gated- and wicket-style dams.** Fish may be able to pass under raised gated-style dams or over lowered wickets. Acoustic deterrents could be placed near any downstream spillway that is periodically open during periods of high water and activated strategically

research would be necessary to determine the possible efficacy of behavioral barriers placed above dams to reduce invasive fish passage during overtopping. Reaching the dissolved $CO_2$ concentrations necessary to elicit avoidance responses may be difficult at the high-flow speeds associated with open-river conditions. Other deterring stimuli like sound would be less limited by flow rate than dissolved $CO_2$. Like any integrated pest management solution, different modalities that may be limited individually can provide synergies when combined (Barzman et al. 2015). Future research should determine the combination of deterrents that is appropriate for the full suite of possible river conditions at deployment sites.

## Response of Invasive Bigheaded Carps to Nonphysical Deterrents

The current literature has shown bigheaded carps avoid both acoustic signals and $CO_2$, although there are various limitations and strengths to each stimulus. Here, limitations for each are reviewed, and the case for developing an integrated deterrent system using both stimuli is presented. In many cases, the weaknesses of one deterrent, such as lacking species selectivity, are covered by the other. Further,

synergies from combining two stimuli may develop, such as reduced habituation to any one modality alone.

## Acoustic Deterrents

Since accidental introduction in the North American Mississippi River, bigheaded carps have negatively affected resident ecological communities. Reducing range expansion will require finding new methods to block upstream migration. A promising method is the use of acoustic carp deterrents at lock chambers to allow vessel traffic to pass freely through dams (Putland and Mensinger 2019). As ostariophysians, carps have specialized bones called Weberian ossicles that directly connect the swim bladder and inner ear. These bones allow for sound pressure detection, giving the fish a greater bandwidth of frequencies detected (up to 5 kHz, Vetter et al. 2018) than many native species that do not possess morphological specializations thought to aid in enhanced hearing.

The historical distinction between species as hearing generalists and specialists is overly simplistic (Popper and Fay 2011). Instead, a continuum of hearing ability is more appropriate, where interspecific variation in hearing is largely dependent on the specialization of structures to assimilate sound pressure into the sense of hearing with the more basally evolved sense of particle motion (Popper et al. 2021). Fishes without known specialized hearing structures, such as the Weberian apparatus, can generally detect frequencies in the range of approximately 100 Hz to 1 kHz at threshold levels around 78–150 dB re 1 μPa (Ladich and Fay 2013). Comparatively, fishes with hearing specializations generally have much lower sound pressure thresholds for detection and respond at higher frequencies (200 Hz to 3 kHz, Ladich and Fay 2013).

Studies on fish hearing can be completed with behavioral experiments and by measuring auditory evoked potentials (AEP), a noninvasive technique to record electrophysiological activity in response to sound. As determined by AEP using high-frequency sound, bigheaded carps can detect frequencies from 100 Hz up to 5 kHz (Vetter et al. 2018). Since hearing as the perception of sound is both sensory and behavioral, behavioral studies of fish hearing thresholds may be more accurate than electrophysiological experiments alone (Popper and Hawkins 2021; Popper et al. 2019; Sisneros et al. 2016). A behavioral audiogram of bigheaded carps would be useful to develop acoustic deterrents finely tuned to their assimilated sense of hearing, although this data is not available at this time. Regardless of whether current AEP data underestimate the upper range of bigheaded carps' hearing, the high end of bigheaded carps' auditory frequency range exceeds that of most native fishes in the Mississippi River Basin. This sensory difference presents ample opportunity to increase the selectivity of acoustic deterrents for invasive bigheaded carps and minimize the effects on native species by using higher-frequency sound (e.g., 1–5 kHz).

## Species Selectivity

Well-designed acoustic deterrents should affect target invasive species while mitigating off-target effects on native species. The same acoustic deterrent could differentially affect species based on interspecific acoustic sensitivities (Jesus et al. 2019). Therefore, acoustic deterrents composed primarily of frequencies above the hearing range of non-ostariophysans could better target bigheaded carps. However, frequency sensitivities alone may not provide complete species selectivity in application at locks and dams. Some frequency detection overlaps with native species, and bigheaded carps are expected. Other native cyprinids and siluriformes (superorder *Ostariophysi*), as well as clupeids (order *Clupeiformes*), would likely still be able to detect high-frequency signals. In a behavioral screening of native and non-native ostariophysan responses to an acoustic deterrent, bigheaded carps were found to demonstrate repeated negative phonotaxis unlike most native species tested (Murchy et al. 2022). While a native cyprinid, bigmouth buffalo (*Ictiobus cyprinellus*), also responded to the acoustic stimulus, fewer consecutive reactions were observed compared to bighead carp. High-frequency acoustic deterrents may be able to target bigheaded carps' hearing while mitigating off-target effects on native species (Murchy et al. 2022).

Bigheaded carps share key behavioral and invasion history similarities (Kelly et al. 2006), which may allow acoustic deterrents to manage both species simultaneously. However, bigheaded carps are distinctly different from common carp (*Cyprinus carpio*), and studies using common carp as a proxy for bigheaded carp have limited value (Murchy et al. 2016). Silver and bighead carps frequently hybridize in the wild, and post-$F_1$ hybrids are not sterile, raising the possibility in the near future of a hybrid being the dominant genotype over parental genotypes (Lamer et al. 2010). In behavioral response to acoustic deterrents, silver (Vetter et al. 2015) and bighead (Vetter et al. 2017b) carps demonstrate consistent negative phonotaxis from complex sounds, but habituate more quickly to pure tones. To date, most studies have demonstrated similar responses between the two, suggesting they can be co-managed by one broadband sound stimulus. Further, broadband sounds will likely be more effective at providing species selectivity, possibly by reducing habituation to any one frequency or spanning a larger swath of the target species' hearing sensitivity (Jesus et al. 2019).

## Limitations of Acoustic Behavioral Barriers

In general, excess anthropogenic sound has the potential to disrupt native fishes' information about the soundscape, to communicate, find prey, or avoid predation (Slabbekoorn et al. 2010). For example, excess noise could mask native fish acoustic communication by reducing the area in which a fish seeking a mate can be heard by conspecifics. Alternatively, masking could reduce the quality of information communicated such as when a signal recipient can no longer detect and/or discern mate quality due to attenuation of the signal. While these concerns for excess anthropogenic noise are generally valid, they are not especially relevant when considering the application of acoustic deterrents for bigheaded carps. Concern about anthropogenic

masking is spatial scale dependent, and the proposed locations of deterrents are highly localized, such as immediately adjacent to dam or lock chamber gates. Further, the temporal effects of acoustic deterrents are constrained, as they can be instantaneously turned on or off, depending on when dam gates are opened, for example.

Acoustic deterrents could be less effective if too intense or implemented for excessive durations. Long-term exposure to high-intensity broadband sound (24 h, 155 dB re 1 μPa) can induce temporary hearing loss in bigheaded carps (Nissen et al. 2019). This suggests that carp deterrents may render themselves less advantageous if designed to produce sound any longer than necessary or if fish remain trapped in the auditory range of the deterrent. However, Nissen et al. (2019) found that short-term exposure for 30 min at the same sound intensity resulted in fewer significant differences in hearing loss between control and treatment fish than the 24 h treatment. Tested immediately after 30 min sound exposure, bighead carp hearing thresholds were elevated above baseline by ~10 dB at 0.4, 0.6, 1, and 2 kHz. After 96 h of recovery, only the shift at 2 kHz remained. In contrast, a silver carp threshold shift was observed only at 0.6 kHz, which returned to baseline after 96 h of recovery. Thus, acoustic deterrent systems should be evaluated in the field to ensure target species do not undergo temporary threshold shifts that impair their ability to hear the deterrent. It will be necessary to assess whether motile fish in the field voluntarily remain in range of an acoustic deterrent long enough for hearing shifts to occur or if fish vacate an area too quickly for hearing shifts to pose any real risk.

Any behavioral deterrent will only be able to act on motile developmental stages that can chose whether to avoid or challenge the barrier. The movement of propagules like eggs by downstream water flow, or by transport from other species, will not be reduced by acoustic barriers. For example, invasive common carp may be able to colonize new waterbodies via the transport of eggs passing through the guts of waterfowl (Lovas-Kiss et al. 2020). Lovas-Kiss et al. (2020) found that only about 0.2% of common carp eggs survived intact through the mallard gut, as indicated by embryo movement in the intact egg after egression. However, given the large consumption of fish roe by birds, 0.2% survival might be sufficient for colonization. The majority of common carp eggs were egressed within 1 h of feeding, suggesting a maximum dispersal range of 60 km given the flight speed of mallards. Any underwater deterrent system will be limited to dispersal through water, and so airborne vectors are not specific limitations for acoustic deterrents.

Bigheaded carps may habituate to an acoustic deterrent as exposure time increases. Habituation to acoustic deterrents needs further and longer-term study. Few studies have examined habituation in fish at all, and many use different definitions or give no definition of “habituation” (Putland and Mensinger 2019). Some studies have found evidence of diminishing responses after repeated exposure to acoustic deterrents. Murchy et al. (2017) found that the percentage of successful repels away from an acoustic deterrent decreased slightly with consecutive sound presentation over a short time period. However, the methodology was unable to discriminate between habituation to sound and fatigue from repeated simulation. Notably, the repulsion rates remained above 80% after the fifth consecutive

application. This long-term limitation could potentially be mitigated by playing the sound only when deterrence is most critical instead of continually. For example, acoustic deterrents could be turned on as the lock gates are opening and then stopped when the vessel is in the lock with the gate shut. By using an acoustic deterrent only as often as necessary, or by varying the sounds used, the risk of habituation can be reduced.

### Pairing Acoustic Deterrents with Other Modalities

Selectively allowing fish passage requires differentiation based on biological traits such as physical capability, morphology, sensory ability, behavior, or movement phenology (Rahel and McLaughlin 2018). It may prove most advantageous to design deterrents that act on multiple species differences at once. For example, to overcome the limitations of acoustic deterrents such as insufficient efficacy and habituation, it will likely be necessary to pair sound with other stimuli that invasive carp can sense and avoid. For example, a bubble curtain paired with an acoustic deterrent was 95% effective at deterring the movement of bighead carp across a channel (Taylor et al. 2005).

Further, nonphysical barrier systems could simultaneously use a "push-pull" strategy to be even more effective (Cook et al. 2007). Deterring stimuli like acoustics or chemical irritants could be used to "push" individuals away, whereas attractive stimuli like food or pheromones "pull" select organisms toward an area for removal (Cook et al. 2007). Adding multiple modalities to any deterrent system builds redundancy and resiliency into the integrative solution.

## Carbon Dioxide as a Deterrent

A localized area of elevated dissolved $CO_2$ could serve as either a behavioral or physiological barrier to fish passage, in the former by reaching levels of avoidance or in the latter by exceeding homeostatic thresholds for dissolved $CO_2$ in the bloodstream, known as hypercapnia tolerance (Noatch and Suski 2012). Invasive carp becoming immobilized by loss of equilibrium (LOE) from hypercapnia will be pushed downstream by the flow of the river, effectively reducing upstream passage. Moreover, by adding dry ice to frozen-over ponds, the partial pressure of $CO_2$ ($pCO_2$) can be increased to lethal levels for controlled winter fish kills (Cupp et al. 2018a). Silver and bighead carps were more susceptible to this under-ice $CO_2$ application than a related native cyprinid, the fathead minnow (*Pimephales promelas*) (Cupp et al. 2018a). Unlike other chemical additives such as conventional piscicides, $CO_2$ does not pose similar toxicity threats (Noatch and Suski 2012). Further, $CO_2$ diffuses spontaneously out of water. For targeted $CO_2$ removal, water can be cheaply aerated to push dissolved gases out of solution.

Free-swimming bigheaded carps avoid regions of elevated $pCO_2$ (Borland et al. 2020; Cupp et al. 2017; Dennis et al. 2015; Donaldson et al. 2016). In a moving water column, schools of bighead carp decide to spend less time in a $CO_2$-injected flume in preference to a non-injected flume (Hasler et al. 2019). When implementing

a nonphysical gas bubble curtain in outdoor controlled ponds, $CO_2$ injection significantly reduced the number of attempted crossings over the barrier but was not 100% effective at reducing passages (Cupp et al. 2017). Spatial analysis of this dataset reveals carps prefer areas as far away from the $CO_2$ injection site as possible (Borland et al. 2020).

A $CO_2$ barrier will likely be best implemented as part of a multi-modal system utilizing multiple integrated deterrents. By acting on multiple sensory modalities, deterrents will be more easily detected than unimodal systems and elicit heightened behavioral responses (Deleau et al. 2020; Putland and Mensinger 2019). For example, light and sound stimuli could be coupled with the bubbling of $CO_2$ gas into the water column to produce an augmented bubble curtain barrier. It will be necessary to determine which of these modalities is most biologically relevant to target species' sensory systems. One study found invasive common carps primarily use sound and hydrodynamic flow cues when avoiding an air bubble curtain instead of visual signals (Zielinski et al. 2014). When light was added to a coarse-bubble curtain, deterrent efficacy decreased significantly from 87% to 75%, opposite to the expected direction if light were perceived as a visual deterrent. Instead, the common carp may have used the light to visually identify and exploit gaps in the curtain (Patrick et al. 1985). Again, however, caution must be used in extrapolating results from common carp to bigheaded carp.

### Limitations to Carbon Dioxide as a Nonphysical Barrier

A pattern of negative chemotaxis is widely observed across fish species and size classes; therefore, $CO_2$ barriers implemented alone will not afford species-level selective permeability (Dennis et al. 2015; Donaldson et al. 2016; Kates et al. 2012; Rahel and McLaughlin 2018). Juvenile bighead and silver carp, bluegill, and largemouth bass show similar aversion behaviors as well as RNA transcriptome stress responses to carbon dioxide exposure (Dennis et al. 2015). Irregular activities such as surface ventilations, twitching, and equilibrium loss also increase across these four species with $pCO_2$ (Kates et al. 2012).

There is high intra-species variation in $pCO_2$ avoidance thresholds (Kates et al. 2012), and deterrent systems will likely need to select for the upper end of these thresholds. Studies testing individuals repeatedly may explain some of this intra-species variation. Tolerance to $CO_2$ was found to be repeatable within individual largemouth bass (*Micropterus salmoides*), as evidenced by 60% of the variation in time to equilibrium loss being explained by inter-individual differences (Hasler et al. 2017). Therefore, $CO_2$ tolerance may be a heritable trait subject to natural selection, although further work will be necessary to determine if this is the case (Hasler et al. 2017). Notably, inter-individual differences are not the standard test for heritability; rather, changes across generations are, which was not captured in the methodology.

If anthropogenic $CO_2$ deterrents provide a positive selective force for hypercapnia tolerance, populations could potentially evolve increased tolerance (Hasler et al. 2017). This would render $CO_2$ deterrents less effective on generational timescales. However, it is reasonable to expect that the level of dissolved $CO_2$ could be quickly adjusted if any meaningful tolerance to $CO_2$ evolved in invasive populations. Future

research will be necessary to establish whether $CO_2$ tolerance is heritable and whether $CO_2$ deterrents pose meaningful selective pressure at relevant timescales. Future studies should be completed in both native species and target invasive species, as not all species would be expected to undergo equal selective pressure from a given deterrent. For example, if a native species evolves $CO_2$ tolerance as suggested is possible by Hasler et al. (2017), but an invasive species does not, $CO_2$ barriers might affect the deterrent's species selectivity in a beneficial direction.

Management actions are not without risk; a poorly implemented barrier of any type would allow the passage of undesirable species while blocking desirable species and may result in worse outcomes than an unaltered river channel (Rahel and McLaughlin 2018). If implemented alone, a $CO_2$ barrier would likely result in a similarly negative outcome due to the indiscriminate deterrence of all fish species. Increased fragmentation of native fish habitat is a potential cost to consider in tradeoff with reducing bigheaded carps' range expansion. For this reason, $CO_2$ may be best deployed as part of a multi-modal deterrent system only when other sensory modalities provide sufficient species-level selectivity and $CO_2$ is not continually deployed (Deleau et al. 2020; Putland and Mensinger 2019).

In contrast, an especially well-designed barrier will be able to alter upstream and downstream movement independently. The Illinois River flows downstream from Lake Michigan, requiring bigheaded carps to swim upstream to colonize the Laurentian Great Lakes. Modeling of bigheaded carp movement in the Illinois River reducing both upstream and downstream carp movement together suggests populations may increase near the invasion front if fish are not allowed to travel back downstream (Coulter et al. 2018). That is, downstream bigheaded carp movement in the Illinois River exceeds upstream movement, such that if barriers are deployed that reduce bidirectional travel equally, carp populations may increase upstream at the invasion front (Coulter et al. 2018). It will likely only be necessary to place deterrents at downstream lock gates instead of upstream ones. Finally, a $CO_2$ barrier will only be effective when mobile organisms can choose to avoid elevated zones, and it would not deter downstream movement of propagules (Kates et al. 2012). In the Illinois River, $CO_2$ barriers would primarily attempt to reduce upstream passage of mobile adult fish, as downstream propagule movement would be less critical to preventing colonization of Lake Michigan.

## Adverse Effects in Native Species

Fathead minnows, but not silver carp, were found to lose olfactory predator avoidance behaviors when exposed to prolonged $pCO_2$ over multiple hours (Tix et al. 2017). Therefore, the impairment of olfactory ability by $CO_2$ may be species specific. An unintended reduction in predator avoidance in native species, but not invasive species, would oppose the conservation goal to reduce invasive populations' interspecific competitive advantages. However, Tix et al. (2017) did not control for the concentration of skin extract alarm cue used across species; thus, there is not yet definitive evidence that $CO_2$ barriers are more harmful to a native species than these target invasive species. Notably, most fathead minnow predator avoidance behaviors returned to baseline after recovery in ambient water, and

free-swimming native fish will likely move away from $CO_2$ barriers before olfactory impairment occurs (Tix et al. 2017).

Since $CO_2$ is toxic to all fish, it may weaken undesirable individuals that challenge a nonphysical barrier. However, similar effects are also likely to occur in non-target species. Elevated $pCO_2$ induces internal hypercarbia and LOE in fish when blood concentrations exceed physiologically tolerable levels, potentially rendering them unable to swim through a gas bubble curtain (Cupp et al. 2020). Weakened fish may not be able to swim through the high-velocity water found in partially open gated dams if weakened by $CO_2$, for example. In contrast to this suggested toxicity effect on passage through a barrier, elevated $pCO_2$ did not impair largemouth bass burst swimming speed and is not expected to impair the movement of similarly sized fish such as bigheaded carps; however, 25–50% of largemouth bass experience LOE when exposed to $CO_2$ concentrations of approximately 100,000 μatm for 120–600 seconds (Schneider et al. 2019). If concentrations and exposure durations of this magnitude are achievable in the field, inducing LOE could reduce the passage of all fish species. Studies on burst speeds in target invasive species need to be conducted, since the assumption that similarly sized largemouth bass will be equally impaired may be inaccurate.

Except for Tix et al. (2017), no study has found serious adverse effects of $CO_2$ in native riverine fish species that were not shared with bigheaded carps (e.g., LOE at elevated concentrations). However, full characterization of species richness, abundance, and susceptibility to $CO_2$ will be necessary to ensure application is appropriate for management goals before field installments of chemical deterrents (Cupp et al. 2020).

### Effects of Water Temperature and Fluid Dynamics

Rivers are not static systems. Moreover, any potential barrier site will have idiosyncratic environmental properties, and nonphysical barriers must therefore be effective under a variety of field conditions. Temperature, barometric pressure, pH, $CO_2$ uptake via photosynthesis, and hydrodynamics will introduce temporal variation in local $pCO_2$, and field testing will be necessary to characterize these conditions at any potential application site (Cupp et al. 2018b). Aerating downstream from a deterrent to increase $CO_2$ off-gassing could reduce risk to non-target species if field testing suggests there may be residual downstream effects (Cupp et al. 2018b). That is, the zone of elevated $CO_2$ can be highly spatially controlled with simple technology like air bubble barriers to mitigate negative effects on native species or at a minimum reduce the impacted area of the river system.

The amount of $CO_2$ used to deter bigheaded carps will likely need to be higher in summertime than in winter due to temperature-dependent effects on voluntary avoidance behavior; bighead carp avoided $CO_2$ at 41% lower concentrations in cold water compared to warm water (Tix et al. 2018). This likely occurs because cold water can hold more dissolved gas than warm water; water and gas concentration are inversely related. Fish metabolism is slower in cold temperatures, and both intake and elimination of $CO_2$ may retard at low water temperatures (Cupp et al. 2020). Studies should use both seasonally appropriate and physiologically optimal

temperatures when designing selectively permeable barriers based on trait differences between species (Rahel and McLaughlin 2018).

## Other Deterrence Methods

In addition to acoustic and $CO_2$ deterrence, other methods have been investigated to reduce the spread of bigheaded carps. Electrifying lock chambers has been proposed as one method to stop fish movement, but concerns about juvenile fish being able to travel through voltage fluctuations in the electrical barrier remain (Davis et al. 2017). Bubbling air into a curtain underwater creates a nonphysical barrier that has been found to substantially reduce bigheaded carp movement (Zielinski and Sorensen 2016). The process of bubbling air underwater inherently creates sound that may partially explain fish deterrence. Further, bubble curtains have been ensonified and illuminated to increase the modalities of deterrents with varied effects (Zielinski et al. 2014). It almost certainly is necessary to combine multiple deterrent methods to reduce the risk of invasive fish passage down to acceptable levels.

## Discussion

Acoustic deterrents have been proposed to reduce the upstream range expansion of bigheaded carps. As ostariophysians, these invasive fishes have hearing sensitivity in higher-frequency ranges than most native fishes, and so acoustic deterrents may afford species-level selectivity (Murchy et al. 2022). Most nonphysical deterrents will only be able to reduce the upstream passage of motile life stages of invasive populations, and acoustic deterrents will not provide an exception to this limitation. Concerns about habituation to repeated playback of acoustic stimuli will need to be further examined. To reach deterrent efficiencies suitable to prevent invasive species range expansion across long-term timescales, it may be necessary to combine acoustic deterrents with other sensory modalities.

While the specific application of carbon dioxide as an under-ice piscicide has been established as plausible and highly effective (Cupp et al. 2018a), its infusion through bubble curtains has been additionally proposed as a nonphysical behavioral barrier to bigheaded carps by effectively controlling their movement. However, $CO_2$ will require integration with additional stimuli as a behavioral deterrent in the field. In flowing rather than still water, bighead carps demonstrate avoidance behaviors like lateral switching between flumes of high and low dissolved $CO_2$ (Hasler et al. 2019). When considering the application of $CO_2$ plumes in locks and dams as a nonphysical deterrent, inter-species differences will not be sufficiently selective. Native and invasive species respond to elevated $pCO_2$ plumes in similar ways, including behavioral avoidance responses and molecular stress biomarkers (Dennis et al. 2015; Kates et al. 2012). Used in a barrier alone, $CO_2$ will not provide species specificity to target invasive species over native species. Additionally, it is reasonable to expect adding $CO_2$ to a river system would physiologically stress native

species (Dennis et al. 2015). The decision to implement a $CO_2$ barrier in the field will rely on determining whether the possible risks to native species outweigh the potential benefits of reducing the probability of invasive carp range expansion.

As bigheaded carps are highly fecund, if even a few individuals can overcome a barrier due to their exceptional intraspecific abilities, they may be able to produce a strong second generation. Therefore, deterrents should be designed to exclude even the most exceptional individuals of a target species (Rahel and McLaughlin 2018). Due to high inter-individual variance in $pCO_2$ tolerance (Kates et al. 2012), it may be necessary to design deterrents at the upper levels of $CO_2$ tolerance. However, care should be taken to determine whether selective pressure exerted by the deterrent will undermine its efficacy in future generations (Hasler et al. 2017) and prevent this if so.

Simulations suggest that to prevent invasion and establishment of a breeding population of bigheaded carps in new waterbodies, individual nonphysical barrier efficacy must be very high ($> 90\%$), and further, a series of high-efficacy deterrents may be necessary (Putland and Mensinger 2019). Carbon dioxide yields consistent negative chemotaxis in target invasive species and thus has potential to increase the efficacy of nonphysical barriers (Borland et al. 2020; Cupp et al. 2017; Cupp et al. 2018b; Hasler et al. 2019; Tix et al. 2018). However, because most fish species and size classes avoid this dissolved gas, it will not afford inherent species-level selectivity (Dennis et al. 2015; Donaldson et al. 2016; Kates et al. 2012; Rahel and McLaughlin 2018).

Integrating multiple deterrents may reduce habituation to any one deterrent, provide redundancy, increase species selectivity, and improve overall efficacy (Noatch and Suski 2012). Carbon dioxide would likely best be implemented as one of the multiple tools used to prevent bigheaded carps' range expansion, where other modalities such as acoustics provide species selective permeability. Further research comparing the off-target effects on native species of individual and integrated deterrent systems will be necessary before deployment at field sites to optimize deterrent efficacy while minimizing costs to native ecologies. Fully characterizing the effects of elevated $CO_2$ at implementation sites will be required to mitigate potential adverse impacts on native species such as loss of equilibrium or reduced predator avoidance behaviors (Schneider et al. 2019; Tix et al. 2017). No nonphysical barrier technology can deter 100% of fish indefinitely; however, increasing the efficacy of nonphysical barriers will be necessary to prevent the spread of invasive fish species between connected waterbodies (Noatch and Suski 2012). Due to shared avoidance and toxicity with target bigheaded carps as well as non-target native species, $CO_2$ should be limited to multi-modal applications where other modalities allow for species selectivity.

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# Spatial Impact of Wind Farm Constructionon Harbor Porpoise Detectability

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**Abstract**

To combat impacts of climate change, offshore wind farm construction is accelerating globally. Accompanying increases in construction activity have raised concerns about associated noise impacts on acoustically sensitive species, including harbor porpoises (*Phocoena phocoena*). This study investigated spatial impacts from the construction of ScottishPower Renewables' East Anglia ONE (EA1) offshore wind farm (310 pin-piles), located off southeast England, on porpoises belonging to the North Sea Management Unit. Data on porpoise presence and received noise levels were collected from up to 12 monitoring stations around EA1 before, during, and after construction from March 2018 to June 2019, using automated click detectors (C-PODs) and full bandwidth acoustic recorders. Acoustic recordings were used to calibrate a sound propagation model to assess porpoise responses to received noise levels. Analysis of porpoise detection data in the presence and absence of pin-piling revealed negative effects of piling, with an overall decrease in porpoise detection probability out to 14.0 km from piling activity. The predicted frequency-weighted received level at this distance was 103.0 dB re 1 $\mu Pa^2s$ SEL. These observations inform our current knowledge of pin-piling-related impacts on harbor porpoises at the spatial scale of the individual wind farm, which is crucial to develop effective mitigation strategies.

**Keywords**

*Phocoena phocoena* · Pin-piling · Disturbance · Displacement · Effective deterrence range (EDR) · Passive acoustic monitoring (PAM) · Offshore wind energy

## Introduction

Cetaceans, such as harbor porpoises (*Phocoena phocoena*), are known to be sensitive to anthropogenic noise pollution, including noise produced by offshore construction activities involving piling (e.g., Brandt et al. 2016; Carstensen et al. 2006; Teilmann and Carstensen 2012). Considerable amounts of piling related to the continued expansion of the offshore wind energy sector are either ongoing or forecast to occur over the next decade in many parts of the world. In Europe,

activities are continuing to ramp up throughout the North Sea, which has long been a global epicenter for offshore wind deployment (Cruciani 2018).

However, there are concerns about auditory injury, masking, and behavioral disturbance imposed upon animals from this activity, particularly when considering the cumulative impacts of overlapping construction projects and noise from other industries such as shipping and oil and gas exploration. Previous studies have shown temporary reductions in porpoise presence and vocalization rates (indicating behavioral disturbance and/or changes in distribution) around marine construction sites, with reduced impacts at increasing distances from the noise source (e.g., Brandt et al. 2011, 2018; Dähne et al. 2013; Tougaard et al. 2009). Impacts are particularly problematic when they occur inside a protected area such as a Special Area of Conservation (SAC), especially when site-specific conservation objectives must also ensure that disturbance of porpoises "*from a significant portion of the designated site, for a prescribed period of time*" (JNCC 2020) is avoided.

In the UK, several Statutory Nature Conservation Bodies responsible for SAC management (Joint Nature Conservation Committee (JNCC), Natural England, and the Department of Agriculture, Environment and Rural Affairs (DAERA) in Northern Ireland) have specified noise disturbance within a harbor porpoise SAC from noise-generating activities, individually or in combination, to be "significant" if it excludes porpoises from more than:

1. 20% of the relevant area of the site, or any given day.
2. An average of 10% of the relevant area of the site over a season (JNCC 2020; based on ASCOBANS 2000).

Under this approach, there is a need to estimate the extent of the area around a wind farm construction site from which harbor porpoises are temporarily excluded. Undertaking regular vessel-based and/or aerial surveys for porpoises during construction to record spatial displacement due to disturbance is expensive and logistically challenging, and is not demanded of developers by the UK regulator. There is also no current UK requirement for passive acoustic monitoring (PAM) during wind farm construction and operation, such as has been proposed for new wind farm developments in the United States (van Parijs et al. 2021), although some developers have undertaken acoustic monitoring studies on their own accord (present study; Benhemma-Le Gall et al. 2021; Graham et al. 2023). Such data allows valuable *post-hoc* assessments of impacts on sensitive species, such as harbor porpoises.

In the UK, assessing the range of temporary habitat loss within harbor porpoise SACs due to construction-associated noise outputs is achieved by applying fixed "Effective Deterrence Ranges" (EDRs) for different types of marine construction activities (Table 1; JNCC 2020). Here, EDRs are determined based on available empirical evidence of significant reductions in porpoise vocal activity or sightings as reported in the scientific literature, rather than estimated based on predictive noise modelling. The latter was deemed by JNCC (2020) to be insufficiently precise due to: (1) lack of agreed quantitative thresholds for disturbance, (2) predictive outcomes dependent on choice of sound propagation models, (3) a lack of understanding of

**Table 1** Recommended Effective Deterrence Ranges (EDRs; Adapted from JNCC 2020)

| Activity | EDR (km) | References on which EDRs were based |
|---|---|---|
| Monopiling | 26 | Dähne et al. 2013; Tougaard et al. 2013 |
| Monopiling with noise abatement | 15 | Dähne et al. 2017; Rose et al. 2019 |
| Pin-piling (with and without noise abatement) | 15 | Graham et al. 2019 |
| Conductor piling for oil and gas wells | 15 | Graham et al. 2019; Jiang et al. 2015; MacGillivray 2018 |
| Unexploded ordnance detonation | 26 | Based on monopile EDR |
| Seismic (air gun) survey | 12 | Sarnocińska et al. 2020; Thompson et al. 2013 |
| Other geophysical surveys | 5 | Crocker and Fratantonio 2016; Crocker et al. 2019 |

relevant acoustic properties causing disturbance, and (4) lack of behavioral context (JNCC 2020).

JNCC (2020) recognized that their specified EDRs are based on a very limited number of scientific studies, and that they are intended to provide generic advice to cover a range of different types of construction activities. Additionally, JNCC (2020) recommended periodic evaluation of EDRs to consider new available data.

## Current Study

Underpinned by PAM data and propagation modelling, this study aims to quantify the spatial scales over which porpoises were being displaced during construction of ScottishPower Renewables' (SPR) East Anglia ONE wind farm (hereafter referred to as EA1), involving a total of 310 pin-piles with a diameter of 2.5 m. The development-specific "pin-piling impact range" is defined here as the distance (in km) within which porpoise presence (inferred from acoustic detection of vocalizations) is affected negatively by pin-piling activity at EA1 throughout the duration of the construction phase. Additionally, the study aims to obtain the porpoise frequency-weighted received level, calculated as Sound Exposure Level (SEL; in dB re 1 $\mu Pa^2s$), associated with this "pin-piling impact range."

This work was part of a larger project that sought to clarify consequences of construction of the EA1 offshore wind farm on the porpoise population in the North Sea Management Unit (IAMMWG 2022). Specifically, this larger project involved modelling population-level impacts of EA1 pin-piling using the interim Population Consequences of Disturbance (iPCoD; King et al. 2015) and DEPONS (Nabe-Nielsen et al. 2018) population impact modelling frameworks. Full methodological details and outcomes of this project were reported by van Geel et al. (2023a, b, c).

## Methodology

Data analyzed in this study were gathered by a PAM array, which was deployed at the EA1 offshore wind farm, located in the southern North Sea (Fig. 1), within the Southern North Sea SAC designated for porpoises (JNCC and NE 2019).

Operational since summer 2020, the wind farm contains 102 Siemens Gamesa wind turbines with a combined rated capacity of 714 MW (SPR 2021). Construction started in April 2018 and continued until January 2019 (involving 310 pin-piles in total), with unexploded ordnance (UXO) clearing operations by high-order detonation continuing until June 2019. Full pin-piling and UXO detonation schedules were provided by SPR; no pin-piling occurred at multiple piles simultaneously.

## Data Collection

Spatiotemporal presence of porpoises within and around EA1 was monitored using a PAM array from the 11 March 2018 to 21 June 2019, involving eight consecutive deployments. The array of 12 monitoring stations, each containing a C-POD automated porpoise click detector (Chelonia Ltd.), was deployed in and around the EA1 site to monitor porpoise echolocation activity (Fig. 1). C-PODs were configured using default settings, except that devices were programmed to record at all angles relative to vertical. Up to six of these monitoring stations also contained RTSYS EA_SDA14 full bandwidth (FBW) acoustic recorders, connected to either a SPARTON PHOD-1 or a RESON TC4014 hydrophone, used to calibrate the transmission loss model. Lost FBW recorders were not replaced, resulting in a reduction in the numbers of monitoring locations equipped with these recorders during later

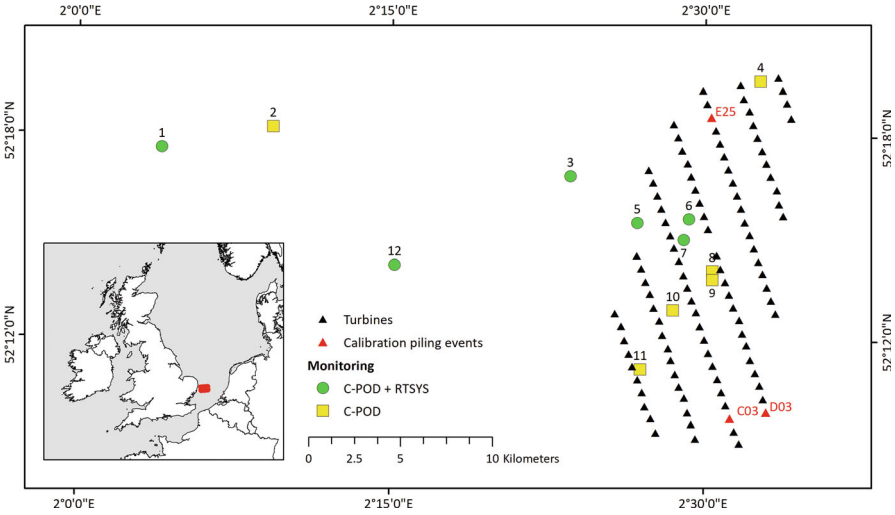



**Fig. 1** Location of the SPR East Anglia ONE wind farm, with positions of individual turbines (black triangles) and acoustic monitoring stations collecting C-POD data only (yellow squares), or both C-POD and full bandwidth data (green circles). Data for calibration of the propagation model were collected during the construction of turbines C03, D03, and E25 (red triangles). European country shapefile data were obtained from European Commission, Eurostat, GISCO @EuroGeographics and UN-FAO (https://ec.europa.eu/eurostat/web/gisco/geodata/reference-data/administrative-units-statistical-units/countries)

deployments. FBW recorders were programmed to collect data at a sampling rate of 156 kHz, adhering to a 10/14 h on/off duty cycle during the first deployment, followed by a 6-h on/off duty cycle in subsequent deployments.

## Data Processing

Following recorder recovery, C-POD and FBW data were checked for issues (e.g., corrupted files, data gaps, and clipping in FBW recordings) before proceeding with in-depth analysis. C-POD data were processed using the proprietary analytical software CPOD.exe (Version 2.044; Chelonia Ltd.), applying the KERNO Narrow-Band High Frequency (NBHF) species classifier and the "Hi" (high) and "Mod" (moderate) train filter detection settings. Data were exported as Porpoise-Positive Minutes per hour. Note that PAM only provides information on acoustic presence of animals. An absence of acoustic detections may indicate an absence of animals, animals not vocalizing, or both.

The C-POD preset recording limit of 4096 clicks per minute was regularly exceeded due to high levels of non-cetacean click-like noise (from movement of waterborne sediment, anthropogenic activities etc.), resulting in loss of monitoring time until the onset of the next minute. Sections of data where C-PODs had logged clicks $\leq$90% of each hour were excluded from further analysis. Processing of the FBW data is described in the next section.

## Propagation Modelling

FBW data, and consequently measured received levels, were not available for each C-POD location (as FWB data were collected at six of the 12 monitoring stations), nor did it they match the full C-POD monitoring period (as FBW recorders operated on a duty cycle, and due to their overall shorter monitoring capability during individual deployments compared to C-PODs). Consequently, a site-specific transmission loss model was developed to estimate received noise levels at the locations of all C-PODs for each pin-piling event, so that these could be related to porpoise presence. This model integrated two techniques for modelling the propagation of underwater noise, each of which was appropriate for specific frequency ranges. For each of these techniques, the transmission loss model incorporated local bathymetry, sediment type, sound attenuation in seawater, piling location, piling schedule, and hammer energetics.

At frequencies up to 3 kHz, the parabolic code RAMSGeo (from the open-source AcTUP suite of codes) was used, but this becomes numerically inefficient as frequency increases. Instead, a ray tracing code, using the commercially available software package COMSOL Multiphysics (COMSOL Inc.), was used at higher frequencies. A sensitivity analysis was conducted to ensure results were not dependent on the frequency at which the modelling approach changed, with transition frequencies between 2 and 5 kHz giving similar results. The RAMSGeo formulation

includes parameters relating to seabed characteristics reflecting the complex interaction between the water column and seabed at low frequency, while the seabed was modelled as an impedance boundary in the ray tracing model. Material properties of water and seabed used in the models are summarized in Table 2. The seabed was modelled as consisting of sandy sediment; its impedance condition was based on a sediment density of 2040 kg/m$^3$ and speed of sound of 1720 m/s (LeBlanc et al. 1992). In shallow water, the water and seabed can form a waveguide that prohibits propagation of low frequency noise (Urick 1983). For the given speed of sound in water and the seabed, frequencies below 40 Hz cannot propagate.

The propagation model was calibrated using acoustic measurements of selected FBW data (>10 min in duration) of three separate pin-piling events from July 2018 (Fig. 1; Table 3). Data selection for this calibration was based on the following criteria:

- Continuous simultaneous recordings from all six recorders.
- Closest sensor at sufficient distance from pin-piling events, such that the acoustic signal did not exceed hydrophone sensitivity (i.e., signal not clipped).
- Recordings not contaminated by noise events unrelated to piling, such as surface vessel noise or UXO detonations.
- Recordings included short piling breaks to allow synchronization of acoustic signals of individual hammer strikes recorded at different sensors to account for different arrival times.

**Table 2** Properties of water and the seabed used in the models. Temperature, salinity, and pH were used to determine the acoustic absorption of sea water, following the method of Ainslie and McColm (1998)

| | Speed of sound (m/s) | Density (kg/m$^3$) | Temperature (°C) | Salinity | pH |
|---|---|---|---|---|---|
| Sea water | 1500 | 1026 | 15 | 35 | 8 |
| Seabed sediment | 2040 | 1720 | – | – | – |

**Table 3** Details of pin-piling events used to calibrate the acoustic propagation model. See also Fig. 1 for corresponding wind turbine and monitoring station locations

| Wind turbine | Maximum hammer energy (kJ) | Start of recording (UTC) | End of recording (UTC) | Distance from monitoring station (km) | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | 01 | 03 | 05 | 06 | 07 | 12 |
| C03 | 737 | 2018/07/13 23:27 | 2018/07/13 23:37 | 34.24 | 15.81 | 11.77 | 11.08 | 10.05 | 20.08 |
| D03 | 643 | 2018/07/24 21:14 | 2018/07/24 21:25 | 35.90 | 16.71 | 12.47 | 11.33 | 10.42 | 21.77 |
| E25 | 617 | 2018/07/22 09:43 | 2018/07/22 09:55 | 29.91 | 8.22 | 7.02 | 5.67 | 6.84 | 19.09 |

For the selected data, peak sound pressure level (peak SPL; $L_{p,\mathrm{pk}}$; in dB re 1 µPa) was computed for 35 1/3-octave bands (base 10; also referred to as decidecade bands) with a nominal central frequency ranging from 25 Hz to 63 kHz over a 1-s window using a customized MATLAB script.

In these selected pin-piling events, piling noise was distinguishable from ambient sound at frequencies up 16 kHz, and as such, broadband noise levels were computed for the 40 Hz–16 kHz 1/3-octave band frequency range. The transmission loss model was applied to predict source level of pin-piling at each turbine location, expressed as SEL ($L_{E,p}$; in dB re 1 µPa$^2$s at 1 m; based on maximum hammer energy), as well as frequency-weighted (for porpoises) received levels based on Southall et al. (2019) and NMFS (2018) at the C-POD deployment locations during construction of each EA1 turbine. The overall frequency-weighted received level below which porpoises were negatively affected by EA1 pin-piling was subsequently derived by using the obtained relationship between distance from piling activity and modelled frequency-weighted received level.

## Harbor Porpoise Presence Modelling

Porpoise presence modelling was undertaken to assess differences in the probability of detecting porpoises in relation to distance from individual pin-piling events, when compared to their distribution in the absence of pin-piling activity. Predicted porpoise presence at increasing distances from pin-piles was modelled to quantify the "pin-piling impact range" for the entire construction phase. Two separate C-POD datasets, covering piling and non-piling periods, were created for this purpose. The piling dataset contained all hours during which pin-piling activity took place. The non-piling dataset consisted of all non-piling days, except for the exclusion of the two calendar days prior to or following a piling day. Both model datasets were analyzed at an hourly temporal resolution. As mentioned previously, only those hours that were effectively monitored >90% of the time were retained in either dataset. Statistical modelling was undertaken in R (Version 4.0.2; R Core Team 2020) based on the "*mgcv*" package (Wood 2017a) using a binomial Generalized Additive Model (GAM) with a logit link function. The binary response variable was defined as porpoise presence/absence (1/0; Table 4). The GAM models for both the piling and non-piling datasets were fitted via Restricted Maximum Likelihood (REML), with model selection based on Akaike's Information Criterion (AIC) values. During the statistical modelling process, the covariate "Distance" was always retained as this was required for the overarching project objectives (van Geel et al. 2023b, c – see also the "Introduction" section).

Numerous studies have provided evidence that porpoise spatiotemporal distribution is related to environmental factors varying in space and time (e.g., Benjamins et al. 2017; Booth et al. 2013; Embling et al. 2010; Marubini et al. 2009; Skov and Thomsen 2008; Williamson et al. 2017). Porpoise presence in the EA1 area, as in the wider southern North Sea, varies both seasonally and annually in response to environmental changes (Gilles et al. 2016; Heinänen and Skov 2015; Paxton et al. 2016).

**Table 4** Overview of candidate parameters considered in the statistical modelling

| Parameter | Description | Values[b] | Modelled |
|---|---|---|---|
| **Response variable** | | | |
| HP presence | Hourly presence/absence of porpoise detections of "Hi" or "Mod" quality | 0 = Absent<br>1 = Present | Response |
| **Covariates of main interest** | | | |
| Distance | Linear distance (in m) between each piling event and each C-POD deployment site | 1179–35,912 | Linear; smooth |
| Received level | Measured frequency-weighted received sound level (in dB re 1 $\mu Pa^2$s SEL; calculated over the 40 Hz – 16 kHz 1/3-octave band frequency range) at C-POD location | 94.1–194.5 | Linear; smooth |
| **Other covariates** | | | |
| Deployment | Identifier of successive deployments | 1–8 | Random effect |
| Location | Identifier of the deployment location | 1–12 | Random effect |
| Julian day | Julian day | 4–365 | Cyclic smooth |
| Month | Calendar month | 1–12 | Cyclic smooth |
| Year | Calendar year | 2018; 2019 | As factor |
| Hour | Hour of the day (in UTC) | 0–23 | Cyclic smooth |
| Diel cycle | Temporal position in the diel cycle; related to changes in daylight levels | 0 – <1<br>0 = 1 = Sunrise<br>0.5 = Sunset | Cyclic smooth |
| Tidal cycle | Temporal position in the tidal cycle; related to changes in the tidal cycle (water level and tidal flow speed) | 0 – <1<br>0 = 1 = High tide<br>0.5 = Low tide | Cyclic smooth |
| Lunar cycle | Temporal position in the tidal cycle; related to changes in changes in spring-neap tidal cycle (water level and tidal flow speed) | 0 – <1<br>0 = 1 = Spring tide<br>0.5 = Neap tide | Cyclic smooth |
| Project day | Numbered sequence of days from the beginning of the project | 1–468 | Linear; smooth |
| Cumulative piling day[a] | Numbered sequence of piling days throughout the project | 1–136 | Linear; smooth |
| Consecutive piling day[a] | Numbered sequence of consecutive piling days without a break (i.e., calendar day without piling) | 1–13 | Linear; smooth |
| Hammer energy[a] | Maximum hammer energy (in kJ) | 475–1169 | Linear; smooth |
| Source level[a] | Modelled frequency-weighted piling source level (in dB re 1 $\mu Pa^2$s SEL) | 178.3–182.2 | Linear; smooth |
| UXO day | Presence/absence of UXO detonation (or attempt) that day | 0 = Absent<br>1 = Present | As factor |

(continued)

**Table 4** (continued)

| Parameter | Description | Values[b] | Modelled |
|---|---|---|---|
| UXO hour | Presence/absence of UXO detonation (or attempt) that hour | 0 = Absent<br>1 = Present | As factor |
| C-POD number | Individual C-POD ID number | 825–3199 | Random effect |
| Angle | Angle (in °) describing C-POD deviation from vertical | 0–84.5 | Linear; smooth |
| Number of clicks | Average number of unfiltered clicks recorded per minute within an hour; index of noise levels | 0–4095 | Linear; smooth |

[a] Parameter only used in development of piling model
[b] Values across the piling and non-piling datasets prior to data exclusions

For both GAM models, a range of covariates were considered describing parameters known or suspected to influence porpoise spatiotemporal distribution, such as diel cycle, tidal cycle, and lunar cycle (Table 4). Sunset and sunrise times were obtained using the sun-methods function of the R "*maptools*" package (Bivand and Lewin-Koh 2020) for monitoring location 08 centered in the EA1 offshore wind farm. Tidal information was extracted from POLTIPS tidal prediction software (Version 3.9.0/16; National Oceanography Centre). In addition, for the non-piling dataset, distances for each monitoring location (the "Distance" covariate; Table 4) were randomly subsampled from the distribution of "wind turbine – monitoring station" distance combinations in the piling dataset. Finally, the piling GAM model also contained various other covariates related specifically to pin-piling activity, namely modelled frequency-weighted source level (see "Propagation Modelling" section), maximum hammer energy, and consecutive number of pin-piling days, which were not relevant to the non-piling model. Information underpinning the various piling-related covariates was provided by SPR.

Initial data exploration included testing for concurvity; covariates were excluded from further modelling when the worst-case concurvity estimates exceeded 0.8. The model fitting approach broadly followed the method outlined in Pirotta et al. (2011). Numeric covariates were optimized in terms of how to best incorporate them into the model (in linear form or as a smooth term), based on AIC values. A full model, including all potential covariates in their optimal form, was then created and stepwise backward model selection was applied to identify the best subset of covariates. Remaining covariates were subsequently included in the model in declining order of importance (i.e., the first covariate caused the greatest increase in AIC if excluded from the model). Significance of each covariate was tested using the Wald's test (using a significance threshold of $\alpha = 0.05$), and nonsignificant covariates were excluded from the model. Finally, the parameter $k$-values (related to the degrees of freedom allocated to a smooth) were optimized in a stepwise process, at each stage excluding those covariates that had become nonsignificant as a result of alteration of the $k$-value.

Following final model selection, posterior simulation was undertaken (cf. Wood 2017b). This allowed for quantification of trends in the probability of porpoise

presence as a function of distance from piling activity (or frequency-weighted received level; see the "Discussion" section), while taking the values of other model covariates, and their uncertainties, into account. Additionally, this approach allowed predictions to be made over covariate value combinations not present in the original dataset.

The prediction matrix was generated over a prediction grid representing ~740,000 (piling) and 1.5 million (non-piling) possible combinations of values for the covariates that were included in the fitted models. This matrix mapped model parameters to the predictions of the linear predictor for any supplied covariate values. Next, 500 replicate parameter vectors were sampled randomly from the posterior distribution of the parameters by taking 500 multivariate normal samples from the posterior distribution of the model coefficients. These parameter vectors, containing coefficients for the model intercept and each model term, were multiplied with the prediction matrix, and the inverse link function was applied to generate predictor values on the response scale (i.e., probability of acoustic porpoise presence) for the given set of covariate values specified in the prediction grid. The median predicted smooth and associated 95% confidence intervals for the relationship between distance and the probability of porpoise presence for the piling and non-piling datasets were visualized using the R package "*ggplot2*" (Wickham 2016). Model outcomes were compared to determine at what distance from piling the median probability of porpoise detection in the piling model achieved parity with the median porpoise detection probability in the non-piling model. This made it possible to establish the perimeter of the piling impact range around the pin-piling activity at EA1.

## Results

### Data Availability

Usable C-POD data were collected during 86 out of 96 individual deployments across the entire monitoring period (11/03/2018–21/06/2019), with gaps in monitoring effort related to a combination of technical problems, loss of devices, and logistical issues. The amount of C-POD data varied between monitoring locations and through time, with fewer data collected during the 2018/2019 winter period, and the amount of pre-piling data available was limited (Fig. 2). Significant amounts of data (up to 38%) were discarded from some stations due to tidally driven noise, leading to hours monitored for $\leq$90% of their duration. This resulted in usable data for these stations being concentrated mainly at periods of reduced tidal flow. Various stations also experienced longer-term periods (i.e., continuously up to several days) of reduced monitoring effort, primarily caused by vessel noise.

FBW data from 24 out of 40 deployments were, on a deployment level, suitable for further analysis, with data excluded or absent mainly due to technical issues, equipment loss, and reduced FBW recording capabilities compared to C-PODs. No

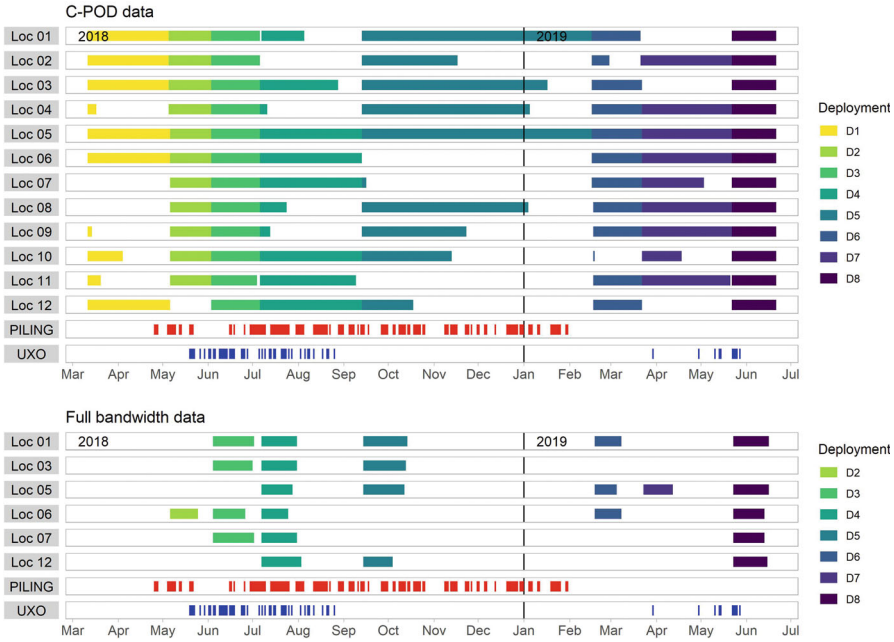



**Fig. 2** Summary of C-POD data (top panel) and FWB data (bottom panel) suitable for subsequent analysis (at deployment level; i.e., not taking into consideration exclusion of hours monitored for ≤90% of their duration, or sporadic corrupted acoustic files), in relation to realized EA1 pin-piling (in red) and unexploded ordnance (UXO) detonation activity (in blue)

pre-piling FBW data were available, and only limited data were available for the 2018/2019 winter months and during the post-construction period.

## Overall Porpoise Presence

Across the entire project duration (11,203 h between 11/03/2018 and 21/06/2019), C-POD monitoring effort (monitoring >90% of each hour) ranged from 3638 to 9668 h, depending on the monitoring station. Pre-construction porpoise detection rates were relatively high but declined considerably during the period of pin-piling activity. Especially for locations within the wind farm, detections remained low post-piling (Fig. 3). While monitoring coverage across both summer and winter was limited, porpoise detections appeared to increase during winter compared to summer.

## Harbor Porpoise Presence Modelling

Following the exclusion of hours that were ≤ 90% monitored, the piling and non-piling datasets contained a total of 6102 and 37,607 h available for modelling of porpoise presence, respectively.

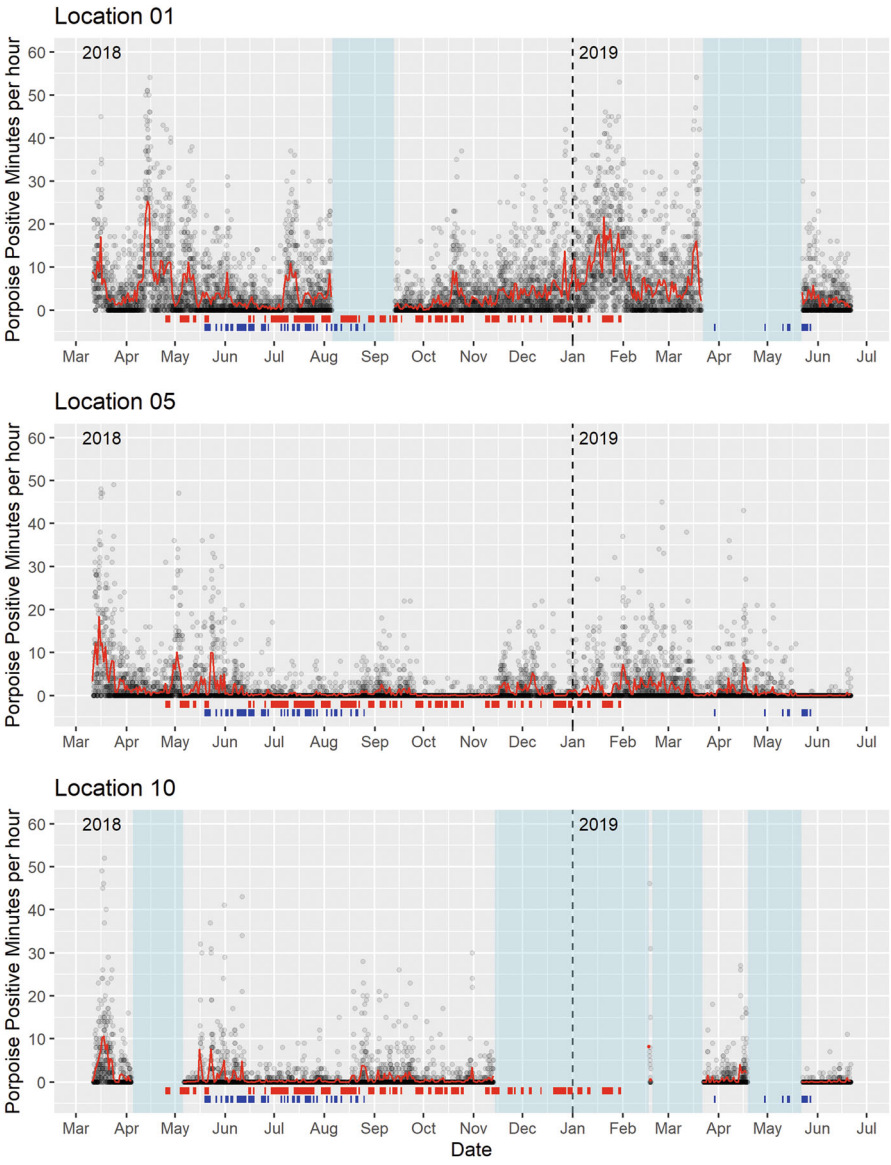



**Fig. 3** Examples of variability in hourly (gray circles) and daily averaged (red line) rates of Porpoise-Positive Minutes per hour throughout the monitoring period, as observed at 3 of the 12 monitoring stations: Station 1 (~20 km outside EA1), Station 5 (on the edge of EA1), and Station 10 (within EA1; see Fig. 1). Light blue areas indicate an absence of data; days with pin-piling and UXO detonation activities are indicated with red and blue markers, respectively

"Distance" was revealed to be the most important (piling model) or second-most important (non-piling model) explanatory variable in predicting porpoise presence. The fitted piling model contained fewer covariates but explained a higher percentage of

**Table 5** Harbor porpoise statistical modelling results. Note that the amount of insufficiently monitored data excluded differed among monitoring stations, and that remaining data represented periods with reduced tidal influences

| Dataset | Fitted model | Deviance explained |
|---|---|---|
| Non-piling | HP presence ~ s(Julian day) + s(Distance) + as.factor(Year) + s(Number of clicks) + s(Tidal cycle) + s(Diel cycle) + s(Lunar cycle) + s(Angle) | 25.1% |
| Piling | HP presence ~ s(Distance) + s(Julian day) + s(Number of clicks) + s(Angle) + s(Lunar cycle) | 38.4% |

s() represents a smooth term

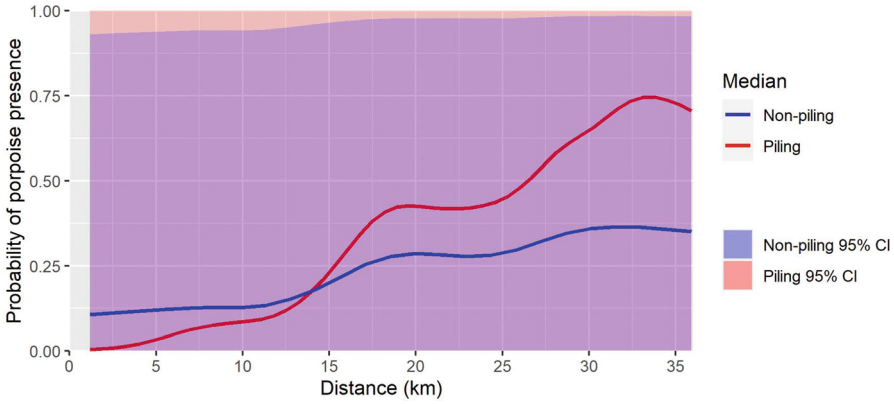



**Fig. 4** Effect of "Distance" on harbor porpoise presence at the EA1 site during non-piling periods, as well as during pin-piling activity. The probability of porpoise presence during pin-piling remains below the probability estimated in the absence of pin-piling out to 14.0 km from the piling activity

deviance in the dataset (38.4% vs. 25.1%; Table 5). The posterior simulation indicated that, despite large 95% confidence intervals (CI), when uncertainty around all model coefficients were incorporated, the median probability of detecting porpoises across the 500 simulations was lower during pin-piling than in the non-piling period up to 14.0 km from pin-piling activity (Fig. 4). The 14.0 km overall "pin-piling impact range" corresponded to a 615.75 $km^2$ impact zone around EA1 piling activity.

## Propagation Modelling and Predicted Received Noise Levels

Using the calibrated transmission loss model, porpoise frequency-weighted source levels were computed for each of the 310 pin-piles. These broadband source levels, calculated over the 40 Hz–16 kHz 1/3-octave band frequency range, fluctuated between 178.3 and 182.2 dB re 1 $\mu Pa^2s$ SEL at 1 m (unweighted source levels ranged between 211.9 and 215.2 dB re 1 $\mu Pa^2s$ SEL at 1 m). The relationship between distance and frequency-weighted received level, as derived from the

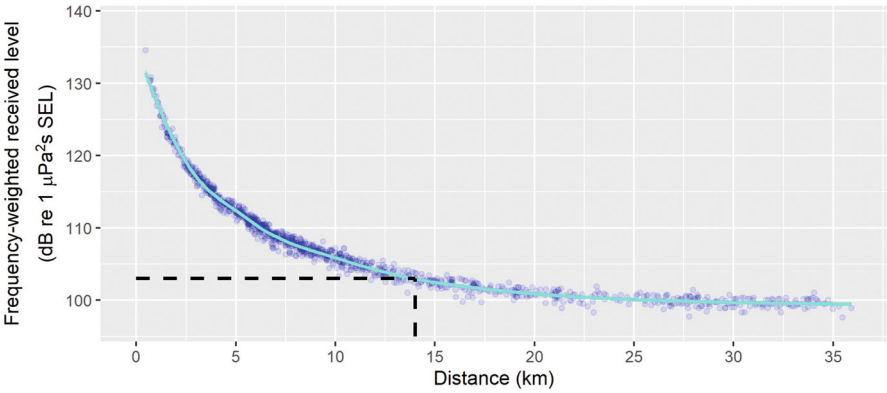



**Fig. 5** Predicted frequency-weighted received levels (dB re 1 µPa$^2$s SEL) in relation to distance from pin-piling activity, for the 40 Hz–16 kHz 1/3-octave band frequency range

propagation model (Fig. 5), indicated that the frequency-weighted received level associated with the pin-piling impact range at 14.0 km is 103.0 dB re 1 µPa$^2$s SEL.

## Discussion

Construction of the EA1 wind farm resulted in a reduced probability of porpoise detection out to 14.0 km from pin-piling activity. At this distance, the frequency-weighted received level from piling was 103.0 dB re 1 µPa$^2$s SEL.

### General Porpoise Presence

Analyses of C-POD data collected prior to, during, and post-construction of SPR's EA1 wind farm revealed that porpoises were present at the site throughout the monitoring period (March 2018–June 2019). Presence varied throughout the year and across the acoustic array, with detection rates generally lower within the wind farm compared to more distant stations. Especially for monitoring locations within, and close to, the wind farm, porpoise detection rates were higher during non-piling periods compared to when pin-piling activities occurred (April 2018–January 2019), and post-piling detection rates remained low compared to the pre-construction period.

Despite some data gaps during autumn and winter, porpoise detection rates appeared to be higher in winter compared to summer. This observation supports previous results showing higher local porpoise densities in the southern North Sea during the winter period (e.g., Gilles et al. 2016; Heinänen and Skov 2015; Paxton et al. 2016; Waggitt et al. 2020), likely reflecting seasonal variability in habitat use. The EA1 offshore wind farm is located within an area of the Southern North Sea SAC known as the "winter area" (October–March), which was designated based on

seasonally persistent, above-average porpoise densities (Heinänen and Skov 2015; JNCC and NE 2019). A general decrease in porpoise detections during the construction period (most of which occurred during the summer months) could thus be anticipated based on natural seasonal variation at the site.

The current study was underpinned by modelling porpoise presence derived from acoustic monitoring and is therefore subject to the same caveats associated with other PAM studies of porpoises, in that it relies on animals vocalizing to become detectable (e.g., Verfuss et al. 2018). Porpoise vocalization detectability is also influenced by multiple factors including ambient noise, source level, and orientation of porpoises' directional echolocation clicks (e.g., Au et al. 1999; Claussen et al. 2019; Macaulay et al. 2020; Nuuttila et al. 2018; Villadsgaard et al. 2007). Reduced detection rates may therefore reflect a decrease in animal presence, altered vocalization behavior, or both. Few studies have investigated the vocal behavior of porpoises in relation to (pin-)piling noise, as well as individual variation in movement patterns in response to noise impacts (but see Graham et al. 2023; Koschinski et al. 2003; van Beest et al. 2018). PAM has been used in a similar way extensively in the past, accepting above caveats (e.g., Brandt et al. 2018; Dähne et al. 2013). Various studies have provided additional support for the application of acoustic detections as a proxy for relative porpoise presence, by comparing acoustic detections to aerial visual line-transect and digital video-strip-transect survey data (Williamson et al. 2016), to vessel-based and aerial visual line-transect survey data (Brookes et al. 2013), to satellite telemetry data (Mikkelsen et al. 2016), and to theodolite tracking data (Kyhn et al. 2012). To further increase confidence that observed occurrence patterns reflected porpoise behavior (rather than changes in detectability due to environmental conditions), data with compromised monitoring efforts were excluded (up to 38.7% for individual C-POD monitoring stations; predominantly comprising of hours with increased tidal flow or persistent vessel noise). Data included in the current study therefore represented only those periods coinciding with reduced tidal flow and vessel noise, the effects of which varied across the array.

## Pin-Piling Impact Range

Statistical modelling of PAM data revealed clear negative effects of pin-piling, with an overall decrease in the probability of porpoise detections out to 14.0 km from the piling locations (associated impact zone of 615.75 $km^2$). At this distance, the overall predicted porpoise frequency-weighted received level was 103.0 dB re 1 $\mu Pa^2s$ SEL as modelled over the 40 Hz–16 kHz 1/3-octave frequency band range over which EA1 pin-piling signals were distinguishable from background noise. Reduced detection probability within the obtained pin-piling impact range, however, does not necessarily equal complete displacement. Likewise, displacement or other behavioral responses to noise by individual porpoises at distances beyond the overall impact range identified here may also have occurred. Individual sensitivity and

motivation, behavioral context, and previous exposure, may all contribute to individual responses (e.g., Ellison et al. 2012).

The GAM modelling revealed the partial contribution of pin-piling activity on the probability to detect porpoises. However, it is likely that porpoises also responded to other EA1 construction-related activities (e.g., increased vessel traffic, acoustic deterrent use, UXO clearance), as well as activities in neighboring wind farm development sites (e.g., geophysical/geotechnical surveys and construction) and other ongoing unrelated activities (e.g., passage of recreational and cargo vessels) in the surrounding area. For example, responses to non-piling activities have been demonstrated for wind farm construction in Scotland, where porpoise occurrence and detection rates of "buzzes" (suggesting foraging activity) were influenced negatively by jacket and turbine installation works and presence of support vessels (Benhemma-Le Gall et al. 2021). Similarly, observed pre-construction declines in porpoise presence in German waters could have occurred in response to construction-related activities like vessel movements and preparations for impact mitigation, i.e., bubble curtain deployment (Brandt et al. 2016). It is reasonable to assume that porpoise distribution and/or vocalization behavior are affected by this wide range of activities occurring alongside pin-piling. Moreover, as demonstrated in the current study and described in the scientific literature (e.g., Brandt et al. 2011, 2016, 2018; Dähne et al. 2013, 2017; Graham et al. 2019; Teilmann and Carstensen 2012; van Geel et al. 2023b), porpoise detection rates are reduced around piling locations, with impacts continuing for some time after construction activities have ceased, indicating that piling activity not only influences porpoise distribution during piling but also during subsequent non-piling periods. Considering the above, the pin-piling impact range revealed in the current study therefore not only reflects porpoises' vocalization and/or distributional responses in relation to pin-piling noise but may also be influenced by porpoises' past experiences, the presence of other anthropogenic pressures, and ecological characteristics of the habitat (e.g., abundance of prey), none of which were evaluated during the present study.

The 103 dB re 1 $\mu Pa^2s$ SEL broadband frequency-weighted received level above which the probability of detecting porpoises was negatively affected during hours of pin-piling, was derived from the relationship between distance from pin-piling activity and frequency-weighted received noise level. These received levels were inferred from propagated source levels (which in turn were based on provided maximal hammer energy applied per pin-pile) and were calculated using an acoustic propagation model that represents a simplification of the actual situation. For instance, a single source level was derived for the installation of an individual pin-pile based on the maximum hammer energy. In reality, however, noise levels vary throughout a piling event due to variation in applied hammer energy (resulting from ramping up protocols or penetration depth-related energy requirements) and changes in the amounts of energy being transferred into the sediment as piles are driven further into the seafloor (Banda and Cook 2018; Martin and Barclay 2019; Thompson et al. 2020). Similar generalizations are often applied to parameters affecting sound propagation, such as seasonally fluctuating water temperatures, stratification of the water column, and differences in weather conditions

(Malme et al. 1995). The transmission loss model developed for this study combined two propagation modelling techniques, incorporated project-specific information on pin-piling and environmental conditions, and was calibrated using acoustic measurements of field data. For these reasons, despite the uncertainties and the simplification, model validation with actual measurements increased confidence that the model predictions are realistic.

The current study did not explicitly incorporate non-piling construction activities and noise from other industries adjacent to the EA1 site into the propagation modelling. Modelling received levels based solely on the propagation of piling noise, while ignoring these other sound sources, may result in predicted sound levels that differ substantially from actual levels experienced by porpoises. To try to account for other noise-generating activities, as well as compare to the frequency-weighted received level of 103 dB re 1 $\mu Pa^2s$ SEL broadband above which the probability of detecting porpoises was negatively affected during hours of pin-piling (as indirectly obtained from the GAM model applying the "Distance" covariate), attempts were made to statistically model porpoise presence/absence using the directly measured frequency-weighted received levels (in dB re 1 $\mu Pa^2s$ SEL calculated over the 40 Hz–16 kHz 1/3-ocatve band frequency range) from the FBW data recorded at the six C-POD monitoring stations where these were collected simultaneously. Despite minimal coincidence between pin-piling activity and FBW monitoring resulting in a limited piling dataset available for the GAM modelling (444 h), the frequency-weighted received level was the most important explanatory variable. The final fitted non-piling model included received level as the 6th term, and it explained a very limited amount of the variation present in the data. As such, comparison with the probability of porpoise presence under piling conditions was impeded and the approach did not work. For this reason, measured (rather than modelled) frequency-weighted received levels were not used to derive the noise level at which the probability of porpoise detection was lower during pin-piling activity than in the absence of piling. Instead, this was obtained indirectly via the relationship between "Distance" and "modelled received level" as described in the "Propagation Modelling" section.

## Pin-Piling Impacts within UK Harbor Porpoise SACs

The results of the current study contribute new information on porpoise disturbance by pin-piling in addition to those presented by Brandt et al. (2016) and Graham et al. (2019). At present, the recommended impact range that should be applied when assessing the impacts of pin-piling on porpoises within English, Welsh, and Northern Irish SAC boundaries, as stated in JNCC (2020), is based on a single study by Graham et al. (2019) involving porpoises in the vicinity of a wind farm in the Moray Firth (Scotland). Construction in that study involved the piling of four 2.2 m pin-piles per turbine, and a deterrence response was defined as a >50% reduction in the number of Porpoise-Positive Hours (PPH) in the post-piling response period compared to those in the pre-piling baseline period. These authors reported a ≥50%

probability of such a behavioral response to piling at distances up to 7.4 km (95% CI 5.7–9.4 km) in the 24 h following cessation of piling, with a ≥0.25 probability of such a response to distances approximately 18 km from the piling location. Graham et al. (2019) also reported that porpoises appeared to habituate to ongoing piling activities, with response distances decreasing over the course of wind farm construction. Based on these results, JNCC (2020) recommended an EDR for pin-piling of 15 km (Table 1) on the premise that most of the effect would occur within this range, acknowledging that a 100% deterrence/disturbance was not expected to occur within the associated area and that this EDR also did not represent the maximum range at which effects may be detected.

JNCC (2020) has recommended periodic evaluation of EDRs as new data become available. The current lack of UK requirements for monitoring and assessing the impacts of offshore wind farm construction based on empirical data means that, unless developers themselves decide to undertake such a campaign (e.g., current study; Benhemma-Le Gall et al. 2021; Graham et al. 2023), the UK will rely on new insights originating from elsewhere in Europe or further afield. For example, data collection under US license requirements (e.g., van Parijs et al. 2021) may contribute to this process.

The pin-piling impact range obtained in the current study for the EA1 offshore wind farm provides useful insights from another UK development area compared to the Graham et al. (2019) study, and can be considered when evaluating the "Guidance for assessing the significance of noise disturbance against Conservation Objectives of porpoise SACs in England, Wales and Northern Ireland" (JNCC 2020), and in broader noise management within the UK and elsewhere. The 14.0 km impact range identified in the current study does not deviate widely from the recommended EDR of 15 km; however, the 1 km difference translates into a smaller overall impact area (615.75 $km^2$ vs. 706 $km^2$), which could have implications for future management of piling schedules in relation to SAC noise management.

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# SanctSound: Building Data Systems for Sound Decisions

# 103

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**Abstract**

In April 2022, the US National Oceanic and Atmospheric Administration, the US Navy and partners launched a Sanctuary Soundscape Monitoring Project (SanctSound) web portal (https://sanctsound.ioos.us/). The portal allows the public to learn about the importance of ocean sound and explore the findings from this collaborative effort by over 50 scientists spanning 3 years. Here, we aim to increase awareness of this new platform and discuss lessons learned and next steps for development of such assets in the future among international colleagues. The portal provides a tangible reference for discussing community-level progress in addressing the priorities that have guided the SanctSound project: (1) enhanced

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standardization through data collection and analysis steps; (2) development of new metrics, indicators, and visualization techniques to inform local to system-wide management needs; (3) centralization and public access for both raw recordings and data products; (4) data-driven communication and education products for a wide array of audiences; and (5) creation of sustainable data pipelines that can be leveraged broadly across projects. The portal also showcases how underwater sound monitoring helps resource managers design methods to better protect national marine sanctuaries. Examples ranging from informing large whale protections to coral reef restoration to fishery management will be provided to highlight both the achievements of the project and emerging opportunities through large-scale data collections to answer questions of interest to managers.

**Keywords**

Marine soundscapes · Monitoring · Marine protected areas · Cyberinformation · Science communication

## Introduction

The Sanctuary Soundscape Monitoring Project (SanctSound) was a 3-year underwater sound monitoring effort in the waters of the USA that began in the fall of 2018 and ended in spring 2022. It was co-led by two federal government agencies, the US National Oceanic and Atmospheric Administration (NOAA) and the US Navy, and engaged experts from 24 institutions from all over the country. The project collected underwater recordings from 30 locations in seven US National Marine Sanctuaries and one National Marine Monument distributed off the US East and West Coasts and off the Pacific Islands.

Recordings aimed for continuous sampling up to 24 kilohertz and moorings were maintained by divers and using acoustic releases. The project resulted in several hundred terabytes of data. This chapter outlines the web interfaces that were built to access, explore, and interpret these data in order to support their applications to managing protected areas.

The SanctSound project is accessed on the web through NOAA's main launch site for ocean observation datasets, known as the Integrated Ocean Observing System. Selecting the project's tile provides a single access point (https://sanctsound.ioos.us/) for three main web services: the project's landing page (which serves a general user audience), the project's data portal (which serves a scientific and managerial user audience), and the project's archive (which provides metadata and download access for the project's data).

## Touring the Portal

The project's landing page provides six tutorials, accessed through a panel of six tiles, with educational graphics and videos explaining the project's specifics and underwater sound recording in sanctuaries generally (Fig. 1). It provides a sound

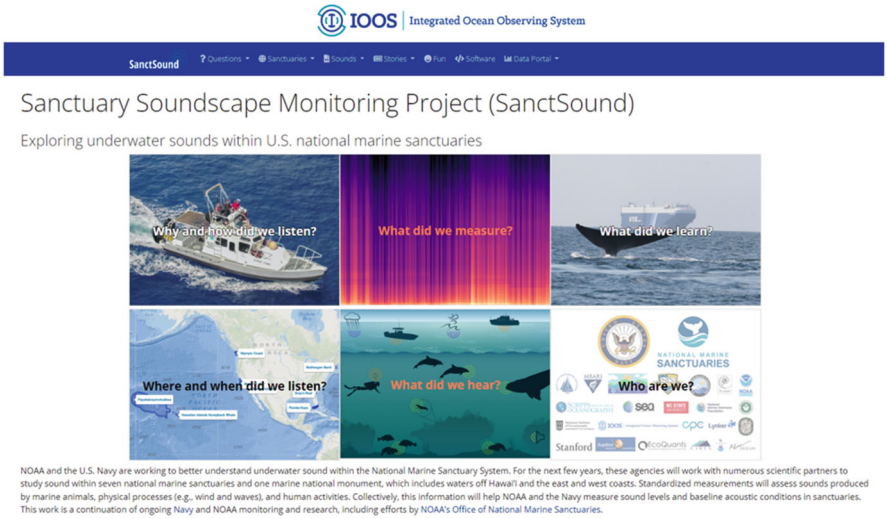



**Fig. 1** SanctSound project landing page (https://sanctsound.ioos.us/)

library for a subset of the specific sources that were heard and identified in sanctuaries, including human made, marine animals, and environmental. A larger library of almost 200 sound clips is available through the project's data portal.

The main menu at the top of the landing page allows users to tour the project's results by sanctuary. This menu also provides access to web stories highlighting project results around the country, a list of software and tools that were used or generated for the project and direct links to the data portal.

In the data portal, green download and citations buttons allow users to access results by recorder deployment. Each dataset has a DOI and is archived at NOAA's National Centers for Environmental Information (NCEI). Download access for raw recordings and results is provided through NCEI's Google Cloud services following selection using a map viewer (https://www.ncei.noaa.gov/maps/passive-acoustic-data/). Download access for results is also provided via another NOAA service called ERDDAP (https://coastwatch.pfeg.noaa.gov/erddap/griddap/).

## Tour 1: Sound Sources in Sanctuaries

The sanctuaries menu item provides users with a clickable map of the sanctuaries where monitoring occurred. For each sanctuary, under the "what did we hear" tab, users are provided with an interactive acoustic scene that shows all of the sources of sound that were identified during the project in this sanctuary (Fig. 2). Selecting an element of the acoustic scene provides a picture and video spectrogram of that source recorded in that sanctuary. For those that were detected over time, users can look at examples of their temporal patterns using tabs that show results both as histograms

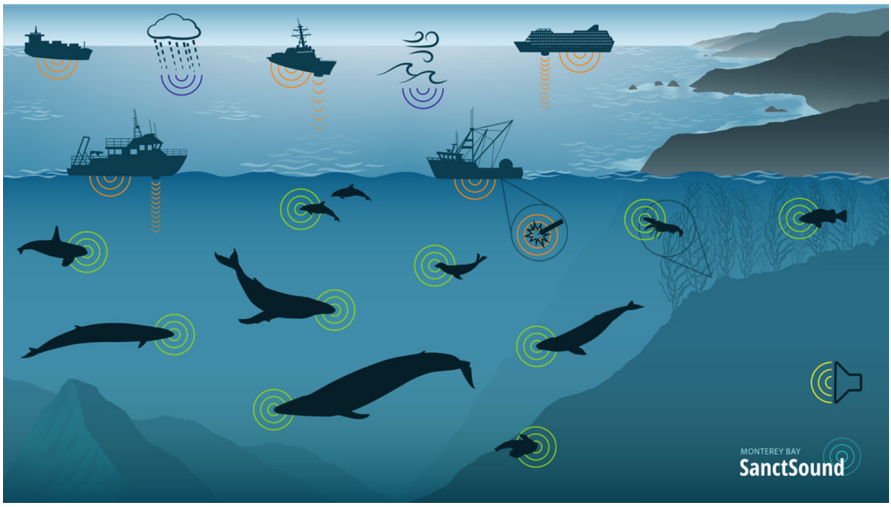



**Fig. 2** Clickable soundscape for Monterey Bay National Marine Sanctuary on the US West Coast (https://sanctsound.ioos.us/s_mbnms.html)

and as seasonal clock or rose plots. In these project page graphics, the plots are annotated to make the results easy to understand and results showcase examples from individual monitoring locations per sanctuary.

For the user who is interested in further exploring time series results, there are links provided into the portal. The data portal also organizes all results around sanctuary locations, which are also accessed via a map browser (Fig. 3). However, in the data portal, the user can explore results from each monitoring location within each sanctuary in more depth. For results that were collected at finer granularity the user can explore sound detection by hour and by month.

## Tour 2: Sound Levels in Sanctuaries

For each sanctuary selected on the project's landing page, the user can also explore the second tab to understand "where we listened" in this sanctuary. This provides information on the recording locations in the sanctuary, including why each was chosen and an example of the sound propagation modeling results that were completed for each monitoring station. Charts show the time periods for which data are available.

The "what did we measure" tab provides a visual overview of the sound level results for this sanctuary with annotations for each visualization to help the general user understand first the long-term spectrogram for the 3 years of recording data, a stacked histogram showing hours above each octave's median over time, and then a power spectrum and anomaly plots. The project landing pages explain what each of these visualization types are showing, using captions to help the user interpret what

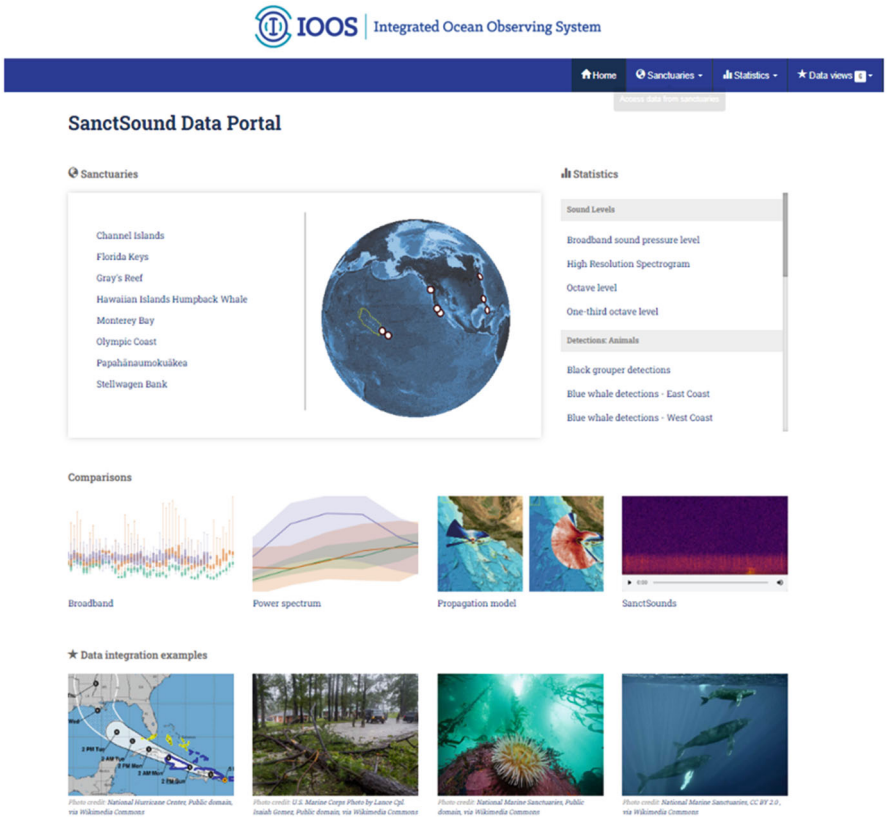



**Fig. 3** Landing page for SanctSound Data Portal. (https://sanctsound.portal.axds.co/)

each tells us about a soundscape (Fig. 4). Selecting this same location in the data portal, under "sound levels overview," provides the same plots but here they are fully explorable. The user can select time and frequency bounds of interest and hover to get summary statistics. Sound levels data summaries are also provided for third octave and octave band levels. Each plot can be expanded and exported.

## Tour 3: Comparing Sound Attributes Across the Sanctuary System

The data portal also supports comparison of sound level statistics across all monitoring locations. The user can hand select which locations and frequencies they are interested in visualizing on the same axes (Fig. 5), or they can review a panel of plots. Histograms and rose plot patterns for sounds that were detected across sanctuary locations (selected through a drop-down menu) can be compared as a panel of plots.

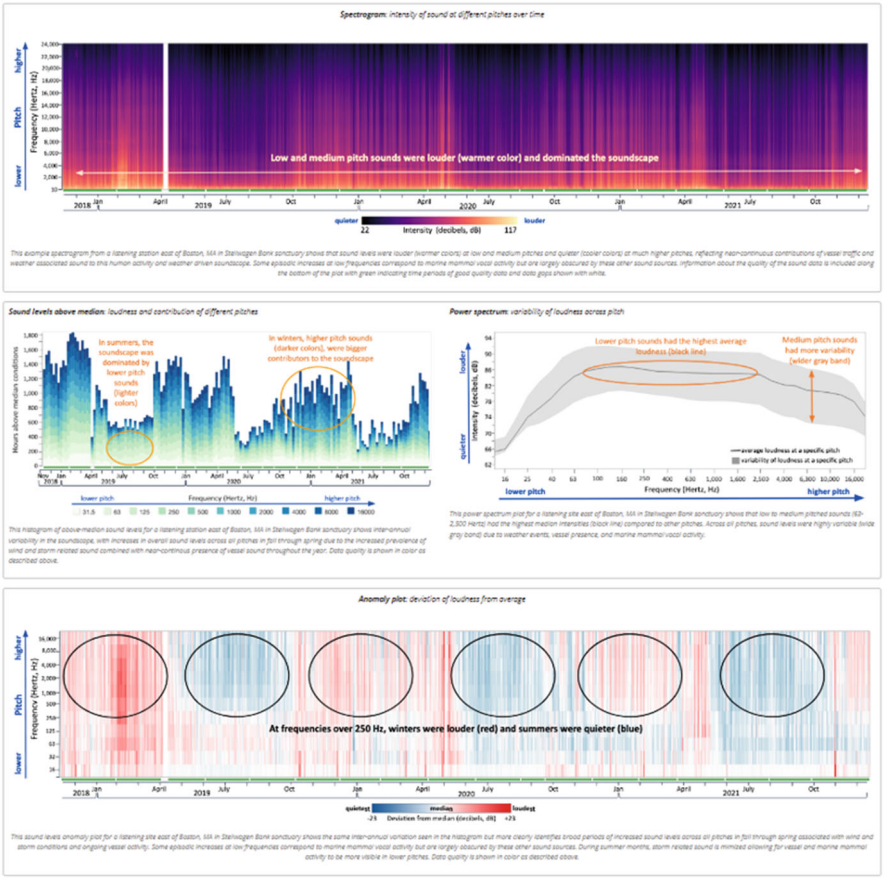



**Fig. 4** "What did we measure" results for Stellwagen Bank National Marine Sanctuary. (https://sanctsound.ioos.us/s_sbnms.html)

Comparisons are also supported for sound propagation modeling results. Estimates of signal to noise and received levels can be selected for standardized sound sources, selecting frequencies and months of interest. Map and histogram results can be reviewed (Fig. 6).

## Tour 4: Integrating Sound Results

The project worked to interpret its data relative to questions of interest in each sanctuary and across the sanctuary system. The "stories" tab on the project's landing page provides access to 17 visually compelling web presentations, accessed through a panel of tiles (Fig. 7). Most of these stories include maps of sound propagation for

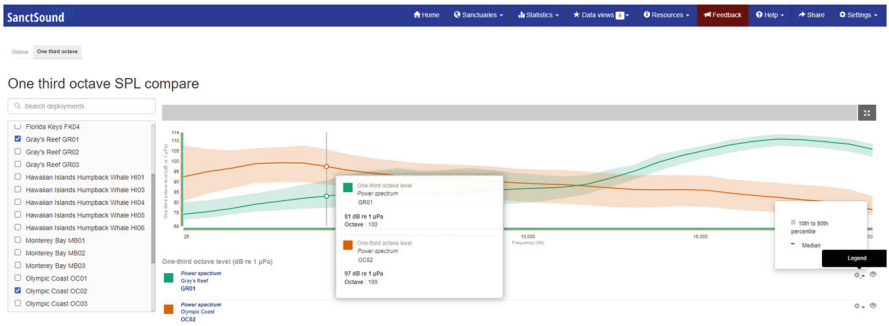



**Fig. 5** Power spectrum data for a shallow water temperate reef site (Gray's Reef National Marine Sanctuary site GR01) and a deeper water site near a busy traffic lane (Olympic Coast National Marine Sanctuary site OC02) showing near-mirror image results, with snapping shrimp dominated higher frequencies versus large commercial shipping traffic lower frequencies. (https://sanctsound.portal.axds.co/#sanctsound/compare/powerspec)

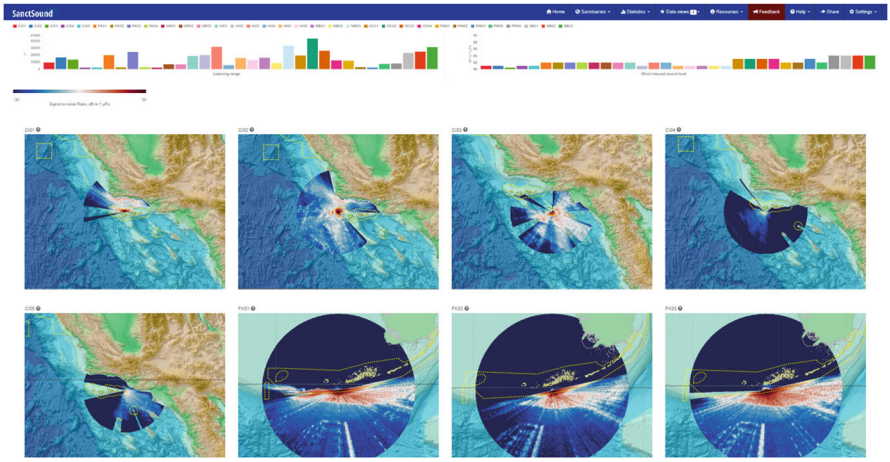



**Fig. 6** Signal-to-noise ratio modeling results across multiple recording locations. (https://sanctsound.portal.axds.co/#sanctsound/compare/prop-model)

the sources that they are examining to help readers understand the relevance of sound measurements to areas within and beyond the sanctuary.

The data portal provides the user with the ability to build their own "story" through curated data views. Selecting datasets of interest to co-visualize presents coincident time series with multiple axes. This visual can be saved, exported, and modified. The data portal also highlights the ability to integrate sound information with other ocean observation variables collected in the same areas and time periods from different mooring systems. The curated data views provided at the bottom of the home page provide examples of some integrations of interest.

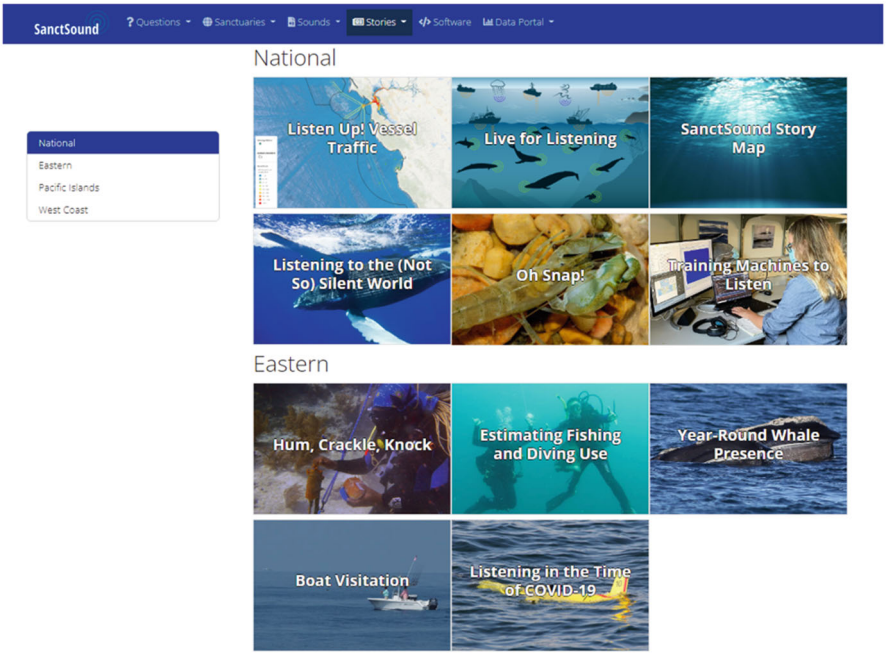



**Fig. 7** Panel of tiles that provide access to web stories published by NOAA's Office of National Marine Sanctuaries highlighting applications of SanctSound results to site- and system-level management questions (https://sanctsound.ioos.us/stories.html)

## Applications

SanctSound focused on the accessibility and interpretability of its data results as critical to ensuring their utility for conservation decision-making. The project applied its data to a diversity of management contexts of marine protected areas, ranging from supporting restoration efforts in reef systems, enforcement of zones within sanctuaries (including detection of illicit activities and compliance monitoring), tracking the status of resources in changing climatic conditions, evaluating use of sanctuaries and their valuation by people, and human use risk evaluation and impact reduction (strike, disturbance, lost communication, and stress). Across all applications, common attributes that determined utility were: longevity, comparability, and accessibility.

Several applications highlighted in the "stories" made available through the portal continue to see development and investment by NOAA and partners. For example, the SanctSound data supported comparison of the influence of vessels on diverse and nationally distributed soundscapes, and showed variance from very little or rare vessel noise dominance to higher vessel noise exceedance and

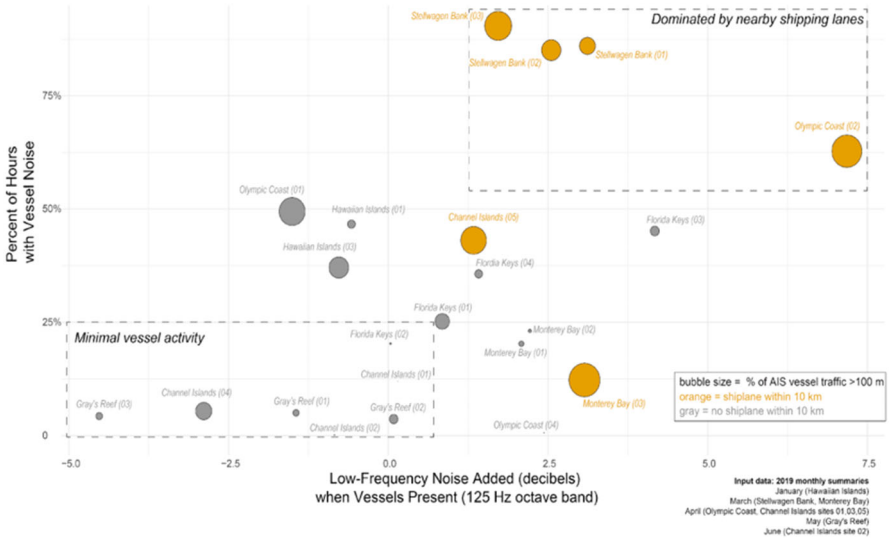



**Fig. 8** The national story on vessels in sanctuaries explored the presence of vessels across sanctuary soundscapes through acoustic detection of vessels, levels of sound in vessel-dominated bands, and comparison with other tracking methods like Automatic Identification System data. (https://sanctuaries.noaa.gov/science/monitoring/sound/listen-up.html)

prevalence (Fig. 8). These results continue to be evaluated relative to the conservation priorities in each sanctuary and to assess the effectiveness of different mitigations, including mandatory and voluntary vessel speed reduction zones that overlap monitored areas.

These data are also continuing to support our understanding of the effectiveness of existing zones within sanctuaries, including those designated to protect spawning fishery stocks and support research. Monitoring for fish spawning sounds in and around these zones as well as detecting illicit fishing activity helps sanctuary managers evaluate their efficacy.

As a final example, these data have provided important baselines for coral reef restoration efforts. Indicators range from snapping shrimp activity levels for sites off Hawaii and off the US Southeast Coast to detections of keystone fishes in and around the iconic reefs of the Florida Keys National Marine Sanctuary that are undergoing active restoration. This is also a research area where training datasets available through SanctSound are supporting the extraction of more holistic metrics of change in soundscapes that capture the multidimensional features of soundscapes (acoustic diversity and ecosystem indicators). Ground-truthing these methods are critical to our ability to continue to integrate sound within restoration monitoring frameworks.

## Next Steps

Although SanctSound ended in spring 2022, NOAA has continued efforts to monitor underwater sound throughout the sanctuary system. Monitoring in 2022–2024 includes 11 sanctuaries and 1 proposed sanctuary with a total of 26 recording locations. NOAA continues to work across diverse communities to build methods for people with a range of interests to access sound information services.

**Acknowledgments** This work depended on the collective efforts of many determined people from the coauthoring institutions who were dedicated to this project during extraordinary times.

# Characterization of Noise Abatement Systems for Protecting Aquatic Invertebrates and Fishes

104

Tobias Bohne, Tanja Grießmann, and Raimund Rolfes

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**Abstract**

Offshore wind energy plays an important role in the future energy supply. But its expansion poses a risk to the marine environment. Especially, during the construction of the foundations high sound pressure levels occur in the water column due to impact pile driving. Noise abatement systems are used to reduce sound pressure levels, primarily to protect marine mammals. However, there is little information on how effective these systems are for protecting aquatic invertebrates and fishes that are sensitive to particle motion. In order to quantify the reduction of sound particle motion due to a noise abatement system a metric is proposed according to the one for sound pressure. For the simple case of a plane wave, the metrics for describing the sound particle motion and sound pressure are compared.

**Keywords**

Pile driving · Particle motion · Noise abatement systems · Bubble curtain

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## Introduction

Offshore wind energy represents an important pillar of the future energy supply. Due to the construction and the pile driving in particular, high sound pressure levels occur in the water column. These are potentially dangerous for the marine environment. In the last decade, different noise abatement systems have been developed. Following Popper et al. (2022), the use of these systems focus on the protection of marine mammals and thus on sound pressure reduction. In various measurement-based studies, c.f. Bellmann (2014) and Bellmann et al. (2017), and model-based studies, c.f. von Pein et al. (2021), Peng et al. (2021), and Bohne et al. (2019), the influence of noise abatement systems on the sound pressure level have been investigated. But there is much less information about how effective noise abatement systems are for protecting fishes and aquatic invertebrates, which often live close to the ground and can be sensitive to sound particle motions and substrate vibration, c.f. Hawkins et al. (2021). With the aim to investigate the influence of different noise abatement systems and piling energies on the particle motion, Sigray et al. (2022) performed, as one of the first groups, measurements during the pile driving of an offshore wind turbine foundation. They observed a clear reduction of the acceleration exposure level when using noise abatement systems.

The question arises if there is a correlation between the reduction of sound pressure and particle motion and thus a possibility to transfer the comprehensive results. This work will first give an overview of the functioning of the bubble curtain as a frequently used noise abatement system (c.f. section "Working Principles of the Bubble Curtain as a Noise Abatement System"). In order to prepare ongoing studies concerning the given research question, a metric is proposed to quantify the reduction of particle motion (c.f. section "Introducing a Metric to Describe the Reduction of Sound Particle Motion"). Three field quantities, i.e., displacement, velocity, and acceleration, exist to describe the particle motion. For the simple case of a plane wave, the metrics describing the sound particle motion and sound pressure are compared (c.f. section "Investigating the Simple Case of a Plane Wave"). Finally, the results are summarized and conclusions are drawn (c.f. section "Conclusion").

## Working Principles of the Bubble Curtain as a Noise Abatement System

To create a bubble curtain, a nozzle hose is laid on the seabed around the pile. In the next step, air is pumped through the hose. Bubbles form at the nozzles and rise, enveloping the pile. The sound waves emitted during the pile driving process strike the bubble curtain and interact locally with the bubbles. These act as oscillators depending on their size and the surrounding static pressure field. The local interaction of the bubbles with the sound wave represents the local acoustic properties of the bubble curtain. These strongly depend on the local air fraction and bubble size distribution. These in turn are determined by the fluid dynamics in the bubble curtain. Within a great part of the water column, the bubble curtain behaves like a

plane buoyant flow. The bubbles induce a buoyancy force in the surrounding water. The water-bubble mixture rises and expands continuously (Bohne et al. 2019). The bubble formation process occurs within a short distance to the nozzle. In this process, large bubbles form at the nozzles and break up into smaller bubbles with increasing distance. This process is driven by the local bubble-induced turbulence. With greater distance, the turbulence decreases significantly, the process stops, resulting in a bubble size distribution which no longer varies over the water depth (Bohne et al. 2020).

The acoustic properties within a certain water depth determine the local sound transmission of the bubble curtain. In the lower frequency range (100–300 Hz), the impedance difference between the water and water-bubble mixture is the decisive factor for the sound transmission and reflection. The width of the bubble curtain, which varies with water depth, produces locally increased sound transmission due to $\lambda/2$ conduction. In the higher frequency range (300–1000 Hz), the frequency of the incident sound wave is closer to the natural frequencies of the bubbles, and the effect of sound absorption increases. Due to the dependence of the natural frequencies on the static pressure, the sound absorption is higher in the upper water layers than at greater water depths. With increasing frequency, absorption also increases in greater water depths. Thus, the bubble curtain closes from the top to bottom.

## Introducing a Metric to Describe the Reduction of Sound Particle Motion

To the author's knowledge, no metric exists describing the reduction of sound particle motion due to a noise abatement system. For sound pressure, the noise reduction is typically described by the differences between the sound exposure levels, writing

$$\Delta L_{E,p} = L_{E,p,\mathrm{ref}} - L_{E,p,\mathrm{BC}} \tag{1}$$

with the sound exposure level of the reference case $L_{E,p,\mathrm{ref}}$ and the sound exposure level of the case with bubble curtain $L_{E,p,\mathrm{BC}}$. The definition of the sound exposure level $L_{E,p}$ follows ISO (2017). It should be noted that the $\Delta L_{E,p}$ corresponds to the insertion loss. A useful form of Eq. 1 can be derived using Parseval's theorem (ISO (2017)), writing

$$\Delta L_{E,p} = 10 \log_{10} \left( \frac{\int_{-\infty}^{\infty} |P_{\mathrm{ref}}|^2 df}{\int_{-\infty}^{\infty} |P_{\mathrm{BC}}|^2 df} \right) \tag{2}$$

with the double-sided sound pressure spectrum $P_{\mathrm{ref}}$ of the reference case and the double sided sound pressure spectrum $P_{\mathrm{BC}}$ of the case with bubble curtain.

According to $\Delta L_{E,p}$, we propose to describe the reduction of the sound particle motion by means of the difference of the time-integrated squared sound particle velocity levels, writing

$$\Delta L_{E,u} = L_{E,u,\mathrm{ref}} - L_{E,u,\mathrm{BC}} \tag{3}$$

with the time-integrated squared sound particle velocity level of the reference case $L_{E,u,\mathrm{ref}}$ and the time-integrated squared sound particle velocity level of the case with bubble curtain $L_{E,u,\mathrm{BC}}$. Contrary to the recommendation of Nedelec et al. (2021), the sound particle velocity has been chosen as the field quantity to describe the particle motion. With the aim to compare the metric for sound pressure reduction with that of sound particle motion, this seems reasonable as shown in the next section.

## Investigating the Simple Case of a Plane Wave

For the case of a plane undamped forward traveling wave, the sound pressure spectrum writes

$$P(x,\omega) = p^{+} e^{-ikx} \tag{4}$$

and the velocity spectrum writes

$$U(x,\omega) = \frac{k}{\omega\rho} p^{+} e^{-ikx} \tag{5}$$

with the amplitude of the pressure wave $p^{+}$ and the real valued wavenumber $k = \frac{\omega}{c}$. Inserting Eq. 4 into Eq. 5 leads to

$$P(x,\omega) = c\rho\, U(x,\omega) \tag{6}$$

Inserting Eq. 6 into Eq.2 shows that for the simple case of a plane undamped wave the reduction of sound pressure is identical to the reduction of sound particle motion in terms of sound particle velocity $\Delta L_{E,p} = \Delta L_{E,u}$.

## Conclusion

With the aim to describe the reduction of sound particle motion by noise abatement systems, a metric has been proposed similar to the one for sound pressure. For a plane wave, it has been shown that the reduction potential of a noise abatement system for sound particle velocity is equal to the one of sound pressure. This chapter proposes to use the sound particle velocity as field quantity in following studies, which refer to the reduction of particle motion by noise abatement systems, to allow for comparability with the reduction of sound pressure.

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# Broadband Excess Noise Estimation via Sample Distribution Field Calibration

105

Sérgio M. Jesus, Cristiano Soares, Miriam Romagosa, Irma Cascão, Ricardo Duarte, Friedrich Zabel, and Mónica A. Silva

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**Abstract**

The Marine Strategy Framework Directive (MSFD) of the European Union currently requires meaningful noise level-based indicators to be obtained either through field data observations or computer modeling. Acoustic recordings in the 20–1000 Hz frequency band were performed 6 h a day during June 2018 in three

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moorings to the southwest of the Islands of Faial and Pico (Azores). Anthropogenic noise was modeled due to ship traffic drawn from AIS data, and environmental sound was attributed to wind. Comparing field and model data at these three locations allowed to introduce a field calibration procedure based on the linear transformation of sample distribution probability density functions in the sound pressure-level domain. The broadband version of the excess noise level (ENL) appears as the sum of the 1/3-octave bands transformed power over the data recording frequency band. The results show that field-calibrated broadband ENL sound maps are in line with the main shipping routes, fishing activity, and leisure paths in the area revealing; however a significant noise spread due to sound propagation conditions to the east and the southwest of the Faial-Pico Islands. Field-calibrated broadband ENL may be seen as a reliable quantity for establishing continuous anthropogenic noise pollution MSFD indicators.

**Keywords**

Ocean soundscape · Shipping noise · Excess noise level · Field calibration

## Introduction

Ocean noise level worldwide was found to be increasing at an estimated rate of 3 dB/decade between 1950 and 2010, in line with the increase in shipping and GDP growth (Frisk 2012). The full impact of this ocean noise increase is still unknown, but evidence is piling up toward a deterioration of the marine habitat, the disruption of the life cycle of a number of endangered species, and biodiversity loss (Hildebrand 2009; Hawkins and Popper 2014; Williams et al. 2015; Erbe et al. 2019; Duarte et al. 2021).

Until recently, the focus of ambient noise studies was on its spatial correlation, coherence, and directivity, with the scope of reduction or elimination. The rising concern of impacts of ocean noise on marine species shifted the focus of ambient noise studies to the ocean sound level, classically defined as the sound pressure level (SPL) mapped into geographical coordinates, time, and frequency – the so-called sound maps. In practice, sound maps are obtained through computer modeling, in order to cope with the required time-space resolution for a meaningful representation, impossible to obtain otherwise by direct observation. Sound maps are, therefore, a fundamental tool for marine policies development/enforcement, identifying areas impacted with high sound levels when compared to others relatively protected, and to follow sound stress dynamical changes along time. Ocean sound was recently declared as an essential ocean variable by the Global Ocean Observation System (GOOS). In the recently published United Nations World Ocean Assessment report, a new full chapter was dedicated to trends on anthropogenic noise inputs into the marine environment (Sirovic et al. 2021).

Conceptually, there are at least two approaches for estimating ocean noise: one is by summing all the individual anthropogenic source contributions in any given point

in space, and the other is by measuring ocean sound and "subtracting" the natural background sound level. As we will see, both approaches have their own difficulties, flaws, advantages, and drawbacks.

The Marine Strategy Framework Directive (MSFD) is the European Union's main policy for marine life protection and conservation, aiming at achieving the good environmental status and its monitoring, along with protection policies which require identifiable and meaningful indicators (European Commission 2010). Descriptor 11 of the MSFD defines two indicators: one for impulsive noise and another for continuous noise. Depending on the context, these two components are often difficult to separate, which is a requirement for building suitable MSFD indicators. A recent proposal defines the excess noise level (ENL) which is the anthropogenic noise in excess of the natural background ambient sound as a possible basis for continuous noise indicator (Farcas et al. 2020). The idea behind ENL is that of determining the anthropogenic noise on top of the natural ocean sound normally present in the ocean. This, of course, requires a measurement (or an estimate) of the reference or baseline sound level without noise contamination which, nowadays, is practically impossible. This is particularly difficult in north Atlantic, which is the basin size region with the most intense and constant ship traffic in the world (Sirovic et al. 2021).

It is commonly assumed that the two main contributions to ocean sound in the band 20–1000 Hz are shipping noise and the sound generated by wind on the ocean surface. Therefore, sound maps use computer models to generate realistic approximations of these two sound fields taking as input environmental information such as bathymetry, water column temperature and salinity profiles, bottom characterization and wind speed estimates, as well as ship traffic information drawn from the Automatic Identification System (AIS). Acoustic propagation models have been used for obtaining three-dimensional (actually better described as N $\times$ 2D) deterministic representations of the shipping noise field (Colin et al. 2015; Soares et al. 2015; Skarsoulis et al. 2016). This is a very computation-demanding task, in particular for large areas with intense traffic, so an alternative was devised to represent ship location through a statistical distribution and thus reduce the computational burden (Kinda et al. 2017).

Sound maps obtained through numerical modeling are error prone due to inherent parameter uncertainties, model mismatch, and environmental/AIS data dropouts. Any direct acoustic observation should be a valuable resource, first for sound map validation and then for an eventual model adjustment. The process of integrating model and field data for sound maps' correction is known as "field calibration." The most common approach for field calibration assumes that environmental data used in acoustic propagation models is erroneous or incomplete and its adjustment allows for matching data and model (Farcas et al. 2020). Another alternative is to consider that the mismatch comes from the source level assumed for the ships cruising in the area and its estimation from the data itself would allow for a better match, as proposed in a simulation exercise in Jesus et al. (2017). Recently, the third approach directly calibrates the statistical distributions of the modeled field as proposed in Jesus et al. (2022). The validation of the gain provided by each field calibration

approach, and how this gain extends for time periods and locations where there are no ground truth observations is a challenging task.

The objectives of this study are as follows: one is to provide evidence for broadband ENL estimation methodology and explore the possibility of its usage to support building a continuous noise indicator, and the other to propose a sample field calibration method to integrate measurements and modeled data, for extending a localized observation into a wider area, where there were no observations.

This work is supported on acoustic recordings performed in the 20–1000 Hz frequency band, 6 h a day during June 2018 at three locations close to the Faial-Pico Islands, in the Azores archipelago, a well-known habitat for over 20 cetacean species (Silva et al. 2014). This data set was evaluated and found to be biased in some frequency bands, so its usage for model calibration is questionable and only made possible after filtering and conditioning. Ocean sound modeling was performed for ship traffic and surface wind action. The linear probability density functions' sample field calibration procedure presented in Jesus et al. (2022) was used and applied to this data set, extending the results with complimentary material in the following aspects: time and spatial evolution of water column; field calibration procedure is detailed; statistics for the worst-case recorder (MG) are shown; non-calibrated and calibrated model data detailed for all bands; and distribution means and variance before and after calibration.

## Methods

### Environmental Data

The Azores archipelago is formed by a total of nine islands in three groups: the western group of two islands, the eastern group of also two islands, and the central group composed of five islands, four of which are shown in Fig. 1 with the island of Graciosa to the north (not shown). The bathymetry data was obtained from the GEBCO (General Bathymetric Chart of the Oceans, www.gebco.net) database

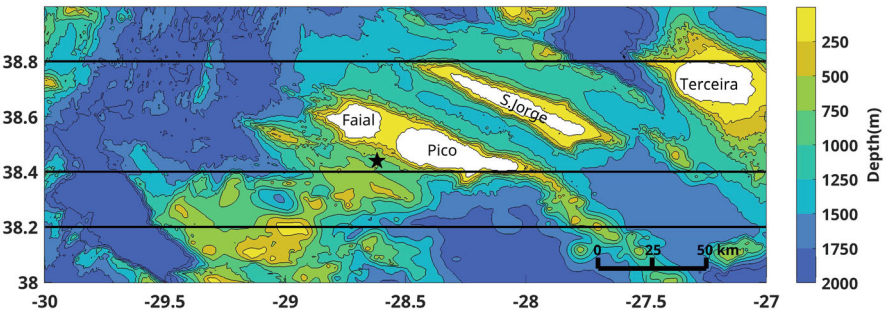



**Fig. 1** Central group of islands of the Azores archipelago. The horizontal lines and black star indicate positions for extracting water column information as shown in Fig. 2 (Adapted from Jesus et al. 2022). Bathymetry data credits: GEBCO database (Group, G.B.C 2020)

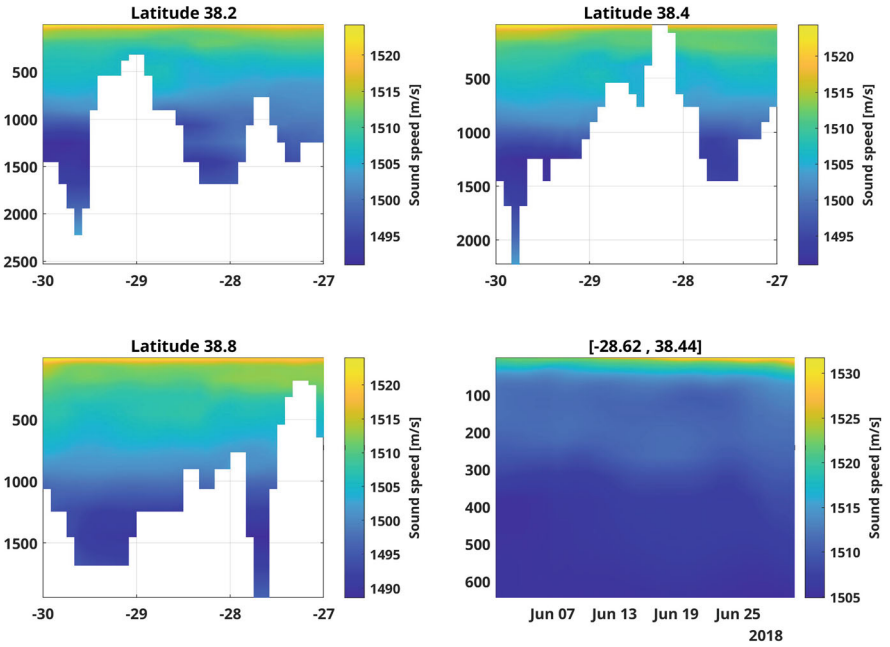



**Fig. 2** Depth-longitude sound speed for latitudes of 38.2°, 38.4°, and 38.8° North calculated from temperature and salinity profiles, upper left, right, and bottom left plots, respectively, and sound speed time evolution through the month of June 2018 at location [−28.62 W – 38.44 N] (bottom right). (Source: CMEMS-Copernicus Marine Service)

(Group, G.B.C 2020). The horizontal lines and the star symbol denote the locus for water column temperature and salinity information as shown in Fig. 2. The star symbol is in the vicinity of the acoustic recorders' location (see Fig. 5 for a close-up). The area under study is characterized by deep water with two main features: the shallow channel between Faial and Pico islands, with an average depth of approximately 80 m, and the shallow banks (<200 m depth) to the southwest of Faial Island, known as an important fishing area. Figure 2 shows computed sound speed using the temperature and salinity profiles extracted from the CMEMS (Copernicus Marine Environment Monitoring Service) (www.copernicus.eu) database, along constant latitude lines 38.2, 38.4, and 38.8, upper left, right, and bottom left plots, respectively, and for the sound speed evolution through time (bottom right plot) at the position indicated with the star symbol. Water column sound velocity variation with range is mild, as well as throughout the month of the study.

Figure 3 shows the wind speed over the month of June 2018 as predicted by the ECMWF (European Centre for Medium-Range Weather Forecasts, www.ecmwf.int) (Hersbach et al. 2018) for the mean (a) and the standard deviation (b). There is a significant deviation of more than 2 km/h from south to north of the considered area, while the standard deviation slightly decreases from west to east.

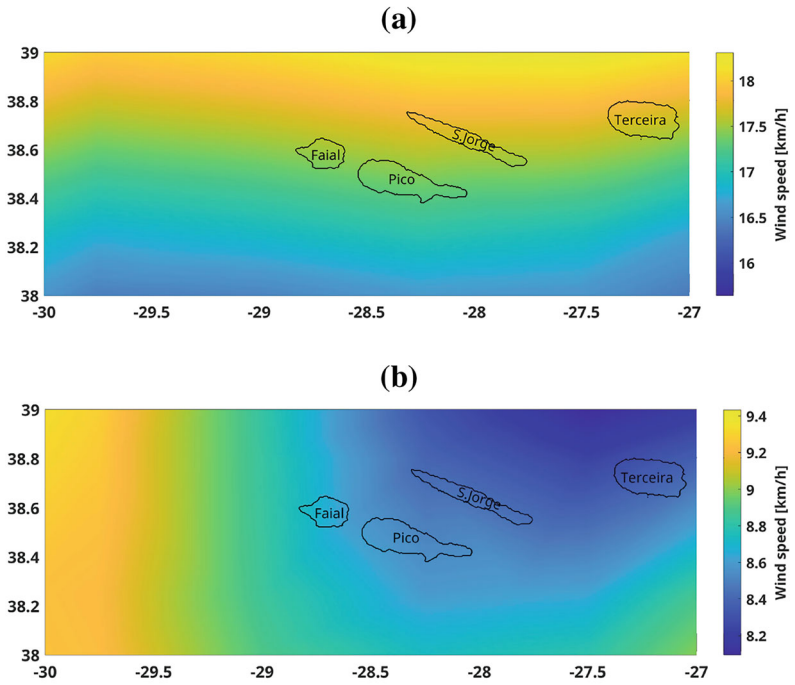



**Fig. 3** Azores wind speed for June 2018: mean (**a**) and standard deviation (**b**) (Source: ECMFW (Hersbach et al. 2018). Plot (**a**) reprinted from (Jesus et al. 2022))

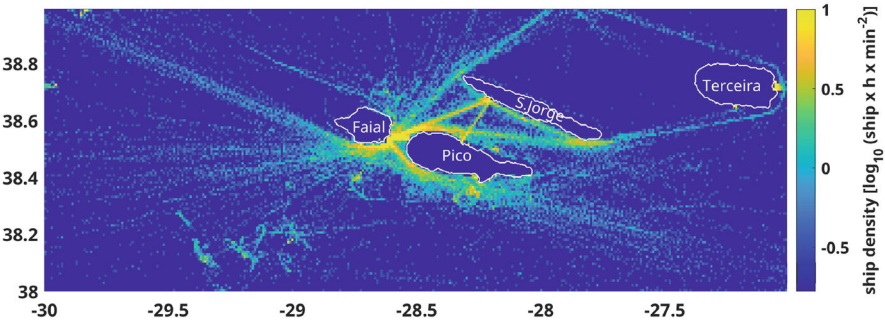



**Fig. 4** Cumulative ship density for the month of June 2018 (Source: AISHub. Figure reprinted from Jesus et al. (2022))

## Ship Traffic

Figure 4 shows the cumulative ship distribution as recorded on the community-shared platform AISHub (ship Automatic Identification System data exchange, www.aishub.net) for the study area during the whole month of June 2018. The

month of June was chosen since it approximately coincides with a period of high probability of the presence of cetaceans in the area. Data is shown as logarithm base 10 of ship density evaluated in ship x hour per square arc minute. That is, two ships for half an hour each on a square of approximately 1 nautical mile side will give rise to a ship density of 1 and will therefore appear as 0 in this figure.

## Acoustic Recordings

Three ecological acoustic recorders (EARs) (Lammers et al. 2008) were positioned at the locations CA, IN, and MG shown in Fig. 5. The EARs are deep water-rated recorders for long-term deployment with a sensitivity between −193 and −194 dB re 1 V/$\mu$Pa over the band 20–1000 Hz. During this experiment the data was sampled at 2000 Hz, with a 16-bit resolution and a 47.5 dB total chain gain. Antialiasing was achieved thanks to the usual low-pass filter to avoid out-of-band energy contamination. A high-pass filter was set at 20 Hz to attenuate static pressure low-frequency oscillations. SPL was estimated from 1 s duration Hann-windowed intervals with 50% overlap using the PAMGuide tool (Merchant et al. 2015) running under MATLAB® which is commonly used in the soundscape community. (See (Merchant et al. 2015) supplementary information with MATLAB® and R codes.) The three recorders at locations CA, IN, and MG were moored at depths of 484, 200, and 200 m, respectively. The received data was averaged in 10-min intervals and integrated over 1/3-octave (base 10) sub-bands in the 40–1000 Hz frequency band.

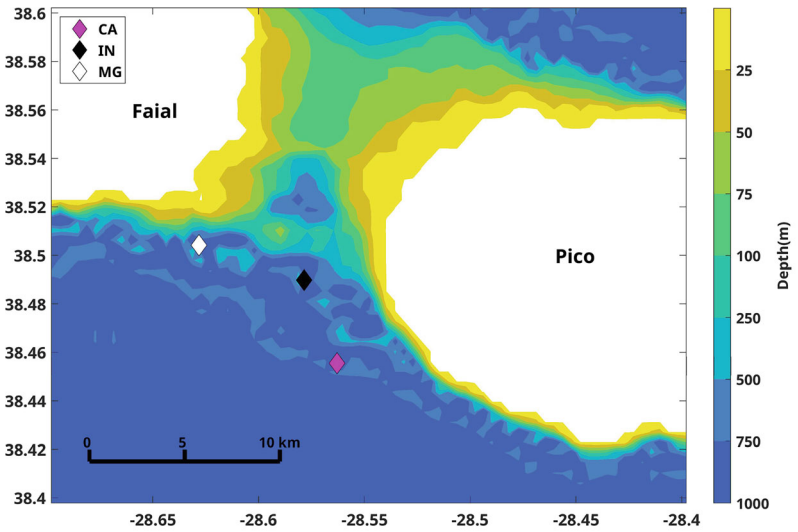



**Fig. 5** Detailed bathymetry with recorders' location on the Pico side (CA), in the channel (IN), and on the Faial side (MG). Recorder's depths are 484 m for CA and 200 m for IN and MG

## Ocean Sound Modeling

Ocean sound modeling was performed for the shipping noise and wind sound components in the frequency band 40–1000 Hz. Thus, model sound-level output $L_m$ may be written as

$$L_m(t,f,\mathbf{r}) = 10 \ \log_{10}\left[\sum_{q=1}^{Q_t} 10^{\left[L_{Hq}\left(t,f,\mathbf{r};\mathbf{r}_q\right)+L_{Sq}(f)\right]/10} + 10^{L_w(t,f,\mathbf{r})/10}\right], \qquad (1)$$

where the first term represents the noise level due to a distribution of $Q_t$ contributing ship sources of the individual level $L_{Sq}(f)$ (in dB) propagated from the $q$th source with a propagation loss $L_{Hq}(t, f, \mathbf{r}; \mathbf{r}_q)$ (also in dB) between source location $\mathbf{r}_q$ and receiver location $\mathbf{r}$ and $L_w$ is the surface wind-generated sound level also in dB, all for time $t$, frequency $f$, and location $\mathbf{r}$ (latitude, longitude, and depth). Note that in this formulation $L_{Hq}$ is taken as negative, since it represents the sound loss along the propagation path. Even acknowledging that the RANDI model is now widely used for source-level modeling (Breeding et al. 1996; MacGillivray and de Jong 2021) for a matter of simplification a lookup table-style source-level model for coefficient $L_{Sq}$ was used, deduced from historical measurements according to ship type from (McKenna et al. 2012). A high number of contacts of sailing vessels was found in the AIS data, and the adopted procedure was that of assigning them to a source level of 1% of that of cargos ≈ 20 dB (see Soares et al. (2020) for details). Source depth for the various ship types was set according to Scrimger and Heitmeyer (1991). Transmission loss is estimated with the Kraken normal mode model (Porter 1991) using the environmental information described in section "Environmental Data." Due to the typical deep water propagation conditions of the area, bottom properties are expected to play a minor role. Therefore, a generic bottom composed of a 10-m-thick sediment layer over a semi-finite hard rock half space was used (see Soares et al. (2015) for detailed properties).

The wind sound model was deduced from the ECMWF database (Hersbach et al. 2018) using the data set shown in the previous section, and the method proposed by Kewley et al. (1990) assumes that the sound generated at the sea surface results from a two regime mechanism: below 8–10 km of wind speed at low sea state and one above that speed involving the formation of white caps. The laborious step of estimating the wind source level is described in detail in the companion paper (Jesus et al. 2022).

## Sample Field Calibration

According to the sound model constraints and simplifications set above, the standard procedure for field calibration follows by estimating sound maps in 1/3-octave (base 10) frequency bands, represented by their center frequencies. At each

center frequency, the SPL time samples over the whole month are sorted into 1 dB bins to form histograms for both data observations and model predictions.

The sample field calibration procedure proposed in Jesus et al. (2022) is based on a linear variable transformation from the model to the data distribution, as described in more detail as follows. Let us assume that $p(L_m)$ and $p(L_d)$ are the model and the data distributions, with means and variances, respectively, given by $\left(\mu_m, \sigma_m^2\right)$ and $\left(\mu_d, \sigma_d^2\right)$. We seek to determine a model-calibrated distribution $p(L_{mc})$ drawn from the model samples, but with the same mean and variance than $p(L_d)$, by the linear scale transformation

$$L_{mc} = AL_m + B. \tag{2}$$

It can be easily found, from the identification between means and variances of $p(L_{mc})$ and $p(L_d)$, that $A^2 = \sigma_d^2/\sigma_m^2$ and that $B = \mu_d - A\mu_d$ which when inserted in (2) allow to determine the change scale of $L_{mc}$ and therefore the calibrated distribution $p(L_{mc})$. Calibration coefficients will be determined for each 1/3-octave band, so that the full calibration map will be obtained through the coefficient set $\{A(f_k), B(f_k); k = 1, \ldots, K\}$, where $K$ is the number of frequency bands. However, as we will see below, it is not sufficient to have the calibrated distributions, since in order to obtain a broadband estimate, the actual sample pressure power field needs to be summed across band. Replacing the levels in (2) by their respective pressure power definitions $L_* = 10\ \log_{10} P_*$ and solving for the calibrated power $P_{mc}$ give

$$P_{mc} = P_m^A 10^{B/10}, \tag{3}$$

which allows to transform power samples according to the calibration mapping.

When the acquired data is hardware calibrated, validated, and free of interferences and self-noise, the proposed sample field calibration may be useful for correcting model systematic biases through the adjustment of the model data distribution mean and variance. This applies to the time intervals and recording locations where the data is gathered, which gives no guarantees for the validity of the correction coefficients for other areas or time frames. The domain of validity of the calibration set depends on a number of factors and, in particular, on the origin of the data-model mismatch.

## Broadband Excess Noise Level

ENL is defined as the difference (in dB) between the measured and the baseline SPL for a given time, frequency, and space coordinates, expressed as

$$L_E(t,f,\mathbf{r}) = L_T(t,f,\mathbf{r}) - L_b(t,f,\mathbf{r}), \tag{4}$$

$$= 10 \ \log_{10} \frac{P_T(t,f,\mathbf{r})}{P_b(t,f,\mathbf{r})}, \tag{5}$$

where in (4), $L_E$ is the ENL, $L_T$ is the total sound level, and $L_b$ is the baseline ocean sound level without anthropogenic pressures. Relation (5) is readily obtained by replacing sound levels $L_T$ and $L_b$ by its respective pressure power counterparts $P_T$ and $P_b$, using $L_* = 10 \ \log_{10} P_*$. The total sound level $L_T$ is drawn either directly from observation or as the modeled sound level $L_m$ given in (1). The baseline (or background) ocean sound level is normally obtained from the modeled wind sound level $L_w$ (or the $P_w$ pressure power). Thus, replacing $P_T = P_S + P_w$, where $P_S$ is the shipping term power, and $P_b = P_w$ in (5) allows to rewrite ENL as

$$L_E(t,f,\mathbf{r}) = 10 \ \log_{10} \left[ 1 + \frac{P_s(t,f,\mathbf{r})}{P_w(t,f,\mathbf{r})} \right]. \tag{6}$$

Under these assumptions, the instantaneous ENL given by (6) will be always $\geq 0$, as expected, and takes into account noise and baseline estimates modeled at a given time and location. Alternatively, the baseline level could be defined as a mean over time, in which case $P_w(t,f,\mathbf{r})$ in (6) would be replaced by the mean $\overline{P}_w(f,\mathbf{r})$. Even in that case $L_E$ remains always $\geq 0$.

The expression above denotes ENL for a single frequency or for a narrow 1/3-octave band around a center frequency $f_k$. Extending this to the broadband case implies power summing (and not power averaging as was proposed in Farcas et al. (2020)) over the required bands so, in that case, the definition for broadband ENL would be given by Jesus et al. (2022)

$$L_E(t,\mathbf{r}) = 10 \ \log_{10} \left[ \frac{\sum_{k=1}^{K} P_T(t,f_k,\mathbf{r})}{\sum_{k=1}^{K} P_b(t,f_k,\mathbf{r})} \right], \tag{7}$$

where $K$ was already defined as the number of frequencies or bands. This definition allows to take into account a ratio of energies which are the preferred quantities for determining indicators of harm to marine life.

## Results and Discussion

### The Acoustic Data Set

Figure 6 shows the spectral probability densities for the received data at the three locations CA, IN, and MG in plots (a) to (c), respectively. Each vertical line is a normalized sample count color-coded image of a data histogram in 1 dB slots, for 1 Hz bin. Each frequency bin is obtained as the sample power spectrum (without averaging) of 1 s duration intervals with 50% overlap throughout the whole month. With this setting, clear colors denote most probable levels and dark colors the least

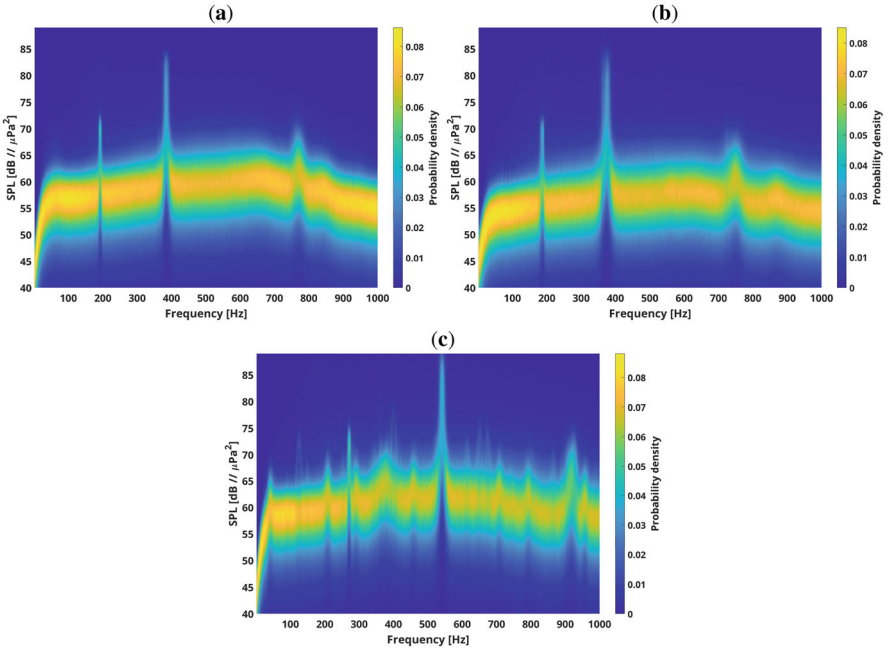



**Fig. 6** Spectral power distribution with 1 s, 50% overlap time slots between 14:00 and 20:00 UTC in June 2018 for recorders at sites CA (**a**), IN (**b**), and MG (**c**) (Figure reprinted from Jesus et al. 2022)

probable, across the frequency band. This figure shows a clear cut-off frequency at 20 Hz for the three recorders, followed by a slight and progressive SPL increase with frequency until approximately 300 Hz. This is the most relevant frequency band for shipping noise. The few spurious high-level peaks present in recorders CA (a) and IN (b) and in large number in recorder MG (c) are believed to be associated to recorders' self-noise. The origin of these peaks is actually unknown but believed to be related to some defecting amplifier or other components in the signal chain. This is clearly an issue on the quality of this particular data set and suggests precautions before its usage for field calibration. Therefore, the focus of this paper is not so much on the final calibrated field result but on showing and detailing the methodology employed for obtaining the result. Nevertheless, the impact of these peaks may be mitigated by filtering the frequency bands clearly off the mean level. The following procedures were adopted: (1) first detect the two highest peaks and (2) filter out the bins of the surrounding frequency band for each recorder of Fig. 6, leading to the result in Table 1. The proposed filtering is not a definitive response to the self-noise impairments found in the data and just attempts to mitigate its effects without compromising the field calibration broadband ENL estimation technique.

**Table 1** Recorder's self-noise filtered bands

| Recorder sites | Bands [Hz] |
|---|---|
| CA | [188–197], [374–393] |
| IN | [184–192], [357–387] |
| MG | [267–274], [528–553] |

## Experimental Data and Model Statistics

Figure 7 shows histograms of the observed data (red), shipping and surface wind sound-modeled data (blue), and field-calibrated data (green) for the MG recording location at the 1/3-octave (base 10) frequency bands between 40 and 1000 Hz. The corresponding broadband field histograms (bottom right plot) are also shown. The MG recorder corresponds to the worst-case self-noise according to the spectral power distributions of Fig. 6. Similar results were obtained for locations CA and IN (not shown). The field-calibrated distributions (in green) were obtained with the proposed method (see section "Sample Field Calibration") with a modification explained as follows. For this case, the data adjustment coefficients $\{A, B\}$ were calculated not independently for the data sets gathered at each location but, instead, from the data samples merged from all three data sets at locations CA, IN, and MG pulled together. This allows for the inclusion of all the observed data in the calibration method, thus providing a higher spatial and temporal robustness than by using sensors individually. This procedure also solves the problem of deciding which area or time interval to calibrate with each sensor. The data merging is also justified by the fact that the three sensor's location may be seen as a single point in the full study area (Fig. 1). The direct result of this merging is that the calibrated distributions are not exactly matched (mean and variance) to any of the data sets. This is illustrated in Fig. 7, for the case of MG, as a slight mismatch between green and red curves. That mismatch is also clear in the broadband case (bottom right plot). Note that in that case, the calibrated broadband distribution is obtained as the sample's sum over frequency and not from a direct calibration between the (broadband) model and data histograms.

The observed data statistical SPL distributions for the 1/3-octave (base 10) bands (Fig. 7 red curves) show a relatively narrow probability density function with a variable mean in frequency and self-noise limited low levels, either due to insufficient gain or poor resolution. As mentioned above, these histograms were obtained after filtering the frequency bands of the two main peaks of the spectral probability densities for each recorder of Fig. 6. Empirical probability density functions for the modeled ocean sound field were obtained for the same setup parameters as the data, i.e., for the recorder coordinates (latitude, longitude, and depth) and for the same time frames (Fig. 7 blue curves). In general, the model SPL distributions tend to overestimate the data in the low-frequency band until roughly 150 Hz, while a better model data fit is obtained above that frequency. In some cases, the model data distribution is bimodal or multimodal, which is also found in the observed data, but with a much larger variance in the model than in the data. Whether that multimodality comes from ship traffic at different times, distances, or ship types is

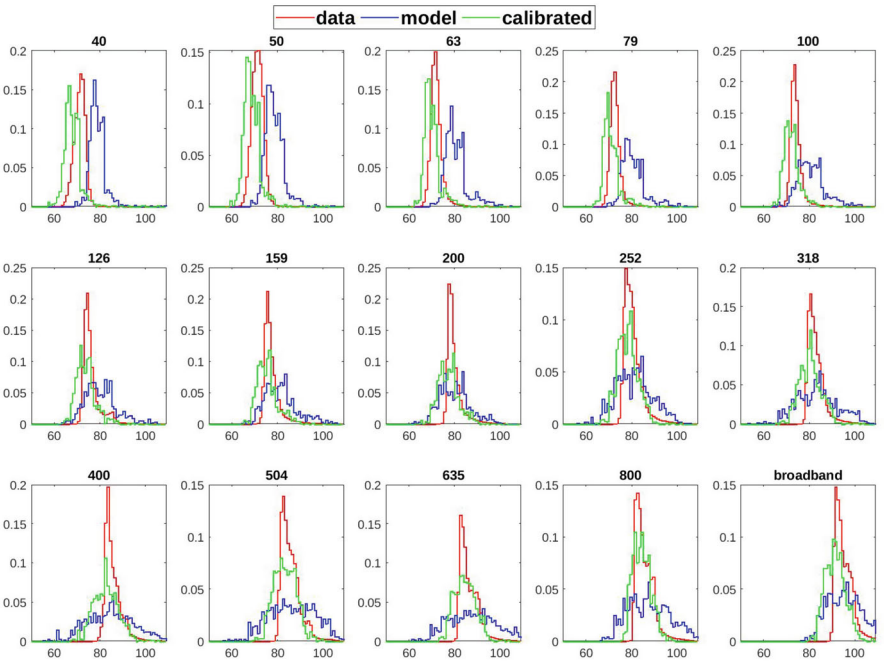



**Fig. 7** SPL histograms for 1/3-octave bands (base 10) and broadband (bottom right), using 1 s time slots with 50% overlap, between 14:00 and 20:00 UTC in June 2018 for recorder location MG positioned at 200 m depth: recorded data (red), shipping and wind model-generated data (blue), and field-calibrated model (green) [axes: probability density (y) in 1 dB re 1 μPa, SPL bins (x)]

unclear. No apparent low-level threshold is found in the modeled data. The high probability peaks larger in the data than in the model are simply due to probability density surface normalization. The differences between data and model were used as the basis for the proposed field calibration method for the wider area even if the real data recording conditions were not ideal.

The next step is to enlarge the modeling area to the four Islands shown in Fig. 1. A sampling interval of 10 min and a spatial grid of square size of 500 m were chosen for computing the wide area sound map. There is no consensus regarding receiver depth for sound models, so 10 m was chosen to represent a depth near the surface at which large whales spend a significant amount of time, and it was also noticed that, for this data set, this depth generally corresponded to the worst-case depth (highest level in general) among the various tests performed. Figure 8 shows the model's estimated mean SPL field for each 1/3-octave band (base 10) in 40–1000 Hz and the broadband case (bottom right plot) for the whole area. The same color scale is used for all frequency bands, including the broadband plot (bottom right). As expected, levels are lower for the lower bands and tend to increase with frequency. Noise peaks up to over 120 dB can be noticed in the channel between Faial and Pico and out of the port of Terceira. Shipping routes are apparent between Faial or Pico to S. Jorge

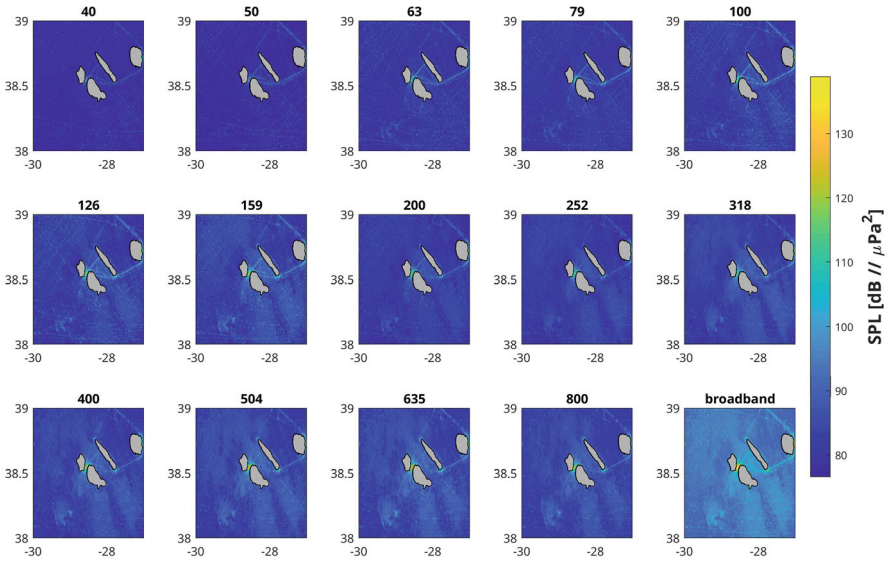



**Fig. 8** Mean SPL for the 1/3-octave band's (base 10) center frequencies in 40–1000 Hz and broadband field (bottom right plot), at 10 m depth for June 2018 [axes: latitude (y), longitude (x) both in decimal arc degrees] (Figure reprinted from Jesus et al. 2022)

and Terceira Islands, as well as other paths to other islands (out of the area). High SPL areas spreading out from these routes can be seen to the southeast of Faial-Pico-S. Jorge and in the shallow banks to the southwest, which is possibly due to the already mentioned fishing activity in the area.

## Field Calibration

The sample calibration method outlined in section "Sample Field Calibration" with the data-dependent modifications set forth in the previous section, illustrated for the MG location in the distributions of Fig. 7, was applied to the modeled data at the recorder locations for verification and then extended to the wide area and for the complete month. Figure 9 shows the frequency evolution of the mean (a) and standard deviation (b) of the sample distributions averaged over the three recorders for data (continuous blue), model prediction (continuous brown), and calibrated separately for CA, IN, and MG (dashed). As expected, there is a clear adjustment in mean and standard deviation from model to data procured by the calibration method. This adjustment is slightly better for IN than for MG and CA recorders. Model predictions overestimate the data over the whole band but more at low frequency (below 200 Hz). The mean absolute error between model and data is in the order of 6.8 dB for the whole data reduced, after calibration, to 3.9 dB.

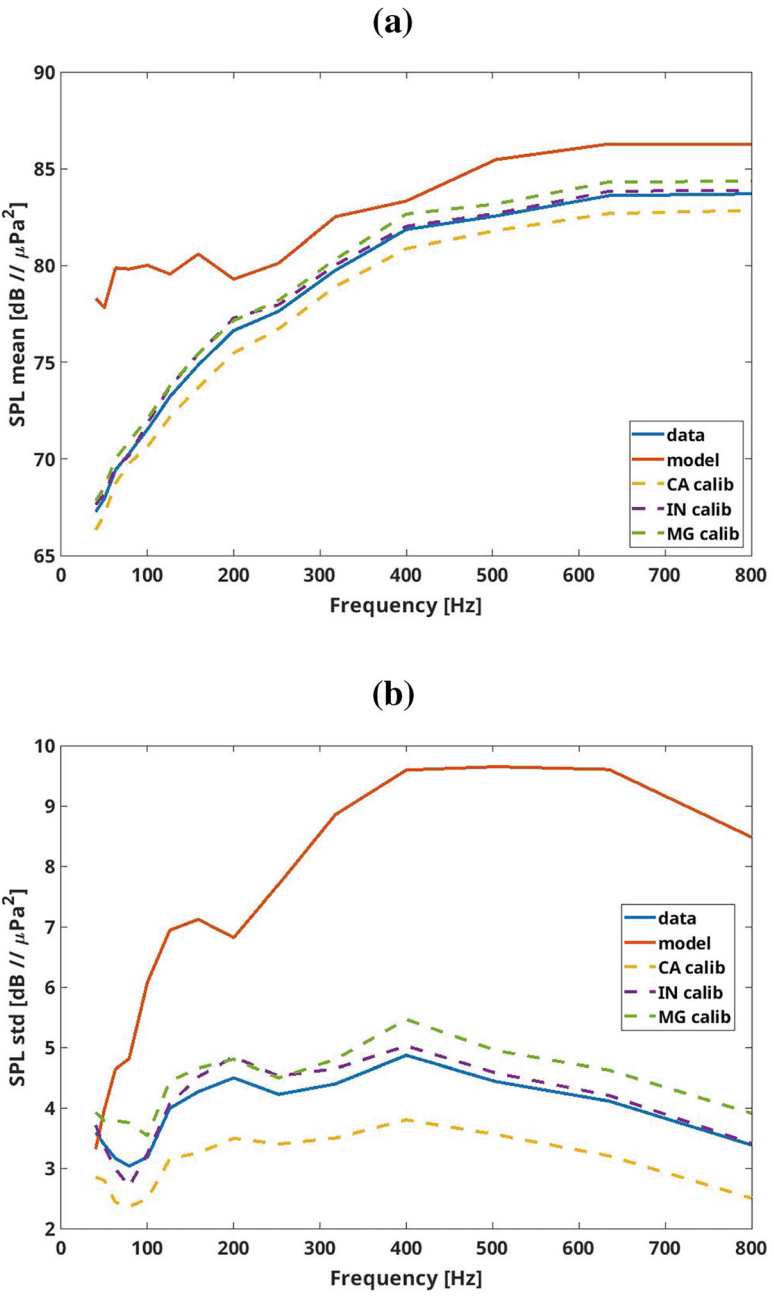



**Fig. 9** Sample distribution's SPL mean (**a**) and standard deviation (**b**) averaged over the three recorders for data (continuous blue), model prediction (continuous brown), and calibration separately shown for CA, IN, and MG (dashed) versus 1/3-octave band (base 10) center frequencies

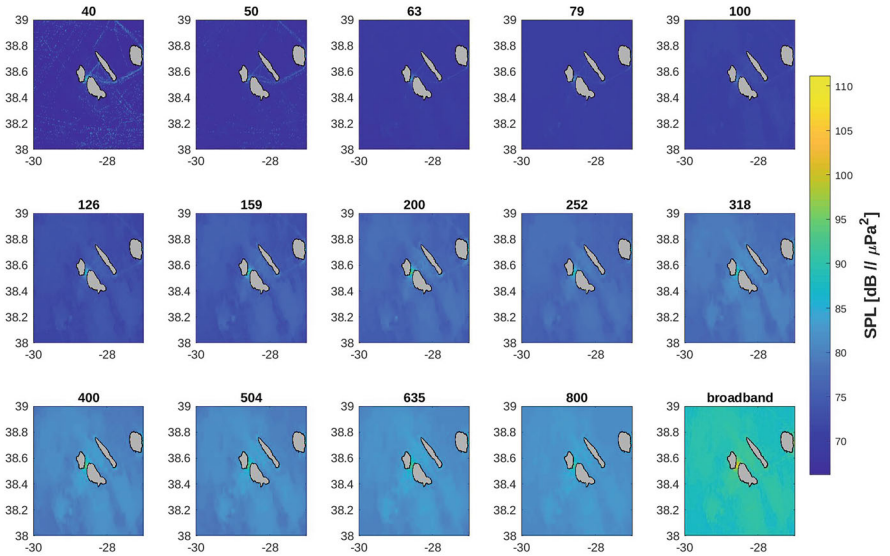



**Fig. 10** Field-calibrated mean SPL for the 1/3-octave band (base 10) center frequencies in 40–1000 Hz using (6) and broadband using (7) (bottom right plot), at 10 m depth for June 2018 [axes: latitude (y), longitude (x) both in decimal arc degrees]

Applying the calibration to the wider area allows to obtain the mean SPL results shown in Fig. 10 for the 1/3-octave band center frequencies and the whole month of June 2018. As expected, levels are overall lower than before calibration, and surfaces are flatter, because variance has, in general, been reduced.

## ENL Estimation

The wind model is used as baseline for estimating the instantaneous ENL using definition (6) and in its broadband form using (7). Figure 11 shows the 95, 75, and 50 percentiles of the broadband instantaneous ENL field at 10 m depth before calibration. ENL values up to 35 dB re 1 $\mu$Pa$^2$ were estimated in the channel between Faial and Pico Islands and from 15 to 25 dB in extensive regions away from the islands but only for 5% of the time.

Figure 12 shows the predicted ENL statistics for the same parameters as in Fig. 11 but using the $\{A, B\}$ field calibration coefficients deduced from the observed data merged for the three recorders, as explained above. The calibration process tends to approach the model to the data, imposing lower mean levels in the lower-frequency bands and narrower distributions (smaller variances), which is consistent with the differences between Figs. 11 and 12. There are a few ENL high peaks reaching over 15 dB at the usual locations (channel between Faial and Pico) and only for small time periods (5% and 25% of the month). Comparison of

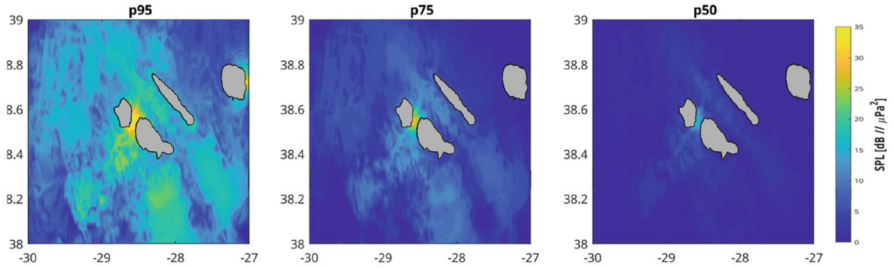



**Fig. 11** Model predicted instantaneous broadband excess noise level (ENL) 95, 75, and 50 percentiles in the 40–1000 Hz band at 10 m depth for the whole area, in June 2018 with a spatial and time resolution of 500 m and 10 min, respectively [axes: latitude (y), longitude (x) both in decimal arc degrees] (Figure partially reprinted from Jesus et al. 2022)

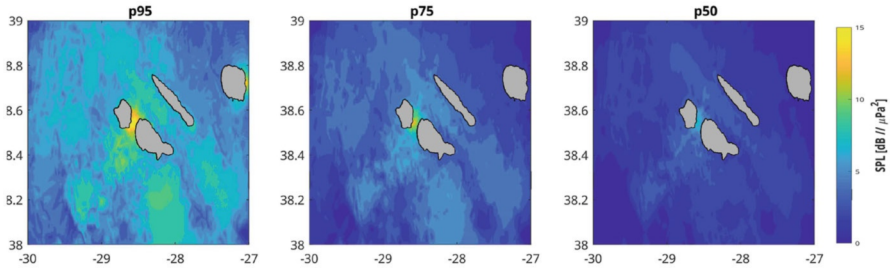



**Fig. 12** Field-calibrated predicted instantaneous broadband excess noise level (ENL) 95, 75, and 50 percentiles in the 40–1000 Hz band at 10 m depth for the whole area, in June 2018 with a spatial and time resolution of 500 m and 10 min, respectively [axes: latitude (y), longitude (x) both in decimal arc degrees] (Figure partially reprinted from Jesus et al. 2022)

figures before and after calibration shows an ENL mean decrease of approximately 8 dB re 1 $\mu$Pa. The validation of the calibration process is not an easy task due to the lack of observations both in time and space. For example, the area that extends from the channel between Pico and S. Jorge to the south and northeast shows prevailing noise levels that are difficult to objectively validate. We may speculate that these levels, which occur in a small portion of the time, might be associated with the shipping lane to Terceira that encounters favorable propagation. Thus, it can be put forward that the proposed calibration methodology provides a balanced adjustment for systematic modeling errors coherent with patterns of the study area. One of the features of the proposed calibration method is that it is based on a statistical behavior of the data in user-defined time lapses of the available data sets and thus perfectly adaptable to other areas and periods of the year.

## Conclusions

The Azores is an important habitat for cetaceans, but it is also a crossroad of several long-haul shipping routes, and a significant part of its economy is based on marine traffic around and between islands (ferries, leisure boats, small cargo and fishing). The ship traffic density around the islands shows some long well-identified routes between islands and from the islands to other destinations outside of the study area. There is also some traffic scattered well away of the islands, probably due to intercontinental traffic. Local ships connect close by ports for small leisure trips, ferries, or fishing. Noise propagates away from intense shipping lanes if and when acoustic propagation conditions are favorable. Marine traffic produces ocean noise that impacts on marine species. The Azores is also challenging for acoustic monitoring because it is a vast deep ocean area with a few islands for supporting permanent monitoring buoys or platforms.

Acoustic data recordings at the three locations to the southwest of the islands of Faial and Pico are in general coherent with AIS ship traffic density in the area. Assuming that model data may be systematically biased, field calibration is extended to a larger area covering four islands of the Azores archipelago for the full month of June 2018. The results show that field-calibrated broadband ENL sound maps are coherent with the ship traffic in the study area, revealing a number of potentially threatening hotspots due to sound propagation conditions off shipping routes and vessel traffic patterns between islands.

The validation of the results is an impossible task without the help of additional direct observations in space and time. However, the proposed broadband ENL formulation seems to give a good superposition with the estimated marine traffic activity in and around the central Islands of the Azores archipelago. More surprising were the vast areas of high ENL to the southeast of the central islands that can be asserted to sound spreading from the shipping routes from these islands to/from Terceira. However, intense spreading in this area only occurs for a small portion of the time, at least during the considered month. The attempt for using localized data for calibrating model predictions in the wide area produced two effects: one was the overall ENL reduction and the other a decrease in the dispersion for most of the time. However, the areas with higher ENL are consistent with direct observation of shipping. Although there is no other comparable study for this area, excess levels seem to be relatively low ($<$10 dB, 50% of the time) during the study period. It is possible that the wide area might be too large for an effective calibration with only three recorders relatively close to each other.

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# Revised Danish Guidelines for Underwater Noise from Installation of Impact or Vibratory Driven Piles

106

René Smidt Lützen, Søren Keller, and Jakob Tougaard

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**Abstract**

Since 2016, underwater noise from the construction of offshore wind in Danish waters has been addressed by guidelines from the Danish Energy Agency. These guidelines were updated in 2022 and specify technical methods for performing numerical prognosis and on-site measurements. Acoustic criteria are stated for compliance, including Permanent Threshold Shift (PTS) and behavioral impact. These criteria are based on auditory frequency-weighted levels.

Impact pile driving and vibratory pile driving installation techniques are addressed, with separate methods for modeling and measurements. Concession holder is required to carry out a prognosis to estimate the environmental impact using the given sound source and propagation properties and calculate the acoustic metrics experienced by a receptor (marine mammal) while it is fleeing away from the noise source. The prognosis must be carried out for two to three scenarios, on fully numerical or semi-empirical basis: reference case, planned construction case, and if relevant a specific case including acoustic deterrent device (ADD).

For comparison with measurements during pile installation, the prognosis is required to provide predictions for certain acoustic metrics that are suited for direct measurements.

On-site measurement of underwater sound during installation is required for two purposes: verification of the propagation model used for the prognosis and demonstration of compliance with the acoustic criteria.

**Keywords**

Acoustic criteria · Pile driving · Acoustic deterrent device · Noise prognosis · Frequency weighting · PTS · Behavioral response · Marine mammals

## Introduction

Underwater noise from percussive (hereafter "impact") pile driving is recognized as having a substantial impact on marine mammals. For this reason, the Danish 2016 Guidelines for underwater noise from installation of impact-driven piles were made (Danish Energy Agency 2016). In that first version of the guidelines, only impact in the form of permanent injury to marine mammal hearing was addressed. The reason for this was that at the time of writing, the empirical evidence of other forms of impact was found to be insufficient to establish regulatory thresholds. However, as new empirical evidence became available, a revision of the guidelines became pertinent. The relevant new material was extracted in a series of technical reviews (Tougaard 2021a, b; Tougaard et al. 2021, 2022) that served as background for the

revised guidelines. The most important changes from the 2016 guidelines are inclusion of criteria for behavioral disturbance of marine mammals and introduction of auditory frequency weighting for different functional hearing groups. Furthermore, the guidelines now specify criteria and procedures for the use of acoustic deterrent devices (ADDs), adapted procedures for impact and vibratory driving, and procedures for calculation of distance-to-threshold in assessments.

The guidelines relate to a set of standard conditions, which are usually included in the construction permit for Danish offshore windfarms. Standard conditions, the guideline document itself, and the background reports are available from the Danish Energy Agency website, www.ens.dk. This text presents an overview of the guidelines. For correct application of the guidelines, the reader must consult the guideline document itself (Danish Energy Agency 2022).

## Regulations for Underwater Noise

Over that past several decades, a steady increase in anthropogenic underwater noise has been observed in the seas, e.g., McDonald et al. (2006). The associated risk of a negative impact on a broad range of marine life has motivated regulatory initiatives worldwide. Regulation for underwater noise is an evolving topic and is handled differently among regulators. A few examples of international (European) regulatory frameworks, according to (Lucke 2020), include:

- EU Habitats Directive for conservation of natural habitats and of wild flora and fauna (European Commission 1992). Does not explicitly mention underwater noise but regulates activities and projects that can potentially injure and/or disturb animals, in this context in particular marine mammals.
- Marine Strategy Framework Directive (MSFD) (European Commission 2008). European legislation that considers a multitude of anthropogenic "stressors" and cumulative effects. Specifically addresses underwater noise and requires that EU member states establish thresholds for good environmental status with respect to underwater noise that can impact marine organisms negatively (European Commission 2017).
- The Baltic Marine Environment Protection Commission (HELCOM) has adopted a Regional Action Plan for underwater noise (Helcom 2021), which specifies actions to reduce impact from, among other sources, impact pile driving.

In Danish waters, the legal framework for offshore wind farms is the Act on Promotion of Renewable Energy (Danish Government 2008). On the technical side of regulation, significant variation exists among countries and frameworks. A multitude of acoustic metrics are referred to, and in some cases, auditory frequency weighting of metrics is mandated. In recent years, international standardization on this topic is gradually coming into shape (International Organization for Standardization 2017a,b), but the application differs greatly from one regulator to another.

For regulation of underwater noise in the context of offshore wind farms, a few country-specific examples are included to illustrate the diversity:

### USA

The seminal review by Southall et al. (2007) introduced the onset of permanent hearing loss (PTS) as a precautionary criterion for acoustic injury in marine mammals. For this purpose, marine mammals were divided into functional auditory groups, with different sets of criteria for impulsive and non-impulsive types of noise. For each type of noise and each functional hearing group, a double criterion was established, such that for each functional hearing group two levels were specified, and none of which are allowed to be exceeded. The first criterion is a cumulative sound exposure level (abbreviated as $SEL_{cum}$), frequency weighted with an auditory frequency weighting curve specific for the functional hearing group, and the second criterion is an unweighted zero-to-peak sound pressure level. These guidelines were updated recently (National Marine Fisheries Service 2018) (Southall et al. 2019).

### Germany

For more than a decade, German Federal Maritime and Hydrographic Agency (BSH) have imposed a dual-metric criterion for pile driving (BSH 2011), aligned with recommendations from the Environmental Protection Agency (German Federal Ministry for the Environment and Nuclear Safety 2013) and specifically targeted protection of harbor porpoises (*Phocoena phocoena*). Evaluation is at 750 m range from the piling, in terms of the single-strike sound exposure level and peak sound pressure level without frequency weighting. For measurements during construction, percentile statistics are required, calculated from measurements on a large number of individual pile driving strikes. The single-strike sound exposure level (unweighted) is not to exceed 160 dB re 1 $\mu Pa^2$ s and peak-to-peak pressure not to exceed 190 dB re 1 µPa.

### Belgium

In Belgium, the underwater noise from piling must comply with a zero-to-peak sound pressure level threshold (185 dB re 1 µPa) at a range of 750 m (Rumes, 2016). This criterion is without frequency weighting.

### Denmark

In 2016 the Danish Energy Agency published its first version of the guidelines for underwater noise, addressing offshore pile installation (Danish Energy Agency 2016). The main metric considered was the sound exposure level, cumulated over all pile strikes within piling of a single foundation. Corresponding threshold values were not stated directly in the guidelines, as the intention was for these to be specified in the conditions of the construction permit. No frequency weighting was required in the 2016 guidelines.

As a novel concept in a regulatory context, the first version of the Danish guidelines introduced a numerical framework for calculating the $SEL_{cum}$ of a moving receiver (animal fleeing from the noise source at a constant speed).

In the current version (Danish Energy Agency 2022), the concept of the constant speed-fleeing animal has been kept, and several new features have been implemented. The most significant being frequency weighting according to species group-specific weighting functions with associated group-specific acoustic criteria.

## Scope of the Revised Guideline

For installation of offshore wind turbine foundations, the requirements of the Danish Energy Agency for minimizing the environmental impact from underwater noise form part of the conditions of the construction permit. On this background, the concession holder is obliged to demonstrate plans for how to comply with these requirements. This means the concession holder must prepare a prognosis for underwater noise generated during construction. This prognosis must then be used as the starting point for an environmental impact assessment addressing the potential impact of underwater noise on marine mammals and selection of adequate mitigation measures. In addition, it is mandatory for the concession holder to conduct a verification measurement program. The corresponding methodologies, requirements, and criteria are described in the 45-page guideline and presented in a summarized version in this chapter.

The revised version of the guidelines presented here covers both impact pile driving and vibratory pile driving. While other installation techniques exist, these are not within the scope. Similarly, noise from operational wind turbines and service vessels is also not covered. The primary application of the guidelines is therefore installation of large-diameter monopiles and multi-pile foundations such as jackets and tripods. As an additional new aspect, the guidelines present procedures for permitting and impact assessment of acoustic deterrent devices (ADDs) when deployed as part of the mitigation strategy.

The core of the guidelines is a criterion for the onset of Permanent Threshold Shift (PTS) taken as a precautionary proxy for injury, in line with recommendations of Southall et al. (2007, 2019) and following the same grouping of species according to hearing abilities. Only species relevant in the context of Danish waters are considered, thus covering harbor porpoise (VHF cetaceans), small delphinids (HF cetaceans), minke whale (*Balaenoptea acutorostrata*, LF cetaceans), and phocid seals in water (PCW). In terms of metrics, these criteria are stated as $SEL_{cum}$ (sound exposure level) cumulated over all the impulsive sounds received by the animals during piling of one foundation, weighted by the appropriate auditory frequency weighting functions (Southall et al., 2019, Tougaard et al., 2022).

In addition, the guidelines provide threshold values for the evaluation of behavioral reactions to underwater noise in harbor porpoises. These thresholds are expressed as root-mean-squared sound pressure levels over 125 ms ($SPL_{125\ ms}$),

weighted by the appropriate auditory frequency weighting functions (Tougaard 2021a).

During pile installation, measurements must be taken with the purpose of comparison with the prognosis. To that end, and to enable direct comparison, the prognosis must provide predicted values for the following metrics at distances 750 m, 1500 m, and 3000 m:

- Single-strike sound exposure level ($SEL_{ss}$) and single-strike root-mean-squared sound pressure levels ($SPL_{125\ ms}$) for impact driving, unweighted and frequency weighted.
- Sound pressure level (SPL) for vibratory driving, unweighted and frequency weighted.

Currently, behavioral impacts are only addressed in the guidelines to the extent that indicative frequency-weighted thresholds for onset of behavioral reactions (fleeing) are provided. No hard limit to the spatio-temporal extent of the disturbance is provided. Instead, the assessment of the likely impact of the disturbance on the local and regional populations of animals is left to the environmental impact assessment. Temporary habitat loss is thus not addressed in detail by the guideline. However, for example, the reader may refer to Tougaard (2021a). It is, however, expected that this topic becomes further developed in future revisions.

The concession holder must carry out an on-site validation of the model, regardless of how the acoustic model was established. Similarly, it is a requirement to perform on-site measurements and demonstrate that the acoustic criteria are fulfilled.

## New Knowledge Since the First Guideline Version

The empirical foundation for regulation of underwater noise expands rapidly these years, which means that guidelines should always be considered interim and should be revised whenever there is significant new information about the impact of noise, biology of the animals, or change in the overarching legislation. Since the previous guidelines (2016) were established, two important developments occurred, resulting in significantly improved knowledge about prediction of the impact and means for mitigation. These developments were revised recommendations for frequency weighting (Southall et al., 2019) and the availability of efficient noise abatement technologies, e.g., Bellmann et al. (2020).

### Frequency Weighting

The concept of auditory frequency weighting of noise in assessment of its potential to cause auditory damage is well-established for humans (see Houser et al. (2017) for a review) and was introduced in regulation of underwater noise impact on marine mammals by Southall et al. (2007). The proposed M-weighting curves were very

**Table 1** Species of relevance in Danish waters, with associated auditory frequency weighting functions and limit values for cumulative sound exposure level $SEL_{cum}$

| Species group | Weighting function | $SEL_{cum}$ limit (dB re 1 $\mu Pa^2$ s) |
|---|---|---|
| Phocid seals (harbor seal, grey seal) | PCW | 185 |
| Mysticetes | LF | 183 |
| Odontocetes, except porpoises | HF | 185 |
| Harbor porpoise | VHF | 155 |

*PCW*, phocid carnivores; *LF*, low-frequency cetaceans; *HF*, high-frequency cetaceans; *VHF*, very-high-frequency cetaceans

wideband, which meant that the difference between regulation based on unweighted and M-weighted levels in regulation of pile driving noise was very small (Tougaard and Dähne 2017). However, new empirical data on temporary threshold shifts (TTS) produced after the review by Southall et al. (2007) challenged this, as data were more consistent with sharper weighting functions, resembling inverted audiograms (Tougaard et al. 2015). Subsequently, this led to a revision of the weighting functions (Southall et al. 2019). Thus, the revised Danish guidelines now follow recommendations of Southall et al. (2019), which means that the cumulative sound exposure level ($SEL_{cum}$), weighted with the species-appropriate auditory weighting function, cannot exceed levels that are considered to represent a significantly elevated risk of inducing permanent hearing impairment (Permanent Threshold Shift, PTS). For pile driving noise, classified according to Southall et al. (2019) as an impulsive sound, the species group-specific levels of relevance in Danish waters are given in Table 1. The associated auditory frequency weighting curves are shown in Fig. 1.

Southall et al. (2019) operates with a dual criterion, one based on the cumulative sound exposure level, frequency weighted, and one on the unweighted peak pressure. As the $SEL_{cum}$ criterion is expected always to be exceeded before the peak pressure criterion for pile driving noise, only the $SEL_{cum}$ criterion is applied in the guidelines. The guidelines may be extended to other types of impulsive sources, such as seismic air guns, but not to all types. Most important exception is explosions, where the peak pressure is likely to be the better predictor of the risk of injury.

## Disturbance of Behavior

New in the guidelines compared to the previous version is that the impact from behavioral disturbance of marine mammals must now be included in an assessment, and a threshold for evaluating disturbance is provided. This threshold, valid for harbor porpoises, is expressed as an auditory frequency-weighted sound pressure level, calculated as a root-mean-squared (RMS) average over 125 ms, corresponding to the integration time of the mammalian auditory system (Tougaard et al. 2015, Tougaard 2021b). Recent empirical data supports the notion that frequency-weighted sound pressure levels are better predictors of behavioral response to pile driving noise than are unweighted levels (Kastelein et al. 2022). Based on a review

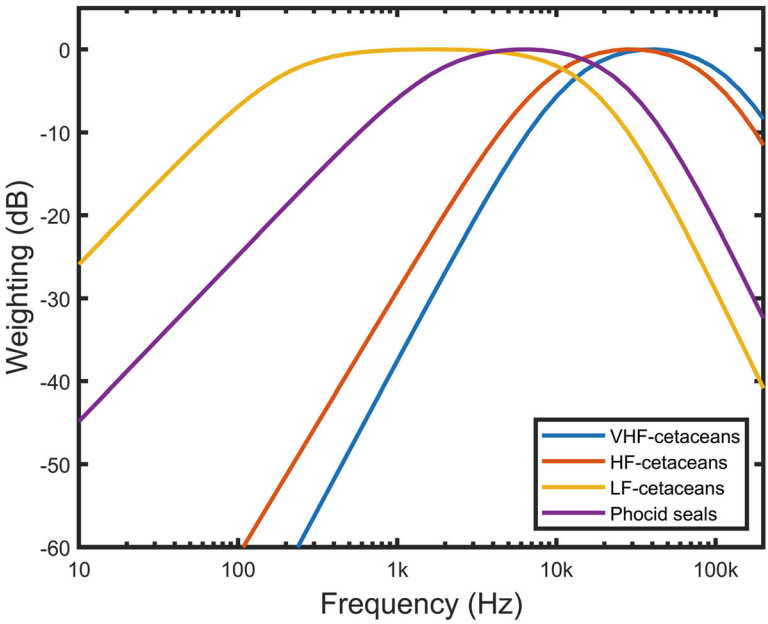



**Fig. 1** Frequency weighting functions proposed by Southall et al. (2019) and Tougaard (2021b) for auditory groups relevant to Danish waters

of experimental data (Tougaard 2021b), including both experiments on animals in captivity and studies of wild porpoises' reactions to full-scale pile driving, a threshold for behavioral reactions equal to 103 dB re 1 µPa, VHF-weighted, is provided.

## Metrics and Terms in the Revised Guideline

Metrics used in the guideline follow the definitions given in ISO 18405 (International Organization for Standardization 2017a) and ISO 18406 (International Organization for Standardization 2017b). A few additional metrics and terms are:

### Max Over Depth Across Water Column

For a fixed range step $r_i$, and considering the entire water column, the maximum metric value is determined, i.e., max over depth (MOD). With $j$ being the vertical grid-point index, MOD of a given metric $L$ is:

$$L_{MOD}(r_i) = \max_j L_j(r_i)$$

Here, all values of $j$ inside the water column are considered.

### Distance to Threshold

Distance to threshold is the largest expected distance from the source where the received level of noise equals the threshold (for onset of PTS or onset of behavioral reactions, whichever relevant). This distance is most often evaluated from a max-over-depth parameter. As such, the distance to threshold (DTT) is used for evaluating the range-dependent variation based on a given acoustic threshold value.

It is worth noting that the sound field in a shallow-water acoustic environment usually decays with distance in a non-monotonous manner. On that background, the numerical evaluation must be implemented carefully to avoid the risk of identifying local features as the global DTT of the transect.

### Exceedance Level

For an acoustic-level metric $L_x$, it is common to define an exceedance level in dB. Applying statistics to $L_x$, the percentage x is the level which during the observation period is exceeded x% of the time. One common observation period is the piling sequence. Taking as an example $L_{90}$, this refers to the metric's level that is statistically exceeded 90% of the time. Analogously, $L_{50}$ corresponds to the statistic median and indicates the level that is exceeded 50% of the time.

## Acoustic Deterrent Device (ADD)

An acoustic deterrent device (ADD) is a technical device designed with the purpose of emitting an acoustic signal that is unpleasant for the target species. The intention is that the ADD's signal causes any target animals to leave the nearby area as a non-lethal, behavioral response. Historically, ADDs were developed for fisheries and the aquaculture industry for deterring marine mammals, thereby reducing depredation and damage to fishing gear and net pens. Over time, its use spreads to other offshore industries including offshore wind and UXO (UneXploded Ordnance) removal and now includes devices known as "seal scrammers" and "pingers." Deployment of ADDs in this context is then as a mitigation device, with the intention of deterring animals out to safe distances before the onset of the main, potentially harmful noise source. An overview is given in McGarry et al. (2020).

In the context of offshore piling, an ADD serves as a marine mammal mitigation technique. In the ideal case, the ADD deters animals away from the potential injury zone. However, thereby the ADD creates an additional disturbance, on top of the disturbance caused by the pile driving itself. If the ADD is too powerful, the disturbance caused by the ADD may exceed the disturbance from the pile driving itself (Dähne et al. 2017), thereby jeopardizing the very mitigation purpose of the ADD.

In the revised guidelines, it is now a general requirement to use an ADD during the construction activity. This applies to any single foundation, with the only exception being cases that are relatively low noise. The ADD is a significant

underwater source of noise itself. In consequence, it is required to assess its acoustic impact. The revised guidelines include a procedure for this specific purpose. The aim is to assure that the distance to threshold for disturbance for the ADD does not exceed the distance to threshold for the pile driving itself, assuring that the ADD does not cause disturbance over a wider area than the pile driving itself. Furthermore, the revised guidelines now require that the ADD is activated at least 15 min prior to start-up of the pile installation activity. If the installation stops for more than 2.5 h continuously, it must not start again before the ADD has been active for another 15 min. This procedure is in line with suggestions presented in Thompson et al. (2020).

## Prognosis Overview

Figure 2 shows a flowchart representation of the sequence behind the required prognosis scenarios. The reader is referred to the guideline document for details, but a few features are highlighted in the following.

Overall, the prognosis is intended for estimating the environmental impact resulting from a given sound source and propagation properties. This is done by calculating the acoustic metrics experienced by a marine mammal receptor. The prognosis must be carried out for two to three scenarios, on a fully numerical or semi-empirical basis.

## Reference Case

This is a worst-case scenario, without noise abatement techniques and without ADD. As such, it provides a standardized setup that allows regulators to track technology

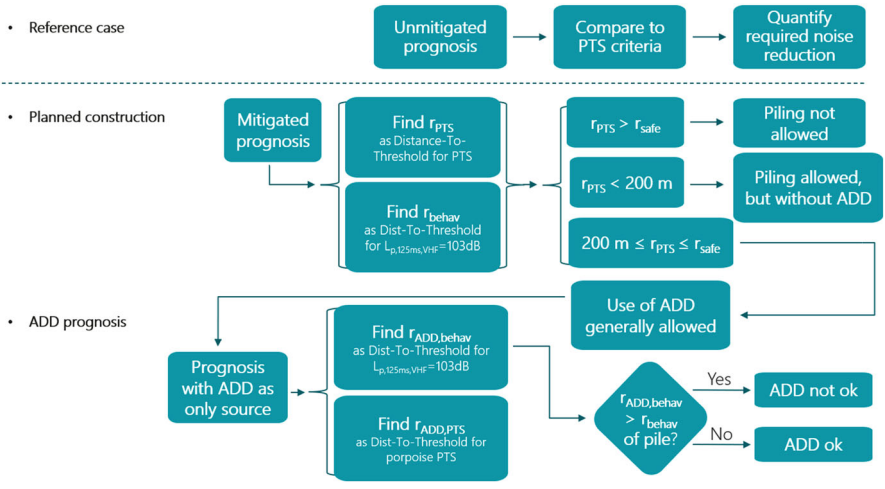



**Fig. 2** Prognosis flowchart. Please refer to the guideline document for full details

improvement across projects. In addition, it represents a worst-case indication of what might happen in case of a malfunctioning noise abatement systems. An example would be bubble-curtain compressor failure.

## Planned Construction Case

In this scenario the prognosis is based on the installation machinery (e.g., hammer) and noise mitigation techniques (if any, e.g., bubble curtain) intended for the construction. It also assumes the use of an ADD prior to the installation (if relevant, see comments below), resulting in a certain minimum displacement of any nearby animals. This distance is called $r_{safe}$ and must be provided for the study by the user. Piling according to the planned construction case may only take place if the $r_{PTS}$, the PTS-related distance to threshold of the piling, is smaller than $r_{safe}$. Furthermore, the resulting $SEL_{cum}$ must comply with the relevant acoustic criteria of Table 1. If these criteria are met, all marine mammals are assumed to be at least at distance $r_{safe}$ at the onset of piling and fleeing directly away from the noise with a constant speed ($v_{flee}$), which in turn means that by the end of the pile driving (for that particular foundation), no animal will have experienced an $SEL_{cum}$ above the PTS onset threshold.

## ADD Prognosis

If the planned construction case results in relatively low noise impact, identified as a very short distance to threshold for the onset of PTS, piling is allowed without an ADD, as there is no significant risk of injury to mitigate. Otherwise, and in most cases for large-diameter piles, a separate prognosis must be done, based on the planned ADD as the only noise source. The intention behind the criteria for this ADD prognosis is to avoid the ADD itself having a larger impact than the piling operation.

## Fleeing Animal Concept

An important feature of the prognosis concept is that of a fleeing animal, introduced to avoid the unrealistic overestimation of predicted impact if it is assumed that the receiver (animal) remains stationary throughout the entire exposure. A numerical framework is given that represents an animal fleeing radially away from the noise source at a constant speed. At present times, a speed of 1.5 m/s is suggested regardless of species, but alternative values (see, e.g., (Tougaard 2021a)) are permitted as long as they are justified. Its practical use has shown that in terms of $SEL_{cum}$ the numerical difference between a stationary vs. constant speed-fleeing receiver is significant and easily more than 10 dB depending on sound field properties. At the same time as being a better approximation to actual behavior, the moving animal assumption may be regarded as reducing an unwarranted but significant worst-case assumption (of a stationary animal) from the prognosis.

## Prognosis Options

It is permitted to base the prognosis on numerical modeling (examples include finite element, parabolic equation, or wavenumber equation schemes) or alternatively using semi-empirical methods. The latter allows for a range of techniques based on artificial sound sources such as airguns.

A general approach stated several places in the guidelines is freedom for choice of method. For several assumptions and methodologies, alternatives are allowed. The requirement is that the user provides adequate justification that the alternative provides similar or better representation than what is stated in the guideline. Examples where this might become relevant are:

- The prognosis formulas for vibratory driving assume constant driving force amplitude over time, which in some cases might be over-simplified. The user might have more detailed information and may modify the $SEL_{cum}$ formulas to accommodate this.
- Similarly, the formulas for impact driving assume a constant time interval between hammer impacts.
- Currently, a fleeing speed of 1.5 m/s is assumed regardless of animal species. If well-documented data justifies other speed(s), these may be used.
- A simple, semi-empirical formula is suggested in the guidelines for correction of differences in applied hammer energy compared to that assumed for the prognosis. The user may apply alternative expressions or approaches for the correction.

## Verification Measurements Explained

It is an inherent challenge of the acoustic criteria of Table 1 that the corresponding $SEL_{cum}$ metric is not easily evaluated from measurements during the installation, particularly because the limit values refer to a non-stationary receiver (a fleeing animal). On that background, it is required for the prognosis to prepare certain sets of data that are more readily derived from field measurements. For the transect identified as having the longest distance to threshold for PTS criteria, the following is calculated:

- Single-strike sound exposure level $SEL_{ss}$ in the case of impact driving.
- Sound pressure level SPL in the case of vibratory driving.

At the construction site, measurements must then be made for two purposes:

- Verification of the propagation model used for the prognosis. This is based on evaluation of the transmission loss, which must be demonstrated to match the prognosis data. The guidelines permit both the use of an artificial underwater sound source and the actual pile driving operation.

- Demonstration of compliance with the acoustic criteria, based on the above-mentioned measurable output from the prognosis. These must be prognosticated in 1/3-octave bands and at specified reference ranges. The measured broadband values corresponding to 5% exceedance levels, with and without appropriate frequency weighting, must be reported.

In both contexts, prognosis and measurements must match within tolerances specified in the guidelines. The idea behind the above sequence is to first establish confidence in the acoustic model, which then leads to confidence in the absolute noise levels resulting from the prognosis.

Finally, there is a risk that the hammer protocol (i.e., impact hammer energy vs. time or vibratory force amplitude) assumed in the prognosis deviates to some extent from that applied during the actual installation. A simple expression is provided in the guidelines to correct for such differences if relevant. After that, the $SEL_{cum}$ of the prognosis must be compared to the limit values of Table 1 to demonstrate compliance.

## Conclusion

New scientific advances in the knowledge of marine mammal hearing led to a revision of the Danish guidelines for underwater noise. The revision kept the basic approach of the original 2016 version, addressing the cumulative sound exposure level of an animal fleeing at constant speed away from the noise source. However, several additions and modifications were implemented, in particular that of frequency weighting based on auditory groups.

It is believed that the current guidelines' approach is an improved representation of the acoustic impact on marine mammals caused by offshore pile driving.

It is expected that future revisions will accommodate updates to reflect new knowledge. In particular, it is foreseen that behavioral response to underwater noise will become better understood from ongoing and upcoming research.

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# Threshold-Based Assessment of Underwater Noise Pollution in French Marine Waters

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Laura Ceyrac, David Dellong, and Florent Le Courtois

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**Abstract**

Since underwater noise has been recognized as a source of pollution, affecting all compartments of underwater ecosystems, efforts have been made to reduce its effects and design management tools to assess and regulate it. Consequently, over the last decades, governance regulations have been implemented internationally, and converging risk assessment methodologies emerged to define threshold values. However, including the complexity of impacts on species and marine habitats at the heart of the process is an ambitious challenge. Given the sparseness of the knowledge, it is difficult to define and set threshold values for common

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regulation and management. The aim of this chapter is to investigate risk assessment conditions in regard to threshold value setting. The assessment framework relies on the definition of a sound pressure-level threshold, related to biological impacts, and temporal and spatial thresholds, related to habitat degradation. As the choice of thresholds still needs to be researched, several ranges of values for each threshold are investigated to estimate their relationships in the decision-making process. This chapter focuses on a practical implementation of a threshold-based assessment over the French coasts. It demonstrates how noise mapping can effectively support regulators to identify and prioritize impacted areas depending on availability of scientific knowledge.

**Keywords**

Shipping noise · Thresholds · Risk · Decision-making · Modeling · Noise map · Underwater noise

## Introduction

Over the last decades, world marine traffic intensity has increased globally, and new navigation routes were established (UNCTAD 2019). Noise generated by the propellers of modern ships can propagate over hundreds of kilometers. It has been characterized as a continuous component of ambient noise at low frequencies, as it can be recorded almost everywhere and all the time (Wenz 1962). Such major modification of the underwater soundscape has been suspected to affect marine species, by masking their communication and the cause of disturbance at the different steps of their life cycle (Erbe et al. 2016; Tyne et al. 2018; Weilgart 2007; Southall et al. 2007; Popper and Hawkins 2016). This observation highlighted by scientists led to a consideration of underwater noise in environmental policies (Duarte et al. 2021; Thomsen et al. 2021; Merchant et al. 2022). The political response to underwater noise requires support from scientific communities to establish (1) reliable assessment methodology and (2) threshold values using the best available knowledge to quantify the risk (Juretzek et al. 2021; Verling et al. 2021; Merchant et al. 2018).

In order to provide the most reliable assessment, stakeholders have to quantify where and when the risk is too important (Ascough Ii et al. 2008). Thus, maps of risks over given periods are usually considered as the main input for management (Halpern et al. 2015). For underwater noise, the pressure covers large areas that could hinder the capacity to make accurate assessment, in particular regarding the lack of consensus in the methodology and in the threshold value definition. It became a major challenge for researchers and stakeholder to provide methodology that reflects the decision-making process to present robustness to uncertainties and data gaps while being sensitive to mitigation measures (Lajaunie et al. 2023).

At the European level, in the Marine Strategy Framework Directive (MSFD 2008/56/EC), a common assessment methodology was adopted in 2021 for underwater noise (Sigray et al. 2021). It aims to be applied across all European waters that includes various bathymetric depths and reliefs, various shipping activities, as well as large biodiversity sensitive to noise. The proposed approach considered that noise levels, time, and space should be assessed according to a decision tree, to be able to identify areas where the risk is too high too often. This risk-based approach follows a DPSIR (driving forces, pressures, states, impacts, responses) framework, by making it possible to measure a phenomenon (noise pollution), identify and monitor pressure (underwater noise), and evaluate the effect of public policies (mitigation measures as speed reduction).

The associated threshold values to quantify the risk were investigated in a second document of the MSFD. Options for spatial, temporal, and noise-level threshold values for continuous noise were adopted in 2022 (Sigray et al. 2022). For all single months assessed over a year, the recommendation has been that the exposed area of the habitat should be equal to or below 20%. The remaining threshold value (or range of threshold values) on the noise level was suggested to depend on additional biological knowledge. The choice of the threshold values on noise level remains to be defined at the subregion level. Still it appeared that defining thresholds requires a deep knowledge of the biology of species which are often lacking or incomplete (Geijzendorffer et al. 2016; Welsh et al. 1996). However, these gaps should not be an obstacle for the assessment of the impact of noise on the environment.

In this chapter, a parametric approach is performed on the assessment methodology for various marine subregions of French Economic Exclusive Zone (EEZ) in 2019. The objectives are twofold: (1) to perform a sensitivity analysis of the methods and impacts on the risk assessment and (2) to provide and validate possibilities for threshold values. For these reasons, an approach integrating scenario multiple threshold values directly related to the pressure makes it possible to test several scenarios of risk. It enables to consider assessment with limited biological knowledge, at large scale and for the three dimensions of noise pressure, the spatial, the temporal, and the noise level. Thus, this chapter proposes to investigate the relationship between noise dimensions and how the risk assessment is impacted by the choice of threshold set on each dimension. The chapter is organized as followed. Section "Material and Method" introduces the area, the noise mapping process, and the decision tree defined in Sigray et al. (2021). Section "Results" presents the results of the parametric assessment and support for the interpretation. Section "Discussion: Implementation of Risk Assessment" discusses about the marine subregion properties for the assessment and concludes on possibilities to set threshold values.

Symbols and notations used in the chapter are presented in Table 1.

**Table 1** Table of symbols

| | | |
|---|---|---|
| $m$ | Cell number | |
| $N(m, t)$ | Noise level in the cell $m$ at time $t$ | (dB re 1 $\mu$Pa) |
| $\mathbf{N}$ | Matrix of noise levels | (dB re 1 $\mu$Pa) |
| $T$ | Assessment period | |
| $M$ | Assessment area | |
| $\lambda$ | Threshold value on the level | (dB re 1 $\mu$Pa) |
| $\tau$ | Threshold value on the relative time | (% of $T$) |
| $\mu$ | Threshold value on the relative area | (% of $M$) |
| $\boldsymbol{\lambda}$ | Test threshold values for level $\lambda$ | **[**55:130**]** dB re 1 $\mu$Pa |
| $\boldsymbol{\mu}$ | Test threshold values for the number of cells $\mu$ | **[**0:100%**]** |
| $\boldsymbol{\tau}$ | Test threshold values for relative time $\tau$ | **[**0:100%**]** |
| $Z$ | Number of days in the cell when $\lambda$ is exceeded | % |
| $Y$ | Number of cells where $\tau$ is exceeded | % |
| $X$ | Status of the habitat | Boolean value |

## Material and Method

### Study Area

This chapter focuses on the French Economic Exclusive Zone (EEZ) off metropolitan waters on the Atlantic coast, composed by three marine subregions, namely, English Channel and North Sea (ECNS), Celtic Seas (CS), and Bay of Biscay (BoB) in January 2019 (Fig. 1). The three subregions are geographically composed of 44° and 51° North, −10.0° West, and 2.0° East.

### Noise Mapping

The shipping noise maps are computed by propagating the noise from shipping activities in the marine environment. The modeling process had been documented in several studies (Le Courtois et al. 2016; Sertlek et al. 2016; Folegot et al. 2016; Jesus et al. 2022). Depending on the environment and the data availability to consider, the process can differ, but they follow the same general principles: (1) computation of the source levels of the ships and (2) computation of the propagation losses between the ships and a receiving position and depth (Aulanier et al. 2016).

Automatic Identification System (AIS) provide the locations of the sources and additional information for the modeling of the source level modelling. Speed, length, and categories of the ships are usually considered as the dominant parameters in the source level (McKenna et al. 2012). As there is no consensus, the RANDI 3 model is used in this chapter (Breeding Jr et al. 1996).

The equivalent source is assumed to emit at a 5 m depth. In spite of being widely used, comparative studies with measurements have criticized results of RANDI 3

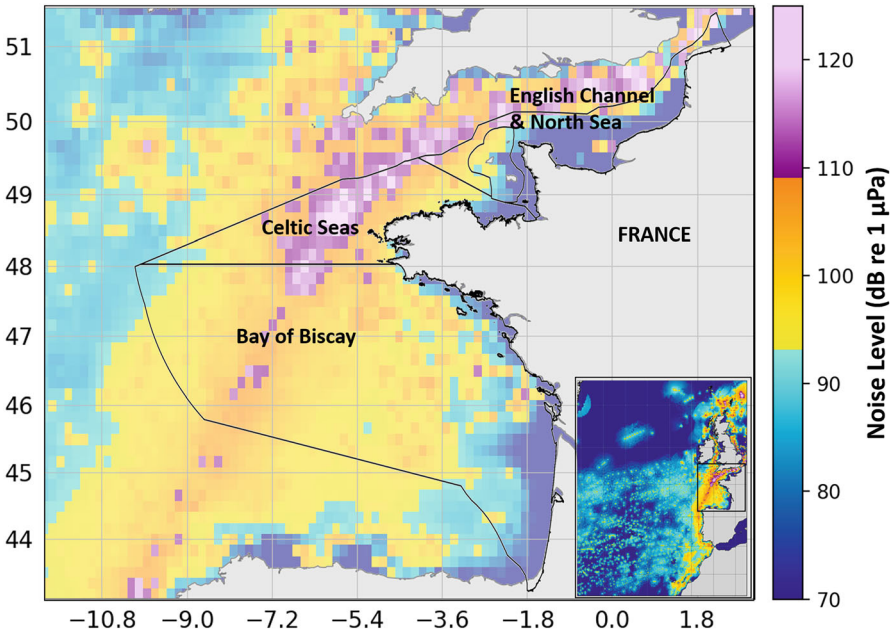



**Fig. 1** Noise levels from for the month of January 2019 and marine subregions of French Economic Exclusive Zone

**Table 2** Environmental and anthropogenic data used for the noise map computation. The seabed is classified according to its geoacoustic properties (density, compressive $c_P$ and shear $c_S$ wave celerity, and attenuation $\alpha_P$, $\alpha_S$)

| Physical quantity | Unit | Provider | Spatial res. | Temporal res. |
|---|---|---|---|---|
| **Bathymetry** | $m$ | SHOM | $5'$ | NA |
| **Sound speed** | $m.s^{-1}$ | SHOM | $0.5°$ | Monthly |
| **Seabed nature** | $\rho$, $c_*$, $\alpha_*$ | SHOM | $15'$ | NA |
| **AIS** | *Density*(*s*/*s*) | Spire | $1'$ | $50'$ |

model and the equivalent source depth, in particular at low frequencies (Wales and Heitmeyer 2002; Sarkoulis et al. 2017; McKenna et al. 2012). Other modeling paradigms were proposed recently, in particular to match a large set of data for newer ships to long-term acoustics measurements (Wales and Heitmeyer 2002; Simard et al. 2016; MacGillivray and de Jong 2021).

The propagation equation is solved using RAMS (Range dependent Acoustic Modeling, with shear wave modeling at the seabed interface) model (Collins 1995, 1998). RAMS model uses a parabolic approximation which is relevant for range-dependent medium at low frequency. Bathymetry, SSP, and seabed data used to compute the propagation losses are listed in Table 2.

The resulting noise levels in a given cell $m$ (whose center is located by its longitude and latitude), at a given time $t$ and depth $z$, are $N(t, m, z)$. In the following,

the maximum value over depth is selected in order to simplify notation and follows a conservative approach. Noise maps are computed with a spatial resolution of 10 arc minutes with a daily interpolated average AIS situation. The resolution is mainly restrained by the computation cost. These units were chosen as a compromise between information and computation cost to support decision making.

Thus, following a matrix formulation, $N$ has two dimensions: cell number $m$ (longitude and latitude) and $t$ (time). In Fig. 1, the whole map is 1,057,069 cells, i.e., 199,454; 165,416; and 692,199 cells for subregions ECNS, CS, and BoB, respectively.

## Decision Tree Aggregation

The risk is assessed following a decision tree. The method was proposed by the technical group for underwater noise (TG Noise) as a common approach for European waters (Sigray et al. 2021) but can be generalized to other scales and locations. It aims to support stakeholders for identifying high-risk areas for the management.

The decision tree can be formalized as the algorithm presented in Fig. 2. The assessment is performed over an area $M$ that contains the known habitats and over a

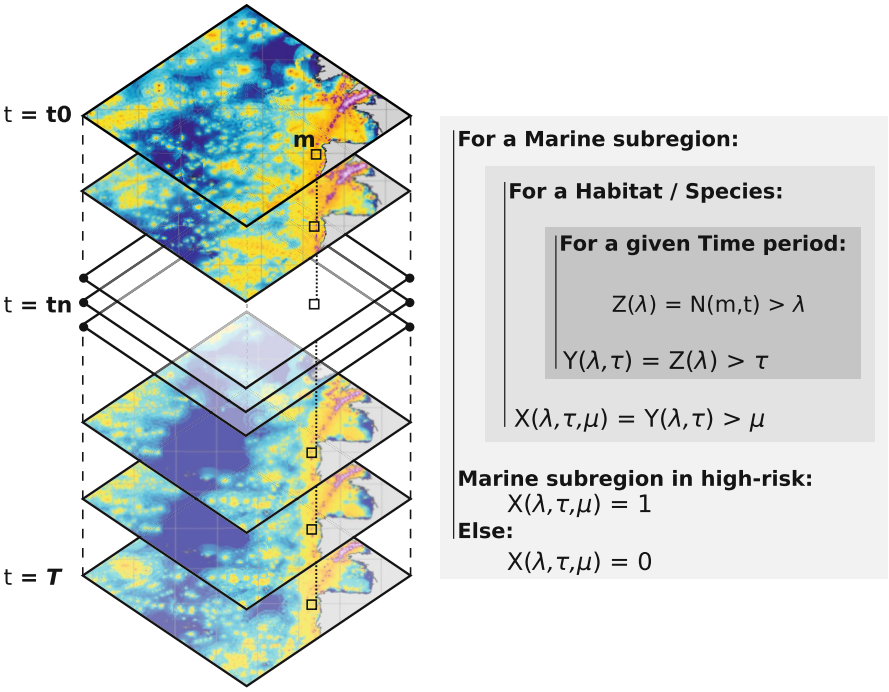



**Fig. 2** Decision tree algorithm (see text for symbol definition)

period $T$. The elementary unit is a cell $m$. The noise level in a cell of the habitats at a time $t$, $N(m, t)$ has to be lower than a level threshold $\lambda$. Let $Z(m, \lambda)$ be the number of days when $\lambda$ is exceeded. The number of days in a cell of noise exposure too loud has to be lower than a temporal threshold $\tau$. The number of cells where $\tau$ is exceeded is $Y(\tau, \lambda)$. The last step to conclude of the risk level on the habitat level is to compare $Y(\tau, \lambda)$ to a spatial threshold $\mu$. The risk is defined by a Boolean quantity $X(\tau, \lambda, \mu) = Y(\tau, \lambda) > \mu$, $X(\tau, \lambda, \mu) = 0$ means the risk is low, $X(\tau, \lambda, \mu) = 1$ means the risk is too high. Additional interpretations are discussed in section "Decision Tree Aggregation."

For the sake of simplicity, in the following, the assessment is performed at the subregion level, assuming that $M$ is fully covered by a single habitat and $T$ is a month. As there is no consensus on any of the threshold values to apply, i.e., the level of noise, the number of days, as well as the spatial extent, the decision tree is investigated by defining uniform distribution of the values, respectively, $\boldsymbol{\lambda}$, $\boldsymbol{\tau}$, and $\boldsymbol{\mu}$. Uniform prior is assumed to be the less informative distribution of every possible combination. Such hypothesis is discussed in section "Perspectives."

## Results

The proposed method produces a set of two-dimensional matrixes of all the combination of threshold values explored in the selected area. The three dimensions represent the values used as test thresholds: noise level, temporal, and spatial. For a set of three thresholds, the risk value is stored as a Boolean value (conditions reached or not for these selected thresholds).

For the sake of simplicity, each graph is obtained by averaging over the third dimension, i.e., marginalization, and displayed by a color scale in Fig. 3 and by averaging over two other dimensions as a curve in Fig. 4. Columns (A), (B), and (C) in Fig. 3 contain the results for ECNS, CS, and BoB subregions, respectively. The line (1) contains results along the temporal and the noise-level threshold values for the spatial threshold values that would lead to high risk. In a similar way, the lines (2) and (3) contain results along spatial and temporal threshold values for the noise-level threshold value and along noise level and spatial threshold values for the temporal threshold value, respectively. In a column, the lines contain the relevant information to link all threshold values and their relation with the decision tree.

Chart B1 in Fig. 3, in the CS subregion, displays the relationship between the three threshold dimensions: the x-axis represents the percentage of time thresholds ($\tau$), the y-axis represents the range of noise-level threshold values ($\lambda$) in dB re 1 $\mu$Pa, and the color range represents the percentage of the selected area ($\mu$). By selecting threshold values for noise level and temporal dimensions, one can determine how much space in the subregion is affected by noise. For example, the figure shows that 30% of the CS subregion is affected 50% of the time by noise levels of at least 100 dB re 1 $\mu$Pa. Chart B2 shows that each noise level is stable spatially and over the month.

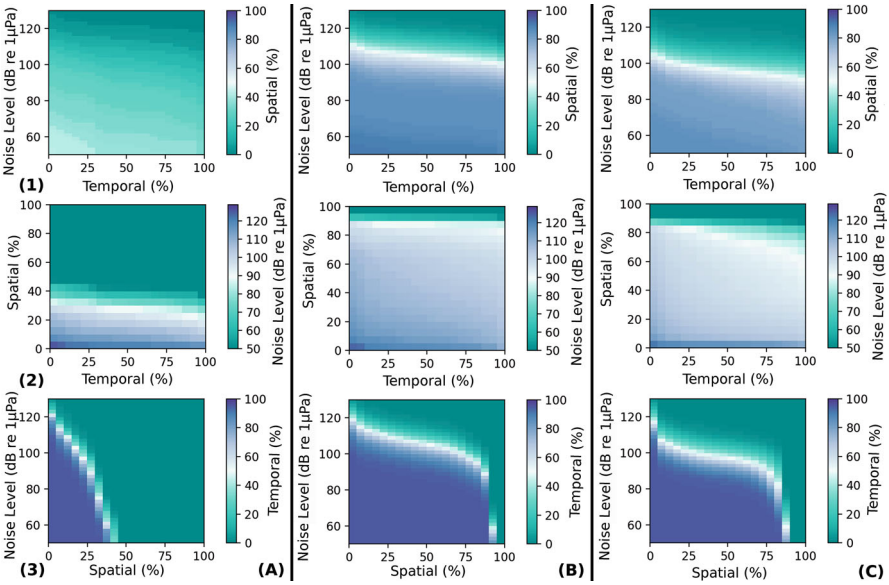



**Fig. 3** Threshold relationships by marine subregion are represented in columns for (**a**) ECNS, (**b**) CS, and (**c**) BoB. Each representation of threshold relationship for the three subregions is represented in lines (1), (2), and (3)

Chart B3 reveals that levels of noise over 120 dB re 1 $\mu$Pa are restricted locally on 5% of the area and are present less than 50% of the time. Levels of noise below 85 dB re 1 $\mu$Pa are distributed also on 5% of the subregion.

Chart A1 indicates that the ECNS subregion is not affected by shipping noise to a significant degree. It appears that high noise levels between 120 dB re 1 $\mu$Pa and 130 dB re 1 $\mu$Pa are present on 20% of the area, when levels between 50 dB re 1 $\mu$Pa and 120 dB re 1 $\mu$Pa are distributed on only 20–45% of the area. It reveals that even if a strict noise-level threshold value of 60 dB re 1 $\mu$Pa is imposed all the time (0% of the time), more than 60% of the area would still meet this low-level standard. This suggests that the shipping noise modeled in this area is relatively low.

The A2 chart shows that all levels of noise continuously occur during the month. It appears that levels of noise over 120 dB re 1 $\mu$Pa are more stable over time than other noise levels. Levels of noise comprised between 50 dB re 1 $\mu$Pa and 60 dB re 1 $\mu$Pa dominate the area (60%) all the time. The A3 chart displays that none of the noise level is distributed on more than 55% of the subregion. All the range of noise levels are represented on 40% of the area.

In chart C1, for BoB subregion, almost half of the areas (45%) is dominated by noise levels over 90 dB re 1 $\mu$Pa all the time. Chart C2 shows that a whole range of noise levels is stable spatially and over the month, except for the 90 dB re 1 $\mu$Pa noise level which displays a spatial coverage from 80 dB re 1 $\mu$Pa to 65%. Chart C3 reveals that levels of noise over 120 dB re 1 $\mu$Pa are restricted locally on 5% of the area and are present less than 40% of the time.

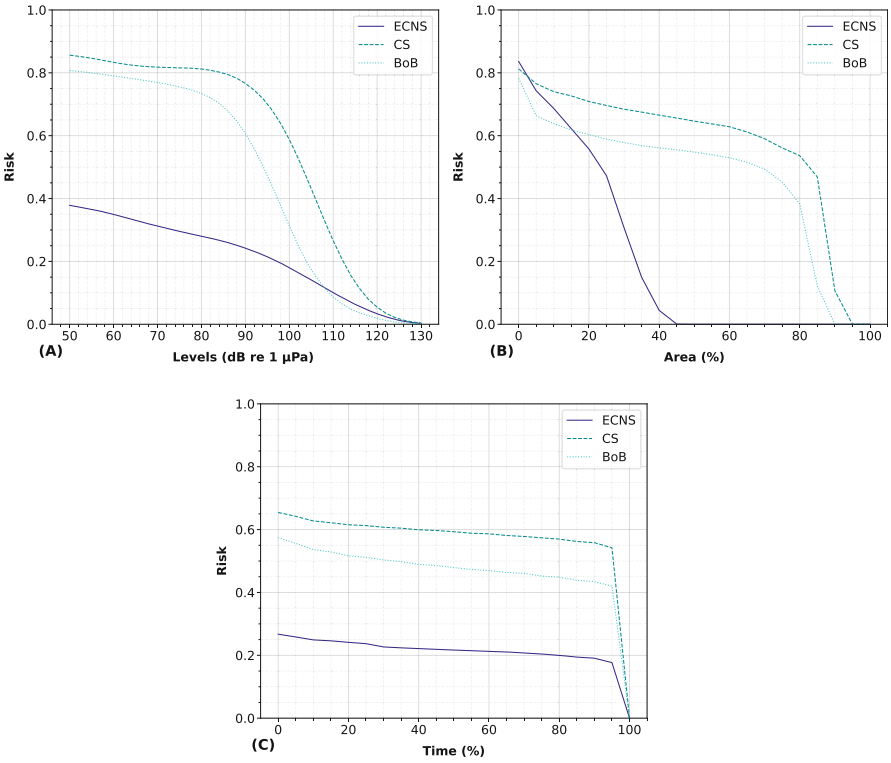



**Fig. 4** Averaged risk levels for ECNS, CS, and BoB marginalized for (**a**) noise level, (**b**) spatial threshold values, and (**c**) temporal threshold values

The marginal curves in Fig. 4 provide an averaging over an additional dimension. This allows studying the evolution of the risk assessment depending on the selection of the threshold value for the noise level, the spatial extent, and the temporal extent, respectively, in Fig. 4a–c. For example, in Fig. 4a, the risk values are expressed as a percentage of the subregion, in terms of space and time, that meet the set noise-level threshold. Here, the lower the noise-level threshold is set, the more difficult is the risk assessment conditions to be meet, which is expressed by higher-risk values (closer to 1). As a general observation, all curves show a similar behavior of decreasing monotonic function of the risk regarding the threshold values. They can have quasi-stationary phases, depending on the subregion and the threshold values. Looking at the subregion differences, it appears that CS and BoB have a similar and close behavior independently of the averaging method chosen, ECNS subregion being an outlier in these representations.

In Fig. 4a, the tensor is averaged over the temporal and spatial thresholds considered. This representation makes it possible to observe quasi-stationary states of the risk assessment for up to a threshold of 85 dB re 1 $\mu$Pa for both CS and BoB subregions. After this threshold, more cells of these subregions (in space and time)

manage to reach the objectives, thus showing a decrease of the risk values. The ECNS subregion is showing a softer decrease of the risk values over all the noise-level thresholds considered. In Fig. 4b, the averaging is performed over the noise-level and temporal thresholds. A similar stationary behavior is observed between spatial thresholds of [0–80%] for both CS and BoB subregions. After this point a steep decrease of the risk values is observed [0.55–0.00]. For the ECNS subregion, the steep decrease of the risk values [0.80–0.00] is observed for relatively low spatial threshold values [2–45%]. In Fig. 4c, the averaging is performed over the noise-level and spatial thresholds considered, which enable to observe a large quasi-stationary state [0–98%] of the temporal thresholds considered. This is closely related to the observations made in the marginal distributions described in Fig. 3.

## Discussion: Implementation of Risk Assessment

### Comparison of Subregional Results

The results from Figs. 3 and 4 suggest that the choice of threshold values influences the risk assessment depending on the selected dimension, affecting the reliability of the decision. Exhaustive analysis of the threshold value combination was investigated.

According to the findings, it appears that high noise levels [110–130 dB re 1 $\mu$Pa] are permanent and spatially concentrated in the three subregions. Moreover, fewer variations are observed over the range of temporal threshold values, demonstrating a certain amount of stability of underwater noise over the time during the studied month. It can be explained in Fig. 1 by the major shipping rails, as Ushant rail, which concentrate the shipping traffic on dedicated routes from English Channel to the south of the Bay of Biscay over the year. The results would probably be similar for longer periods of assessment. Indeed, the traffic rails are stable, and they remain very concentrated on the shelf and the continental slope.

The study found that noise-level thresholds ranging from 60 to 110 dB re 1 $\mu$Pa are more prevalent in the CS and BoB subregions, but not as much in the ECNS subregion. This suggests that the spatial threshold value definition seems to have a different effect depending on the area selected, thus may be influenced by the size of the subregion. This could be explained by different factors such as the proportion of noisy and quiet areas, the physical complexity of the area which includes shipping lanes, and local characteristics of sound propagation such as the depth of the water and environmental parameters.

Marginal representations can serve as a useful addition to noise maps as they provide more general information regarding the distribution of noise values across different thresholds in relation to the decision tree. By comparing the temporal, spatial, and intensity distributions of noise in the three subregions through charts, it becomes evident that CS and BoB subregions have similar characteristics, whereas

the ECNS subregion shows a clear distinction in terms of noise distribution. Indeed, the ECNS subregion is spatially dominated by noise, all levels combined, at almost 45% of the area, all the time. While, in CS and BoB, about one-third of the areas are dominated by the noise level over 100 dB re 1 $\mu$Pa. Furthermore, in the three subregions, high noise levels (over 110 dB re 1 $\mu$Pa) are permanent and concentrated in a reduced part of the areas (5%).

Consequently, the choice of noise-level threshold value and the choice of the size and the complexity of the assessed area may influence the confidence that can be placed in the results of the risk assessment and then on the validity of the decision-making.

## Impact of Threshold Dimensions

The results presented using the marginal distributions (Figs. 3 and 4) make it possible to observe the impact of threshold values setting on the different dimensions studied.

Large differences between temporal threshold values [0–100%] lead to relatively small variations, from 0.05 to 0.15 of the resulting risk assessment (Fig. 4c and Fig. 3a2, b2, c2 of each region). This demonstrates a certain stability of underwater noise over time during the studied month and over the studied areas. When compared to the other dimensions, the choice of the temporal threshold value has the smallest impact on the assessment outcome. The highest impact comes from the value of the noise-level threshold. The spatial threshold value definition seems to have a different effect depending on the area selected. The subregion ECNS is the smallest and the most exposed to low noise levels (below 80 dB re 1 $\mu$Pa) as shown in Fig. 1. Following the results of Fig. 3a3, the choice of the spatial threshold value seems to be important. Here, for different noise-level threshold values, this choice is reduced to a narrow band formed by the sharp transition from low to high temporal threshold values [0–100%]. This narrow band is related to the limited impact of the temporal threshold value when compared to the other two threshold values considered. In the larger subregions CS and BoB (Fig. 3b3 and c3), this same chart presents a different shape, with this sharp transition occurring for a wider range of spatial threshold values. This observation is related to the higher impact of the spatial threshold values on the ECNS area than on the CS and BoB areas. The stability of the temporal dimension is confirmed by Fig. 4c, where the risk is steady for each subregion until 0.95 and then drops to zero for the range of level and spatial threshold values.

Hence, the choice of temporal threshold values has a little impact on the result of the risk assessment, thus allowing a fairly reliable decision for the time period considered in the study. Conversely, the noise-level dimension has a strong influence on the risk assessment, thus on the reliability of the decision. Knowing that noise-level threshold values are directly linked to the biological knowledge of marine species, it becomes decisive in the management process.

## Application for Policymakers

Decision tree has been previously investigated in relation to pollution management, including noise pollution (Singh et al. 2013). The proposed algorithm follows a decision-taking process from the unitary cell to the larger scale of the habitat. If the risk is assumed to be too high in the habitat, the decision of mitigation responses follows and then a reverse order: from the habitat to the unitary cell. Such algorithm provides relevant supports to identify periods and areas that have the highest risk level regarding the defined thresholds. In addition, if the risk at the habitat level is considered as not too high, intermediary steps have to be considered depending on the values of calculation of the steps $Y$ and $Z$. $Y$ informs on the habitat level and could suggest the use of local mitigation measures to avoid the number of impacted cells to be to important. At a local scale, $Z$ informs on risk regarding the noise level, and it can be related to a particular set of activities of shipping in order to investigate dedicated mitigation.

Analysis of the marginal charts informs as well on the covariance between steps of the decision tree. It contains valuable relation for decision-making but requires stronger expertise for the interpretation. In addition, such relation depends on the area because of the traffic intensity and acoustic propagation. A first reading proposition can be to identify areas of the marginal charts where the value of the risk assessment can be strongly impacted by errors. They can be located where

$$\frac{\partial^3 X}{\partial \mu \partial \lambda \partial \tau}$$

is the greatest. In Fig. 4, the strongest gradient value is located at the inflexion point. Thus, the strongest sensitivity can be located around [95–100%] of the time for all areas; [80–95%] of the areas for BoB and CS and [55–60%] of the area for ECNS; and [100–110] dB re 1 $\mu$Pa for BoB, [95–105] dB re 1 $\mu$Pa for MC, and [60–70] dB re 1 $\mu$Pa for ECNS.

By considering the combination of all threshold values, marginalization provides an effective operational tool for the future integration of habitats and target species. It is built on the methodological basis developed at the European level and promotes the integration of threshold values recommended by the TG Noise (Sigray et al. 2022). It can also be implemented with the methodology developed by technical groups of Sea Regional Conventions (Merchant et al. 2018). In case of lack of biological knowledge, it supplies alternative possibilities to set threshold values and contribute to the effectiveness of mitigation measures (Lajaunie et al. 2023). It is essential to have a spatialized approach, in connection with mapping, because the physical environment is at the center of public policies in marine spatial planning (Tew-Kai et al. 2020).

In the event that the threshold values recommended by TG Noise are integrated in the proposed method, this would mean that 20% of the surface is affected by noise levels ranging from 100 dB re 1 $\mu$Pa to 110 dB re 1 $\mu$Pa most of the time in the three subregions. And as the studied areas are affected by a shipping rail, traffic and thus

noise are spatially concentrated. In this case, setting a spatial threshold value at 20% or lower is relevant to effectively manage the noise while also identifying the most at-risk areas.

## Perspectives

This chapter is a preliminary step for further developments for the application of underwater noise assessment and regulation. It relies on investigating a parametric test of all threshold value combinations, assuming to be a poorly informative methodology. It can be seen as a major caveat at the moment, as the threshold value definition has to be supported from the best available science. The construction of the appropriate prior distribution on $\lambda$, $\tau$, and $\mu$ using literature review is a required step. Among other, one can already mention researches on noise levels (Di Franco et al. 2020; Gomez et al. 2016; Southall et al. 2007, 2019, 2021; Weilgart 2018), as well as spatial and temporal extents (Rompré et al. 2010; Swift and Hannon 2010).

In a similar fashion, uncertainties on noise modeling and habitat distribution would expand the analysis of the decision tree. The proposed framework can easily consider additional complexity layers that would help in having a more comprehensive definition of the risk of impact while considering uncertainty at the heart of the decision-making process.

Other approaches have been investigated to assess underwater noise (Merchant et al. 2018), in particular regarding the aggregation of time and space dimension or measuring excess noise levels compared to non-anthropogenic sounds or by considering noise values over depth in a different way (Sertlek et al. 2016). As they rely on dedicated methodology, the framework has to be adapted. The impact of uncertainties of the areas at risk will be the main outcome for the comparison, as it is a major challenge to provide robust assessment.

Even considering the difficulty to define the right prior distributions, the probabilistic assessment has to be considered as a strong framework to extend to other pressures. By increasing dimensions to hypercube, it can be generalize to multi-pressures and multi-impacts. The forward modeling of combined pressure effects remains complicated, but an additive modeling would help stakeholders prioritize the knowledge acquisition.

## Conclusion

In this chapter, a multi-dimensional approach based on modeling and threshold definition for risk assessment was used to study the impact of anthropogenic noise on the marine environment in the French Exclusive Economic Zone. Such assessment remains challenging and strongly impacts the capacity to define relevant threshold values. This chapter investigates a decision tree to define the risk assessment and explore its sensitivity to threshold values and dimensions.

The decision support provided by the study is presented through a three-dimensional chart that highlights the covariance among spatial, temporal, and noise-level dynamics. The results strongly vary based on the environment and traffic intensity. It reflects as well that the decision tree allows considering where and how often the risk level can be too high.

The parametric test of the threshold values for the decision tree provides a comprehensive analysis of the relation among threshold values. Risk levels are strongly related to the selection of thresholds, and certain dimensions have a greater impact on the risk assessment. Considering the spatial, temporal, and noise-level dimensions could aid in prioritizing the acquisition of knowledge and planning of mitigation measures. Furthermore, the chapter identifies combinations of threshold values that may lead to erroneous conclusions.

The chapter provides valuable insights for policymakers and stakeholders in the management of marine noise pollution. The results were consistent with the recent work of setting options of thresholds for continuous noise in the MSFD. This framework can be generalized to include future developments regarding acoustic modeling, habitat, and knowledge on the impact.

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# Sound-Related Effects of Offshore Wind Energy on Fishes and Aquatic Invertebrates: Research Recommendations

# 108

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**Abstract**

Offshore wind (OSW) energy development is a fast-growing international industry. A range of sounds that potentially affect marine life are produced during preconstruction surveys, construction, and operations of offshore wind farms. However, the effects of these sounds on marine life, and particularly on fishes and aquatic invertebrates, are not well understood since there have been few studies of noise-induced OSW effects on these animals. Thus, building from recent efforts to identify research and coordination needs for environmental OSW research, this chapter suggests several priorities for further research to improve understanding of the effects of OSW on fishes and aquatic invertebrates. This chapter also identifies several areas where further coordination to prioritize taxa for study, standardize data collection, explore mitigation measures, and centralize research funding could help fill key gaps in knowledge and inform development decisions and mitigation measures. These gaps are relevant to floating and fixed-foundation OSW, as well as to other marine renewable energy sources such as wave and tidal energy. Filling these research and coordination gaps would help to improve understanding of the effects of sound and vibrations on fishes and aquatic invertebrates at both the individual and population levels and provide important information to help in the assessment of the potential for cumulative impacts to populations and ecosystems.

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**Keywords**

Sound pressure · Particle motion · Substrate vibration · Wind energy · Funding · Offshore · Prioritization · Acoustics · Alternative energy sources

## Introduction/Background

### Responses by Fishes and Invertebrates to Anthropogenic Sound

There are substantial gaps in knowledge of the potential effects of, and responses to, underwater sound by fishes and aquatic invertebrates (e.g., Popper et al. 2014; Hawkins et al. 2015; Di Franco et al. 2020; Chahouri et al. 2021), including anthropogenic (human-generated) sound generated from vessels, geophysical and geotechnical surveys of the ocean floor, and marine renewable energy installations used to harvest wave, tidal, and wind energy, among other sources. Moreover, while significant attention has been given to marine mammals, there is a fundamental lack of knowledge regarding the detection, production, and use of sound by many fishes and aquatic invertebrates, making it particularly difficult to determine how various anthropogenic sources might impact these animals. An additional issue is that most of the earlier work on anthropogenic sound and its effects on these animals focused on the sound pressure component of a sound, whereas most species of fishes and aquatic invertebrates primarily detect and respond to particle motion (e.g., Nedelec et al. 2016; Popper and Hawkins 2018) and/or substrate vibration (Roberts and Elliott 2017; Hawkins et al. 2021) rather than sound pressure. For a definition of terms, see Popper et al. (2022). For an extended discussion of underwater sound directed at nonexperts, see www.dosits.com.

### Background on Offshore Wind and Potential Effects

Offshore wind (OSW) energy development is a fast-growing industry around the world. The amount of planned OSW energy development is substantial, and projections of the scale of development needed to achieve climate change reduction scenarios indicate that a large proportion of some countries' marine waters may be affected by OSW energy development in the coming decades (Freeman et al. 2019). Furthermore, the technologies associated with OSW are rapidly changing to maximize efficiencies and reduce operational risks, resulting in highly variable structural components and installation and operation methods.

OSW energy development drives a variety of ecosystem changes, including in hydrodynamic conditions and primary production (Daewel et al. 2022) and the development of artificial reefs on turbine foundations and scour protection (Degraer et al. 2020). Sounds are also generated by geophysical and geotechnical surveys (primarily during the preconstruction period), pile-driving and other construction activities (including multiple piling technologies with different sound

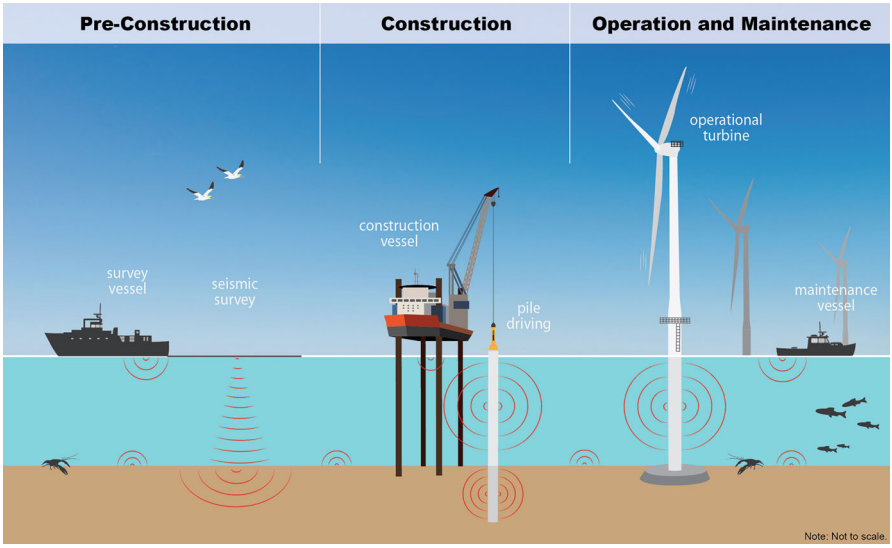



**Fig. 1** Major sources of sound generated during different phases of offshore wind energy development. Sound (shown as red lines) may include pressure, particle motion, and substrate vibration. (Figure © 2022 Iain Stenhouse/Biodiversity Research Institute, all rights reserved, used with permission)

characteristics), the operation and maintenance of offshore wind turbines and associated infrastructure, and vessel activity during all development phases (Fig. 1).

The effects of OSW pile-driving sound on aquatic life have only been studied in detail for a few marine mammal species in European waters including harbor porpoise (*Phocoena phocoena*), harbor seals (*Phoca vitulina*), and gray seals (*Halichoerus grypus*) (Madsen et al. 2006; Thomsen et al. 2006, 2015; Brandt et al. 2018). A growing concern, however, is that OSW-related sound may also affect fishes and aquatic invertebrates. Such effects may include behavioral changes such as movement from areas of spawning or feeding, the inability of animals to detect sounds of biological importance due to masking by the OSW sounds, or changes in migration routes (e.g., Siddagangaiah et al. 2021; van der Knaap et al. 2022). Moreover, even if there are no overt behavioral changes, detected sounds may result in physiological effects (e.g., increases in stress levels) that have the potential to reduce fitness (Filiciotto et al. 2014, 2016).

The geographic scale at which masking and behavioral or physiological effects might occur depends on a variety of factors, including the frequency range and sensitivity of hearing for a given species, ambient noise levels, OSW sound levels (which can vary with weather during operations; Wahlberg and Westerberg 2005), and other factors. Direct physical harm from sounds is also possible (e.g., from high level construction sounds generated when driving turbine pilings into the seabed) in situations where sensitive species are located close to impulsive sound sources or when exposure to sufficient energy occurs over a long duration (e.g., Thomsen et al. 2006; Halvorsen et al. 2012; Dahl et al. 2020).

## Origin of Findings and Recommendations

This chapter is based, in part, on the outcomes of a workgroup established during the 2020 State of the Science Workshop on Wildlife and Offshore Wind Energy Development (www.nyetwg.com/2020-workshop) that identified a set of clear, focused recommendations for research on the effects of OSW-related sound on fishes and aquatic invertebrates (Popper et al. 2021, 2022). The workgroup included researchers, resource managers, offshore wind energy developers, and other stakeholders with relevant technical expertise who identified and prioritized research needs and defined seven short-term priorities (i.e., within the next five years) that include a mix of research and coordination efforts (Popper et al. 2021, 2022).

Several other efforts have further informed the discussion in this chapter and recommendations in sections "Recommendations – Research and Coordination" and "Recommendation – Funding: Net Positive Impact Fund," including recent studies (e.g., van der Knaap et al. 2022) and reports such as those produced by the Regional Synthesis Workgroup of the Environmental Technical Working Group and the Responsible Offshore Science Alliance (Regional Synthesis Workgroup 2022, 2023; ROSA 2022). Additional regionally focused research, monitoring, and mitigation efforts underway in the United States that have influenced the recommendations in section "Recommendation – Funding: Net Positive Impact Fund" include:

- *Draft guidance issued in 2022 by the U.S. Bureau of Ocean Energy Management (BOEM) for mitigating OSW impacts to commercial and recreational fisheries* (BOEM 2022b). This guidance includes the recommendation to develop a compensatory mitigation fund to address unavoidable impacts to fisheries targeting both fish and invertebrate (e.g., shellfish, squid) species. The guidance suggests that such a fund could be specific to each individual wind farm or could encompass the effects of multiple wind farms, but that regardless, the OSW energy developer(s) should consider hiring a neutral third party to manage the fund. Further scoping of a regional compensation fund indicated that there are clear benefits to managing fisheries in a consistent way across the region, rather than handling compensation on an individual basis for each state or OSW farm, as the latter may fail to address cumulative impacts to resources of interest (North Atlantic Coast States 2022).
- *The requirement from several states for OSW energy developers to fund regional environmental and fisheries research.* These requirements are implemented via power purchase agreements between offshore wind energy developers and the states that include a minimum funding commitment to support regional research (e.g., BPU 2020; NYSERDA 2020, 2022).
- *Biodiversity offsetting, otherwise known as compensatory mitigation of ecological losses caused by infrastructure development, which can be used to achieve net positive biodiversity impacts* (Moilanen and Kotiaho 2021). OSW energy developers are increasingly committing to "net positive" gains in biodiversity from OSW energy development (e.g., Dupont et al. 2022; Jedele et al. 2023) and

compensatory mitigation has been identified as a necessary step to achieve no net loss of biodiversity from OSW energy development (Croll et al. 2022).

It is clear from the ideas in Popper et al. (2022) and the additional studies above that focused, well-planned research is needed to target essential data gaps and best improve the state of knowledge on potential effects of sound on fishes and aquatic invertebrates as the OSW industry continues to grow. Thus, this chapter, and particularly the section "Recommendations – Research and Coordination," identifies critical research and coordination needs to improve understanding of the effects of OSW energy development and operational sound on fishes and aquatic invertebrates. The chapter also identifies several commonalities between these data gaps and those that have been recognized for other aquatic energy sources such as marine hydrokinetics, and from this foundation, makes suggestions that may help to direct future research and coordination activities.

## Recommendations – Research and Coordination

To advance our understanding, additional research must be conducted into the effects of OSW sound on detection, production, and use of sound on fishes and invertebrates, as well as on their behavioral responses to anthropogenic sound and vibration. Baseline data prior to sound exposure will be essential to help understand the effects of anthropogenic sound, though such data may become increasingly challenging to obtain as the industry grows and animals become exposed to the development of multiple offshore wind facilities in a region over time. Thus, interactive effects relating to the presence of multiple individual OSW farms in close proximity to each other should also be explored. These research studies will be most effective at increasing our understanding if there are substantial coordination efforts to ensure that new studies are focused on the highest priority data gaps and are coordinated and standardized such that data can be aggregated to maximize gains in knowledge.

Importantly, this work should be supported by a dedicated funding stream that is regional in focus (rather than specific to individual OSW farms) to facilitate both research and coordination efforts. This would also further help to ensure that funding is directed toward regional priorities that best inform a broader understanding of the potential for cumulative impacts to populations and ecosystems from a variety of renewable energy activities (see section "Recommendation – Funding: Net Positive Impact Fund").

The recommendations below build directly from Popper et al. (2022), while introducing or expanding upon several ideas that were not emphasized in that paper. These new suggestions include the use of consequence models as a framework for directing research efforts as well as the formation of a regional fund to help coordinate, prioritize, and conduct necessary research and coordination in a timely fashion. Moreover, these recommendations are inclusive of all aspects of sound (particle motion and substrate vibration, as well as sound pressure). They also

include research on a variety of life history stages and species with differing mobility (e.g., ability to escape sound to which they are exposed). Examination of benthic invertebrates will be important to assess the effects of substrate vibration.

It should be noted that the recommendations are not presented in any particular order, though Popper et al. (2022) do provide a partial list of prioritized recommendations. From a logical sequencing standpoint, it would make sense to address many of the coordination needs before research is conducted – for example, to identify priority taxa for further research first, and then target those priority taxa in studies of behavioral responses to offshore wind energy development. However, the reality of funding availability, the time required for data standardization and other coordination efforts, and the immediacy of OSW energy development in the United States and elsewhere, all preclude an orderly, consecutive approach to addressing priorities. Indeed, it is likely that the best approach would be to address, where possible, these needs concurrently, and to balance needed coordination and standardization with the need to collect field data from imminent commercial-scale OSW projects. In the longer term, the primary end goal of all of these research and coordination efforts is to inform an understanding of the potential for cumulative impacts to populations or ecosystems; as such, this topic is presented as our first recommendation in the below list, to provide context for those following it.

## Inform Cumulative Impact Estimates (Consequence Models)

The end goal of all the suggested research studies in this chapter is to understand the cumulative biological impacts to fishes and aquatic invertebrates from OSW energy development. However, developing this understanding will require substantially more data for more fish and invertebrate species than now available. In the longer term, data from new studies must be aggregated into cumulative impact frameworks for assessing the effects of OSW sound on populations of interest (e.g., Pirotta et al. 2018; Slabbekoorn et al. 2019; Mortensen et al. 2021). Additionally, OSW can affect aquatic taxa via multiple mechanisms (for example, via the development of artificial reefs on hard substrates; Degraer et al. 2020), and a range of studies on such phenomena are needed to provide context for species- and sound-specific studies of OSW effects to aquatic taxa.

## Examination of Animal Displacement and Other Behavioral Responses to Sound

There are some data, primarily from Europe, on the responses of fishes to sound generated by OSW energy development and operations (e.g., Thomsen et al. 2012; Debusschere et al. 2016; Kok et al. 2021). For example, recent tagging studies of free-ranging Atlantic cod (*Gadus morhua*) are adding to our knowledge of the effects of geophysical exploration and pile-driving sound on the behaviors and movements of this species (van der Knaap et al. 2021; van der Knaap et al. 2022).

There are also data from other anthropogenic activities such as the driving of piles for wharfs and bridges (e.g., Hawkins et al. 2014; Iafrate et al. 2016a, b).

From the available data, the OSW sound sources with the highest risk of causing behavioral or physiological changes with the potential to affect animals' fitness are likely to be some of the lower frequency sounds associated with geophysical and geotechnical surveys (primarily occurring during the preconstruction period), as well as impact pile driving during turbine installation. The explosive removal of unexploded ordnance (UXO) on or in the seabed during construction may also be a concern in some locations. If sound causes displacement from important spawning areas, for example, this could affect breeding success and thus impact individual or population fitness. Changes to predator-prey dynamics and changes in survival could likewise have the potential for cumulative impacts.

Thus, it is necessary to first determine whether changes in behavior or physiology are occurring, and then to determine the individual or population implications of such changes. In addition to examining displacement from spawning or foraging areas, studies should also examine other changes in movement behavior, reproductive behavior, predation success, foraging activity, and changes in acoustic behaviors.

To the extent possible, behavioral studies should occur in the field in the presence of actual OSW activities to make the study results more representative and thus useful for management. However, such studies are not always possible since field work requires very high effort and experimental conditions are extremely difficult to control, particularly for more physiologically focused studies or studies of phenomena that remain particularly poorly understood, such as the effects of substrate vibration on benthic invertebrates. Thus, laboratory studies should aim to reproduce real-life conditions as much as possible, for example by employing a real sound source for controlled exposure and designing enclosures that allow experimental animals to respond to sound exposure in realistic ways. Best practice could be to combine field efforts with laboratory studies. Ecological/spatial models can also be utilized to help to identify parameters to be measured in the field (Heinänen et al. 2018).

In any case, focal species should be carefully selected (see section "Identification of Priority Taxa on which to Focus Research") to include species present in OSW areas (e.g., Friedland et al. 2021) and that possess the hearing capabilities to be affected by OSW-generated sound, inasmuch as this is known.

## Hearing Sensitivity Studies Related to Particle Motion, Substrate Vibration, and Sound Pressure

Some of the above research on the effects of OSW-related sound to fishes and aquatic invertebrates can be conducted without a thorough understanding of sound detection capabilities of the animals under study. However, understanding of species' hearing capabilities is focal to informing interpretation of the results of behavioral studies. Moreover, it is crucial to understand the applicability of results to other species, sites, and research questions. Thus, it will be difficult to effectively

assess the potential for cumulative impacts across OSW areas without tying observed behavioral responses to a species' ability to detect the sounds and vibrations to which they are exposed.

Studies of species hearing capabilities, as discussed by Hawkins et al. (2015), must include a focus on the range of frequencies that are detectable as sound pressure and particle motion, as well as frequencies detectable via substrate vibration for those animals living on, in, or near the seabed. Much more also needs to be known about the potential masking of biologically relevant sounds (e.g., those of conspecifics and sounds used to attract mates or detect the presence of predators or food) due to sound generated by OSW and other anthropogenic sources. While some studies might involve examination of electrophysiological responses of the ear or auditory system, these physiological responses do not necessarily represent the sounds to which an animal will change its behavior in the wild, and studies using behavioral approaches will be essential (Popper and Hawkins 2021).

## Identification of Priority Taxa on which to Focus Research

There are a very large number of fish species and an even larger number of aquatic invertebrate species that are likely to detect and use sound in their normal behaviors (e.g., Rice et al. 2022). As such, it is impossible to investigate behavior or hearing in a sizable proportion of extant species. At the same time, because of the extraordinary diversity in the biology of fishes and aquatic invertebrates, and the associated diversity in hearing organs, extrapolation from one species to another requires considerable caution and a solid scientific understanding of their similarities. This issue was discussed by Popper et al. (2014) who recommended focusing on groups of fishes and aquatic invertebrates with differing hearing capabilities, as well as different hearing mechanisms, in hopes that collecting data on representatives from each group could help to inform predictions of effects for like taxa (Popper et al. 2014; Hawkins et al. 2020).

Thus, identification of a suite of preidentified taxa of interest would focus studies on those taxa, so that research can most effectively contribute to the state of knowledge regarding potential effects of OSW. Focal species must: (1) occur in the vicinity of OSW development areas (e.g., Friedland et al. 2021); and (2) have the potential to be affected by OSW-related sound. This vulnerability could draw from the regularity of their occurrence in OSW areas (for example, structure-oriented species; Degraer et al. 2020 or from their use of OSW areas during important life history stages such as spawning and migration toward reproduction locations. Species known to be sensitive to displacement from OSW energy development activities should be carefully considered, not least because species with known sensitivity have by definition already been studied, and thus, there is a foundation upon which to add knowledge. Species about which very little is known could also be a focus, as they may have the potential for severe effects that have not been studied to date. Species should also be drawn from a range of ecological niches and may include species of regulatory or commercial concern, such that research studies can help to inform OSW decision making and potential management actions.

## Standardization of Methods and Sharing of Data

To ensure that results are comparable between studies and can effectively inform our understanding of cumulative impacts over the long term, it would be of particular value if studies were conducted in standardized ways. Data should also be made publicly available. To date, studies on fishes (and the few on aquatic invertebrates) have focused on diverse species and used a variety of methods and metrics, making it difficult to compare results, much less begin to extrapolate to other species. There are several existing standards from which to build (e.g., Dekeling et al. 2014, 2016; Merchant et al. 2022) but these are primarily focused on marine mammals and leave substantial gaps in standards that are applicable to other taxa.

Existing standards, as well as detailed recommendations on further methods and metrics standardization, are discussed in Popper et al. (2022). In general, there is a need for:

- Reporting received signal levels (in the appropriate metrics/units).
- Standardized methods and metrics for studying particle motion and substrate vibration, including the measurement of particle acceleration where predictions of acceleration are not reliable (e.g., near boundaries and in shallow waters).
- Effect size reporting to facilitate future meta-analysis and other efforts to compare or compile data between studies.
- Publishing and acceptance of data where no effects are shown (assuming care is taken to avoid Type II errors) to help guide the direction of future work.

## Development of a Long-Term, Highly Instrumented Field Site to Support Research

Field-based research is an essential component of a research strategy to understand the effects of OSW-related sound on fishes and aquatic invertebrates, as noted above. However, it can be quite challenging to characterize ambient soundscapes and control environmental variables such that animal responses can be properly attributed to causal mechanisms or linked to specific sound characteristics. If an ecologically relevant long-term field site was identified and made available for OSW sound-related research, it could help to address some of these challenges while also facilitating inclusivity/access and standardization of research methods. Investigators could develop a baseline understanding of the site's acoustics and substrate vibration (and ideally control both as much as possible), install a variety of instruments for measuring acoustic conditions and animal responses, and study focal species, substrates, and research questions in a more controlled (but still ecologically and acoustically relevant) environment. Such a long-term research site would need to be carefully selected to be as similar as possible to OSW areas of interest, while allowing control of the local soundscape (or at least effective characterization).

## Preliminary Assessment of Sound Mitigation Options for Fishes and Aquatic Invertebrates

Development of OSW noise mitigation approaches has focused primarily on mitigating effects to marine mammals (e.g., Verfuss et al. 2016, 2019; Thomsen and Verfuss 2019). These approaches have included the development of noise exposure thresholds, noise abatement systems such as bubble curtains, vibratory pile-driving approaches, and animal deterrents. It is unclear the degree to which such mitigation approaches are likely to be effective for fishes or aquatic invertebrates. The frequencies and types of sound (e.g., sound pressure) for which these mitigation methods have been developed may not match those relevant to fish and invertebrate hearing and do not take into consideration particle motion or substrate vibration. One approach, at least at first, might be to do a desktop feasibility study that assesses the utility of existing sound mitigation options for fishes and aquatic invertebrates. Such a study could help to identify key knowledge gaps to inform the development of effective mitigation measures for these taxa.

This assessment must take into consideration several factors. First, there is no purpose to developing or employing mitigation unless there is a significant risk that the animals of interest are potentially impacted by the sounds (Popper et al. 2020; Thomsen et al. 2021). Second, it must be determined whether the mitigation method is likely to be effective in reducing sound levels at frequencies for which the animals in question are sensitive. Third, virtually all OSW sound mitigation methods developed to date focus on sounds in the water. Yet, vibrations and sounds in the substrate can travel considerable distances to be detectable by benthic animals (e.g., Hawkins et al. 2021). Thus, an assessment of sound mitigation approaches must include explicit consideration of substrate vibration and its potential impact on animals living in or near the ocean floor.

## Recommendation – Funding: Net Positive Impact Fund

It will be difficult to address these research and coordination needs on a timeline that allows results to inform OSW energy development in the United States and around the world. Thus, development of a centralized funding mechanism is recommended to support environmental research and mitigation – a "net positive impact fund." As described below, there are several ongoing activities in the eastern United States and elsewhere that indicate the timeliness and relevance of such a recommendation.

The regional OSW fisheries compensation fund currently being planned in the eastern United States is focused on compensating fishermen for effects to their livelihoods from OSW energy development – e.g., if they cannot fish in a given area for some period of time during or after construction. Several European countries as well as U.S. states have already established some kind of compensatory mitigation process for fisheries in relation to OSW energy development (F-TWG 2022).

A similar fund that OSW developers pay into to support environmental research could help ensure that studies are conducted in standardized ways and funds are

directed toward key priorities that can best contribute to understanding cumulative impacts. Moreover, OSW energy developers are increasingly committing to "net positive" gains in biodiversity from OSW energy development (e.g., Dupont et al. 2022; Jedele et al. 2023). Such biodiversity offsetting, otherwise known as compensatory mitigation of ecological losses caused by infrastructure development, can be used to achieve net positive biodiversity impacts (Moilanen and Kotiaho 2021) and has been identified as a necessary step to achieve no net loss of biodiversity from OSW energy development (Croll et al. 2022). As little is currently known about the effects of OSW-related sound on fishes and aquatic invertebrates, the degree of compensatory mitigation that may be required is unclear. As such, in the immediate term funding could be focused on understanding the scale of impacts to inform the degree of mitigation that may be required. There are several models for collective funding of offshore wind environmental research, including the Offshore Renewables Joint Industry Programme for Offshore Wind (http://bit.ly/3XjDDXZ; http://bit.ly/3ITLqqZ).

There are several regional or national collaborative bodies in the eastern United States that could house such a "net positive impact" fund. The Regional Wildlife Science Collaborative for Offshore Wind (RWSC – https://rwsc.org/) and Responsible Offshore Science Alliance (ROSA – https://www.rosascience.org/), collaborative groups focused on OSW and the environment and fisheries, respectively, were both developed with this coordination and funding role in mind. Existing funding mechanisms at the state level (e.g., BPU 2020; NYSERDA 2020, 2022) are also being implemented to increase available funds to support regionally important research and coordination efforts. Additionally, the National Offshore Wind Research and Development Consortium (NOWRDC) could increase their current focus on sound-related research and mitigation measures and expand the scope of this interest beyond marine mammals. The Center for Marine Acoustics at BOEM and the Passive Acoustics Program at the National Marine Fisheries Service Northeast Fisheries Science Center could play a role in the guidance and standardization necessary to ensure that a net positive impact fund would effectively prioritize and coordinate research efforts.

Many of the above-mentioned groups focus largely or exclusively on OSW, but it may be more efficient to broaden the scope of a net positive impact fund to also incorporate research into the effects of other types of marine energy development as well as OSW.

## Application to Other Industries and OSW Technologies

Technologies used within the OSW industry are undergoing substantial advancements that will influence the types of sound to which fishes and invertebrates may be exposed. Turbines in shallow waters typically have steel monopile or jacket foundations that are installed via pile driving, but offshore wind is advancing into deeper waters where floating foundations will become the norm. Floating turbines still require anchoring to the seabed, and are connected to these anchors via at least

three large mooring lines; depending on the design, these lines may drag along the ocean floor, leading to a larger physical footprint (Maxwell et al. 2022). Construction is generally less noisy for floating foundations (Carduner et al. 2022), as the anchors are smaller than the turbine foundation itself, so driving them into the seabed requires substantially less energy. Some types of anchors (such as gravity anchors) do not require pile driving at all (Maxwell et al. 2022). However, movement of the mooring lines during turbine operations represents a new source of sound and vibrations for the lifetime of these offshore wind projects (Carduner et al. 2022; Maxwell et al. 2022). Thus, the specific sounds produced by floating wind energy may differ somewhat from fixed-platform technologies. The locations in which floating technologies will be deployed may also lead to the exposure of new taxa that have not previously interacted with OSW energy development. The above research recommendations hold true regardless of foundation type, but research focused specifically on floating wind may need to adjust objectives, focal taxa, and other specifics to examine this new type of development.

More generally, while the focus of this chapter is on research needs associated with OSW with respect to fishes and aquatic invertebrates, there are a variety of other devices in various stages of development that will potentially harness energy in the marine environment. These include devices to extract energy from tidal or wave motion (Clément et al. 2002; Wolfram 2006). Some devices are similar to fixed-foundation OSW turbines in that they require some amount of pile driving during construction, while others have much reduced sound footprints during the construction period. All such devices are likely to produce some sound during the operational period within the range of hearing of fishes and aquatic invertebrates, and thus have the potential to impact behavior and possibly fitness. Indeed, while the exact nature of the sounds may vary between devices, including (but not limited to) sound spectrum, amplitude, and repetition rate, it is likely that there are more similarities than differences in the sounds produced by marine renewable energy devices. While the habitats in which devices are placed are likely to vary, particularly tidal devices (which are generally targeted in coastal locations) vs. open-water OSW and wave energy generators, there remains a lack of information about sound detection and behavioral responses to sound for all such devices.

Thus, it is reasonable to suggest that what is learned about responses to OSW sounds may also inform understanding for other types of marine renewable energy development, and coordination of research may be valuable not just among different OSW sites but also among studies of fixed and floating OSW as well as other marine renewables. Indeed, in addition to marine renewables, these recommendations apply to a range of sound-producing activities in the marine environment. Much current knowledge of potential effects from OSW draws from studies of pile driving for bridges, wharves, and other structures. While the characteristics of the local environment and species present clearly affect the specifics of how sound may affect aquatic life, there are generalities that can be drawn across sound source type that, with concentrated research and coordination efforts, may further help to improve the knowledge base.

## Conclusions

Several focused, high-priority recommendations were developed by a workgroup in 2021 to improve our understanding of cumulative effects of offshore wind energy-related sound to fishes and aquatic invertebrates. Those recommendations were published in a workgroup report (Popper et al. 2021) and an article in *The Journal of the Acoustical Society of America* (Popper et al. 2022). Since the workgroup's efforts concluded, there has been continued progress on a variety of fronts to further identify research and coordination needs. With several notable exceptions, however, there has been limited progress in actually funding or conducting the necessary research, and so there are still few field studies conducted under real-world conditions that examine behavioral, physiological, and ecosystem effects of sound that may have possible fitness consequences for fishes and aquatic invertebrates. There have been several recent efforts toward standardization in the United States, such as with BOEM's recently released Nationwide Recommendations for Impact Pile Driving Sound Exposure Modeling and Sound Field Measurement for Offshore Wind Construction and Operations Plans (BOEM 2022a). There have also been several recent field studies to improve our understanding of effects from pre-construction surveys, construction, and operation of OSW facilities in Europe (e.g., van der Knaap et al. 2021, 2022).

In the long term, the aim of all such funding and research should be to inform cumulative impact models, thereby substantially enhancing our understanding of possible sound-related impacts of OSW energy development to populations and ecosystems. Effective assessment of potential cumulative impacts of sound and vibration on fishes and aquatic invertebrates is currently hindered by a range of knowledge gaps, including those identified in this paper. Filling such critical gaps in knowledge will improve understanding of possible sound-related impacts of energy development to populations and ecosystems.

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# Effective Underwater Noise Management of Offshore Energy Activities: Use of Monitoring and Adaptive Management

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**Abstract**

The National Offshore Petroleum Safety and Environmental Management Authority (NOPSEMA) is Australia's offshore energy regulator and has a unique perspective on the benefits of adaptive management for addressing underwater noise impacts. This is due to the dual role in assessing the adequacy of proposed management programs for underwater noise-generating activities as well as undertaking compliance inspections to ensure management programs are effectively implemented as intended.

Environmental impact assessment is not a perfect discipline. It relies on a series of systematic impact predictions based on comprehensive considerations of interactions between a proposed activity and the environmental values and application of relevant scientific studies. The predictions of impact then inform the selection of mitigation measures needed to reduce potential impacts to acceptable levels. The effectiveness of mitigation measures and how they will

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function and perform within the range of variables presented by a real-world setting is uncertain in many cases. For underwater noise-generating activities, uncertainty in the effectiveness of mitigation measures can be addressed through the application of adaptive management.

NOPSEMA's regular compliance monitoring of noise-generating activities has revealed the importance of adaptive management for responding to unforeseen circumstances and environmental data during the implementation of activities to address key areas of scientific uncertainty.

This chapter highlights the post approval requirements to implement adaptive management programs to address new information, and scientific uncertainties that remain following project approval. Monitoring and adaptive management are critical for ensuring underwater noise impacts can continue to be managed to acceptable levels throughout the full duration of an activity.

Two case studies are explored in the chapter to highlight how changes in the knowledge base that informed impact predictions and changes in the environmental setting should be evaluated to determine the need for adaptive management.

**Keywords**

Environmental regulation · Monitoring and adaptive management · Continuous improvement · Underwater noise · Environmental regulation · Offshore energy activities · Scientific uncertainty

## Introduction

In Australia, the National Offshore Petroleum Safety and Environmental Management Authority (NOPSEMA) is the independent regulator of environmental management, health and safety and structural (well) integrity for all offshore oil and gas operations and greenhouse gas storage activities in Commonwealth waters, and in coastal waters where regulatory powers and functions have been conferred. NOPSEMA regulates these offshore activities under the *Offshore Petroleum Greenhouse Gas Storage Act 2006*, and more recently NOPSEMA's role has expanded to administer the functions of the Offshore Infrastructure Regulator under the *Offshore Electricity Infrastructure Act 2021* which includes the regulation of environmental management, work health and safety, and infrastructure integrity of future offshore renewable energy activities.

Offshore energy activities can introduce significant sources of anthropogenic underwater noise in the Australian marine environment, with the potential to adversely affect marine receptors if not effectively managed. Common examples of significant noise-generating offshore energy activities that NOPSEMA regulates include marine seismic surveys which are undertaken for multiple purposes such as to identify potential oil and gas reservoirs below the seafloor or for site characterization for renewable energy developments, and pile driving which is undertaken to

install foundation piles into the seabed for the construction of offshore oil and gas or renewable energy developments.

NOPSEMA administers an objective-based environmental management regime in accordance with the Offshore Petroleum and Greenhouse Gas Storage (Environment) Regulations 2009 (Environment Regulations) that aims to ensure environmental impacts and risks of offshore energy activities are reduced as to as low as reasonably practicable (ALARP) and an acceptable level and consistent with the principles of ecologically sustainable development (ESD) set out in section 3A of the *Environment Protection and Biodiversity Conservation Act 1999*. This objective-based regime requires proponents of offshore energy activities to submit an environment plan (EP) to NOPSEMA for approval before their activities can commence. An EP must include an assessment of all the environmental impacts and risks of a proposed activity and set out the evidence base that demonstrates how the activity will be managed in a manner that will ensure the impacts and risks are reduced to ALARP and acceptable levels and consistent with the principles of ESD.

An EP must also set out environmental performance outcomes (EPOs) that are equivalent to or better than the acceptable levels of environmental impact or risk, along with environmental performance standards (EPSs) with associated measurement criteria, which collectively outline a framework of measurable benchmarks for environmental performance to be achieved by a proponent during the implementation of their proposed activity. EPs must also include an implementation strategy with arrangements for ensuring that environmental management will continue to be effective during the implementation phase. Once NOPSEMA is reasonably satisfied the EP meets the acceptance criteria set out in the Environment Regulations, an EP will be approved (NOPSEMA 2022a, b, c).

NOPSEMA's objective-based regime means NOPSEMA does not prescribe "one size fits all" control measures or "set and forget" approaches to environmental management. Rather, activity proponents have the flexibility to design their environmental management arrangements during the preparation of an EP to suit the specific activity circumstances and operating environment with arrangements in place to continuously review and adapt their management arrangements in response to potential changes not foreseen during the EP's preparation. The benefit of this adaptive regulatory approach is that it can address scientific uncertainty, provides for continuous improvement in environmental management over time, and allows for offshore energy activities to rapidly adopt advances in technology, consider new science, and respond to any unforeseen changes in an operational environment.

Activity proponents are legally responsible for complying with the commitments in the approved EP. NOPSEMA can and will undertake compliance-monitoring inspections to ensure activity proponents are effectively implementing key environmental management commitments and effectively adapting management arrangements where necessary, so that EPOs and EPSs will continue to be met and environmental impacts and risks remain ALARP and acceptable.

This chapter outlines how NOPSEMA's regulatory framework provides for the effective management of potential impacts and risks from anthropogenic underwater noise generated by offshore energy activities on an ongoing basis. It provides

insights into the challenge faced by NOPSEMA to make defensible decisions in relation to impacts and risks from anthropogenic noise when faced with scientific uncertainty and explains how this challenge can be overcome through the development and implementation of monitoring and adaptive management approaches. Two case studies have been included to illustrate monitoring and adaptive management in practice and demonstrate how this environmental management approach has successfully resulted in the effective management of impacts and risks from underwater noise in response to unexpected changes during activity implementation.

## The Challenge: Making Defensible Regulatory Decisions When Faced with Scientific Uncertainty

Ecologically important species such as whales, sea turtles, and pinnipeds as well as commercially important species of fish and invertebrates are among the major noise-sensitive marine fauna groups in Australia. Potential impacts to these marine fauna groups that could result from underwater noise generated by offshore energy activities may include displacement from important biological habitats, disturbance to important biological behaviors, stress, diminished health, communication masking, impairment to hearing (e.g., temporary or permanent threshold shifts), physical injury, and mortality (CMS 2017). There are also a number of indirect impacts that may occur, including but not limited to impacts on other trophic levels (e.g., prey displacement or depletion) and reduced reproductive success. The extent, severity, and duration of these impacts will depend on the nature of the noise exposure (e.g., intensity, frequency, and duration), the nature of the environmental receptor (e.g., behavior, life stage, ecology, and physiology), and the level of spatial and temporal overlap between the noise-generating activity and the habitat of the noise-sensitive fauna group (NOPSEMA 2023a).

While the scientific knowledge base is rapidly evolving, the population or ecological level consequences of different levels of underwater noise exposure remain uncertain for a wide range of taxa. Some examples of the key areas of scientific uncertainty that reduce confidence in predictions of the consequences of underwater noise on values of the marine environment include (NOPSEMA 2023b):

- Dose-response relationships for relevant noise effects to inform development of suitable effect thresholds for environmental impact assessment
- The effects of underwater noise emissions on listed threatened whales to inform the management necessary to ensure offshore energy activities do not reduce population viability and /or impede species recovery
- Behavioral, physical, and physiological impacts of noise on fish and invertebrates and implications for the fisheries that target them
- The biological and ecological consequences of underwater noise impacts on ecologically and commercially important marine fauna
- The biological and ecological consequences of underwater noise exposure on marine turtles particularly during important life stages and behaviors

- The effectiveness of marine fauna detection technologies or control measures for managing the impacts of noise from offshore energy activities

During the EP assessment process, NOPSEMA scrutinizes the level of scientific uncertainty associated with predictions of underwater noise impacts and the anticipated effectiveness of management measures in reducing these impacts. NOPSEMA applies greater precaution in the decision-making process when there is a higher degree of scientific uncertainty in the effectiveness of the control measures proposed in the EP by the proponent.

## Addressing Scientific Uncertainty

When making a decision regarding whether the EP demonstrates the environmental impacts and risks of an activity will be of an acceptable level, NOPSEMA will assess the case made by the proponent in the submission in accordance with NOPSEMA's policies and guidance (NOPSEMA 2022a, b, c). This includes, but is not limited to, whether areas of uncertainty in predictions of impact and risk are appropriately identified, acknowledged, and addressed.

Where there is uncertainty as to whether acceptable levels of impact will be met, there is a broad range of measures that will need to be considered to account for this uncertainty. Some examples include:

- Approval of the EP with conditions or limitations that exclude the noise-generating aspects of the activity from occurring within sensitive seasons or habitats
- Ensuring measures are in place for the proponent to identify and review relevant new science prior to and during activity implementation and adjust management measures where necessary
- In situ measurement of received sound levels to validate sound level predictions and inform further mitigation needed to account for deviations from predictions made in the EP
- In situ field monitoring to detect the presence, abundance, and behavior of biota to inform management and mitigation
- Implementation of monitoring technologies and protocols to inform real-time mitigation (Douglas and Brandon 2016)
- Employing adaptive management measures in response to exceedances of management triggers such as higher than expected fauna encounters, changes in the sound exposure regime, or lower than expected performance of important control measures

Monitoring and adaptive management measures form part of a proponent's environmental management system (EMS) designed to help manage environmental impacts and continuously improve environmental performance through the adoption of a systematic approach centered on environmental leadership commitment. The

Australian/New Zealand Standard ISO 14001:2016 sets out the requirements for an EMS and promotes the application of a conventional cylindrical improvement cycle centered around four management elements that includes:

1. **Plan** (i.e., establish the necessary environmental objectives and processes to deliver results in accordance with the organization's environmental policy)
2. **Do** (i.e., implement the planned processes)
3. **Check** (i.e., monitor and measure processes against the organization's environmental policy including its commitments, environmental objectives, and operating criteria and report results)
4. **Act** (i.e., where necessary, take actions to continually improve)

An EMS is an important part of the implementation strategy in an EP and must include measures to ensure impacts and risks continue to be identified and managed, control measures are effective, and defined levels of performance for environmental management are achieved. An example of monitoring and adaptive management cycle considerations within the Australian/New Zealand Standard ISO 14001:2016 EMS framework is provided in Fig. 1.

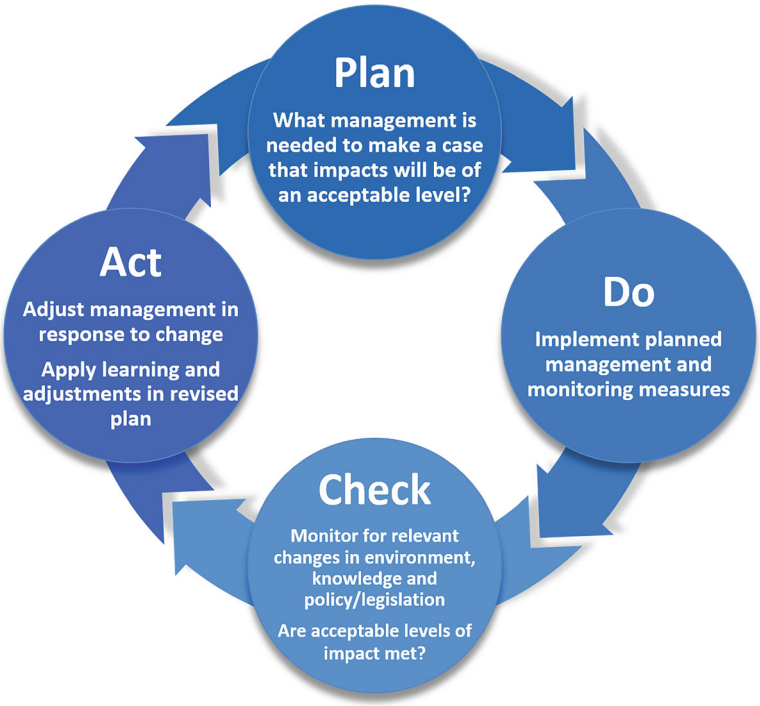



**Fig. 1** Example of the considerations for an effective adaptive monitoring and management process within the Australian/NEW Zealand Standard ISO 14001:2016 environmental management system framework

The benefit of implementing monitoring and adaptive management measures within an EMS is that it not only results in improved environmental outcomes but also allows proponents to have the flexibility to continue to implement their activities within operational schedules and time frames by adapting to changing circumstances. In order to ensure that a monitoring and adaptive management process is effective, the EMS should clearly describe:

- The primary performance indicators of any measures that will be implemented to verify whether impact predictions are accurate (e.g., equipment calibration and/or specifications and competency and/or training of personnel responsible for undertaking verification checks)
- The process for assessing and applying results of monitoring to appropriately inform environmental management
- The criteria to be used to decide if and when a defined action needs to be employed in response to results of monitoring to ensure that impacts to the environment are continually managed to an acceptable level and reduced to ALARP (e.g., implementation of revised or new control measures)

In an underwater noise context, key examples of changes that can trigger adaptive management responses include:

- An unforeseen change in presence, abundance, and behavior of sound-sensitive marine fauna within the area that may be affected by underwater noise.
- The unexpected ineffectiveness of marine fauna detection methods in real-world conditions.
- The ineffectiveness of control measures due to a change in offshore activity or environmental circumstance.
- The received sound levels within important fauna habitat are higher than predicted.

## Monitoring and Adaptive Management Case Studies

Two hypothetical case studies are presented below to illustrate the benefits of an effective monitoring and adaptive management program. The case studies particularly illustrate the efficacy of an effective monitoring and adaptive management program in addressing scientific uncertainty in the extent, severity, and duration of environmental impacts of underwater noise during the implementation of an activity where there are changing circumstances.

Both of these case studies relate to the implementation and environmental management of marine seismic surveys (MSS). MSSs are undertaken for a range of purposes including to identify potential oil and gas reservoirs below the seafloor. During these activities, underwater noise is generated from the use of high-pressure airguns that, when discharged, release highly compressed air that expands and

compresses rapidly to produce a pulse of high intensity and low-frequency impulsive, broadband sound.

## Case Study 1: Adaptive Management in Response to New Science

- MSS activity was proposed to be undertaken within a productive fishing ground for the southern rock lobster (*Jasus edwardsii*), a highly valuable commercial species.
- The EP was approved on the basis that a review of the best available science indicated the expected sound exposure regime would result in impacts deemed acceptable to southern rock lobsters or the commercial fishery. The sound exposure regime was predicted using validated noise-modeling methods tailored to the specific sound source and operating environment. In addition, an EMS was in place that required new information, including newly published science, to be considered in relation to refining the ongoing management of the activity.
- Prior to the commencement of the activity, new scientific evidence was published indicating that noise generated by seismic survey airguns has the potential to result in physiological impacts to southern rock lobsters at noise levels that can be generated by commercial seismic surveys (Day et al. 2016). Given the relevance of this result to the predictions of impact and proposed control measures in the accepted EP, the publication of this scientific paper triggered a formal review as part of the EMS measures.
- New information published in this paper triggered a reevaluation of the impacts of the activity. The formal review undertaken by the proponent resulted in a conclusion that there was a higher potential for the activity to impact southern rock lobsters than was predicted in the approved EP. For example, the paper presented new evidence that noise exposure at levels commensurate to a seismic survey could result in a depressed immune response, impairment of reflex behaviors, and damage to the sensory hairs of the statocyst that may have implications for southern rock lobster ecology such as feeding, predator avoidance, locomotion, and social behaviors.
- In response to the revised assessment indicating that the impacts may be greater than what was initially predicted in the accepted EP, the proponent adapted to the change by designing and implementing an enhanced suite of control measures to ensure the impacts to southern rock lobsters and the commercial fishery continue to be managed to an acceptable level and ALARP. This enhanced suite of control measures included:
  - Utilizing a smaller seismic source to reduce the received noise levels at the seabed
  - Increasing the number of streamers to allow a reduction in survey intensity by increasing acquisition line spacing
  - Establishing exclusion zones to reduce the overlap with southern rock lobster habitat and fishing grounds
  - Designing compensation arrangements for demonstrably affected commercial fishers

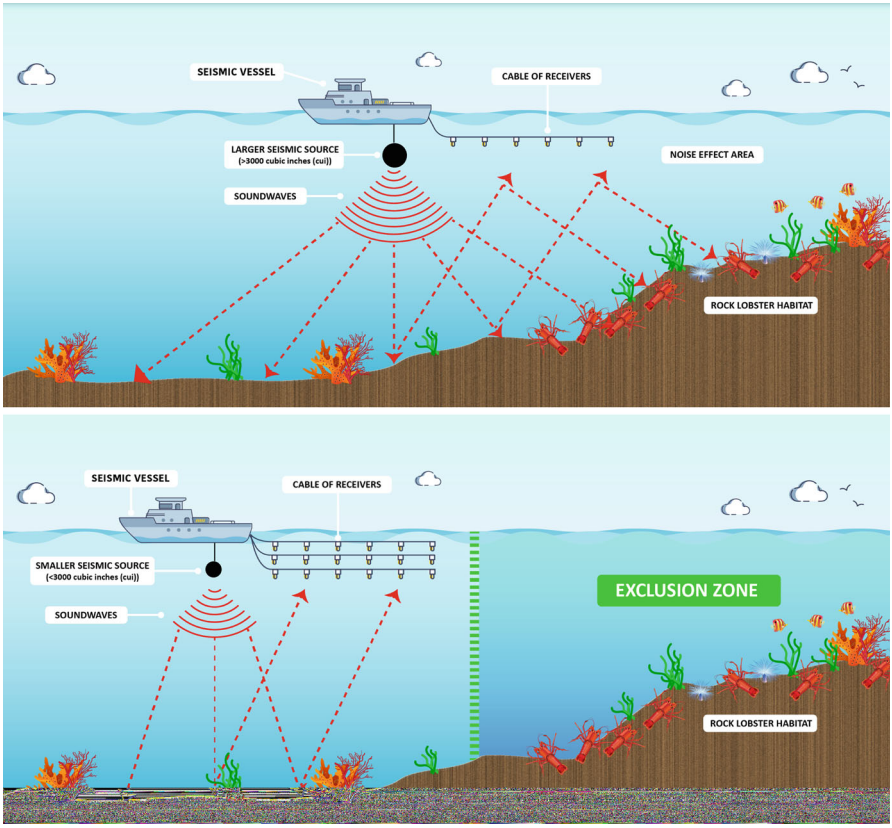



**Fig. 2** Illustration of the approved approach (top) versus post approval adaptive management approach (bottom) in response to new information

- The adaptive management approach implemented provided confidence that underwater noise impacts to a commercially and ecologically important species continued to be managed to acceptable levels and also allowed the proponent to continue with the activity in an environmentally responsible manner (Fig. 2).

## Case Study 2: Adaptive Management in Response to In Situ Fauna Monitoring

- An MSS activity was proposed to be undertaken adjacent to an area of known upwelling that supports elevated primary and secondary productivity formally defined as a biologically important area (BIA) for the pygmy blue whale foraging.

Blue whales are listed as endangered under Australian environmental law, and there is a recovery plan in place to support the recovery of this species (Department of Environment 2015). Anthropogenic noise is identified as a priority threat to recovery that needs to be carefully managed.

- During the assessment process, NOPSEMA required the activity to be restricted spatially so that it did not take place inside the defined blue whale foraging BIA. This was necessary because a case could not be made that impacts to blue whales in this area could be managed to an acceptable level during the foraging season.
- The environmental impact assessment in the EP was based on the prediction that the potential for the activity interact with blue whales was low given that the activity location was outside of the defined BIA boundaries. This assumption was consistent with available data on the distribution of blue whales in the region of the proposed activity.
- However, after the EP was approved and during the implementation of the activity, the proponent observed high numbers of blue whales in the activity area outside the defined BIA. Although the EP anticipated the potential for some blue whales to be present outside the BIA, the observed numbers were higher than what was predicted in the accepted EP, indicating a greater potential for impact to foraging blue whales.
- In response, the proponent was expected to adequately evaluate the change to the operational environment and adapt the management of the activity in accordance with their EMS. In particular, the EMS required the proponent to adopt specific measures to ensure impacts to blue whales continued to be managed to ALARP and acceptable levels.
- The enhanced suite of control measures in this scenario included:
  - Additional observation platforms and enhanced marine fauna detection capability
  - Increasing the size of mitigation zones to account for more effective noise propagation in deeper water
  - The cessation of nighttime operations due to the lack of whale detection capability
- This example of an effective adaptive management approach highlights the importance of having measures in place to allow the proponent's activity to rapidly adapt when the level of marine fauna detection is higher than predicted. The timing and distribution of marine fauna presence and behavior is often highly variable annually and interannually, and incomplete baseline information may mean the expected presence and abundance of noise-sensitive fauna may differ from observations. It is particularly important that management arrangements for activities proposed in sensitive marine fauna habitats include measures to adaptively manage unexpected fauna interactions so that impacts remain acceptable for the life of the activity (Fig. 3).

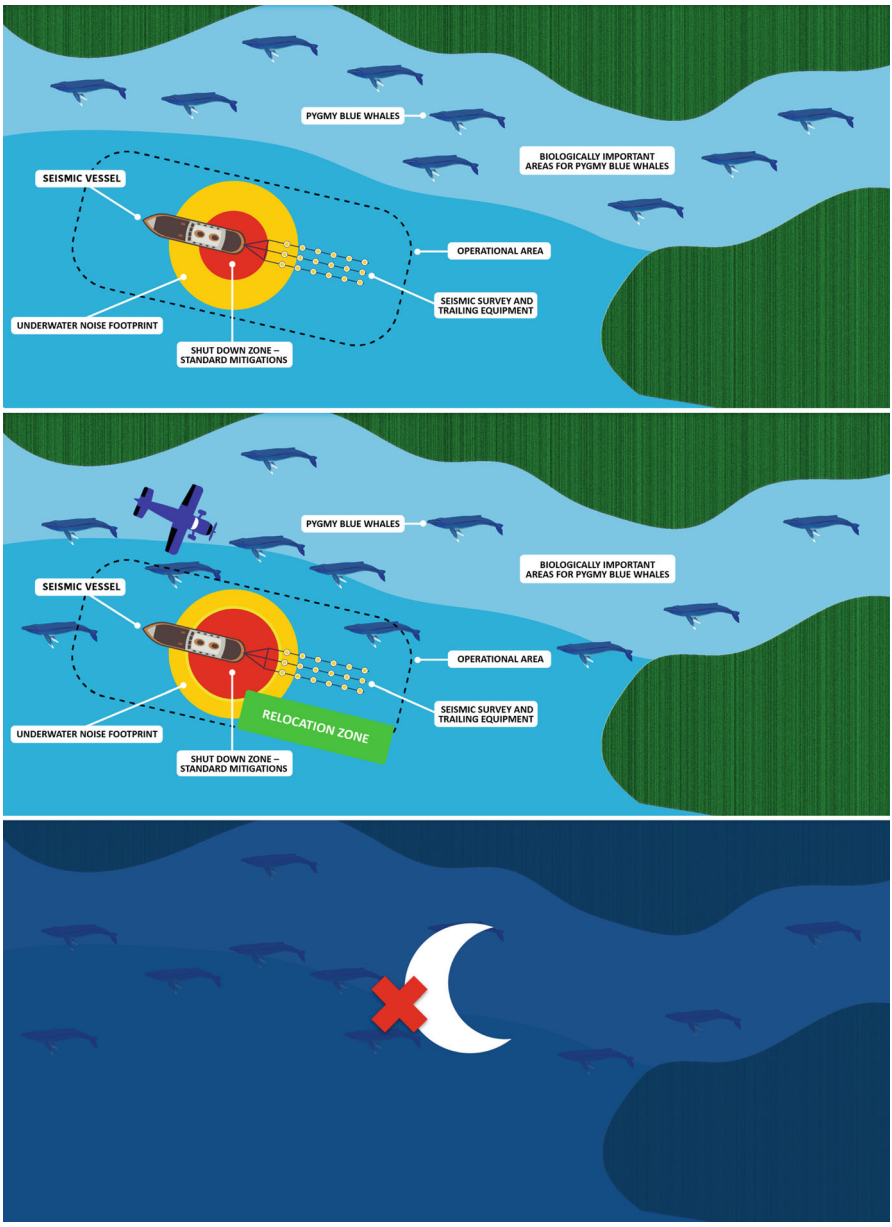



**Fig. 3** Illustration of the approved approach (top) versus post approval adaptive management approach (middle) and further adaptive management in response to a change in the environmental setting – no night-time operations (bottom)

## Conclusion

Industry and the research community need to work together to address research priorities in a timely, cross-disciplinary, collaborative, and integrated way. Varied, balanced, and transparent research outputs with clear pathways to adoption in risk and impact assessment, regulatory decision-making, and activity monitoring and management will reduce uncertainty and improve confidence in the effectiveness of environmental management.

Monitoring and adaptive management will continue to play a vital role in ensuring new science, and new technology, and unanticipated changes in the environment are taken into account by proponents through their EMSs during project implementation. A well-managed and flexible environmental management approach that triggers adaptive management in response to change can be an effective mechanism for addressing some areas of scientific uncertainty and improve confidence that impacts and risks continue to be managed to an acceptable level for the life of the activity.

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# Geophysical Industry Marine Mammal Conservation Under the Gulf of Mexico Proactive Regulatory and Observational Program (GOM-PROP)

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**Abstract**

The United States National Marine Fisheries Service (NMFS) issues permits for incidental marine mammal harassment under the U.S. Marine Mammal Protection Act. Authorizations may be issued on a project-by-project basis via incidental harassment authorizations (IHAs), or on a programmatic basis using rulemaking to issue an incidental take regulation (ITR) for a period of up to five years, with individual projects receiving letters of authorization (LOAs). In January 2021, NMFS issued the first ITR encompassing the programmatic activities of competitive industry interests for the energy geoscience industry in the Gulf of Mexico. The Rule became effective in April 2021, and under the ITR, LOAs are issued for geophysical activities related to oil and gas exploration and development in the region. The ITR requires LOA holders to "collectively be responsible for compilation and analysis of [protected species monitoring] data for inclusion in subsequent annual synthesis reports." The "Gulf of Mexico Proactive Regulatory and Observational Program" (GOM-PROP) was conceptualized to achieve compliance with this provision. The approach to achieving this comprehensive reporting across dozens of companies with various contractors and subcontractors considered issues such as managing confidential business information while facilitating collaboration; achieving consistency in data collection and management; logistics of compiling and storing data; achieving participation from LOA holders; and enabling industry to lead subsequent ITR development. GOM-PROP works closely with U.S. regulatory agencies to ensure reporting will meet the standards and needs and will appropriately inform compliance and future permit applications.

**Keywords**

Seismic · Industry · Protected species observers · Monitoring · Mitigation · Conservation · Marine mammal

## Introduction

In recent years, academia, government, industry, and the general public have paid keen attention to the potential effects of anthropogenic underwater sound on marine life, and particularly as pertains to marine mammals. Exposure to underwater sound has the potential to contribute to hearing threshold shifts or elicit behavioral responses (Southall et al. 2019; Southall et al. 2021). The biological significance of sound exposure, both with respect to individuals and the population, is the subject of significant ongoing research, with methods like Population Consequences of Disturbance (PCoD) modeling showing significant promise in appropriately assessing the significance of exposure over time (Keen et al. 2021).

Anthropogenic sound can generally be divided into impulsive or continuous sounds, which are treated separately under existing regulatory schemes. While

definitions vary, impulsive sounds can be characterized by their kurtosis when the signal is detectable above the effective quiet threshold (EQT) (Martin et al. 2020). Both types of signals may also be characterized as tonal or broadband. Tonal signals are those of a specific frequency, whereas broadband signals contain many frequencies. Marine mammal physiological and behavioral responses vary based on the characteristics of the signals to which they are exposed (National Marine Fisheries Service 2018).

In addition to the type of signal, the context in which the sound exposure occurs can influence behavioral responses. These contextual factors may include the individual and group behavioral state; group composition; age and sex of the individual; and age, sex, and group composition of the larger group (Ellison et al. 2011; Southall et al. 2021). Certain species may also be more sensitive to certain sounds, or may dampen hearing in anticipation of sounds (Nachtigall and Supin 2013; Nachtigall et al. 2016; Nachtigall et al. 2018).

Seismic surveys are used for a variety of applications, including hydrocarbon exploration, determination of appropriate carbon capture and sequestration (CCS) sites, and evaluation of locations for offshore infrastructure (e.g., wind farms). Sound from seismic surveys has been a particular focus for the regulatory, academic, and conservation communities. Whereas in most cases anthropogenic sound is incidental to the activity, the production of sound in seismic survey is essential to obtaining geological images of the subsurface. In the marine environment, compressed air sources, colloquially referred to as air guns, are used to produce a broadband, impulsive signal of approximately 0.1 s every 10–20 s as the vessel tows the source(s) at approximately 5 knots (Gisiner 2016). Compressed air sources are typically necessary to image the deep geological features required for hydrocarbon exploration and CCS storage locations. In order to achieve the acoustic output levels necessary for imaging, commercial seismic surveys utilize an array of multiple compressed air sources.

In the United States, marine mammals are protected by the Marine Mammal Protection Act (MMPA). The MMPA prohibits "harassment" of marine mammals, which is defined as, ". . .any act of pursuit, torment, or annoyance which— (i) has the potential to injure a marine mammal or marine mammal stock in the wild; or (ii) has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering" (16 U.S.C. § 1362). The "take" of marine mammals is defined as attempts to, "harass, hunt, capture, or kill," and divided into two levels: Level A, which indicates significant potential to injure a marine mammal or stock; and Level B, which, ". . .disturbs or is likely to disturb a marine mammal or marine mammal stock in the wild by causing disruption of natural behavioral patterns," (16 U.S.C. § 1362). Take of marine mammals may occur incidentally during the course of anthropogenic activity in the marine environment. Incidental take of marine mammals is allowable provided that the take is of a small number of individuals; has no more than a negligible impact on the species or stock; and has no unmitigable adverse impact on the species or stock for subsistence use.

NMFS' policy associated with assessing harassment that may occur via exposure to sound is based on the concept that exposure to certain levels of underwater sound can constitute a Level A or Level B take of a marine mammal based on the potential resultant hearing threshold shift. Threshold shifts are characterized as either permanent threshold shifts (PTS) or temporary threshold shifts (TTS). Permanent threshold shifts are not fully recoverable, and result in permanent suppression of hearing capabilities around the range of frequencies to which the receiver was exposed. Temporary shifts are recovered within hours to days of the exposure. Only PTS is considered injurious and, therefore, would constitute Level A take.

Whether PTS occurs is dependent upon the hearing capability of the animal receiving the signal. In the United States, marine mammals are divided into five generalized hearing groups to evaluate their hearing ranges, given in Table 1 (National Marine Fisheries Service 2018). The PTS onset thresholds for each hearing group for both impulsive and nonimpulsive sound are estimated in Table 2 based on studies of TTS and general physiological characteristics and hearing studies for species within the hearing group (National Marine Fisheries Service 2018). In the United States, NMFS applies a received level of 160 dB root mean square sound pressure level re 1 μPa as the threshold beyond which behavioral changes that constitute Level B take of marine mammals exposed to impulsive underwater sound would potentially occur. Exposure to this threshold does not necessarily always result in Level B take defined under MMPA (change in behavioral pattern); that is, detection of a sound of a given amplitude does not always elicit a behavioral response. Similarly, exposure to a sound at the Level A take threshold does not necessarily result in PTS for every species or individual in the hearing group. These thresholds should be considered precautionary and intended to be preventative in order to provide sufficient protection to the animals, rather than indications of when a Level A or B take is most likely to occur (Gisiner et al. 2020).

Incidental harassment of marine mammals by underwater sound produced during anthropogenic activities, including seismic surveys, is authorized either via an

**Table 1** Hearing group generalized hearing ranges for marine mammal species. All ranges are given for animals underwater (National Marine Fisheries Service 2018)

| Hearing group | Generalized hearing range |
|---|---|
| Low-frequency (LF) cetaceans (baleen whales) | 7 Hz to 35 kHz |
| Mid-frequency (MF) cetaceans (dolphins, toothed whales, beaked whales, bottlenose whales) | 150 Hz to 160 kHz |
| High-frequency (HF) cetaceans (true porpoises, *Kogia*, river dolphins, cephalorhynchid, *Lagenorhynchus cruciger & L. australis*) | 275 Hz to 160 kHz |
| Phocid pinnipeds (PW) (underwater) (true seals) | 50 Hz to 86 kHz |
| Otariid pinnipeds (OW) (underwater) (sea lions and fur seals) | 60 Hz to 39 kHz |

**Table 2** Onset level of permanent threshold shifts (PTS) for impulsive and nonimpulsive sound by marine mammal hearing group. All thresholds are underwater and therefore referenced to re 1 μPa (National Marine Fisheries Service 2018)

| | PTS onset thresholds (received level) | |
|---|---|---|
| Hearing group | Impulsive | Nonimpulsive |
| Low-frequency (LF) cetaceans | **Cell 1**<br>$L_{p,0\text{-pk,flat}}$: 219 dB<br>$L_{E,p,LF,24h}$: 183 dB | **Cell 2**<br>$L_{E,p,LF,24h}$: 199 dB |
| Mid-frequency (MF) cetaceans | **Cell 3**<br>$L_{p,0\text{-pk,flat}}$: 230 dB<br>$L_{E,p,MF,24h}$: 185 dB | **Cell 4**<br>$L_{E,p,LF,24h}$: 198 dB |
| High-frequency (HF) cetaceans | **Cell 5**<br>$L_{p,0\text{-pk,flat}}$: 202 dB<br>$L_{E,p,HF,24h}$: 155 dB | **Cell 6**<br>$L_{E,p,LF,24h}$: 173 dB |
| Phocid pinnipeds (PW) (underwater) | **Cell 7**<br>$L_{p,0\text{-pk,flat}}$: 218 dB<br>$L_{E,p,PW,24h}$: 185 dB | **Cell 8**<br>$L_{E,p,LF,24h}$: 201 dB |
| Otariid pinnipeds (OW) (underwater) | **Cell 9**<br>$L_{p,0\text{-pk,flat}}$: 232 dB<br>$L_{E,p,OW,24h}$: 203 dB | **Cell 10**<br>$L_{E,p,LF,24h}$: 219 dB |

incidental harassment authorization (IHA), or via a letter of authorization (LOA) under an incidental take regulation (ITR). IHAs allow for authorization of a specific activity for a period of up to one year for activities that meet the MMPA standards: only a small number of individuals affected; no more than a negligible impact on the species or stock; and no unmitigable adverse impact on subsistence use. An ITR allows for the authorization of a given set of activities over a five-year period, where individual activities are authorized via LOAs. Per the National Marine Fisheries Service, the negligible impact standard is applicable across all activities covered under the ITR; the small numbers standard is applied at the LOA level (50 C.F.R. Part 217 §§ 217.180 - 217.188-217.189).

The U.S. Gulf of Mexico is home to a substantial amount of geophysical survey activity to support ongoing hydrocarbon exploration and development in the region. (This activity is expected to continue to increase as the Gulf of Mexico progresses into offshore wind development and carbon capture, sequestration, and storage activities, both of which require geophysical surveys.) Accordingly, in 2002, the U.S. Bureau of Ocean Energy Management (BOEM, then the Minerals Management Service) petitioned NMFS for development of an ITR to govern geophysical activities for hydrocarbon development in the Gulf of Mexico. This petition underwent multiple petitions prior to the filing of a lawsuit challenging the proposed ITR (NRDC v. Jewell 2010). BOEM revised the petition a third time in 2011, and a settlement agreement was implemented in 2013 that allowed for the authorization of geophysical surveys in the Gulf of Mexico provided that these surveys abided by a set of mitigation measures. This agreement was amended in 2016, and extended in 2017, 2018, and 2020 to allow for completion of the ITR. The Final Rule was released in January of 2021, and the final ITR (50 C.F.R. Subpart S – Taking Marine

Mammals Incidental to Geophysical Survey Activities in the Gulf of Mexico, referred to herein as the "Gulf of Mexico ITR") became effective on April 19, 2021, for a period of five years.

Data collection regarding marine mammal detection is conducted by observers aboard survey vessels, and is required under the ITR. Visual monitoring for marine mammals during geophysical surveys under the Gulf of Mexico ITR must be conducted by qualified Protected Species Observers (PSOs). Similarly, acoustic monitoring is conducted by Passive Acoustic Monitoring Operators (PAMOs). LOA holders are required to employ third-party observers, trained via an NMFS-accepted course and who have received NMFS approval, with no responsibilities other than monitoring, data collection, and communication with relevant crewmembers regarding mitigation implementation. The primary purpose and objective of both PSOs and PAMOs is to monitor for marine mammals within a designated proximity to the survey and initiate mitigation measures as necessary. It is imperative to recognize that data collection is a secondary responsibility; observers' primary objective is implementation of required mitigation measures.

Specific mitigation requirements vary based on the operational details of the survey. Compressed air surveys can be subdivided into deep penetration surveys – those using arrays of compressed air elements with a total volume of greater than 1500 cubic inches – and shallow penetration surveys, which use small arrays or a single compressed air source with a volume under 1500 cubic inches, or an electro-mechanical acoustic source (e.g., sparkers). At least 30 min prior to activating a deep penetration acoustic source, an area of 1000 m around the array must be monitored and remain clear of marine mammals (200 m for shallow penetration). For both deep and shallow penetration surveys, seismic survey operators are required to "ramp-up" the acoustic array by slowly increasing the number of active elements over a period of at least 20 min. If, during the course of operations, a marine mammal approaches within 500 m of the acoustic source (100 m for shallow penetration surveys), the source must immediately be shut down. This shut down requirement is waived for small delphinids (genera *Steno*, *Tursiops*, *Stenella*, and *Lagenodelphis*), which are known to approach and bow-ride vessels even during periods when the acoustic source is active.

During operations, PSOs and PAMOs must collect data via an electronic reporting form. However, while the United States requires the collection of certain data, there is currently no standardized reporting format. LOA holders must submit a report to NMFS within 90 days of survey completion or following the expiration of an issued LOA, or annually if the LOA is for a period of more than one year. These individual reports must include operations conducted and marine mammals detected; documentation methods, results, and interpretation; summarize all operations and sightings; provide information regarding where the acoustic source was used; and demonstrate validity of the PAM system. LOA holders must also submit track lines of the specific locations in which the survey was conducted.

A requirement unique to the Gulf of Mexico ITR is development of a comprehensive annual report analyzing all geophysical activity in the Gulf of Mexico during the ITR year. This requirement stipulates that, "LOA-holders must contribute to the

compilation and analysis of data for inclusion in an annual synthesis report addressing all data collected and reported through annual reporting in each calendar year. The synthesis period shall include all annual reports deemed to be final by NMFS in a given one-year reporting period" (50 C.F.R. § 217.185(d)(2)). The annual report is independent of individual reporting requirements described above. The comprehensive annual report is intended to provide a summary of: geophysical survey activity (including type, location, month, and source status); monitoring effort including source status, location, and visibility; mitigation measures implemented by survey type and location; sighting rates of marine mammals during periods with and without source activity including initial sighting distance, closest point of approach, observed behaviors and movements, and analysis of effects of factors influencing detectability; total take across all activities; conclusions from monitoring during the year; and recommendations for adaptive management.

The adaptive management provision of the Gulf of Mexico ITR is intended to allow NMFS to consider new information to determine whether modification of mitigation and/or monitoring measures may be appropriate without undergoing a full rule revision. Adaptive management may consider the annual reports, research on the potential effects of sound on marine mammals, and any information that may indicate take inconsistent with what was authorized by the LOA. The annual adaptive management process includes other applicable regulatory agencies (BOEM and the Bureau of Safety and Environmental Enforcement, BSEE) and industry.

The lack of a standardized reporting format has created challenges for analysis of PSO and PAMO data in the U.S. Detection data varies widely between regions, making comparisons within the United States and abroad difficult (Milne et al. 2019). Even within the U.S. Gulf of Mexico, detection data may not be comparable across the entire PSO dataset (Barkaszi and Kelly 2018). The U.S. Gulf of Mexico is currently home to 28 cetacean species and one species of manatee; three additional species have been sighted nearby, but are not considered endemic in the Gulf region (Würsig et al. 2000). Many of these species are difficult to distinguish, particularly during the typically brief glances afforded to PSOs.

The diversity of species, coupled with the mutable nature in which PSO data have historically been collected, presented a significant challenge for compliance with the annual reporting of the Gulf of Mexico ITR.

## Development of the Gulf of Mexico Proactive Regulatory and Observational Program (GOM-PROP)

Finalization of the Gulf of Mexico ITR highlighted the need for industry to proactively collaborate toward a structure that would facilitate satisfying the requirements associated with producing a comprehensive annual report. This posed a particular challenge, as companies with an operational interest in the region are competitors and therefore are understandably cautious about data sharing. As the global trade association for the energy geoscience industry, the EnerGeo Alliance was

well-positioned to serve as the intermediary through which to centralize the data necessary to achieve reporting goals. GOM-PROP members for year 1 included the following companies (in alphabetical order):

- Beacon Offshore Energy
- BHP
- bp
- CGG
- Chevron
- EnVen
- ExxonMobil
- Hess Corporation
- Houston Energy
- ION
- Kosmos Energy
- LLOG Exploration
- Magseis Fairfield
- Occidental Petroleum Corporation
- PGS
- PXGEO
- Quarter North Energy
- slb
- Shell
- Talos
- TGS

ION withdrew from the program at the end of year 1 due to unrelated business challenges. Equinor joined the program for year 2, maintaining membership at 21 participants as of November 2022.

## Data Collection

Data were collected at sea during operations conducted under an LOA by qualified PSOs and PAMOs, as described above. Data included effort, environmental conditions, operational status of the seismic source, cetacean acoustic detection and sighting events, and sea turtle visual detection. Herein, acoustic events are referred to as "detections" and visual sightings as "observations." Collectively, visual and acoustic events are referred to as "sightings."

For each detection or observation, PSOs recorded the following information: time; vessel position; vessel course; water depth; species; number of animals; group age/sex composition; sighting/detection distance and bearing; animal heading and movement; behavior of animal(s); vessel activity; source status; and environmental data. Not all data could be accurately recorded for each event, and PSOs reported the identification "unknown" if a determination could not be made at the time. Protected

species were identified to the species level if possible, or to the closest identifiable group (e.g., unidentifiable delphinid).

These data were then imported into a bespoke database by syncing the observer forms returned to GOM-PROP members by contractors with the database and importing the data.

## Database Development and Use

The database endeavored to provide more than simply satisfying regulatory objectives and would also provide operational value to GOM-PROP members. Marine Remote Services LLC (MRS) developed and deployed an independently administered, central Protected Species Observer (PSO) database solution to support a comprehensive reporting system across dozens of companies with various contractors and subcontractors. The database was designed to unify PSO sightings collected by multiple PSO service providers, and to consider complex Industry issues such as: (1) managing confidential business information while facilitating collaboration; (2) achieving consistency in data collection and management; (3) logistics of compiling and storing data; (4) ensuring data collection meets analysis requirements; and (5) achieving participation from LOA holders.

The resulting database, firstKnowledge™, is a standalone, fully functional, and independent web-served PSO time-series database which is designed to contain user-submitted protected species detection data in an industry-supported format. In its internet-connected mode, the product is referred to as SaaS or "Software as a Service." In unconnected scenarios, the database defaults to an "on-prem" database instance allowing all sightings and events to be recorded in a file for later uploading to the SaaS portal. In the cloud configuration, the database is hosted on a secure and redundant web server with data integrity, protection, and duplication built-in. The access portal allows for portal sign-in and a specific database access authentication which determines what functionality is available and which dataset details are accessible to different users.

Basic data entry into the firstKnowledge™ database is performed by import widgets that authorized users are assigned. For users that have current data collection systems in place, the system provides for data ingestion with the use of custom widgets programmed to field map those datasets over to firstKnowledge™ data fields and tables from a variety of formats (including the frequently used .xls and .csv) by the batch processing method. For those users without existing systems, the system provides an intuitive input screen that performs content and format validation when populating the collection of data tables, including time stamping and other common database functions on an individual sighting or event record basis. All records were scrutinized for duplication and an audit trail established for all subsequent data manipulation performed.

The database is based on an Elastic Stack and used Elasticsearch which allows for rapid searching for all types of data. Data are stored in Elastic Stack, enabling the system to interactively explore, visualize, and share insights into the data collected.

Elasticsearch stores and indexes data to facilitate searches. A basic set of use cases were created that describe stakeholder data needs. Use cases incorporate the following:

- Multiple authentication levels addressing the basic and advanced stakeholder requirements.
- A secure data collection portal with secured individual contributor access.
- User dashboard serving metrics from user-specific datasets.
- Basic reporting capability.
- Fulfilling the custom needs of industry analysis.
- Managing, integrating, and analyzing spatial information using Elasticsearch as a geographic information system.

The reliability and trustworthiness of the data throughout its lifecycle must be maintained, and therefore, it is essential that any corrections or revisions are validated and traceable. Additionally, data security must be closely monitored to ensure only authorized users are able to access information. To ensure data integrity, data input and validation are continuously monitored, as are data security and backup. Finally, access control is maintained and an audit trail is established.

Finally, firstKnowledge™ is designed so that specific details of data contributions from system users will be maintained in separate tables, ensuring anonymity, while still being able to connect to the relevant record and allow for the use of table and foreign keys during specific data mining and reporting operations.

## Results from Year One

During the first year of GOM-PROP, from April 2021 to April 2022, more than 95% of LOA activity days authorized under the ITR were issued to GOM-PROP members. The limited number of LOAs issued to nonmembers was largely small-scale activities (i.e., single source or small array over few days). The extensive participation by industry allowed for a particularly comprehensive review of protected species detections and observations during the first year of ITR implementation. The completeness of this dataset will allow for valuable comparisons during subsequent program years.

### Summary of Geophysical Activity

Effort data were analyzed across the seven geographic management zones identified in the GOM ITR (Fig. 1). These zones were defined for management purposes. Sightings were assigned to zones based on the geographic coordinates. Next, activities were sorted based on source activity (Table 3). Survey effort when the source was not active (excluding transit) represented 20.2% of all survey time. For all vessel activities recorded in the database and representing total time on the water,

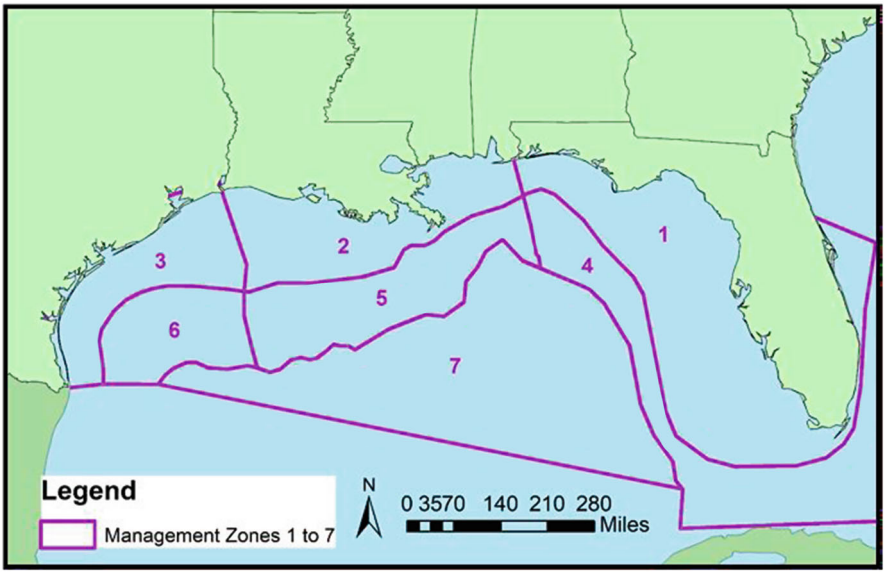



**Fig. 1** The U.S. Gulf of Mexico with the seven management zones used in the ITR highlighted with purple lines

**Table 3** Vessel activity codes reported in effort data sorted by source activity category

| Vessel activity code | Category |
|---|---|
| Data acquisition | Active source |
| Deploying/retrieving equipment | Inactive source |
| Line change[a] | Active source |
| Standby | Inactive source |
| Testing[a] | Inactive source |
| Transit | Transit; inactive source |
| Other[a] | Active source |
| No data[a] | Active source |

[a]Indicates that the source activity status is uncertain based on code. The source was assumed to be active unless otherwise indicated

transit represented 5.4% of the total time on water; source active codes represented 75.5%; and source off represented 19.1% of the total time on water.

PSO effort data were collected from July 2021 to February 2022. While the ITR year extended from April 2021 through April 2022, the first survey undertaken under the ITR did not begin until July, and surveys in progress at the conclusion of the ITR year were not included in the analysis at the request of NMFS. Data were collected in zones 2, 3, 5, 6, and 7 (Fig. 1), with acquisition occurring in all of these areas except zone 2. Coordinates for ten observations placed the sightings as outside the Gulf of Mexico. Because operations were only authorized within the GOM, these coordinates are most likely erroneous, and therefore, the effort was categorized as "unknown zone." Hours of effort by source status are show in Table 4.

**Table 4** Effort data from July 2021 to February 2022 divided by zone. Geospatial coordinates for ten sightings erroneously placed the observations outside the Gulf of Mexico; these are listed as "unknown zone"

| Zone | Total active effort (hours) | Total transit effort (hours) | Total deploying/standby/other/no data (hours) |
|---|---|---|---|
| 2 | 0 | 8.8 | 0 |
| 3 | 7.8 | 31.3 | 20.6 |
| 5 | 404.1 | 16.0 | 44.5 |
| 6 | 251.4 | 34.8 | 194.2 |
| 7 | 606.6 | 0 | 60.6 |
| Unknown | 4.2 | 0 | 1.7 |
| Total | 1274.1 | 91.0 | 321.5 |

The distance at which animals can be detected is affected by visibility. Visibility conditions were documented during daylight operational hours. For both active and inactive survey time, visibility exceeded 5 km for 91.1% of the recorded time; was between 2 and 5 km 5.3% of the time; between 0.5 and 2 km 1.8% of the time; and less than 500 m 1.8% of the time. For effort during the time that the source was active, passive acoustic monitoring (PAM) was used 99.9% of the time (day and night). When visibility was recorded while the source was active, visibility exceeded 5 km 95.0% of the time; was between 2 and 5 km 2.6% of the time; between 0.5 and 2 km 1.2% of the time; and less than 500 m 1.2% of the time. The use of PAM allowed for continued operation during nighttime hours and when visibility was limited to distances smaller than the exclusion zone required to be monitored for mitigation.

Beaufort (BF) sea state also affects the ability of PSOs to detect marine life and was recorded during visual monitoring effort. Sea state is measured on a scale of 1 to 8, and the ability to detect certain species (e.g., beaked whales) decreases with an increase in sea state, particularly above Beaufort 5 (Barlow 2013, 2015). BF sea state ranged from BF1 to BF7 during periods of visual monitoring. Visual monitoring includes both active source time, as well as inactive time (e.g., during transit and preclearance watch). The most frequently reported sea states were BF2, BF3, and BF4, which represented 26.5%, 24.6%, and 27.2% of recordings, respectively, for a total of 78.3% of reported sea states. The remaining 21.7% of observations were divided between BF1 and BF5–BF7.

## Mitigation Measures

Mitigation measures, specifically shutdown of the seismic source or delay of ramp-up, were implemented in compliance with each survey's LOA. There were 26 instances of marine mammal visual observations recorded, and an estimated 106 individuals. Four observations of sperm whales (eight individuals), one obser-

vation of unidentified dolphins (two individuals), and one observation (two individuals) of unidentified cetaceans. Ten acoustic detections of an estimated 25 individuals were recorded, all of which were unidentified dolphins. (Total effort is reported in Table 4.)

When animals were sighted within the exclusion zone while the source was powered, the source was shut down until the animal(s) were observed moving outside the exclusion zone or were not sighted for 30 min. Soft start was similarly delayed when animals were sighted in the exclusion zone.

Three of the four sperm whale observations occurred while the source was powered, resulting in shutdowns ranging from 12 to 36 min for a total mitigation time of 1 h and 28 min. For context, total effort was 1686.6 h, with the source active or deploying/standing by for 1595.6 h. The remaining sperm whale observation was outside the exclusion zone and mitigation was not required. A single unidentified cetacean observation resulted in a shutdown of 12 min. Sperm whales require an exclusion zone of 1500 m, compared with 500 m for other species sighted. All visual observations, including those that occurred while the source was inactive or the vessel was in transit, are shown in Table 5.

**Table 5** Summary of all marine mammal detections by source or vessel activity

| Species | Active source (Groups/ individuals) | Inactive source (Groups/ individuals) | Transit (Groups/ individuals) | Total (Groups/ individuals) |
|---|---|---|---|---|
| Atlantic spotted dolphin | 0/0 | 0/0 | 1/6 | 1/6 |
| Clymene dolphin | 0/0 | 1/3 | 0/0 | 1/3 |
| Common bottlenose dolphin | 0/0 | 1/3 | 2/10 | 3/13 |
| Gervais' beaked whale | 0/0 | 1/2 | 0/0 | 1/2 |
| Pantropical spotted dolphin | 0/0 | 1/7 | 1/23 | 2/30 |
| Risso's dolphin | 0/0 | 2/20 | 0/0 | 2/20 |
| Sperm whale | 4/8 | 4/5 | 0/0 | 8/13 |
| Unidentifiable cetacean | 1/2 | 0/0 | 0/0 | 1/2 |
| Unidentifiable dolphin | 1/2 | 4/11 | 1/2 | 6/15 |
| Unidentifiable whale | 0/0 | 1/2 | 0/0 | 1/2 |
| Total | 6/12 | 15/53 | 5/41 | 26/106 |

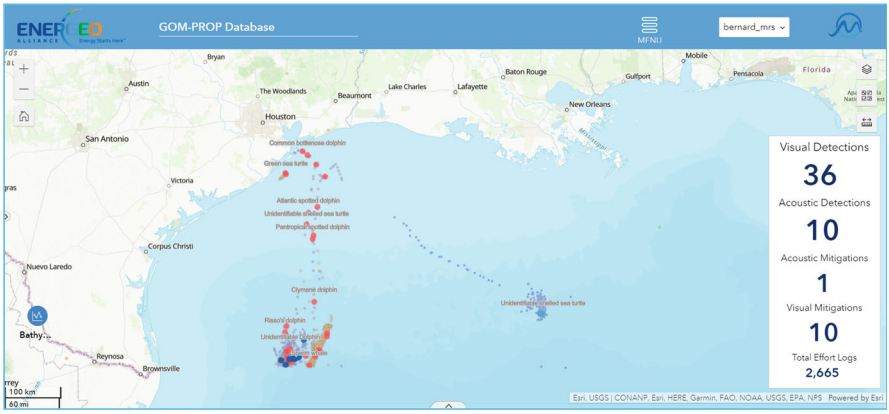



**Fig. 2** First-year GOM-PROP detections of protected species groups recorded in the firstKnowledge™ database, detailing effort logs (small dots), acoustic detections (blue dots), and visual detections (red dots)

One acoustic detection resulted in a 38-min delay to soft start. The remaining acoustic detections occurred while the source was powered and did not result in shutdowns, as small delphinid detections do not require shutdown of the source.

Visual and acoustic detections, along with effort logs, were recorded in the firstKnowledge™ database (Fig. 2).

## Detection Rate

Visual detections were evaluated for frequency over both time and space. Common bottlenose dolphins were the most frequently sighted species in Zone 3 at a rate of 0.22 individuals per hour or 0.02 individuals per kilometer. In Zone 6, Risso's and Pantropical Spotted Dolphins were the most commonly sighted delphinids at 0.042 and 0.048 individuals per hour respectively. Sperm whales were the most commonly sighted whale and were visually detected in Zones 6 and 7 at approximately 0.01 individuals per for both zones. Detection rates by zone are reported per hour (Table 6) and distance surveyed (Table 7).

Due to the low number of detections during the first year of the program, it was not possible to examine whether environmental factors influenced detection rate.

## Summary

The majority of the monitoring effort (75.5%) during the first year of the program occurred while the source was active. Inactive source time while the vessel was in transit or the source was deactivated (for maintenance, preclearance watch, etc.)

**Table 6** Visual detection rates per hour by zone and source activity status. Zeros indicate effort but no visual detections; dashes indicate no effort for that zone and activity combination

| Species | Active source Groups/ individuals | Inactive source Groups/ individuals | Transit Groups/ individuals | Total Groups/ individuals |
|---|---|---|---|---|
| Zone 3 | | | | |
| Atlantic spotted dolphin | 0.0000/0.0000 | 0.0000/0.0000 | 0.0319/0.1916 | 0.0168/0.1006 |
| Common bottlenose dolphin | 0.0000/0.0000 | 0.0487/0.1460 | 0.0639/0.3193 | 0.0503/0.2179 |
| Pantropical spotted dolphin | 0.0000/0.0000 | 0.0487/0.3406 | 0.0000/0.0000 | 0.0168/0.1174 |
| Unidentifiable dolphin | 0.0000/0.0000 | 0.0973/0.3893 | 0.0319/0.0639 | 0.0503/0.1676 |
| Zone 6 | | | | |
| Clymene dolphin | 0.0000/0.0000 | 0.0052/0.0155 | 0.0000/0.0000 | 0.0021/0.0062 |
| Risso's dolphin | 0.0000/0.0000 | 0.0103/0.1030 | 0.0000/0.0000 | 0.0042/0.0416 |
| Pantropical spotted dolphin | 0.0000/0.0000 | 0.0000/0.0000 | 0.0287/0.6606 | 0.0021/0.0479 |
| Sperm whale | 0.0000/0.0000 | 0.0206/0.0258 | 0.0000/0.0000 | 0.0083/0.0104 |
| Unidentifiable cetacean | 0.0040/0.0080 | 0.0000/0.0000 | 0.0000/0.0000 | 0.0021/0.0042 |
| Unidentifiable dolphin | 0.0000/0.0000 | 0.0103/0.0155 | 0.0000/0.0000 | 0.0042/0.0062 |
| Zone 7 | | | | |
| Gervais' beaked whale | 0.0000/0.0000 | 0.0165/0.0330 | – | 0.0015/0.0030 |
| Sperm whale | 0.0066/0.0132 | 0.0000/0.0000 | – | 0.0060/0.0120 |
| Unidentifiable dolphin | 0.0016/0.0033 | 0.0000/0.0000 | – | 0.0015/0.0030 |
| Unidentifiable whale | 0.0000/0.0000 | 0.0165/0.0330 | – | 0.0015/0.0030 |

represented only about a quarter of monitoring effort, and not all surveys engaged in off-duty monitoring except where required such as during preclearance.

Effort was concentrated predominately in Zones 5, 6, and 7. Given the areas under development and the areas under consideration for future development, it is likely that hydrocarbon exploration will be concentrated in these areas for the remaining years of the current ITR and potentially in future years as well.

GOM-PROP aims to evaluate behavioral responsiveness, or lack thereof, of various species and marine mammal hearing groups to geophysical surveys, as well as quantify the effectiveness of the mitigation measures employed, which are largely aligned with international industry best practices. However, due to the limited number of activities in the first program year, it was not possible to make inferences regarding these objectives.

In four instances, a shutdown of the source was required due to sightings of animals within the regulatory shutdown radius. One shutdown was documented as an "unidentified cetacean," while the other three were sperm whale groups. Given that source shutdown is required for sperm whales at 1500 m rather than the 500 m

**Table 7** Visual detection rates per kilometer by zone and source activity status. Zeros indicate effort but no visual detections; dashes indicate no effort for that zone and activity combination

| Species | Active source Groups/ individuals | Inactive source Groups/ individuals | Transit Groups/ individuals | Total Groups/ individuals |
|---|---|---|---|---|
| Zone 3 | | | | |
| Atlantic spotted dolphin | 0.0000/0.0000 | 0.0000/0.0000 | 0.0019/0.0114 | 0.0015/0.0088 |
| Common bottlenose dolphin | 0.0000/0.0000 | 0.0092/0.0277 | 0.0038/0.0191 | 0.0044/0.0191 |
| Pantropical spotted dolphin | 0.0000/0.0000 | 0.0092/0.0645 | 0.0000/0.0000 | 0.0015/0.0103 |
| Unidentifiable dolphin | 0.0000/0.0000 | 0.0184/0.0738 | 0.0019/0.0038 | 0.0044/0.0147 |
| Zone 6 | | | | |
| Clymene dolphin | 0.0000/0.0000 | 0.0010/0.0029 | 0.0000/0.0000 | 0.0003/0.0009 |
| Risso's dolphin | 0.0000/0.0000 | 0.0020/0.0195 | 0.0000/0.0000 | 0.0006/0.0063 |
| Pantropical spotted dolphin | 0.0000/0.0000 | 0.0000/0.0000 | 0.0017/0.0395 | 0.0003/0.0072 |
| Sperm whale | 0.0000/0.0000 | 0.0039/0.0049 | 0.0000/0.0000 | 0.0013/0.0016 |
| Unidentifiable cetacean | 0.0006/0.0013 | 0.0000/0.0000 | 0.0000/0.0000 | 0.0003/0.0006 |
| Unidentifiable dolphin | 0.0000/0.0000 | 0.0020/0.0029 | 0.0000/0.0000 | 0.0006/0.0009 |
| Zone 7 | | | | |
| Gervais' beaked whale | 0.0000/0.0000 | 0.0031/0.0063 | – | 0.0002/0.0005 |
| Sperm whale | 0.0011/0.0021 | 0.0000/0.0000 | – | 0.0010/0.0019 |
| Unidentifiable dolphin | 0.0003/0.0005 | 0.0000/0.0000 | – | 0.0002/0.0005 |
| Unidentifiable whale | 0.0000/0.0000 | 0.0031/0.0063 | – | 0.0002/0.0005 |

required for most species, the prevalence of sperm whale shutdowns may be related to the larger mitigation zone and the frequency with which the species was detected. The frequency of observation may be due to the greater total effort in Zones 5, 6, and 8, which represent the continental shelf and abyssal area (see Fig. 1). In three of the instances where a shutdown was required, the first observation of the animal occurred within the exclusion zone. In the fourth instance, a group of sperm whales was observed outside the exclusion zone for 69 min before an animal entered the exclusion zone and a shutdown was required. This observation suggests that particular group was not actively avoiding the survey area. While this behavior cannot be extrapolated beyond this anecdotal instance, GOM-PROP presents a unique opportunity to aggregate data over an extended period to consider questions about avoidance and approach as activities authorized under the Gulf of Mexico ITR progress.

Although the number of observations were insufficient to draw substantive conclusions regarding animal responses to seismic surveys, the data collected illuminated a number of areas for potential improvement in the collection of PSO

data. First, widely used source codes for vessel activity do not always clearly indicate whether the source was active, whether certain individual elements were active, or if the source was shutdown (see Table 3). Project proponents may consider adding clarity via metadata definitions for activity codes or adding a "source status" column to more clearly delineate whether the source was engaged. Next, as recommended by RPS (2019), it would be beneficial to include the time and/or distance of infill due to mitigation for protected species. This information would be useful in evaluating whether mitigation measures may increase the number of encounters with protected species, as well as assessing the additional operational costs that may be incurred in certain areas based on historical protected species sightings. Finally, unidentified cetacean sightings be should be refined for use within the Gulf of Mexico. For example, adding an "Unidentified cetacean – not large whale" category would help to definitively rule out sightings of baleen or sperm whales when evaluating unidentifiable observations. PSOs may consider adding further categories as appropriate in order to define unidentifiable observations as clearly as possible and reduce ambiguity in the PSO dataset during analysis without the necessity of reviewing the "notes" section for each encounter.

NMFS is encouraged to consider implementation of a standardized set of columns, including specified units, for PSO data collection. This would not preclude the inclusion of additional data as PSO providers deem relevant and important but would standardize the most essential information (such as time, geographic coordinates, source status, etc.) to facilitate broad-scale analysis of PSO data.

## Future Directions

While only a small number of surveys occurred during the first year of GOM-PROP, the program nonetheless represents an important step forward in conservation efforts in the Gulf of Mexico. Twenty-one member companies, ranging from regional operators to international energy corporations, partnered to ensure the continued safe, sustainable implementation of geophysical surveys in the region. This significant investment demonstrates the Industry's desire to lead the science on important conservation questions.

GOM-PROP was developed with a long-term focus in mind; not only for the current five-year ITR, but for many subsequent ITRs to come. As operations continue, this program will support the development of a robust PSO dataset that has been thoroughly vetted and quality controlled, and, through use of the database, been preprepared for comprehensive analyses and review by scientists and regulators. The use of independent PSOs, and analysis by an independent group of scientists for this report, further demonstrate the industry's commitment to ensuring that operations are rigorously evaluated to ensure compliance with required monitoring and mitigation, as well as critically assessing the potential impacts of operations on marine mammals.

The database also presents a new tool for project proponents to use in planning, allowing for the evaluation of historical protected species sightings within a given

geography. As the dataset grows over time, the GOM-PROP database will represent a unique opportunity to quickly review sightings and plan for potential downtime based on prior sightings. This will assist project proponents in evaluating the likely duration of surveys and inform future ITRs.

The first year of data collection via GOM-PROP successfully developed a bespoke database with necessary data standards and management, and analyzed the first year of data to address questions associated with ITR goals and beyond. This program has established a first-of-its-kind, industry-led effort to ensure that mitigation during operation is implemented and documented and that data collected during operations are reviewed on an annual ongoing basis.

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# Real-Time Monitoring of Underwater Radiated Noise Levels from Ships

111

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**Abstract**

Commercial ships are the most common anthropogenic sound sources in the ocean today. Only a few monitoring stations exist where opportunistic measurements and estimations of ships' underwater radiated noise (URN) can be performed. This chapter describes the development and operation of a cabled real-time hydrophone station, located outside a major port on the west coast of Sweden. The station was designed to record the local ambient noise and obtain opportunistic measurements of large numbers of commercial ships passages in relatively shallow, 46 m, waters. The propagation loss was measured twice to reduce the uncertainty in the SL estimations. During the measurement period, the source level (SL) for 254 opportunistic commercial ship passages was estimated. One particular cargo ship passed the station 23 times, resulting in multiple URN estimations and the possibility to investigate the effect of different propagation loss conditions on the calculated SL. Finally, the calculated SL was compared to SL models commonly used in the production of soundscape maps.

**Keywords**

Source level · Hydrophone station · Ambient noise · Propagation loss · AIS · Commercial vessels · Port · Gothenburg

## Introduction

To get a good estimation of anthropogenic sound sources in the ocean, their source characteristics need to be described. In many oceans, radiated noise from commercial ships is the most common, and often dominating, anthropogenic sound source of ambient noise, and there are international standards and class notations that describe how to measure it (ANSI 2009; ISO 17208-1:2016; Bureau Veritas 2018; ISO 17208-2:2019). The method used to calculate the radiated noise level (RNL) or source level (SL) differs, and the accuracy varies between the standards due to their different requirements regarding, e.g., the measurement setup and estimation of the propagation loss (PL). Additionally, most of them are not designed to work in shallow water (Ainslie et al. 2022). However, the Bureau Veritas (2018) notation covers both shallow and deep waters. It is likely that future international regulations will require knowledge of the radiated noise from commercial ships (IMO 2014), and both the shipping industry and authorities foresee a need for a simple and cost-effective estimation of the SL from commercial ships. One way forward is to have monitoring stations close to ports and shipping lanes, relying on opportunistic measurement as the ships pass by (Hannay et al. 2016). However, it should be noted that a simplified method to estimate the SL with ship passages of opportunity would include uncertainties, typically related to the estimations of PL (Ainslie et al. 2022). The PL depends on environmental parameters, such as the bottom type(s), topography, and sound speed profile, and show a large variation over time but can be measured or modelled. Depending on the purpose of the SL estimations, estimates

based on opportunistic ship passages could still be adequate and answer many questions for industries, scientists, and managers.

Multiple SL estimations from ships can be used to develop and evaluate SL models of various ship categories such as cargo, tankers, and containerships (MacGillivray and de Jong 2021). One of the main usages of such SL models for different ship categories is to make soundscape maps (Farcas et al. 2016; Jalkanen et al. 2018; MacGillivray and de Jong 2021). Soundscape maps are an important tool for regulators to evaluate the environmental impact of noise over large regions. To improve the source models and maintain accuracy of the models as ships are modernized, a large number of ship URNs needs to be estimated each year. To date, however, very few attempts have been made to estimate the SL from a large number of ships, but see Hannay et al. (2016), Karasalo et al. (2017), and Wladichuk et al. (2019).

The aim of this work was to design and deploy a real-time monitoring station close to the inlet to the Port of Gothenburg in Sweden for recording the ambient noise as well as SL estimations from opportunistic passing commercial ships. Further, the aim was also to develop a systematic method for estimating the SL from commercial ships passages in near real-time inspired by Hannay et al. (2016). The Bureau Veritas (2018) measurement and analysis guidance were used when possible. Data on ship movements was recorded locally.

This work was conducted within the EU INTERREG North Sea project Joint Monitoring Programme for Ambient Noise North Sea (JOMOPANS). The project was also funded by the Swedish Agency for Marine and Water Management and the Swedish Transport Administration.

## Design and Operation of the Hydrophone Station

### Technical Description the Hydrophone Station

The hydrophone station was developed by the Swedish Defence Research Agency (FOI) and consists of three main parts. First, a sea unit located at the sea floor, at 46 m depth, consisting of two hydrophones and a transmitter, a data collection system, and a network switch that sends the recorded data to shore. Second, a hybrid fiber optic cable that connects the sea unit with the third part, a land unit that supplies power to the sea unit and receives data and stores it on a hard drive (Fig. 1). The system described here was in operation from November 2021 to March 2022. An AIS antenna and a receiver were also connected to the computer in the land unit, storing the AIS data locally. The land unit was remotely controlled through a 3G modem and included the possibility to reprogram the data collection at the sea unit. A first version of the station was in operation at the study site from May 2019 to February 2020.

The sea unit consisted of a waterproof Delrin plastic housing mounted on a 125 kg concrete foundation. Two hydrophones (Neptune D70h) and one transmitter

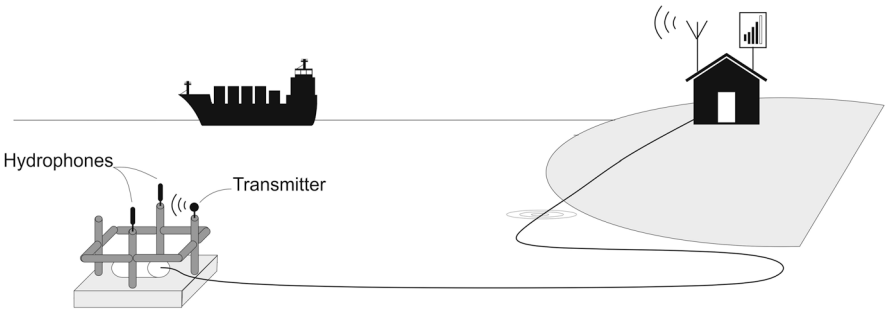



**Fig. 1** A schematic drawing of the three parts of the Vinga station. To the left, the sea unit with the white cylindrical Delrin housing on a 125 kg concrete foundation (light gray) and the hydrophones and transmitter in a PVC structure (gray) is shown. A hybrid fiber optic cable connects the sea unit to the shore where a land unit, located in a house, received and processed the data

(Neptune D45) were mounted in a PVC structure 1.5 m above the seabed with a 0.75 m separation (Fig. 1).

The sea unit contained a programmable real-time unit from National Instruments, NI cRIO 9063. A NI 9250 data acquisition card and a NI 9263 signal generation card was connected to the real-time unit. The main purpose of the real-time unit was to redirect the data to the output of the tcp/ip network for data transmission to the land unit. The Neptune D70h hydrophones were connected to the inputs of the NI 9250 card. The card had a user selectable sampling frequency of up to 102.4 kHz. The signal resolution was 24-bits. The hydrophones, having built in 40 dB preamplifiers, had a sensitivity of −153 dB re 1 V/μPa below 10 kHz. From 10 kHz up to 25 kHz, the sensitivity dropped monotonically to −157 dB re 1 V/μPa. The 2σ of the sensitivity was approximately +/−1 dB. The hydrophones were wired to separate amplifier units with 0 dB gain settings for optimal impedance matching with the data acquisition unit.

The 2-kilometre hybrid cable had 12 fiber optical wires for signal transmission and 2 AWG16 copper wires supplying the sea unit with power. Only two of the fiber optical wires were used. The communication between the land part and the sea part takes place with the help of "Network stream" technology provided by NI LabVIEW software package.

In the Delrin housing, temperature and humidity sensors were added to monitor the climate within the sea unit to warn if the temperature gets too high or if there is any leakage.

A Neptune D45 transducer was connected through a cable to the NI 9263 signal generation card in the sea unit. It was used to function test the hydrophones regularly in situ when it was located on the sea floor. At set times (daily at 05:05 local time), fifteen 1 ms long calibration pulses in the frequency range 2–20 kHz were transmitted with the transducer and captured with the hydrophones. The purpose was to investigate if there were any measurement variations (short or long term) giving indications of fouling or other more sudden disturbances or damage to the hydrophones.

The land unit was controlled by a computer through a LabVIEW generated program. Incoming data was stored on an external hard drive which disk capacity limits the maximum sample rate. The system was programmed to collect data with a 51.2 kHz frequency. The selected sample rate allows for analysis up to approximately 20 kHz, including radiated noise from most ships, sound from wind and waves, as well as biological sound from fish, seals, and some whales.

A system calibration was done in FOI's water tank in Kista, Sweden. Sequent, the system was evaluated in a marine environment at FOI's test site in the Stockholm archipelago before it was deployed at the study site.

At the end of the measurement period, the sea unit malfunctioned due to water leakage in one of the cable connectors resulting in a short circuit. An autonomous logger, a DSG LS1 from Loggerhead Instruments with an HTI hydrophone (−165 re 1 V/µPa), was deployed to finalize the measurements from April to June 2022.

The hydrophone station was deployed 2 km outside the island of Vinga where many ships pass within 1 km on their way to or from the Port of Gothenburg on the Swedish west coast (Fig. 2). The number of possible opportunistic passages of commercial ships with an active AIS transponder was about 30 ships every day within 5 km from the station. There are also smaller vessels without AIS transponders, which will add noise to the area. The seafloor in the area consists mainly of mud with patches of hard rock.

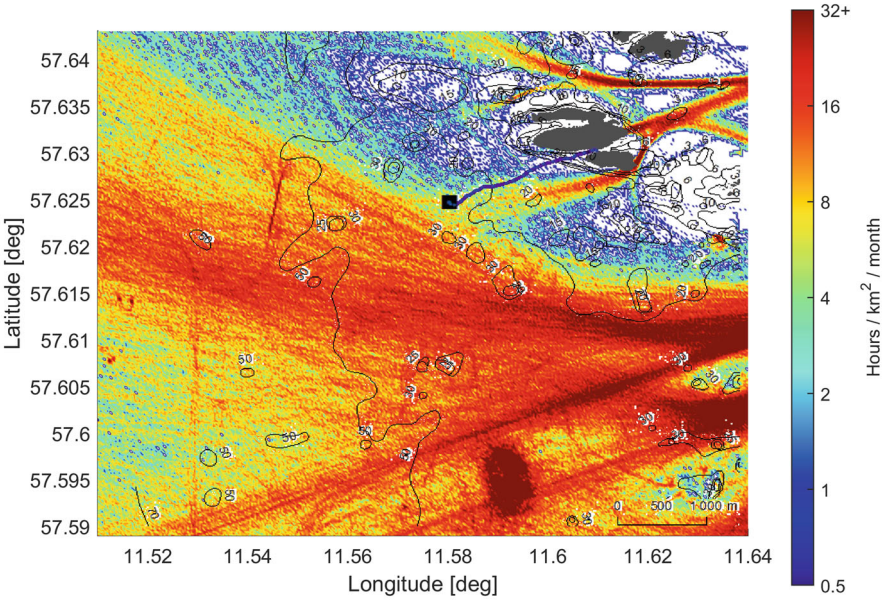



**Fig. 2** Monthly averaged ship density from November 2021 to June 2022. The position of the Vinga station is marked with a black square with a connecting dark line showing the fiber optic cable path to the island of Vinga

### Propagation Loss Measurements

Propagation loss (PL) measurements were conducted in March and June 2022 by towing an underwater acoustic transducer (Lubell Labs 1424HP) at a depth of approximately 5 m from the position of the hydrophone station and 4 km towards south west. Seven tones ranging from 160 Hz to 1060 Hz were continuously played throughout the measurement. As the hydrophone station was online during the March measurement, the decay of the signals could be monitored in real time and the measurements were conducted until the signals decreased to background levels. The received levels were compared to the received levels from a reference hydrophone located on the ship about 7 m from the transducer.

The sound speed profile was measured with a Swift Sound Velocity Profiler (SVP) from Valeport Instruments, both in March and June 2022.

## Ambient Noise, PL and SL Analysis

Opportunistic SL estimates of commercial ships were based on detecting individual ship passages close to the hydrophone station. Ship passages were detected using automatic identification system (AIS) data recorded locally, and suitable time periods in the recorded acoustical data were selected based on the closest approach distance as well as the distance to other ships. This section describes the way the recorded data was analyzed to achieve as accurate an estimation of the SL as possible.

### Ambient Noise

Hydrophone data from the station was processed using the regional standards developed by the EU project JOMOPANS (Ward et al. 2021) in 1/3 octave bands for frequencies ranging from 20 Hz to 20 kHz. Monthly and weekly statistics were thereafter calculated for three different frequency bands 126, 501, and 5012 Hz. This allows for a broader understanding of the soundscape of the study area. The ambient noise will affect the SL estimation and how this is handled can be found in section "Source Level Calculation Based on AIS Data," point 5.

### Propagation Loss

The PL was estimated for each frequency individually. A simple $A \cdot \log_{10} R$ formula was used for fitting the data, where $A$ is the PL coefficient to be estimated and $R$ is the total distance between the underwater acoustic transducer and the hydrophone station.

## AIS-Based Detection of Single Ship Passages at a Hydrophone Station

The subsequent steps were followed to extract time periods suitable for opportunistic SL estimates. This method has been used previously in Lalander et al. (2021).

1. Decode AIS data within a suitable area and time period.
2. For each ship with a unique identification number (MMSI) within the area, detect time periods where the distance from the ship to the hydrophone station is below 1000 m and the ship speed is larger than 1 m/s.
3. Keep only passages with time periods longer than 30 s.
4. Keep only time periods where all other moving ships (ship speed $>0.2$ m/s) are at least 1200 m away from the hydrophone station.
5. Keep only time periods where the closest point of approach (CPA) is below 650 m.
6. Keep only time periods where the longest duration between AIS messages is less than 20 s.

## Source Level Calculation Based on AIS Data

The described approach is based on the guidelines of the classification society Bureau Veritas (2018). It is recognized that the methods themselves and the meaning of some key terms (for example, source level, propagation loss, and radiated noise level) may differ from those adopted by recent ISO standards such as ISO 18405, and ISO 17208 parts 1 and 2. However, in the absence of an ISO standard on measuring ship noise in shallow water (ISO 17208-3 being at only Committee Draft stage) and for consistency with the approaches in the classification society guidance, the terminology and basic methodology used in this chapter and indicated in the text where adaptations have been made.

Data from the hydrophones were processed in several sequential steps to calculate the SL of a passing ship, based on the guidelines from Bureau Veritas (2018). In accordance with the guideline, the SPLs measured in different directions from a ship are treated equally and averaged to form the final SL from the ship. The signal processing steps are given in detail in Svedendahl et al. (2021) and are summarized below:

1. From AIS data, the GPS position of the ship were extracted and the distance, $R$, and angle, $\alpha$, between the hydrophones and the assumed acoustic center of the ship were calculated. The acoustic center was assumed to be $L/3$ in front of the stern of the ship and 2/3 of the draught, where $L$ is the length of the ship. This data was combined with the data from the hydrophone to synchronize acoustic data and ship position.
2. Each passage was divided into 19 time segments, all of which centered around a given set of $\alpha$. The set was given in steps of 5°, ranging from $\pm 45°$ to $\pm 135°$, where the sign depends on if the starboard or port side was facing the hydrophone. In order for the time segments to contain a certain amount of data, they correspond to 100 m long distances or the ship length, whichever is greater.

3. Subsequently, for each time segment, the power spectral density (PSD) was calculated using Welch method (1 s segments with 50% overlap and a Hanning window function).
4. For 1/3 octave band analysis, the narrow band spectrum was converted into 1/3 octave bands by taking the total narrow band energy in the respective band and normalizing with the 1/3 octave bandwidth. Thus, the bands do not correspond to the traditional 1/3 octave band metric, instead they correspond to the frequency averaged spectral density level in each band.
5. The calculated spectrum was compared to the background noise spectrum. If the signal-to-background ratio (SBR) was larger than 10 dB, the spectrum was used directly without any modifications. For a SBR of 3–10 dB, the background was subtracted from the signal spectrum. Lastly, if the SBR was less than 3 dB, this data was removed from further analysis.
6. The PL was corrected for in each time segment, by taking into account the distance from the hydrophones to the location of the acoustic center at the midpoint of each time segment. A simple formula was used in this case: PL = Alog(R). $A$ can be a frequency dependent variable or a constant value and was either estimated from PL measurements or fixed to 17, a value that often yields relatively good agreement with measurements in the study area, based on previous FOI experiences (Bergström et al. 2013; Andersson et al. 2015). Nevertheless, the estimation of PL is the largest source of uncertainty when calculating the SL. The environmental parameters are known to effect the estimation of a ships SL and are not included in this model. However, measurements or more refined modelling of the PL was not feasible.
7. If several hydrophone channels were used, the power spectrum densities calculated from each channel were averaged in all time segments.
8. After the PL correction, all source spectrum density levels were averaged to create a single spectrum for the run.

## Factors That Influence URN Estimation

The international standards and class notations highlight the importance of knowledge of the ambient noise, calibrated equipment, and the estimation of PL. If these are not known, the SL estimation can have large uncertainties. In this chapter, the projects efforts to reduce these uncertainties are described.

### Ambient Noise

The highest SPL over time was recorded in the 1/3-octave band 126 Hz (Fig. 3). The ambient noise showed a large variation in short timescales, minutes to hours. This is caused by the high number of ships passing within a few kilometers from of the station and appears as peaks in the time series (light gray lines in Fig. 3). Both the weekly median and the weekly arithmetic mean (AM), i.e., the mean value calculated in the linear scale and not in the decibel scale, are more stable over time

(Fig. 3). The AM shows more variation and is significantly higher than the monthly median value for all three frequency bands, suggesting that the sound sources that pass the station have a very high sound level, even if they only occur a small part of the time. In May to June, a reduction in SPL is noticed in the lower frequency bands, most likely caused by the change in the PL rather than changed ship density.

## In Situ Function Test

The transmitted calibration pulses were most of the time picked up by the hydrophones. The weekly averages of the calibration ping show a stable result above

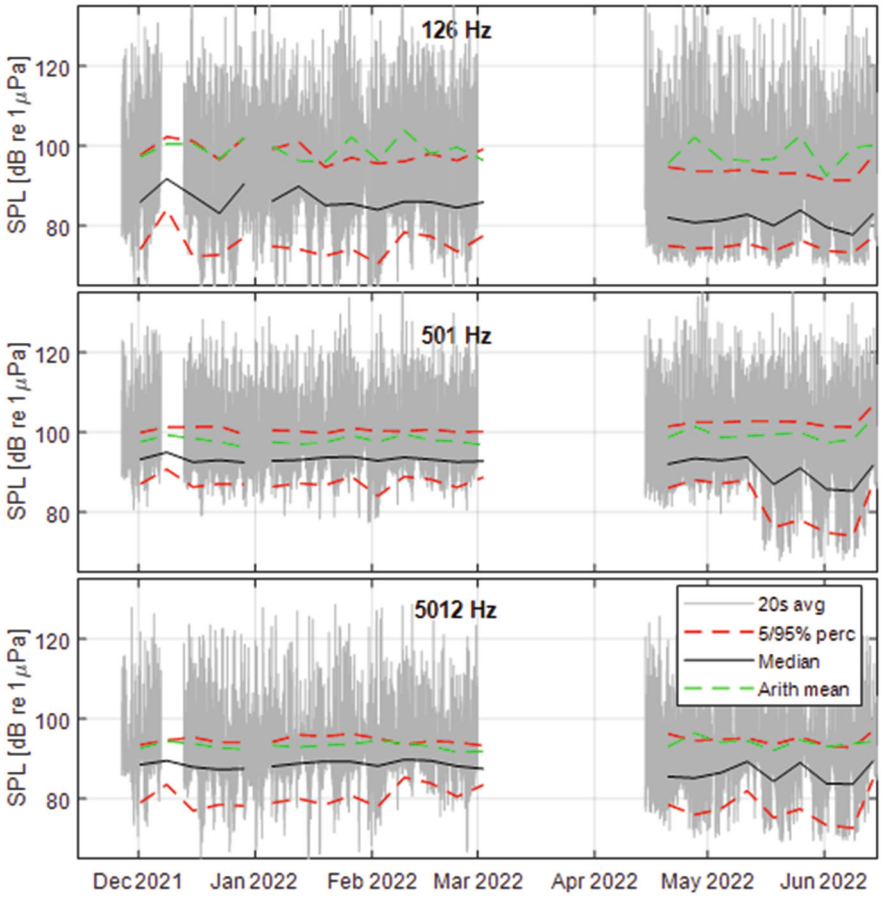



**Fig. 3** Sound pressure level (SPL) at Vinga 2021–2022 over time for different 1/3-octave bands. Light gray lines show 20-s averages. The median, 5%/95% percentile, and arithmetic mean are calculated per week. The statistics are calculated on weeks that have at least 3 days of audio recordings. Gaps in the data series was due to technical failure

6 kHz (Fig. 4), while below the measured pressure, variation is high. This is due to a low signal-to-noise ratio of the transmitted sound waves and variable ambient noise.

## Propagation Loss Estimations

By fitting the received levels in the hydrophone station to a simple PL = Alog(R) formula, and using the reference hydrophone located approximately 7 m from the underwater acoustic transducer to estimate the source levels, the PL coefficient, *A*, could be estimated for all frequencies. Figure 5a shows the results for a single frequency (660 Hz). The measurement data have clear fluctuations that likely stem from interference effects originating from multiple acoustic pathways from the source to the hydrophone station.

In March, the sound speed was lower close to the surface (around 1445 m/s) compared to the sea floor (around 1475 m/s), while the sound speed profile varied much less with depth in June (1485–1495 m/s). The sound speed profile likely affected the PL estimates, seen in Fig. 5b. The PL coefficient was found to vary less with frequency in March, compared to June.

## Estimated SL from Opportunistic Ship Passages

During the measurement period, 254 opportunistic commercial ship passages were recorded from 100 individual ships that passed the Vinga station under the right circumstances to pass the quality check for a SL calculation. The SL estimations were largely done automatically, but the data was also reviewed manually.

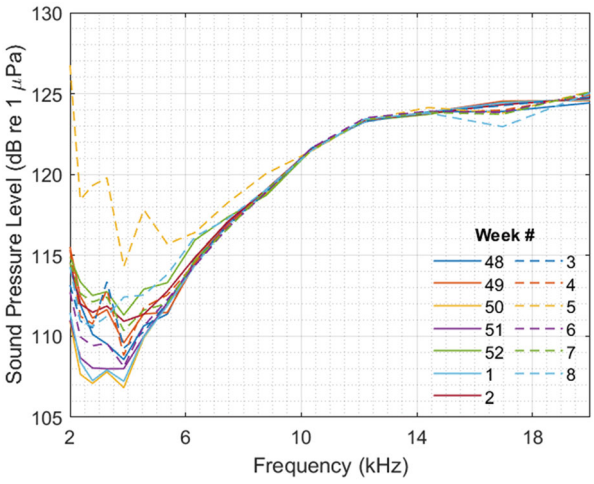



**Fig. 4** Weekly averages of received calibration pings in one of the two hydrophones on the sea unit in the frequency range of 2–20 kHz for November 29th (week nr. 48) 2021 to February 21st (week 8) 2022)

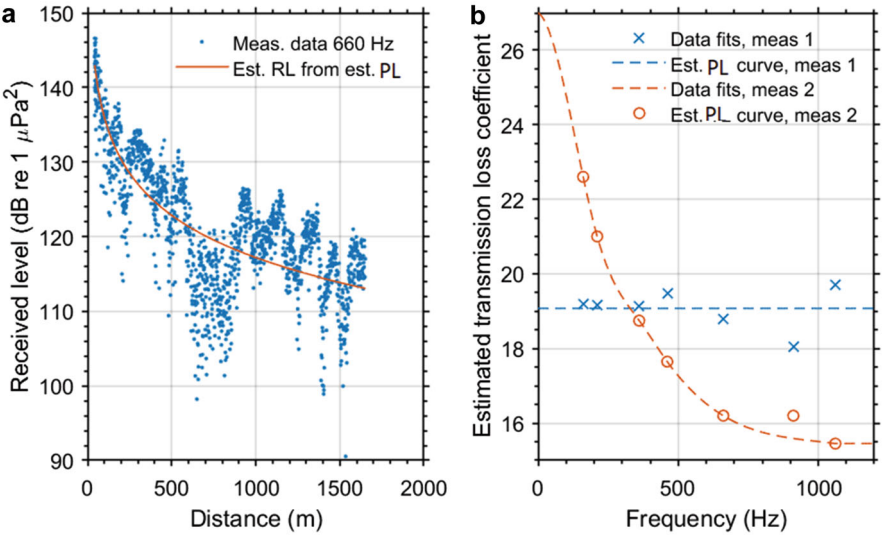



**Fig. 5** Results of the propagation loss (PL) measurements in March 2022 (measurement 1) and June 2022 (measurement 2). (**a**) Received level (RL) from measurement 1 at 660 Hz including a best-fit curve based on a PL-coefficient, the estimated source level, and the distance between the source and the hydrophone station. (**b**) Calculated PL-coefficient as a function of frequency and an estimated PL-coefficient curve based on these calculations for measurement 1 (blue) and 2 (red)

A 238 m long Ro-Ro ship (roll-on/roll-off ships are cargo ships designed to carry wheeled cargo) passed the station 23 times at distances ranging from 310 to 480 m during the measurement period. It mostly travelled at speeds of 17–23 knots, once passing at 10 knots. The PL estimation has a substantial effect on the calculated SL of the ship. For a passage of the ship at 20.4 knots, there is a 5 dB difference in the frequency range 10 Hz–25 kHz depending on the PL estimation used (Fig. 6, left). The uncertainty that the assumed simple model for sound propagation attenuation leads to is estimated to be a couple of decibels.

The measurements show that the ship makes the most noise below 100 Hz with two peaks around 18 and 40 Hz for the speed presented here. For speeds between 19 and 21 knots, the estimated SLs are very similar (Fig. 6, right). The calculated SL for this Ro-Ro ship was also compared with the SL for a similar ship and for the speed of 20 knots, which was produced according to the JOMOPANS-ECHO model for the AIS code vehicle carrier (MacGillivray and de Jong 2021) (Fig. 6, right). The model fits relatively well down to 20 Hz where the measured values increase.

As mentioned, about 100 individual ships SL were calculated from several different ship categories, taken the AIS data. Since the URN estimations are from almost an 8-month period, the PL will have changed during this period. In order to reach as good estimation as possible, the data of PL from the two measurement in March and June 2022 was used, with a cut-off date on the 14th of April. Specifically, the PL measurement from March was used for ship passages before the cut-off and

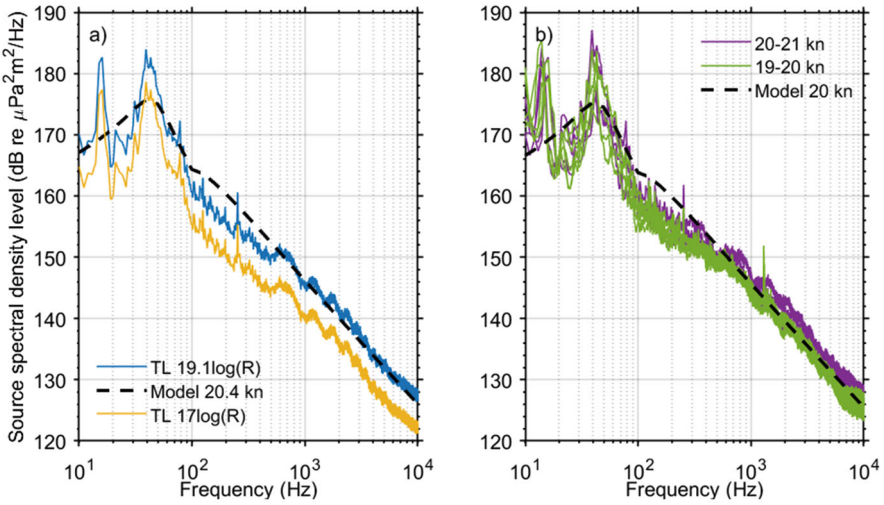



**Fig. 6** Estimated source level (SL) at 1 m from a 238 m long Ro-Ro vessel passing the Vinga station compared to the JOMOPANS-ECHO model for a vehicle carrier (black) (MacGillivray and de Jong, 2021) for a ship of equivalent category, size and speed. (**a**) SL at 20.4 knots with two different propagation loss (PL) – coefficient estimations. (**b**) Calculated SL at 1 m for various speeds compared the model

the one from June was used for ship passages after the cut-off. The cut-off date was chosen naturally due to the two different hydrophone stations used in the project and noticed change in the sound speed profile.

The ship category cargo was the category that made the most noise (ship length over 200 m, Fig. 7). These ships were also the fastest. The top right ship in Fig. 7, with a length of 238 m, is the same ship presented in Fig. 6. Smaller and slower ships make less noise. However, there were a number of passages of small fast pilot boats, which were some of the quietest vessels recorded (Pilot vessel, Fig. 7). The bandwidth of the broadband SPL presented in Fig. 7 was chosen to relate to the article by McKenna et al. (2012).

## Summary and Recommendation

The development and operation of a cabled real-time hydrophone station on the Swedish west coast were successful and generated high-quality data on both ambient noise and SL estimations of over 254 opportunistic commercial ship passages. However, several were from the same ship.

The in situ function test worked well throughout the measurement period and did not indicate any damage to the hydrophones. Damage to the hydrophone elements would indicate variation in the higher frequencies, which was not seen in the data. The high variation below 6 kHz is caused by low signal-to-noise ratio, which is a

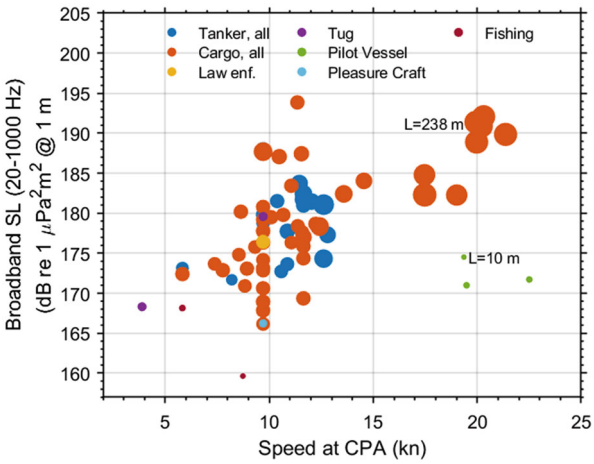



**Fig. 7** Estimated broadband (20–1000 Hz) sound pressure level (SPL) of the SL for unique ship passages near the Vinga station. The ship classifications were based on AIS data. The marker sizes indicate the length of the ship, with the longest (238 m) and shortest (10 m) ships indicated in the figure

consequence of the low signal output by the transducer and higher ambient noise. Having a transducer that can produce high enough output energy at lower frequencies could not be used in this station design due to the size and power requirement of such transducer. By having this function test running, unnecessary and costly retrievals of the station can be avoided.

The ambient noise level measured showed a high variation in the short time scale of minutes to hours. This is to be expected from a location close to the entrance of a busy port. In addition, the highest SPL was noticed in the 1/3-octave band 126 Hz compared to the other two bands (501 and 5012 Hz). This is to be excepted since ship noise has most of the energy below 500 Hz (McKenna et al. 2012; Karasalo et al. 2017; Wladichuk et al. 2019). More results from the ambient noise measurements can be found in Putland et al. (2022).

The PL measurements include relatively large uncertainties that are not completely covered by this chapter. Both the acoustic transducer and the reference hydrophone were located on the same ship, but on different rigs, thereby naturally varying in depth and relative distance between each other throughout the measurement. A more detailed knowledge of the time-varying distance and depth data, possibly by additional sensors, mounting the reference hydrophone and acoustic transducer to a single, more rigid, construction, or using a calibrated source could decrease some uncertainties in the propagation loss coefficient estimates. Moreover, the acoustic transducer transmit voltage response (TVR) and the electrical power available on ship were limited, which lead to a compromise regarding the number of tones that were projected into the water and the amplitude of each tone. The TVR of the acoustic transducer was significantly weaker for lower frequencies, for which the ambient noise is larger, which limited the feasible frequency range of the measurement. Otherwise, it would certainly have been of interest to measure frequencies below 160 Hz as well. In this case, a single tone per 1/3 octave band was chosen,

while also avoiding multiples of other tone frequencies and multiples of 50 Hz. Furthermore, an oversimplified model for the PL was used here, which did not take into account multipath or similar effects from the environment. If detailed knowledge of the measurement site is available, a more complex model could be used.

With the above comments in mind, it may be of interest to work towards a PL measurement standard, which could indicate not only how to conduct such measurements but also what frequency ranges that should be included. This is of particular importance for shallow water measurements. Some URN guidelines promote PL measurements but do not specify how to conduct them nor the frequency range(s) that need to be covered.

As a possible outlook, it would be interesting to study if an individual ship could be used to track changes in the PL. As the site was near a pilot station, and the same pilot vessels pass by the station relatively often, it may be possible to adjust for some environmental changes without the need to invest in the large time consuming effort of making PL measurements. By tracking the changes in the estimated SL of the ship, given certain ranges in spatial location, heading, and speed, it may be that PL measurements can be done less frequently.

The Ro-Ro ship with multiple passages showed similar SLs for 19–21 knots during the measurement, no matter the time of passage period. Disregarding any possible difference in cargo load and drive mode, the PL correction works well even though the closets point of approach is different for most passages. The estimated SL fits well with the JOMOPANS-ECHO SL model for vehicle carrier (MacGillivray and de Jong 2021) for a ship with the same length and at the same speed. Local PL effects are most likely the cause of the discrepancy between the model and the SL estimation data below 100 Hz.

The calculated broadband levels are in the same range as previously published (McKenna et al. 2012; Karasalo et al. 2017; Wladichuk et al. 2019). The AIS category cargo, which is a broad category of ships, was the most common, the largest and the most noisy ship category. The second most common and noisy category was tankers, which in general travel a bit slower. A more in-depth analysis of each subcategory of cargo can be done but was out of the scope in this chapter.

When choosing a measurement position for opportunistic ship passages, there is a trade-off between having enough passages to get good data and too many passages that could disturb the SL measurement of each individual ship. The position chosen proved to be a good compromise.

In conclusion, having a hydrophone station at the entrance of a major port, as described in this chapter and in Hannay et al. (2016), the SL of specific ships can be obtained in a cost-effective manner. The radiated noise information are useful for shipping companies and managers if noise levels for individual ships are regulated in the future, and silent class notification becomes more common. Although the economic gains are low for this particular benefit at the moment, silent ships has been proven to increase fuel efficiency due to good maintenance and/or technical solutions (Gassmann et al. 2017), which in turn results in lower running cost.

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# Measurements of Underwater Radiated Noise from Mobile Offshore Drilling Units

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Melanie E. Austin, S. Bruce Martin, and Craig R. McPherson

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**Abstract**

Mobile offshore drilling units (MODUs) are used to drill oil and gas wells in the ocean. For deeper water operations, there are two common forms of MODUs: semisubmersible rigs and drillships. Over the past several years, JASCO Applied Sciences measured MODU drilling operations by semisubmersibles and drillships, including platforms that were moored with anchors or that held position using dynamic positioning (DP) thrusters. The MODUs were accompanied by support vessels on standby, with additional vessels conducting resupply operations during the drilling campaigns. The measurements were performed in response to regulators requesting verification of the vessel source levels and the distances where the sounds were expected to either injure or disturb marine life.

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Overall, drillships had higher source levels than semisubmersibles. Moored platform sound levels decreased with drilling depth and hole diameter. The signatures of moored platforms featured tones from different types of rotating machinery. When platforms or nearby support vessels were holding station using DP, the tonal signatures were obscured by DP thruster sounds. All measurements of underwater radiated noise from platforms using DP contained sounds from ultrashort baseline (USBL) beacons at 25–27 kHz.

**Keywords**

Offshore drilling noise · Drilling noise

## Introduction

Nearly 45% of global gas reserves and 15% of global oil supplies are found in the offshore environment (IEA 2018). To access these resources, mobile offshore drilling units (MODUs) are used to drill gas and oil wells in the seafloor. The most common types of MODUs for offshore operations are semisubmersible rigs and drillships. Both can hold position over offshore wells using dynamic positioning (DP) thrusters or can be moored using anchors. Characterizing the underwater sounds emitted by these MODUs is important for understanding the potential underwater noise impacts that these offshore drilling activities can have on marine fauna, including the distances over which these sounds have the potential to injure (via permanent or temporary hearing threshold shifts) or disturb marine fauna. To this end and in fulfillment of local regulatory requirements, JASCO Applied Sciences (JASCO) has measured underwater sounds of MODU operations for several semisubmersibles and drillships (Table 1) at locations around the globe. This chapter provides a high-level overview of the results of these measurements, with respect to the nature and levels of underwater noise associated with MODU activities.

This chapter considers underwater noise measurements of the semisubmersible *West Aquarius* while on DP as well as the moored semisubmersibles *Ocean Onyx* and *Polar Pioneer*. Underwater noise measurements of four drillships are also considered: *Stena Forth* and *Stena IceMax* while on DP as well as the moored drillships *Noble Discoverer* and *Kulluk* (Austin et al. 2018).

Measurements were recorded at locations in the Alaskan Chukchi and Beaufort Seas, in the Atlantic Ocean off the east coast of Canada, and in the Otway Basin off Southeastern Australia (Table 1). These locations span different environmental regimes for water depths and seafloor types. In the Chukchi and Beaufort Seas, the water depths are shallow (35–45 m) and the seabed is relatively flat with a sandy silt bottom. The sites on the Scotian slope of the Atlantic are deep (2000–3000 m) with a clay seabed. At the site in the Otway Basin, the water depth is 70 m with a relatively flat seabed consisting of well-cemented carbonate caprock, overlying semicemented carbonate rock.

**Table 1** Measured mobile offshore drilling units

| MODU type | | Vessel | Operator | Location | Length (m) | Width (m) | Draft (m) |
|---|---|---|---|---|---|---|---|
| Semisubmersible | Moored | *Ocean Onyx* | Diamond Offshore Drilling | Otway Basin | 106 | 105 | 9 |
| | | *Polar Pioneer* | Transocean Ltd | Alaska | 85 | 82 | 23 |
| | DP | *West Aquarius* | Seadrill Limited | Scotian Slope | 118 | 97 | 23 |
| Drillship | DP | *Stena Forth* | Stena Drilling | Scotian Slope | 228 | 42 | 12 |
| | | *Stena IceMax* | Stena Drilling | Scotian Slope | 228 | 42 | 12 |
| | Moored | *Noble Discoverer* | Noble Drilling | Alaska | 157 | 26 | 8.2 |
| | | *Kulluk* | Shell | Alaska | 81 | 81 | 10 |

## Background

Drillships are large vessels equipped with a drilling derrick and drilling equipment such as pumps, mixers, shakers, pipe racking, and pipe movement equipment. A drill-string generally passes through a moonpool in the ship's hull. In addition to the drilling equipment, as with other large ships, the list of sound-generating equipment also includes the vessel's engines, generators, propulsion systems, and hydraulics for cranes and winches. Noise is transmitted to the ship's hull through mechanical coupling of the machinery with the ship's structure, as well as through coupling of airborne noise. All the mechanical equipment and machinery is housed on or below the ship decks and mechanical vibrations are well coupled to the ship's hull. The vibrations are then transmitted from the hull into the surrounding water and radiated throughout the water column.

Drillships float and are self-propelled so they have the advantage of ready mobility. They are well suited to deepwater drill sites where mooring can be a challenge. Because they are mobile platforms that float on the sea surface, they require advanced dynamic positioning capabilities or sophisticated moorings to maintain their positions in the presence of waves and currents.

Semisubmersible MODUs consist of a main platform above columnar legs attached to pontoons. The pontoons float at the water surface while the unit is towed to a well site. Once at the drilling location, the pontoons are flooded to ballast the structure and lower its center of gravity. The main platform remains elevated above the sea surface and away from waves, which provides increased operational stability compared to a drillship. Pathways for sound coupling for a semisubmersible include vibration transmissions through the drill-string and from the columns and pontoons below the water.

Each MODU employs similar types of equipment and machinery, so their acoustic footprints differ principally in the mechanisms by which the sound couples from the structures to the surrounding water. It is generally assumed that drillships emit higher underwater sound levels than semisubmersibles, given the greater surface area that is coupled with the water.

The drilling programs JASCO measured took between 1 and 4 months per well. The duration varied based on the well type (exploration or development), the depth, and operational complexity. MODUs are typically accompanied by support vessels on standby. These vessels include supply vessels, fuel tankers, anchor handlers, spill response vessels, ice management vessels, etc. In remote locations where there are no nearby ports, these vessels remain on standby for extended periods, typically within a few kilometres of the MODU. They use operationally relevant power levels, ranging from idle to underway, and may use DP to hold station. During the measured programs, stationary vessels within 500 m of the MODUs were always under DP for navigational safety. DP thrusters, similar to ship propellers, cause cavitation noise, i.e., the noise created through vibration and creation of low-pressure points and bubbles, and their signatures differ for the various thruster types and power levels.

## Results

### Otway Basin, Australia

In the Otway Basin, JASCO measured underwater sound levels of the moored semisubmersible *Ocean Onyx*, while it was undertaking an exploration drilling program for Beach Energy Limited. Measurements were collected to validate JASCO's modeling predictions used in the Beach Energy Otway Environmental Plans for exploration and development drilling activities. The model validation is not discussed in this chapter.

Sound levels were measured at 0.3, 1, 5, and 25 km ranges (Fig. 1). Continuous recordings at 24-bit resolution and 64 kHz sample rate were collected using seafloor-mounted Autonomous Multichannel Acoustic Recorders (AMARs; JASCO). The AMAR at the furthest recording distance was configured with three hydrophones such that the originating direction of the received sounds could be computed; a single hydrophone was used at the other recording locations. The most prominent feature in Fig. 1 is the decrease in sound level with distance from the drill rig. Across all frequencies, recordings at the two nearest stations indicate that the mean and all data above the median exceed the upper limit of expected ambient sound levels, based on the Wenz curve, owing to noise from the MODU. At the 5 and 25 km stations, the sound levels are much reduced, with sound from the MODU not notable at the 25 km station. The soundscapes at the farthest two stations exhibit frequency distributions characteristic of noise from vessel traffic originating from commercial shipping. Operational details for the variable activities of the MODU and support vessels were obtained from the MODU operator's activity logs. The recordings were

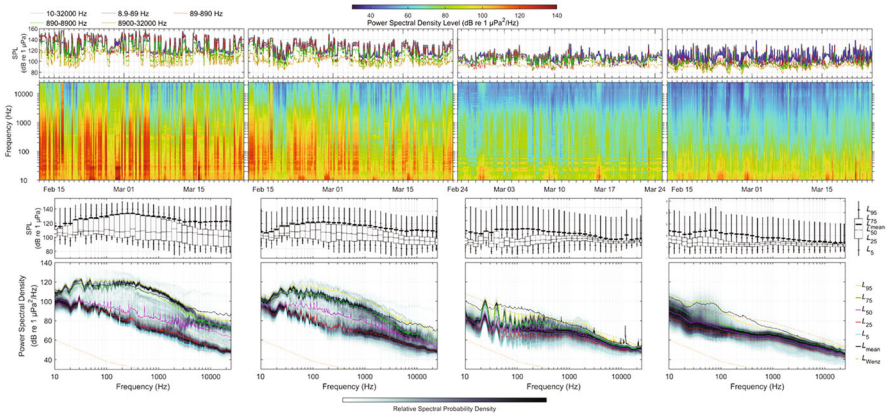



**Fig. 1** (Top row) In-band sound pressure level (SPL) and spectrogram (or long-term spectral average; LTSA) of underwater sound, (bottom row) percentiles and mean of decidecade SPL and percentiles and probability density (grayscale) of 1-min power spectral density levels, by station, compared to the Wenz curve limits (colored lines) of prevailing noise (Wenz 1962). Stations (left to right) were located 0.3, 1, 5, and 25 km from the semisubmersible

also influenced by weather conditions and merchant shipping unrelated to the MODU program.

The operational activity logs were referenced to identify periods when specific drilling activities occurred. The spectrogram on the left in Fig. 2 was generated from acoustic data collected at 300 m range during drilling. Several distinct tones are present in the data, and broadband sound levels are around 120 dB. The spectrogram on the right in Fig. 2 is from a period during drilling when there was a resupply vessel alongside. Vessels used DP whenever they were within 500 m of the *Ocean Onyx*, and vessels in DP mode were the dominant source of noise in this example. Here the received broadband SPL is much higher, reaching nearly 160 dB at the loudest time, and the tones evident in the spectrogram on the left are obscured by the more broadband noise from the DP thrusters on the support vessel.

It is worth noting that to characterize sounds from an MODU operating in isolation, it is crucial to be able to select time periods not contaminated by noise from ancillary support vessels. AIS records or navigational logs for the vessels can be cross-referenced with the time stamped acoustic recordings for this purpose. Acoustic recordings should also be collected over long durations (days to months) because there are often only short periods of time when no vessels are alongside the MODU. Activities associated with mooring operations, cementing, installing the blowout preventer (BOP), or running casing were not considered. Twenty time periods for drilling were identified, and seventy 10-minute periods out of the 6-week drilling operation were considered suitable for characterizing noise from the *Ocean Onyx* while drilling. Source level measurements were matched with drill logs and the source levels were grouped by the drilling activities.

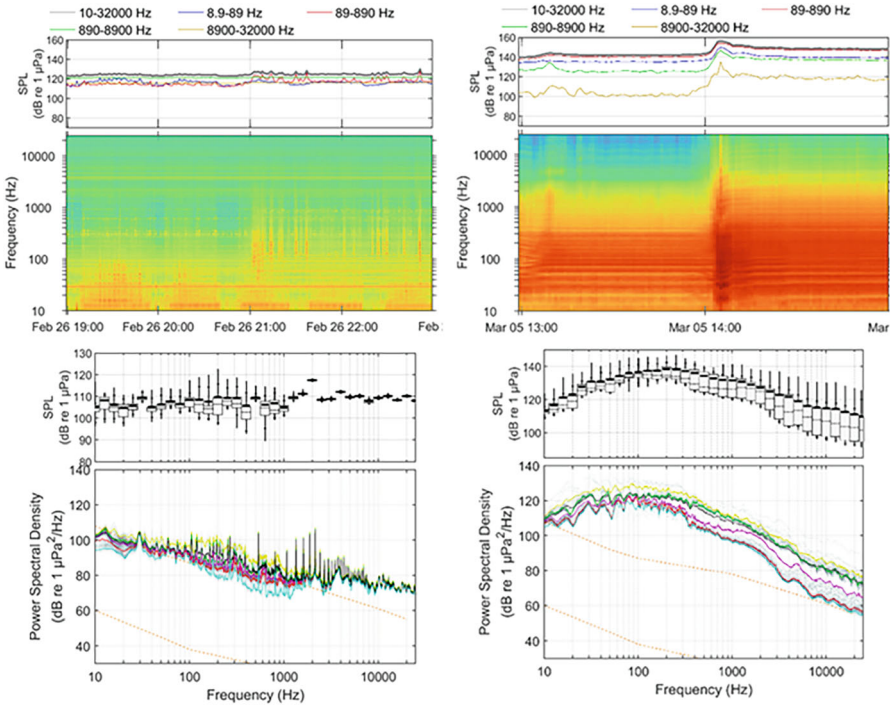



**Fig. 2** (Top row) In-band sound pressure level (SPL) and spectrogram (or long-term spectral average; LTSA) of underwater sound, (bottom row) percentiles and mean of decidecade SPL and percentiles and probability density (grayscale) of 1-min power spectral density levels, by station, for a (left) drilling period on 26 Feb 2021 with no support vessels nearby and (right) drilling period on 5 Mar 2021 which included resupply and support vessels, compared to the Wenz curve limits (colored lines) of prevailing noise (Wenz 1962)

Detailed examination of periods during drilling with no vessel noise indicated the presence of tones associated with machinery with variable rotation rates, diesel generators and pumps with variable loads, and a consistent high-pressure pump. The monopole source level (MSL; dB re 1 µPa m) for each drilling period sample was calculated using JASCO's ShipSound software. The mean MSL for the *Ocean Onyx* was determined for three drilling depth ranges (Fig. 3). The mean MSL was 175.2, 169.3, and 162.7 dB re 1 µPa while drilling at depths between 95–172, 365–621, and up to 1851 m, where the hole diameter was 26″, 17.5″, and 12.25″, respectively.

## Alaskan Beaufort and Chukchi Seas, USA

Measurements of drillships in the Alaskan Chukchi and Beaufort Seas (obtained for Shell Offshore Inc.) were more readily obtained without contamination from sounds of ancillary support vessels. For these drilling programs, support vessels were

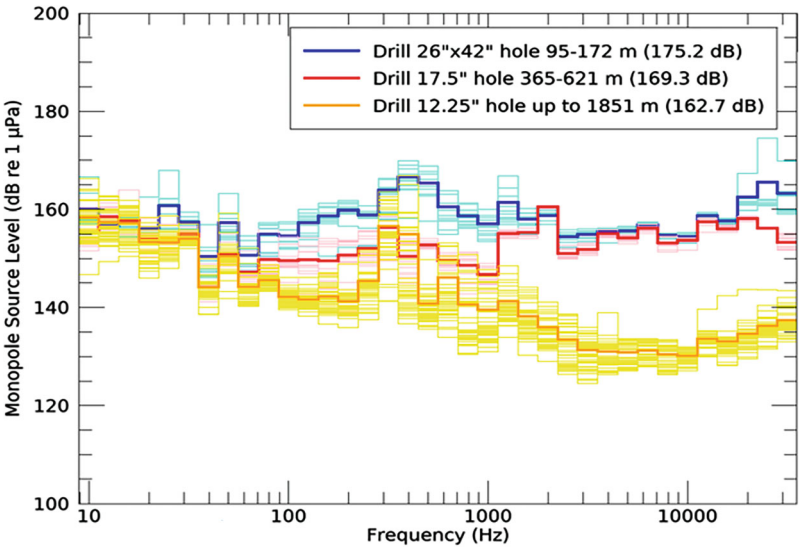



**Fig. 3** Monopole source level (MSL; dB re 1 µPa m) and spectra for the moored semisubmersible *Ocean Onyx* drilling in the Otway Basin, Australia, from ShipSound processing of data measured at 300 m range, averaged over measurements for three drilling depths

required to be farther than 25 nmi from the drillship unless the vessels were actively engaged in an activity that required them to be closer. This provided long recording periods of drilling noise without vessels present. The image on the left in Fig. 4 shows drilling noise measured while three support vessels were within 5 km of *Discoverer* while it drilled a pilot hole at the top of a well in the Chukchi Sea. These data were measured 1 km from the drillship with a seafloor-mounted AMAR recording at a 64 kHz sample rate. The period with drilling is not easily discernible from the vessel noise in the panel on the left. In this case, an ice management vessel and a spill response vessel were both between 0.5 and 1 km from the drillship, and an anchor handler was within 4 km. The spectrogram on the right was recorded when there were no vessels within 10 km of the drillship. Low-amplitude tonal noise is evident in these data, caused by the machinery on the drillship when it is not actively drilling. A period while the drillship drilled a 26″ hole is annotated on this figure.

The left panel of Fig. 5 shows the power spectral density levels for sounds recorded during the period of drilling the 26″ hole as measured at 1, 2, 4, and 8 km from the drillship. Many discrete tones are evident. The right panel in Fig. 5 is a similar plot, for the drillship *Kulluk* drilling in the Beaufort Sea.

There was one activity during these drilling programs that was notably louder, and that was the drilling of the mudline cellar (MLC); a large pit excavated from the seafloor, 12 m deep by 6 m wide, to house the blowout preventer at the top of the well. This was dug using a 6 m diameter drill bit, consisting of multiple rotating disks angled to displace the excavated sediment. During this activity, the sound levels jumped up by approximately 20 dB, as can be seen in the spectrogram and

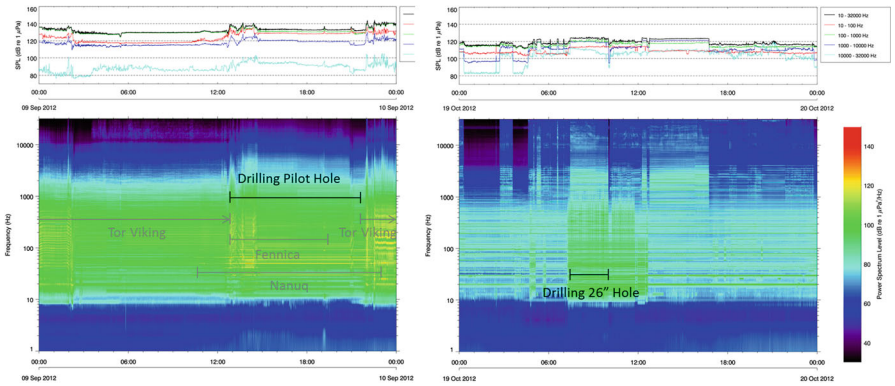



**Fig. 4** In-band sound pressure level (SPL) and spectrogram (or long-term spectral average; LTSA) of underwater sound measured while the *Discoverer* was (left) drilling a top hole with three support vessels nearby and (right) drilling a 26″ hole with no vessels within 10 km

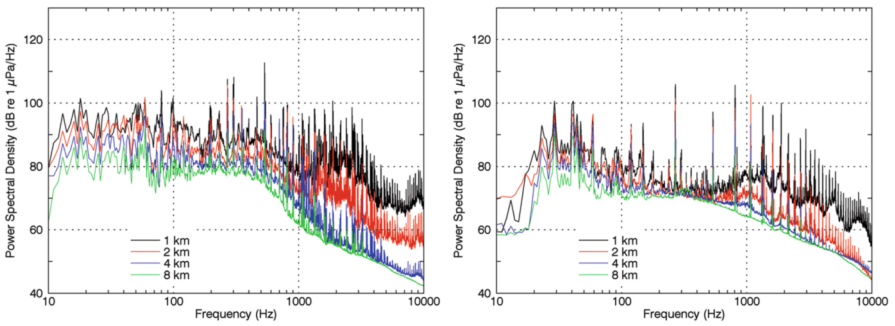



**Fig. 5** Power spectral density for (left) the *Discoverer* drilling a 26″ hole in the Chukchi Sea and (right) the *Kulluk* drilling in the Beaufort Sea as received at seafloor acoustic recorders at four ranges

band level plots in Fig. 6. JASCO measured this activity in the Chukchi Sea in 2 years, once the activity was carried out from a drillship and once from the semisubmersible platform the *Polar Pioneer*. Figure 7 shows a plot of the power spectral density for drilling and for mudline cellar excavating in these 2 years; the left panel is the drillship and the right is the semisubmersible. In both cases, the MLC excavation (which is the upper black curve) is not only louder than the drilling, but in both cases many of the tones in the spectrum were obscured and the noise was more broadband in nature. Source levels were estimated by back-propagating received levels measured at a range of 1 km. Mean broadband source levels were calculated as 168.6 dB re 1 µPa·m for the *Kulluk* drilling in the Beaufort Sea, 174.9 dB re 1 µPa m for the drillship *Discoverer* drilling in the Chukchi Sea, and 170.1 dB re 1 µPa m for the semisubmersible *Polar Pioneer* drilling in the Chukchi Sea. Source level

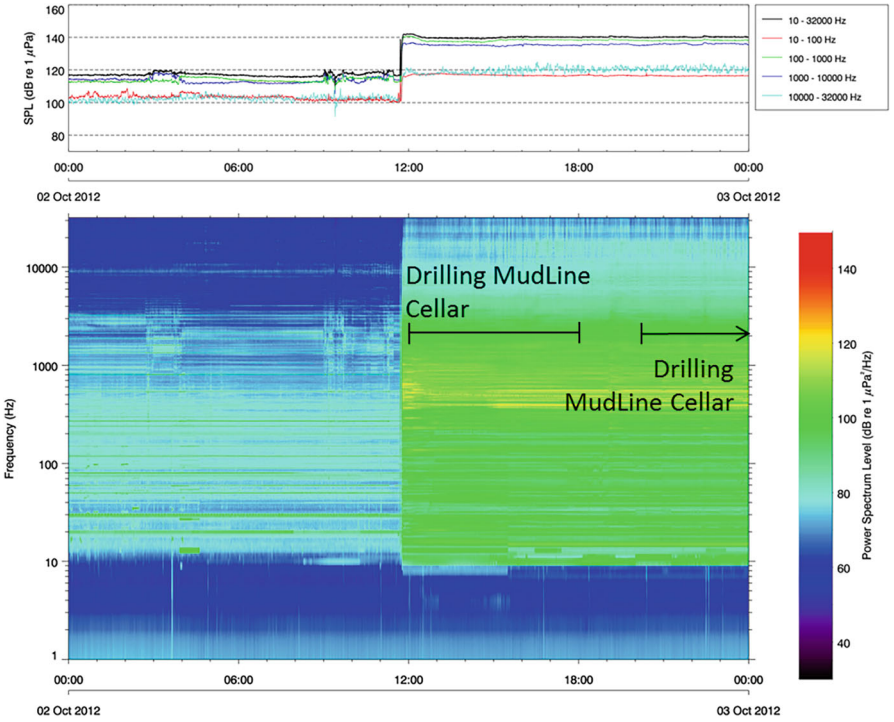



**Fig. 6** In-band sound pressure level (SPL) and spectrogram (or long-term spectral average; LTSA) of underwater sound measured while the drillship *Discoverer* was drilling a mudline cellar with no vessels within 10 km

estimates for MLC excavation were more consistent among the sources: 191.8, 193.0, and 193.3 dB re 1 µPa for the *Discoverer*, *Kulluk*, and *Polar Pioneer*, respectively. These measurements are all detailed in Austin et al. (2018).

## Scotian Slope, Canada

JASCO measured two drillships (the *Stena IceMAX* at the Shelburne Basin Venture Exploration Project for Shell Canada Limited, and the *Stena Forth* at the Pelles A-71 Exploration Well in the Flemish Pass for the Chinese National Offshore Oil Corporation [Kowarski et al. 2021]) and one semisubmersible (the *West Aquarius* at an exploratory drilling well in the Flemish Pass for Exxon Mobil Canada [Delarue et al. 2021]) and during an exploratory drilling campaign off the coast of Nova Scotia for BP Canada Energy Group ULC (Martin et al. 2019) at four drilling locations off the Canadian east coast. All measurements were collected using AMARs, with (project-dependent) samples rates of 16, 32, 64, or 128 kHz. All these MODUs were operating on DP during the drilling programs.

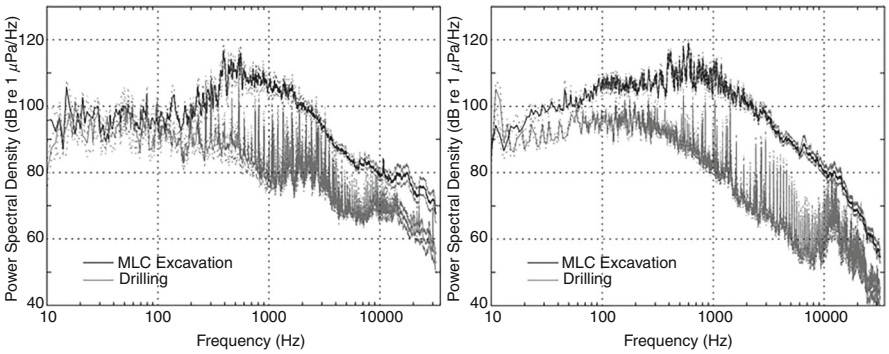



**Fig. 7** Power spectral density for drilling and mudline cellar (MLC) excavation in the Chukchi Sea by (left) the drillship *Discoverer* and (right) the semisubmersible *Polar Pioneer*, as received at seafloor acoustic recorders at 1 km range

Figure 8 is an example of the data collected at (left) 2 km and (right) 20 km during drilling by the semisubmersible *West Aquarius* on DP in the Scotia Basin. There are fewer discrete tones discernible in the data then was observed for the moored semisubmersibles measured in Alaska and Australia. In this case, the dominant sources of noise were the DP thrusters constantly engaged to maintain position over the well, nearby vessels, and wind. There was no obvious trend in overall sound levels relative to the operational drilling activities. This was also the case for data measured at 2 km range from this semisubmersible while drilling in the Flemish Pass (Fig. 9).

Similarly, measurements of drillships on DP on the Scotian Slope were dominated by noise from the platform's DP thrusters. Fig. 10 is an example of this from measurements made at 2 km range while the *Stena IceMAX* was drilling in the Shelburne Basin and using DP. Fifty-two hours of data were collected during this measurement program, and approximately 33 h of drilling activity occurred in this period. Throughout all 52 h, the *IceMAX* and four support vessels were operating in dynamic positioning (DP) mode. The full period spectrograms and band-level data in Fig. 10 do not show changes in measured sound levels that correlate with changes in the drilling state. This suggests that DP system noise and mechanical noise from the *IceMAX* and OSVs were larger contributors to the sound field than the actual sound generated from drilling equipment. Over the entire measurement period, the 100–1000 Hz band had the highest SPL on both static recorders. This band is where most sound from large vessels occurs. The spectrogram shows lines varying in frequency, forming many "V" shapes. This effect normally indicates vessels transiting past a recorder, but a vessel using DP in changing current or wind could also generate this pattern as the power output of the thruster increases and decreases. In the case of the fixed-pitch propellors on all thrusters on the platforms measured, this varied in association with the variation in revolutions per minute (RPM) of the thruster.

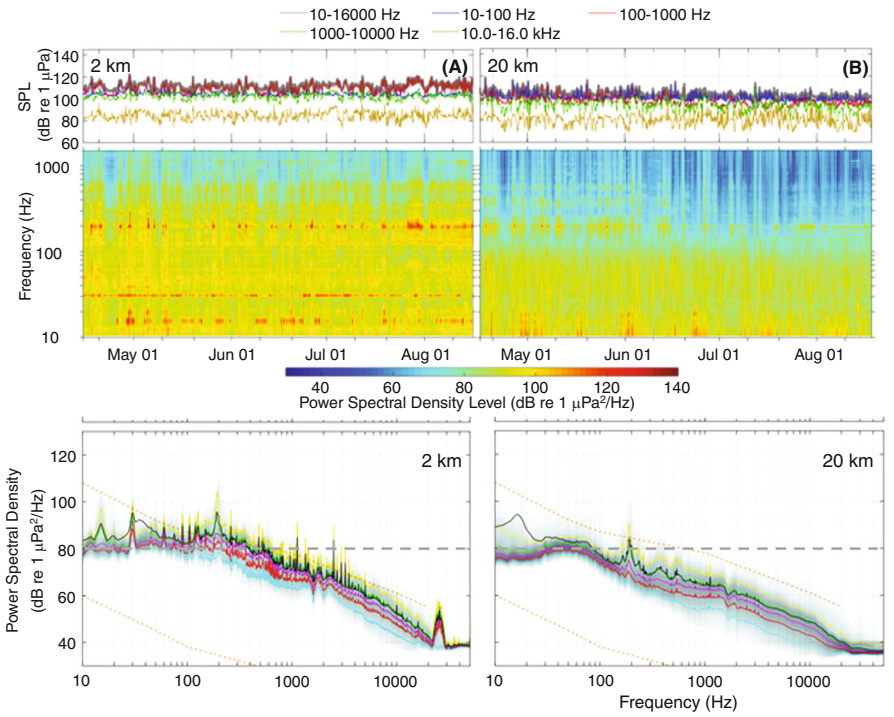



**Fig. 8** (Top row) In-band sound pressure level (SPL) and spectrogram (or long-term spectral average; LTSA) of underwater sound, (bottom row) percentiles and probability density (grayscale) of 1-min power spectral density levels recorded at (left) 2 km and (right) 20 km from a semisubmersible rig drilling while under dynamic positioning in the Scotia Basin, offshore Eastern Canada

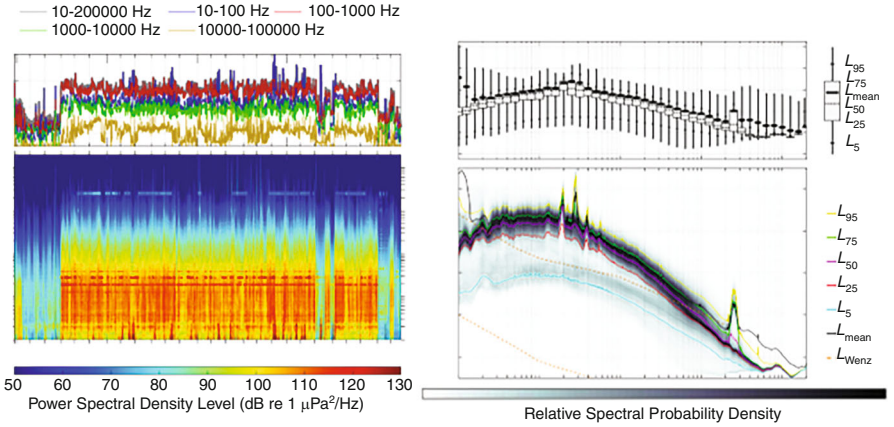



**Fig. 9** (left) In-band sound pressure level (SPL) and spectrogram (or long-term spectral average; LTSA) of underwater sound, (right) percentiles and probability density (grayscale) of 1-min power spectral density levels recorded at 2 km from a semisubmersible rig drilling while under dynamic positioning in the Flemish Pass, offshore Eastern Canada

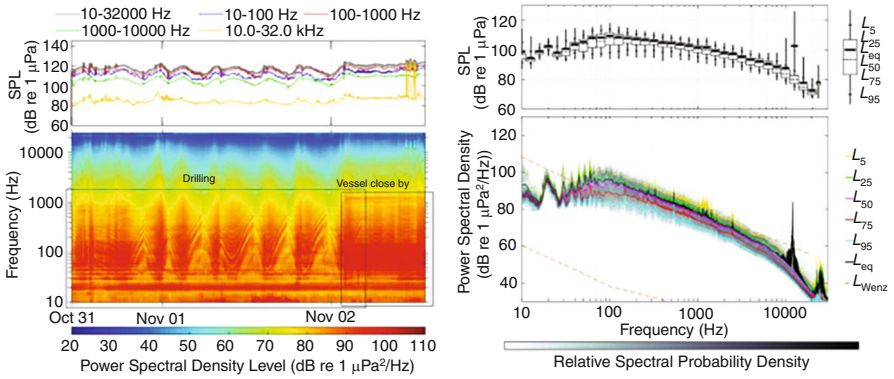



**Fig. 10** (left) In-band sound pressure level (SPL) and spectrogram (or long-term spectral average; LTSA) of underwater sound, (right) percentiles and probability density (grayscale) of 1-min power spectral density levels recorded at 2 km from a semisubmersible rig drilling while under dynamic positioning in Shelburne Basin, offshore Eastern Canada

All recordings of drillships and semisubmersibles consistently contained a notable high-frequency tone around 25 kHz. This was determined to be associated with the high-precision acoustic positioning system associated with the DP system and an echosounder used to monitor the drillship distance from the seafloor. Estimated mean MSLs for each MODU measured while using DP varied between 183 and 187 dB re 1 µPa m. It is worth noting that faint tones were recorded by JASCO in the Orphan Basin offshore Newfoundland, Canada, at times corresponding to when the semisubmersible *West Hercules* was operating while on DP at a 209 km range from the recorder. This suggests that in some environments, these DP sounds can extend over long distances.

## Summary

The measurement campaigns that characterized the MODU drilling operations presented here were often logistically challenging, with significant effort expended for, in most cases, relatively limited periods of usable data once attending vessels were removed from the datasets. The loudest part of MODU operations, considering the platforms themselves and the attending vessels, is the operation of the dynamic positioning thrusters. The summary of the mean MSLs in Table 2 demonstrates that the quietest estimate for the mean MSL for a DP MODU is approximately 8 dB greater than the loudest mean measured MSL for moored MODUs. If there is a regulatory or best practice requirement to minimize underwater noise emissions from a drilling operation, the first consideration should be whether the MODU can be moored rather than using DP. Similar logic applies to support vessels, which can minimize underwater noise emissions by maintaining slow transit speeds rather than

**Table 2** Summary of mean monopole source levels for the measured mobile offshore drilling units (MODUs)

| MODU type | Vessel | Location | Mean monopole source level (dB re 1 µPa m) |
|---|---|---|---|
| Moored semisubmersible | *Ocean Onyx* | Otway Basin | 162.7–175.2 |
| | *Polar Pioneer* | Alaska | 170.1 |
| Semisubmersible on DP | *West Aquarius* | Scotian Slope | 183–187 |
| | *Stena Forth* | | |
| Drillship on DP | *Stena IceMax* | | |
| Moored drillship | *Noble Discoverer* | Alaska | 174.9 |
| | *Kulluk* | | 168.6 |

holding station under DP. The reported measurements likely do not represent the full range of potential emissions by DP vessels and thrusters, and certain thrusters and operational conditions may result in both lower and higher MSLs. For context, JASCO has characterized Anchor Handler Tug Support vessels, which were supporting the *Ocean Onyx*, as having mean MSLs of 193.9 dB re 1 µPa (McPherson et al. 2021). Additionally, all measurements of noise from platforms using DP contained sounds from ultrashort baseline (USBL) beacons at 25–27 kHz. The possible impacts of the beacons on the presence of odontocetes near the MODUs is considered in a separate chapter.

The measurement studies in this chapter did not consider the emissions during other stages of a drilling operation, such as mooring, cementing, installing the blowout preventer (BOP), or running casing. While the underwater noise emissions from the platforms using DP varied depending on the speed of the fixed-pitch thrusters, no engine data were available to complete an analysis of this relationship. The speed, which depends on operational requirements and weather conditions, predominantly masks any tonal signatures of rotating machinery. For the moored platforms, sound levels decreased with drilling depth (and hole diameter), and the sound levels were primarily determined by the tones from different types of rotating machinery.

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# Management of Noise in a Busy Harbour Porpoise Protected Area

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Sónia Mendes, Abi Goulding, and Lisa Mogensen

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**Abstract**

The Southern North Sea Special Area of Conservation (SAC) is the largest of a suite of sites designated to protect harbor porpoise (*Phocoena phocoena*) and its habitats in UK waters. It was identified as having high persistent densities of porpoise despite the area being highly industrialized and therefore noisy. Activities in and around the site include geophysical surveys, clearance of unexploded ordnance, and construction activities involving pile driving.

Given the potential conflict between maintaining a favorable habitat status and continuing noisy activities, in 2020 the UK nature conservation agencies published guidance on how to assess and manage operations that could cause significant noise disturbance in porpoise SACs. The approach provides an equitable way of managing noise across industries, by defining area/time noise limits. The limits are used as references in impact assessments when planning activities, and measures such as noise abatement and scheduling are agreed between industry and regulators to ensure limits are not exceeded. At the core of this approach is the UK Marine Noise Registry, a database recording when and where impulsive noise occurs, for both past and future activities. This not only helps with planning but also allows for retrospective monitoring and has already provided useful insights for management.

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**Keywords**

Noise management · Protected area · Noise registry · Seismic · Piling · Porpoise · Marine mammal

## Introduction

A suite of Special Areas of Conservation (SACs) for harbor porpoise were designated in the United Kingdom in 2019, due to the persistent high densities of the species found in those areas. The SACs have been identified based on elevated densities of harbor porpoise in either summer (April–September) or winter (October–March) (Heinänen and Skov 2015), or year-round. The sites are assumed to provide good foraging opportunities, and it is therefore important that porpoise can access and utilize the habitats within the SACs.

Some of these areas were identified despite a certain level of local disturbance by underwater noise. This is particularly so for the Southern North Sea (SNS) SAC, the largest of the SACs, spanning 36,951 $km^2$, having both summer and winter areas and a small year-round area. The SNS SAC is a particularly industrialized and therefore noisy site. There is considerable oil and gas exploration and associated seismic surveys. There is also a need for clearing of unexploded ordnance (UXO) to make the seabed safe for placing infrastructure. Also, the installation of offshore wind and cables has continued to increase in the area (JNCC 2022). There is an emerging conflict between trying to maintain a favorable habitat for the species and persistent and increasing industrial activities. The site is protected for a species sensitive to noise, and therefore the bar must be set higher in these areas.

In 2020, after several years of stakeholder discussion and trial, the UK nature conservation agencies published guidance (JNCC 2020) on how to assess and manage operations that could cause significant noise disturbance in porpoise SACs. The guidance applies to all regulated activities that result in impulsive noise and therefore have the potential to disturb or injure harbor porpoise. It introduces spatial and temporal limits to disturbance from noise within the sites to protect those important habitats for porpoise: (1) a daily limit, where no more than 20% of the site (or seasonal area) is subjected to disturbance in any given day, and (2) a seasonal limit where no more than an average of 10% of the relevant seasonal area of the site is disturbed over a season. The approach provides an equitable way of managing noise across industries and incentivizes less noisy operations and that those be undertaken in the times of the year with lower densities of porpoise. For the purposes of impact assessment and monitoring, the guidance also advises the use of operation-specific disturbance ranges where it is assumed most of the disturbance will occur. In the absence of dose-response curves for each operation and site-specific environmental conditions, an alternative, semiempirical approach is advised based on published evidence of disturbance of harbor porpoise. The largest of those ranges is 26 km for monopile driving noise (without noise abatement) and high-order

**Table 1** Recommended effective deterrence ranges (EDRs) (JNCC 2020)

| Activity | EDR (km) | References in which EDRs were based |
|---|---|---|
| Mono-pile | 26 | Tougaard et al. 2013; Dähne et al. 2013 |
| Mono-pile with noise abatement | 15 | Dahne et al. 2017; Rose et al. 2019 |
| Pin-pile (with and without noise abatement) | 15 | Graham et al. 2019 |
| Conductor piling for oil and gas wells | 15 | Jiang et al. 2015; MacGillivray 2018; Graham et al. 2019 |
| UXO | 26 | Based on mono-pile EDR |
| Seismic (airguns) survey | 12 | Thompson et al. 2013; Sarnocińska et al. 2020 |
| Other geophysical surveys | 5 | Crocker and Fratantonio 2016; Crocker et al. 2019 |

UXO clearance, and the smaller one 5 km is for geophysical surveys involving sub-bottom profilers (Table 1).

The purpose of this chapter is to introduce a noise management approach that is being adopted in a protected area for marine mammals in the UK, and provide examples of its application, both when planning operations inside the SAC and also in terms of retrospective monitoring providing useful insights for management.

## The UK Marine Noise Registry (MNR)

To assist with the management of noise and assess the effectiveness of the spatio-temporal approach, it is essential to monitor the occurrence of noisy events and understand their distribution, patterns, and potential for disturbance. The MNR (https://jncc.gov.uk/our-work/marine-noise-registry/), established in 2015, is a custom-built database where data is collected on activities in UK waters resulting in impulsive noise (frequency 10 Hz–10 kHz) such as seismic surveys, some sub-bottom profiling equipment, impact pile driving, and explosives/ordnance clearance. Data for planned activities is submitted at the time of license applications, and once the activity has taken place, closeout data is submitted within 3 months or less of the activity takes place. For some sub-bottom profiler surveys exempt from the need for a license, this information is reported on a voluntary basis. The data collected refer to the type of activity, and the day it occurred and where, with parameters such as sound levels and frequencies being optional. The MNR's main purpose is to monitor the distribution and intensity of impulsive noise events retrospectively, but from its inception it was also intended that it could be used in the management of noise. The MNR is currently being developed to be able to provide a planning and management tool using the elements in the SAC noise management guidance to help industry and regulators plan and coordinate to not exceed the noise limits advised.

## Planning Noisy Activities Inside a Protected Area

Projects planned within or in the vicinity of an SAC need particularly stringent environmental impact assessments (habitats regulations assessment, HRA) in order to ascertain beyond reasonable doubt whether the planned activities could have an impact on the integrity of the site considering its conservation objectives (https://www.gov.uk/guidance/habitats-regulations-assessments-protecting-a-european-site). One of objectives for harbor porpoise SACs is to avoid significant disturbance of the species within the site. By using the guidance's two elements (the 10/20% limits and the disturbance ranges) in assessments for both projects alone and in combination, developers and regulators can identify "crunch" seasons where those noise limits could be exceeded.

In the SNS SAC, HRAs have identified seasons where limits could be exceeded, for example, when two wind farms were due to construct in the summer of 2021 in the high-density summer seasonal area (https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/753026/RoC_SNS_cSAC_HRA_5.0.pdf). Both would be using monopiles, and therefore the 26 km buffer was used in the assessment which concluded that the 10% seasonal limit could be exceeded. In this case, rescheduling construction or using noise abatement systems with pile driving would bring the impact to within limits. Another example was in the summer of 2019, when another HRA (https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/799521/Spectrum_Seismic_Survey_HRA_Rev_1.3.pdf) concluded that both the 20% daily and 10% seasonal limits would be likely exceeded through several operations planned to occur simultaneously. Regulators and industry agreed on a series of measures to avoid that, including that UXO clearance in a single day had to be within 5 km of a central point to contain the noise footprint, also limiting the number of UXOs detonated each day, and delaying part of a seismic survey to the winter, when lower densities of porpoise were to be expected. This ensured that neither limit was exceeded (daily: 7.27%, seasonal average: 1.42%) (JNCC 2022). Since noisy activities are planned to increase in the site in the short to medium term, it will be imperative to use not just scheduling but also less noisy alternatives and noise abatement systems to keep within the limits.

The upgraded UK MNR, currently in development, will help with HRAs, planning, and management by providing a series of functions that will help with exploring disturbance scenarios by calculating activity "noise disturbance footprints" in areas of interest such as SACs, and providing in-combination totals to compare with advised noise limits. In addition, it will provide a coordination hub for industry to upload, share, and see the work programs of others, allowing live communication for scheduling work to avoid exceeding limits. As of 2022, oil and gas and windfarm operators have new license conditions which require them to cooperate and coordinate their activities so as to not occur on the same day, or if they do operate simultaneously, they must evidence that the noise disturbance limits will not be breached.

## Retrospective Monitoring of Noise

The MNR data has been used to retrospectively investigate the prevalence of impulsive noise events in the SNS SAC, to determine impulsive noise distribution in time and space and the contribution of the different activities to the overall disturbance footprint as well as whether advised limits have been exceeded. Data from 2015 to 2020 were extracted from the MNR and imported into GIS for further analyses. Only data from within the SNS SAC and from within a 50 km radius around the SAC (to capture noise events outside the SAC that could potentially cause disturbance inside it) were used. The data stored in the MNR can include geographic information in points (latitude/longitude) or polygons (the oil and gas licensing blocks). The points mostly show the locations where piling or explosives took place and the blocks where geophysical surveys (seismic and sub-bottom profilers) took place. To estimate the area around the noise source potentially affected by noise disturbance, buffers were added according to the activity type and respective Effective Deterrence Range (EDR) as provided in the noise guidance (Fig. 1). The buffer areas were used to calculate:

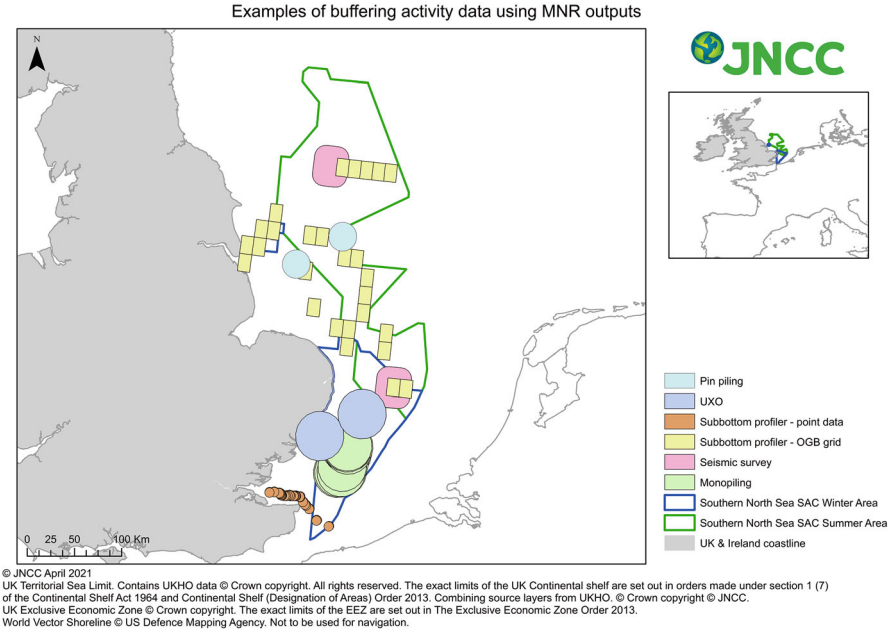



**Fig. 1** Illustration of spatial disturbance footprints for different activities according to EDRs in JNCC (2020). For UXO and mono-piling point locations, the buffer has a 26 km radius. For pin-piling point locations, the buffer has a 15 km radius. For seismic surveys, a 12 km radius buffer was applied around the oil and gas blocks. For sub-bottom profilers, with locations given in oil and gas blocks this was used unbuffered as a proxy for the disturbed area. For sub-bottom profilers' locations given in latitude and longitude, a 5 km radius buffer around the point was used

- Daily disturbed area from all activities combined for each day of each season/year for the relevant seasonal area.
- Seasonal average disturbed area from all activities combined for the relevant seasonal area. For example, a daily footprint of 19% for 95 days would result in an average of 19 × 95/183 days (summer) = 9.86%.

The areas of each buffer overlapping the SAC's summer and winter areas were calculated for activities that occurred in each relevant season (i.e., only winter activities are accounted for in the winter area and the same for the summer area). Plans or projects potentially resulting in noise disturbance in the SAC but operating outside of the season for which the area was identified as having elevated harbor porpoise densities are unlikely to result in significant effects to the site. Since during the "off season" the area is no different in terms of average densities than the rest of the population range, the noise management approach does not apply outside the relevant season (JNCC 2020).

The number of days with impulsive noise events in each season/year period ranged from 2 (winter 2014–2015) to 107 (summer 2018) (Table 2). Seven out of the twelve periods had impulsive noise events in less than 40 days (i.e., less than a quarter of days in a season). Two of the top four periods with the most days with noise were in winter, and two were in summer. The four periods with the lowest number of days were in the winter.

The daily 20% threshold was exceeded on 1 day only, in summer of 2015 (21.01% of the relevant seasonal area). On this day, three geophysical companies were undertaking seismic surveys in the north of the summer area, resulting in a large area potentially disturbed. The second highest daily maximum was recorded on 1 day in winter of 2016–2017 with 19.08%, because of two adjacent mono-piling

**Table 2** Total days with noise and estimated area disturbed for each of the 12 periods (summers and winters from 2015 to 2020) in the respective seasonal SAC area

| Period | Days with noise out of 182 days/winter and 183 days/summer | Maximum daily area disturbed (%) | Seasonal average area disturbed (%) |
|---|---|---|---|
| Winter 2014–2015 | 2 | 9.25 | 0.09 |
| Summer 2015 | 45 | 21.02 | 1.34 |
| Winter 2015–2016 | 0 | 0 | 0 |
| Summer 2016 | 16 | 0.67 | 0.05 |
| Winter 2016–2017 | 83 | 19.08 | 7.38 |
| Summer 2017 | 11 | 5.33 | 0.16 |
| Winter 2017–2018 | 9 | 3.82 | 0.09 |
| Summer 2018[a] | 107 | 10.23 | 1.08 |
| Winter 2018–2019 | 93 | 16.47 | 1.73 |
| Summer 2019 | 93 | 7.27 | 1.42 |
| Winter 2019–2020 | 5 | 10.09 | 0.22 |
| Summer 2020 | 26 | 8.00 | 0.55 |

[a]The figures for Summer 2018 are an underestimate as at the time of this analysis the MNR did not have data on the installation of mono-piles for a large wind farm installed during that summer.

events. This was the season/year period with the highest seasonal percentage, 7.38%, so the seasonal limit of 10% was never exceeded between 2015 and 2020. In both the summer and winter seasons, the maximum daily footprint was due to single activities, mono-piling, and seismic surveys, which due to their large EDRs (26 km and 12 km, respectively) resulted in a large percentage area affected. This shows that on days where mono-piling or seismic activities occur without any noise abatement in place, there is little buffer for other activities to be able to occur without exceeding noise limits, particularly in the winter area which is smaller than the summer area.

Figure 2 shows the contribution of different activities to the estimated area disturbed by noise within the SNS SAC, together with the total number of days in which the activities took place for the relevant season/area for all years combined (2015–2020). It shows that in the summer (summer area), seismic surveys contributed the most to the noise footprint (65.6%, 99 days), with sub-bottom profiler surveys occurring in more days but due to the estimated smaller potential disturbed area contributed less to the total footprint (14.7%, 167 days). UXO clearance using explosives accounted for 19.5% of the total area and occurred in 57 days over the 6 years. As mentioned before, there are several wind farms due to be installed in the SNS SAC's summer area in the next 10 years. If piling campaigns take place in the summer, they will likely account for much of the future disturbance in this area. Scheduling piling, geophysical surveys, and explosives in the "winter" (October to March) in this area will avoid the elevated harbor porpoise densities during the "summer" months and limit disturbance within the SAC. In the winter (winter area), mono-piling contributed the most to the noise footprint (50.8%, 53 days), followed

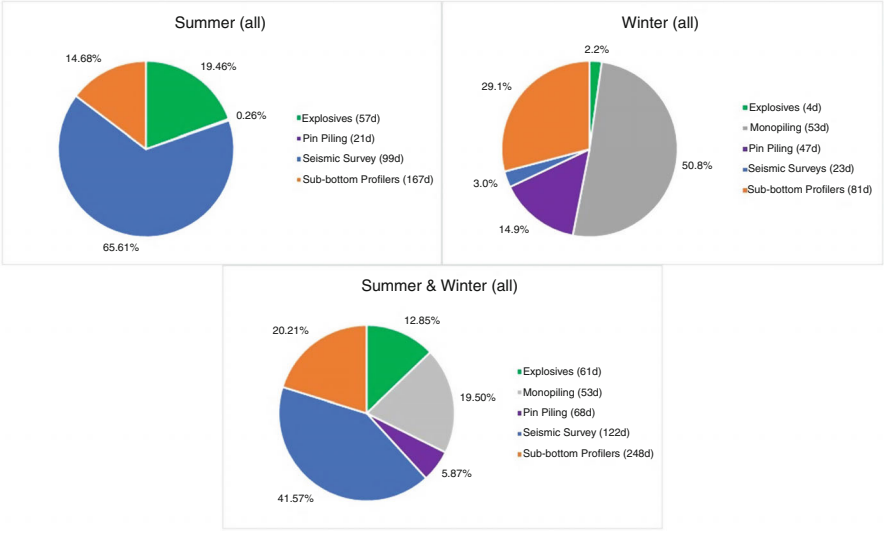



**Fig. 2** The contribution of different activities to the estimated area disturbed by noise within the Southern North Sea SAC (2015–2020). All summers, all winters, and all data combined. Total days of noise per activity also given in brackets

by sub-bottom profilers (29.1%, 81 days) and pin-piling (14.9%, 47 days). It is likely there will be further wind farms being installed in the winter area in the next 10 years, and scheduling piling in the "summer" months (April to September) will avoid the elevated harbor porpoise densities during the "winter" and limit disturbance within the SAC.

While seismic accounted for the largest spatial footprint overall, this is likely an overestimation due to the way the noise is reported in oil and gas blocks rather than survey lines. Sub-bottom profilers had the largest temporal footprint (248 days over the 6 years). This is an underestimate given that many such surveys are not yet licensed and therefore do not necessarily report to the MNR.

## Discussion

For highly industrialized marine areas that contain important habitats for species sensitive to noise, it is a challenge for regulators to keep noise to levels low enough to avoid significant disturbance to species in these areas. In the UK, a new approach advised by nature conservation agencies has been adopted for managing noise inside harbor porpoise SACs. The approach is based on the setting of spatial and temporal limits to regulated activities that result in impulsive noise. The appeal of the approach has been that it applies equally to all industries and helps to standardize cumulative assessments. It is also pragmatic recognizing that porpoise habitats within these sites need to continue to provide those important functions for which they were designated but are also able to cope with a certain level of noise disturbance. The approach is relatively straightforward to interpret and apply, but it relies on good environmental impact assessments, on good regulation and communication, on good reporting, and on monitoring of both noise and harbor porpoise distribution. Initially, there was some resistance by regulators and industry in adopting it since they could see issues in implementation, the potential restrictions to operations, and the incomplete evidence base. But by keeping the approach under scrutiny, monitoring its application, and through good communication between regulators and industry, the approach has kept noise footprints contained in the area. In the UK, different industries are regulated by different departments so for the approach to work, it has also been essential for regulators to communicate with each other, which they have done through a working group that meets regularly (https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/1114313/SAC_Noise_Management_Regulators_Working_Group_Terms_of_Reference_2022.pdf).

It has been possible to undertake retrospective monitoring of noise inside the SNS SAC, through the data collected via the UK MNR. This has helped to identify spatial and temporal patterns of past activities, and also identify persistent hotspots of noise, information that can help predict future noise scenarios and better plan and regulate future activities. For example, the data showed that seismic surveys account for most of the spatial footprint on the site since they occur mainly in the summer area in the summer, but it is sub-bottom profilers that have the highest temporal footprint. It is

clear also that when there is either monopiling, UXOs, or seismic surveys, the daily noise quota is rapidly reached.

While the advised limits have only been exceeded on 1 day in the 6-year period looked at, it is increasingly likely in the future that the daily or seasonal thresholds could be exceeded in the SNS SAC, with at least ten winds farms, planned in or around the SAC, likely requiring pile-driving and underwater explosive clearance campaigns. This highlights the need for careful planning. In the future, it is likely that in addition to scheduling to avoid large periods of concurrent operations, the use of noise abatement systems such as bubble curtains (Bellmann et al. 2020), alternative methods of turbine installation, and low-order UXO clearance will be needed to ensure that limits are not exceeded and that disturbance in the protected site is kept low.

Data that is routinely reported to the UK MNR on planned activities can be used in disturbance risk assessments, and the new functions currently being implemented in the upgraded MNR will enable the exploration of future disturbance scenarios, which can be used to support assessment, planning, and management.

It is hoped that this noise management approach for use in marine mammal protected areas will, alongside other measures, incentivize industry to employ less noisy alternative operations and technologies and find other ways to reduce their disturbance footprint in time and space. The approach will be reviewed regularly to see if it is working at reducing noise in the sites and if it continues to be based on the best available evidence.

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# Balancing the Use of Acoustic Deterrent Devices (ADDs), Slow-Start, and Soft-Start for Offshore Impulsive Piling

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**Abstract**

Injury to marine mammals due to sound from impact piling of the foundations for offshore wind farms has become a significant concern in the UK and worldwide. With increasing pile sizes and hammer energies being proposed for future developments, this concern remains a significant consenting risk.

The use of soft-start (where the initial hammer energy is reduced for a period at the start of piling) and the use of slow-start (where the hammer strike rate is reduced) are common mitigation strategies for reducing the potential for injury to marine mammals in the UK. More recently, the use of Acoustic Deterrent Devices (ADDs) has become commonplace as an additional mitigation strategy for deterring marine mammals from the area where piling will take place.

The range at which injury can occur depends upon numerous different factors, including the source level of sound radiated by the pile into the water column, hammer energy, strike rate, swim-away speed, timings for each phase, and propagation of sound away from the pile. For the mitigation strategy to be most

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effective, any near-by animal needs to swim some distance from the pile before full-power piling occurs. However, at present there is no definitive methodology for determining the most effective combination of ADD, soft-start, and slow-start in order to reduce the risk of injury to as low as practicable.

This chapter examines the various factors that affect the efficacy of a combined ADD, slow-start, and soft-start piling mitigation protocols, and makes recommendations as to how to design the most effective mitigation strategy.

**Keywords**

Sound · Impact · Injury · Pile driving · Mitigation · ADD · Soft-start

## Introduction

Installation of offshore wind turbine foundations using percussive piling can lead to temporary or permanent threshold shift (TTS/PTS) in marine mammals (NMFS 2018; Southall et al. 2019).

During offshore impulsive pile driving, the early stages of the construction operations are managed to give near-by marine mammals the opportunity to move away from the pile and reduce the potential for injury. This may be via a combination of:

- Acoustic Deterrent Devices (ADDs)
- Slow-Start (low hammer strike rate)
- Soft-Start (operational speed, low hammer energy)

An ADD can be employed to promote flight away from the foundation before potentially harmful peak-sound levels are experienced. Given that animals will experience the greatest proportion of the energy when closest to the pile, hammer strikes early in the pile sequence are designed to be quieter and have a lower contribution to the cumulative effects of sound energy (slow- and soft-starts). The intended combined effect of these mitigation protocols is that any near-by marine mammals could move away from the sound source at a rate such that the increasing transmission loss offsets the increasing hammer energy, reducing the potential for injury.

The dual injury thresholds set out in Southall et al. (2019) include both unweighted peak-sound pressure level and cumulative (marine mammal hearing weighted) sound exposure level (SEL) metrics. Mitigation strategies therefore need to account for both instantaneous injury (as reflected by the peak threshold) and cumulative SEL over the entire pile installation period. While it is relatively straightforward to determine the required mitigation for instantaneous injury (by estimating the radius over which the peak threshold is exceeded), the specification of mitigation using cumulative SEL is more complex.

The effects of the ADD duration, and the strike rate/blow-count of the slow- and soft-start periods can be chosen to minimize the impact region. This can then assist with suggestions for an effective mitigation strategy.

## Outline of Methodology

There are numerous factors which can affect the calculated injury range for a marine mammal including:

- source sound level and frequency spectrum;
- site-specific propagation losses;
- hammer strike rate (in each phase of piling);
- slow- and soft-start durations;
- hammer energy reductions during soft- and slow-start;
- species group hearing weighting;
- swim speed and direction.

In order to focus on the objective of investigating the mitigation strategy, it has been necessary to focus on those parameters which are of most interest to this study. Because low-frequency (LF) cetaceans are often predicted to experience the largest PTS zones in recent noise modelling studies for piling (based on cumulative SEL), compared to high-frequency (HF) cetaceans which typically experience the largest instantaneous peak-sound pressure threshold, the LF-cetacean hearing group was chosen as a focus for the noise modelling. Likewise, since the swim speed and direction of travel of a marine mammal is not within the control of the developer, the study has removed the swim-away speed from the parameters investigated, although it is worth noting that the value chosen will have a significant effect on the calculated PTS range. Likewise, the starting range (prior to ADD activation) is also an important parameter on the cumulative exposure but is outside the control of the developer.

The range at which the Southall et al.'s (2019) PTS threshold for LF cetaceans was estimated based on simple, empirical propagation modelling. The various input parameters were then varied as described in Section 55 in order to provide a large dataset of predicted PTS ranges. This dataset was then used as an input to a statistical analysis model to determine the relationship and significance of the various input parameters to the PTS range.

Numerous simplifications were made in the study, and it is important to understand that the real-world piling scenarios will be much more complex. Examples of simplifications made include a constant hammer energy during each phase of piling, no ramp up of energy, marine mammals swimming directly away from piling for the entire duration, and simplified propagation modelling.

## Definition of Parameters

Sets of parameters for swim speeds and pile-driving schedules were taken as pseudo-random combinations of timing (e.g., ADD duration) and piling properties (e.g., strike rate). Each set of parameters, or *experiment*, was then passed to a simple exposure model to give a PTS range for LF cetaceans. The input parameters and their distributions across 125,000 of these experiments are shown in Fig. 1. Note that some of the parameter configurations were removed from the analysis, such as when the total piling duration was shorter than the combined slow-start and soft-start durations. Also, the strike rate and energy were set to zero when there was no soft-/slow-start duration, hence the deviation from the uniform distribution in the below histograms.

The full list of input parameters investigated is as follows:

- ADD activation duration;
- Total piling duration;
- Hammer strike rate at full-power piling;
- Duration of slow- and soft-start periods;
- Hammer strike rate and hammer energy during slow- and soft-start periods; and
- Percentage of the full-power hammer energy used for slow- and soft-start.

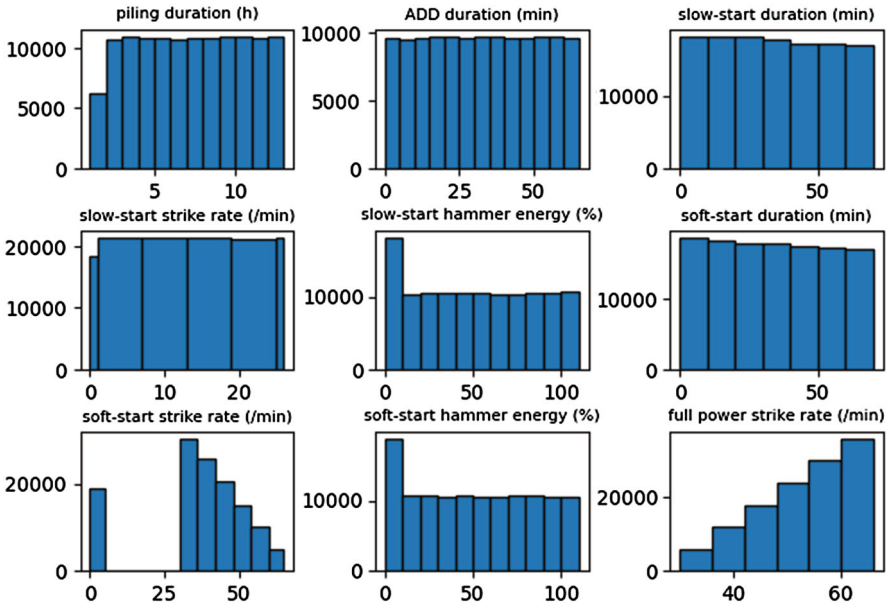



**Fig. 1** Parameter spread of model inputs used in the analysis (125,000 experiments)

The experiment parameters were used to form a complete hammer schedule for each experiment: a series of strike times (in seconds after the first strike or, if used, ADD); and pile-driving source levels (increasing after the slow- and soft- start).

The cumulative sound exposure level was calculated for a moving marine mammal starting at a given distance by finding the distance travelled at the time of each strike (accounting for an assumed LF-cetacean swim-away speed of 2.3 m/s) and taking the sum of all the received single strike SELs based on the equivalent monopole source level and an estimated propagation loss. The sound propagation loss is taken from a simple empirical logarithmic-loss equation

$$RL = SL - A\,\log(r) - \alpha\, r - B,$$

where $r$ is the range from the piling foundation in meters, and the constants $A$ , $B$, and $\alpha$ are taken from representative results of sound propagation modelling of pile driving accounting for LF-cetacean hearing sensitivities, taken here as $A = 16.0$, $B = 3.0$, $\alpha = 0.00012$. In the equation, $SL$ denotes the source level and $RL$ the received level.

In order to model the period of ADD usage, the starting distance of the marine mammal from the pile includes a distance equivalent to the ADD duration multiplied by the swim speed. Sound from the ADD is assumed not to have a significant contribution to the sound exposure, based on the findings of Boisseau et al. (2021).

The above steps describe the process of taking a starting distance and calculating the sound exposure level (for a given set of model experiment parameters). It is relatively simple to use a root-finding algorithm on this process to find the starting distance which corresponds to a specific sound exposure level; in particular, the threshold range for and permanent threshold shift (PTS) in LF cetaceans can be derived using the PTS onset metric from Southall et al. (2019).

## Analysis

For each selected source level, approximately 125,000 different hammer schedules were produced. These were analyzed using the R package *relaimpo*, which calculates the relative importance of each metric toward a total $R^2$ value, which represents the proportion of variance that is explained by the model. The fit to the model was overall strong and increased with source level up to 227.5 dB re 1 $\mu Pa^2s$ re 1 m and then reduces (Table 1).

Because the relative importance of the combined mitigation time is so much greater than the other variables, and since they represent the same unit, these were totaled and a single regression for each source level was plotted. The linear regression model for each sound exposure level was used to estimate the overall mitigation time required. Using the rearranged equations from the regression plots (example can be seen in Fig. 2), the total mitigation time is calculated to maintain an impact radii of 500 m. This distance was chosen since the Joint Nature Conservation

**Table 1** Relative importance of controllable variables to the impact ranges of low-frequency cetaceans

| | Source sound exposure level (dB re 1μPa$^2$s re 1 m) | | | | | |
|---|---|---|---|---|---|---|
| | 220 | 222.5 | 225 | 227.5 | 230 | 235 |
| **Overall model fit ($R^2$)** | 94.1% | 96.1% | 97.3% | 99.5% | 99.3% | 98.6% |
| | **Relative importance ($R^2$)** | | | | | |
| **Piling Duration** | 0.01% | 0.02% | 0.04% | 0.1% | 0.3% | 1.5% |
| **ADD Duration** | 29.4% | 32.6% | 33.6% | 33.0% | 32.4% | 31.7% |
| **Slow-Start Duration** | 30.8% | 30.1% | 30.7% | 31.7% | 31.5% | 30.6% |
| **Slow-Start Strike Rate** | 2.0% | 1.8% | 1.1% | 1.1% | 1.1% | 1.1% |
| **Slow-Start Hammer Energy** | 3.0% | 2.6% | 1.8% | 1.7% | 1.7% | 1.7% |
| **Soft-Start Duration** | 22.2% | 23.4% | 24.8% | 26.6% | 26.8% | 26.4% |
| **Soft-Start Strike Rate** | 4.4% | 3.8% | 3.9% | 3.7% | 3.6% | 3.5% |
| **Soft-Start Hammer Energy** | 2.0% | 1.6% | 1.2% | 1.2% | 1.2% | 1.2% |
| **Full-Power Strike Rate** | 0.2% | 0.3% | 0.3% | 0.5% | 0.6% | 1.0% |

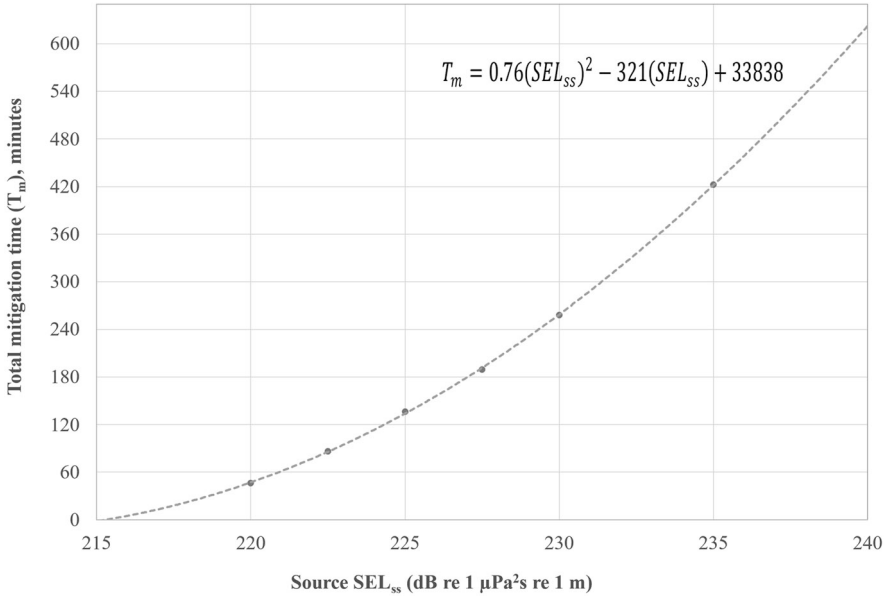



**Fig. 2** Modelled mitigation time for different source $SEL_{ss}$ levels, with polynomial regression applied

Committee (JNCC) will stipulate that the area within 500 m of the sound source must be clear of marine mammals prior to the commencement of activity (JNCC 2010). The area will be monitored by dedicated marine mammal observers (MMOs) and often complimented with passive acoustic monitoring (PAM) prior to commencement of piling. The analysis shown in Table 2 shows that there is an increase in overall delay as the source $SEL_{ss}$ is increased to maintain a mitigation zone of 500 m.

**Table 2** Modelled delay duration to restrict marine mammal PTS due to cumulative effect to 500 m

| Source SEL, dB re 1μPa$^2$s re 1 m | Total mitigation time, Minutes[a] |
|---|---|
| 220.0 | 46 |
| 222.5 | 86 |
| 225.0 | 136 |
| 230.0 | 258 |
| 227.5 | 190 |
| 235.0 | 422 |

[a]These mitigation times are examples for this specific modelling case and input parameters

When the total mitigation time is plotted against the unweighted source $SEL_{ss}$ it is possible to fit a polynomial model very closely to the data ($R^2 \sim 99.99\%$). The model predicts that if source sound level reaches 240 dB re 1 μPa$^2$s re 1 m or more, the total mitigation time required to maintain a 500 m radius PTS zone would be greater than 10 h. Such a mitigation time would be an impractical and expensive burden on the developer. In any case, these mitigation measures assume a behavioral response of a marine mammal which, if the animal is assumed to be swimming at 2.3 m/s for 10 h, by this point would be 83 km from the source (and therefore may not continue to flee from the sound source). Consequently, it is concluded that it is unlikely that use of ADD, soft-start, and slow-start combined can practically mitigate sound from these high sound levels sufficiently to avoid PTS to LF cetaceans outside the 500-m mitigation zone.

## Conclusions

The majority of piling in the UK to date has utilized mitigations such as ADDs, soft-, and slow-start methodologies, combined with a mitigation zone monitored by MMOs and PAM, as the primary means of minimizing the potential for PTS to marine mammals.

Based on the results of the modelling and data analysis, the total length of the mitigation sequence (i.e., the total time of ADD activation, soft-start and slow-start combined) is the most important factor with all three mitigation measures being of similar effectiveness. The analysis shows that the exact parameters used for each phase of mitigation are less important than the total mitigation time, although by fine-tuning these parameters the PTS zone can be further reduced once the total mitigation time has been defined. It should be noted that some variables are closely tied, and despite the apparent low $R^2$ value, should not be ignored (e.g., soft-start hammer energy).

It can be seen in Table 1 that the strike rates during the slow- and soft-start periods are higher contributors to the total importance than, for example, the energy reductions. However, the strike rates show greater contributions to the total for the lower full-power hammer energies than the higher, with the percentages steadily reducing as the energy is increased. It is therefore evident that although the strike rate is an

important factor to consider during the slow- and soft-start periods, it is more useful for reducing the impact to marine mammals for hammers with lower maximum sound levels.

For pile driving with a higher source level, the analysis shows that the required total mitigation time may make the use of ADD, soft-start, and slow-start alone impracticable. Therefore, the trend for larger piles and higher energy hammers means that it is more likely that physical sound reduction methods will be required instead of, or in addition to, these other measures.

Finally, it is worth reiterating that the analysis presented in this chapter is based on a simplistic spreading model and simplified input assumptions. Consequently, while the general findings might be useful in informing an approach to mitigation, the precise relationships in the chapter will not be directly transferable to specific projects.

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# Evaluating the Benefits of Noise Reduction Mitigation: The ECHO Program

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**Abstract**

Mitigation measures to reduce underwater levels of radiated noise are of increasing interest. An example implementation is the Vancouver Fraser Port Authority's ECHO

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(Enhancing Cetacean Habitat and Observation) Program that has initiated voluntary, seasonal vessel slowdowns since 2017. ECHO's goal is to reduce the acoustic disturbance of commercial shipping on endangered whales, whose critical habitat coincides with major shipping routes. This initiative requires the cooperation of many stakeholders, and it is therefore imperative that acoustic monitoring along shipping routes suitably evaluates the effectiveness of such mitigation measures. However, doing so is not straightforward, as levels of ambient noise fluctuate not just in response to vessel slowdowns but also with confounding noise sources of environmental, biological, or anthropogenic origin (e.g., noise from currents, wind, marine mammals, small boats, etc.). Here, methods are presented for how to appropriately compare noise levels during baseline and mitigation periods, using data from one acoustic monitoring station as an example. Factors to consider when evaluating the effectiveness of mitigation measures are highlighted, including investigating suitable covariates and thresholds of these, both of which may vary between sites and seasonally at the same site. Useful ways of communicating results to a variety of stakeholders are also presented.

**Keywords**

Slowdown · Vessel speed reduction · Passive acoustic monitoring · Noise mitigation · ECHO · Shipping noise · Haro strait · Southern resident killer whale

## Introduction

The acoustic signature of shipping significantly contributes to ambient noise levels in the ocean (Hildebrand 2009). Shipping noise is broadband, can extend to echolocation frequencies (Veirs et al. 2016), and can be of high level (with broadband-radiated noise levels sometimes exceeding 200 dB re 1 μPa at 1 m; MacGillivray et al. 2019). Such noise pollution impacts marine life (such as marine mammals, fish, and invertebrates) by masking communication, disrupting foraging and traveling behaviors, causing habitat displacement, and increasing physiological stress (e.g., Nowacek et al. 2007; Duarte et al. 2021). As such, there is interest in mitigation measures to reduce levels of underwater-radiated noise from vessels. Such levels are positively correlated with vessel speed, leading to proposals that noise may be mitigated by lowering vessel speed (McKenna et al. 2012; Veirs et al. 2016). Indeed, initial vessel slowdown initiatives near Vancouver, Canada (e.g., MacGillivray et al. 2019; Grooms et al. 2022), and in the Santa Barbara Channel off California, USA (ZoBell et al. 2021), have demonstrated just this. Modeling explorations into other regions, such as the Canadian Arctic, have also demonstrated the soundscape benefits of vessel slowdowns (Pine et al. 2018). As reducing source levels reduces the spatial extent of affected habitat (Risch et al. 2021), not only do vessel speed reductions reduce the extent to which acoustic space is affected by marine traffic (MacGillivray et al. 2021) and decrease acoustic masking space (Putland et al. 2018), but they also compliment other conservation issues via the co-benefits of

reducing the risk of vessel strikes on whales and lowering greenhouse gas emissions (Ebdon et al. 2020; Leaper 2019; Rockwood et al. 2020).

An example implementation of a vessel speed reduction initiative, led by the Vancouver Fraser Port Authority, is the ECHO (Enhancing Cetacean Habitat and Observation) Program in British Columbia, Canada. This program has encouraged voluntary, seasonal vessel slowdowns since 2017, into and out of Canada's largest port. ECHO's goal is to reduce threats to at-risk whales from commercial shipping, notably the acoustic disturbance on the endangered Southern Resident killer whale (*Orcinus orca*; SRKW), whose critical habitat coincides with major shipping routes (Cominelli et al. 2018). SRKWs are listed as endangered and protected under Canada's Species at Risk Act (SARA) and the United States' Endangered Species Act (ESA). Both Canadian and American governments recognize vessel noise as a threat to the survival of the of SRKW population of only ~73 individuals (Centre for Whale Research 2022; Fisheries and Oceans Canada 2018; National Marine Fisheries Service 2008), alongside chemical pollution (e.g., Desforges et al. 2018) and shortages in prey availability (e.g., Williams et al. 2021). However, unlike mitigating contaminant loads or efforts to manage prey resources, mitigating underwater noise levels and disturbance is an immediate way to facilitate the recovery of the SRKW (Lacy et al. 2017; Burnham et al. 2021).

Previous studies specific to the SRKW have shown that vessel traffic disrupts foraging behavior (Lusseau et al. 2009), that the probability of successful prey capture decreases with increasing vessel speed (Holt et al. 2021), and that high noise levels reduce their probability of foraging in the first place (Williams et al. 2021). Modeling work connecting the ECHO slowdown with SRKW bioenergetics predicted that the voluntary slowdown led to a 22% reduction in "potential lost foraging time" (Joy et al. 2019); this value changed to a 40% reduction in potential lost foraging time when the model was set to a 100% participation rate of vessels travelling under 11 knots. Even though slower vessel speeds mean that vessel transits are of greater duration and vessels are in critical habitat for longer, underwater noise levels are still lower, leading to vessel slowdowns being discussed as an effective way to improve SRKW habitat quality (Joy et al. 2019).

During seasonal ECHO slowdowns, vessels are encouraged to travel at either $<$11 knots (bulkers, tankers) or $<$14.5 knots (containers, cruise ships, car carriers). Slowdown behavior is encouraged in Haro Strait and Boundary Pass (Fig. 1), which vessels pass through as they enter and exit the Port of Vancouver, and has, in recent years, extended to include a slowdown at Swiftsure Bank (Burnham et al. 2021). A related incentive is that vessels that hold a quiet ship notation from an international ship classification society are rewarded with reduced harbor dues at the Port of Vancouver (Vancouver Fraser Port Authority 2022a). The 90% participation rate of the ECHO slowdown period in 2021 in Haro Strait and Boundary Pass (Vancouver Fraser Port Authority 2022b) is aided by cooperation with the British Columbia Coast Pilots Ltd. and the Pacific Pilotage Authority – a similar combination also increased vessel slowdown participation in the Saguenay-St. Lawrence Marine Park (Parrott et al. 2016).

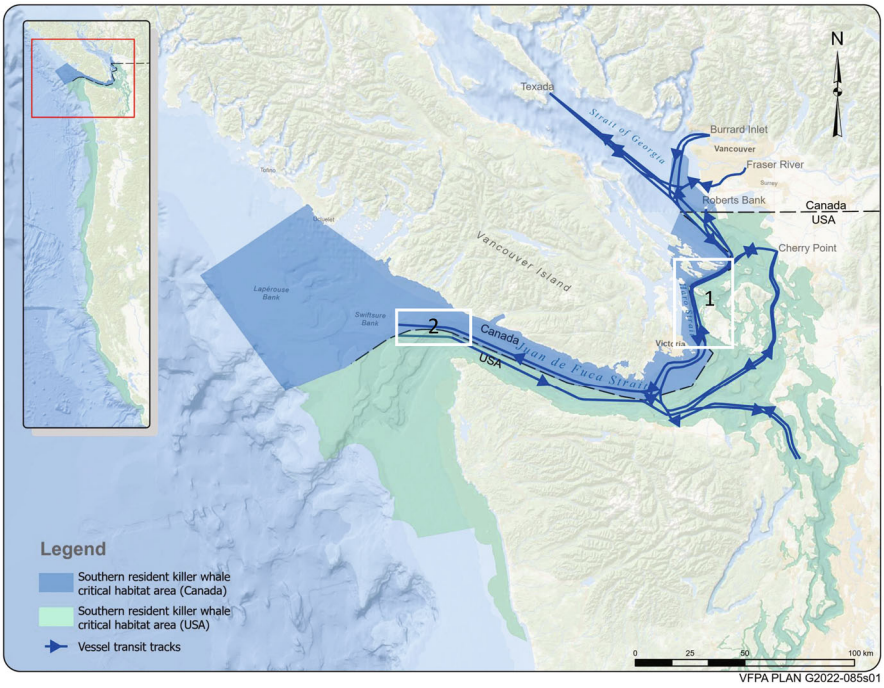



**Fig. 1** Map of shipping lanes into Vancouver and Seattle, alongside the ECHO Program slowdown areas in Haro Strait and Boundary Pass (area 1 in box) and Swiftsure Bank (area 2 box). The dark blue area denotes SRKW critical habitat in Canadian waters, and the green denotes SRKW critical habitat in US waters. (Map modified from Vancouver Fraser Port Authority Plan G2022-085s01, with permission from the Vancouver Fraser Port Authority)

As the ECHO Program is a collaborative effort that requires the cooperation of many stakeholders, from vessel owners and operators to pilotage and port authorities, it is imperative that acoustic monitoring along shipping routes appropriately evaluates the effectiveness of such mitigation measures (MacGillivray et al. 2019). In this chapter, the acoustic data from the 2021 ECHO Program slowdown season (Grooms et al. 2022; VFPA 2022b) are used as a case study with which to explore and evaluate how noise mitigation efforts are evaluated. Annual reports detailing acoustic findings for each ECHO slowdown season are freely available online (https://www.portvancouver.com/echo) and consistently show that broadband (10 Hz–100 kHz) noise levels are reduced by ~3 dB when slowdowns are in place. In other words, a ~ 50% reduction in underwater sound intensity has been consistently achieved through the ECHO Program's seasonal underwater noise reduction initiative. The focus here is not to present these results again but instead to explore how sensitive our approach to evaluating noise mitigation is to natural soundscape variation and different noise filtration criteria metrics. Below, results from several scenarios are presented for how the acoustic data are handled and how this impacts the interpretation of noise mitigation effectiveness.

## Methods

The seasonal ECHO slowdown aligns with historical peak SRKW presence, and the slowdown dates vary annually as they are initiated by the first SRKW sighting on or after June 1, and running until (usually) Oct 31. The ECHO 2021 season consisted of a 2-month baseline (May–Jun 2021) leading up to the start of the slowdown, followed by a 4-month slowdown period (Jul–Oct 2021). Due to animal presence, there was an extension of the slowdown effort into November 2021 (see Grooms et al. 2022); however the extension period was evaluated separately and is not included in this chapter.

Ambient noise was measured at cabled underwater acoustic listening stations along the slowdown route. SMRU Consulting monitors acoustics from Lime Kiln, on San Juan Island, on the east side of Haro Strait, and JASCO Applied Sciences monitors acoustics from Boundary Pass. For details on the locations, depths, acoustic recording settings, etc., refer to Grooms et al. (2022). The ECHO 2021 baseline and slowdown data from Lime Kiln are here used as an example dataset with which to explore various noise filtration metrics used to exclude periods with confounding noise.

## Thresholds and Filtering for Acoustic Data Inclusion

To robustly measure the probability of noise exceeding a given level in baseline versus slowdown conditions, periods of confounding noise need to be excluded from the analyses. This is to ensure that any differences in the noise levels of baseline and slowdown periods arise from the slowdown behavior of participating vessels. In other words, periods with elevated noise levels due to factors external to the vessel slowdown (e.g., nearby speedboats, strong winds, etc.) need to be excluded from acoustic analyses evaluating the effectiveness of the noise mitigation measure. Similarly, periods with low noise levels due to factors external to the vessel slowdown (e.g., when there are no vessels nearby) also need to be excluded.

To identify periods with confounding noise, thresholds were set for measured variables that had correlations to noise levels. Specifically, criteria for inclusion of data collected at Lime Kiln are detailed below.

### Excluding Periods When Participant Vessels Are Absent

Acoustic data were included in analyses when potential vessel slowdown participants were within 6 km of the hydrophone at Lime Kiln. This criterion excluded periods that were quiet, but where this quietness was unrelated to vessels participating in the slowdown. The effect of a slowdown on ambient sound levels can only be measured when the vessels that are potentially slowing down are present, otherwise the data could be skewed by differing amounts of low-amplitude noise levels. Six kilometers was selected as the typical range at which potential slowdown vessels can be heard by human analysists and at which acoustic vessel detectors are triggered.

### Only Including Periods when Potential Participant Vessel Is the Closest AIS Vessel

Acoustic data were included in analyses when potential vessel slowdown participants were the nearest of all automatic identification system (AIS)-broadcasting vessels to the hydrophone and therefore dominating the received noise field. In other words, data were included only when the closest AIS-broadcasting vessel was the potential participant. This was to ensure that noise measurements at Lime Kiln were driven by a potentially participating vessel and not another vessel type, such as recreational boats.

### Excluding Periods with Noise Unrelated to Slowdown Behavior

Acoustic data were included in analyses when an automated acoustic detector did not flag the presence of a small, non-AIS-broadcasting boat. (For details of this acoustic detector, including thresholds at different frequency bands, see Grooms et al. (2022)). This criterion excluded periods that were noisy, but whereby this noise was unrelated to the slowdown behavior of potentially participating vessels.

Acoustic data were included in analyses when wind speed measurements from a weather station located near the underwater acoustic recording station was $<|5$ m/s|. This criterion excluded periods where elevated noise levels were independent of slowdown participation.

Finally, acoustic data were included in analyses when water current speed readings from a near-seafloor current meter co-located with the underwater acoustic recording station were $<|25$ cm/s|. Again, this criterion excluded periods where elevated noise levels were independent of slowdown participation.

Both the wind speed and current speed threshold levels were set by exploring correlations between 1-min SPLs (dB re 1 μPa) in a lower decadal frequency band (10–100 Hz) as a function of current speed, as in Eickmeier et al. (2021). Sound pressure levels (SPLs) tend to have an obvious positive correlation after a particular current/wind speed is reached, and it is this value which Eickmeier et al. (2021) suggest choosing as a threshold level, beyond which data are excluded from noise analyses evaluating noise mitigation measures. Below, SPLs are plotted as a function of both wind speed and current speed, demonstrating how thresholds were selected for acoustic analyses at Lime Kiln (Figs. 2 and 3).

Note that particular thresholds for environmental covariates are site-specific and varied even between nearby Lime Kiln (current speed threshold at 0.25 m/s) and Boundary Pass (current speed threshold at 0.5 m/s) along the ECHO slowdown route (Grooms et al. 2022). Exploring these correlations is encouraged to be site-specific if possible, as soundscapes can vary on small spatial scales.

It is worth noting that the majority of acoustic data is excluded from evaluating the noise mitigation effectiveness. This is because not all the inclusion criteria are being met, whereby thresholds or criteria detailed above are exceeded. In the ECHO 2021 slowdown season, only 7.4% of the available acoustic data were retained for assessing the baseline period, and 10.8% of available acoustic data were retained for assessing the slowdown period (Grooms et al. 2022). It is worth noting that the

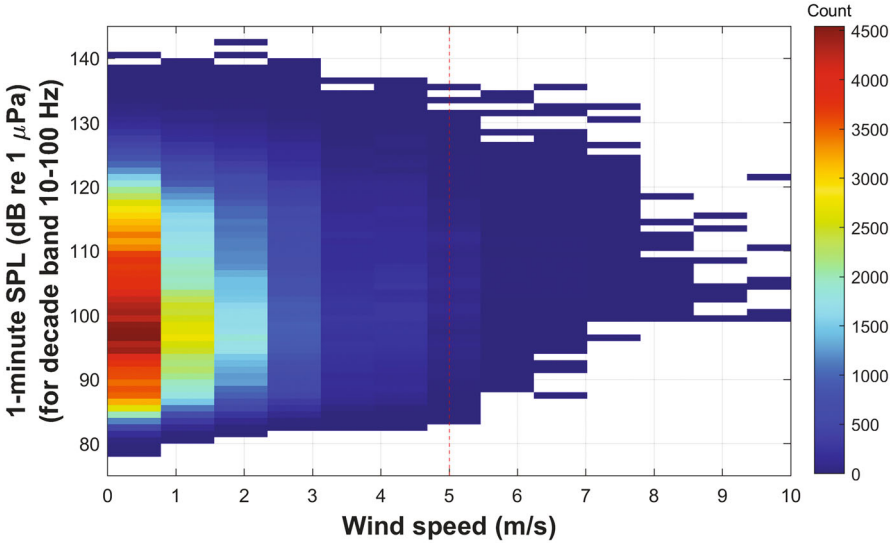



**Fig. 2** Sound pressure levels (1-min averaged SPLs in decade band 10–100 Hz, in dB re 1 μPa) as a function of wind speed (m/s) at Lime Kiln. A threshold value of |5 m/s| was used, beneath which acoustic data were included in noise mitigation analyses. (Data shown cover acoustic and wind data collected for the entire duration of the 2021 baseline and slowdown periods)

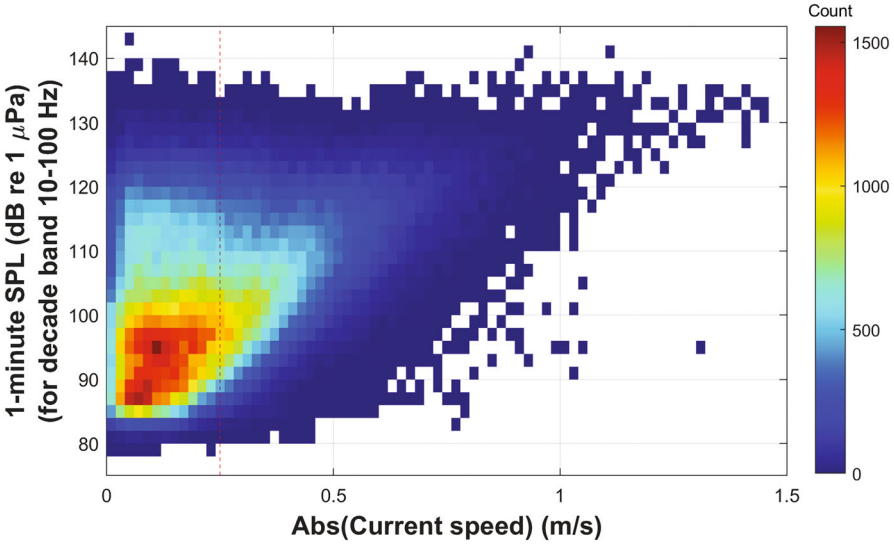



**Fig. 3** Sound pressure levels (1-min averaged SPLs in decade band 10–100 Hz, in dB re 1 μPa) as a function of current speed (m/s) at Lime Kiln. A threshold value of |0.25 m/s| was used, beneath which acoustic data were included in noise mitigation analyses. (Data shown cover acoustic and current data collected for the entire duration of the 2021 baseline and slowdown periods)

majority of data exclusion is due to quiet times between passes of large commercial vessels (slowdown participants). It is from these sub-selections of data – where periods with confounding noise or quiet have been removed – that the noise mitigation evaluation is based on.

## Example Dissemination

In the past years of ECHO reporting (e.g., Grooms et al. 2022), overall results have been presented in the form of cumulative distribution function (CDF) plots. Here, the curves show the probability that a 1-minute averaged sound pressure level is exceeded, in the specified frequency range. CDF plots were created for each decadal frequency band (10–100 Hz, 100–1000 Hz, 1–10 kHz, and 10–100 kHz), broadband (10 Hz–100 kHz), as well as in frequency bands relevant for SRKW communication (500 Hz–15 kHz) and echolocation (15–100 kHz) (Heise et al. 2017). Here, the focus is on a CDF plot for the broadband frequency range, as included in the final ECHO 2021 slowdown evaluation report (Fig. 4).

CDF plots provide a helpful way of simply disseminating slowdown evaluation results, making them more readily interpretable by a wider audience. In particular, such a plot allows for the evaluation of noise mitigation to be relayed as one number,

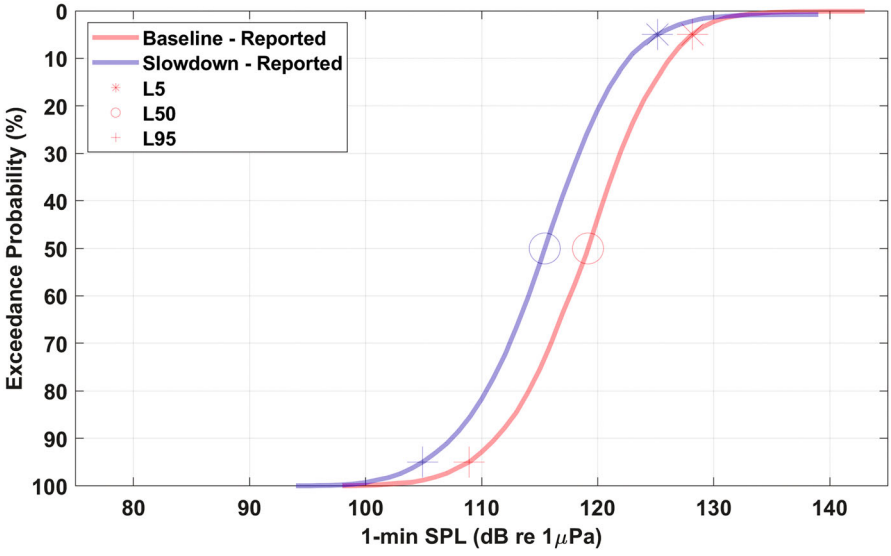



**Fig. 4** Example CDF plot, as reported for ECHO 2021 data, showing the cumulative distribution function for broadband acoustic data (10 Hz–100 kHz) at Lime Kiln, calculated using ECHO 2021 baseline and slowdown data. The baseline $L_{50}$ was 119.2 dB re 1 µPa, and the slowdown $L_{50}$ was 115.5 dB re 1 µPa, leading to a broadband $\Delta L_{50}$ of 3.7 dB. $L_{50}$ values are circled, $L_5$ values are denoted with an asterisk, and $L_{95}$ values are shown with a plus sign

while allowing concurrent graphical evaluation of other metrics. One can focus on the $L_{50}$ value, for example, which here is the median broadband SPL. In other words, the SPL value corresponding to the $L_{50}$ indicates that this value would be exceeded 50% of the time. Similarly, the $L_{95}$ value indicates where broadband ambient noise levels are exceeded 95% of the time, and the $L_5$ value indicates where broadband ambient noise levels are exceeded 5% of the time. The $L_{50}$ exceedance level is commonly used when comparing noise levels (e.g., Merchant et al. 2012). Here, the $L_{50}$ during baseline conditions was 119.2 dB re 1 μPa, and during the slowdown, the $L_{50}$ decreased to 115.5 dB re 1 μPa (Fig. 4). This visualization readily shows how median broadband SPLs were lower during seasonal slowdowns. The "one number" useful for dissemination is the difference between the $L_{50}$ values during periods where vessels acted as normal and periods where vessels were encouraged to slow down, leading to a $\Delta L_{50}$ of 3.7 dB and a concluding remark that median broadband noise levels had a reduction in underwater sound intensity by ~50% when vessels were encouraged to slow down.

## Modifying Filtration Metrics

To examine how appropriate and/or necessary the thresholds and metrics for acoustic data to be included in noise mitigation evaluations, variations on these thresholds were explored using the dataset from Lime Kiln. These included asking the following:

1. What if no data were excluded, and instead, all available acoustic data were used to compare noise levels during baseline and slowdown periods (i.e., including all data)?
2. What if only times with confounding quiet periods were excluded from noise mitigation evaluations (i.e., including all data, except data where AIS >6 km)?
3. What if, instead, only times with confounding quiet and confounding noisy periods from noise mitigation evaluations were removed? In other words, acoustic data can remain in the analysis regardless of current speed or wind speed (i.e., including all data, except data where AIS >6 km and data with acoustic detections of small boat presence).
4. What if the durations of baseline and slowdown periods become more equal (i.e., using only data that pass all inclusion criteria and drawing from a slowdown period that is reduced so that it is the same duration as the baseline period)?
5. How much data are needed for comparisons between baseline and slowdown periods to be robust (i.e., using only data that pass all inclusion criteria and drawing from smaller and smaller slowdown and baseline durations)?

The results of these variations are presented below, in terms of the effect on median $L_{50}$ differences, with the question numbers above corresponding to the subsections in section "Results."

## Results

### Findings If No Data Are Filtered

Using all available data in the CDF analysis meant that the acoustic analyses used to evaluate whether the noise mitigation was effective included all possible confounding covariates, and this skewed the results derived from the CDF plots. This skew is evident in Fig. 5, where the curves for baseline and slowdown periods have shifted to the left (relative to the final reported values shown for broadband noise in Fig. 4 and shown in Fig. 5 in muted tones) because several instances of confounding quiet times have been purposefully included (for the sake of example). While several instances of confounding noisy times are also included, the shift of the curves to the left indicates that it is the inclusion of the confounding quiet times that has a more predominant effect on the overall result due to there being more total minutes where potential participants were >6 km of the hydrophone at Lime Kiln than ≤6 km of Lime Kiln. The crossing pattern of the unfiltered baseline and slowdown CDF curves hints that this strategy of no filtration is too unstable to draw conclusions from. Nevertheless, the $\Delta L_{50}$ of 0.7 dB under this scenario leads to the conclusion that unfiltered slowdown is slightly quieter than the unfiltered baseline.

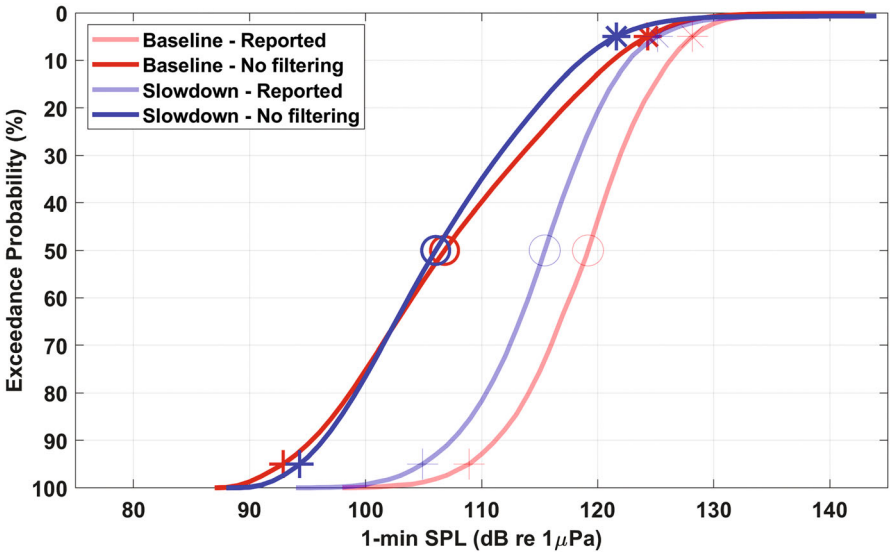



**Fig. 5** CDF plot on all available acoustic data (i.e., no data have been excluded for the noise mitigation evaluation), showing the cumulative distribution function for broadband acoustic data (10 Hz–100 kHz) at Lime Kiln, calculated using ECHO 2021 baseline and slowdown data (bright lines). The original CDF curves presented in Fig. 4 (as reported by the ECHO Program) are shown in muted tones. $L_{50}$ values are circled, $L_5$ values are denoted with an asterisk, and $L_{95}$ values are shown with a plus sign

## Findings If Only Confounding Quiet Times Are Removed

Following on from the finding that including all acoustic data from Lime Kiln in the noise mitigation evaluation skewed the cumulative distribution functions to the left, the results were then explored for a scenario where only the exclusion of confounding quiet periods was applied (Fig. 6). In other words, all periods with confounding noise (from small boats, from high current speeds and high wind speeds) were still included in the analysis. Here, the resulting $\Delta L_{50}$ of 3.4 dB in this scenario leads to the conclusion that partially filtered slowdown is quieter than the partially filtered baseline, though this result is not far from the final reported $\Delta L_{50}$ of 3.7 dB (Fig. 4).

## Findings If Only Confounding Quiet and Confounding Noisy Times Are Removed

Following on from the finding that excluding only confounding quiet periods from the noise mitigation evaluation at Lime Kiln yielded a $\Delta L_{50}$ of 3.4 dB (Fig. 6), the results were then explored for when the exclusion of periods with confounding noisy periods due to the presence of small boats was also applied (Fig. 7). In other words, periods where the automated small boat noise detector flagged the presence of a

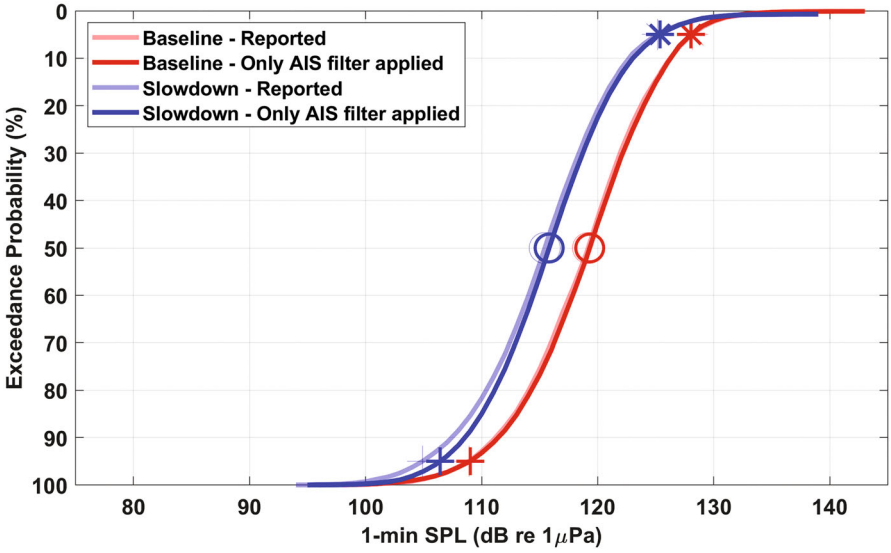



**Fig. 6** CDF plot when the only exclusion criteria was to remove confounding quiet periods, showing the cumulative distribution function for broadband acoustic data (10 Hz–100 kHz) at Lime Kiln, calculated using ECHO 2021 baseline and slowdown data (bright lines). The original CDF curves presented in Fig. 4 (as reported by the ECHO Program) are shown in muted tones. $L_{50}$ values are circled, $L_5$ values are denoted with an asterisk, and $L_{95}$ values are shown with a plus sign

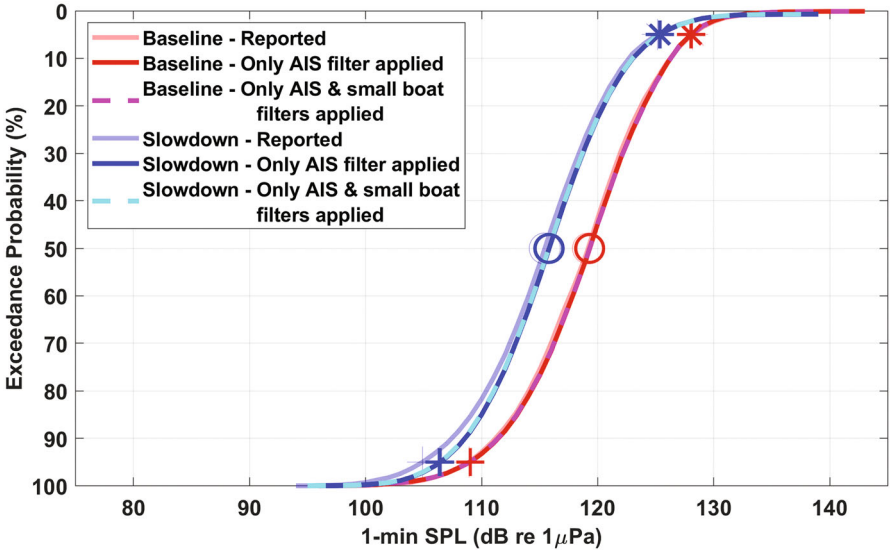



**Fig. 7** CDF plot when the only exclusion criteria was to remove confounding quiet and noisy periods, showing the cumulative distribution function for broadband acoustic data (10 Hz–100 kHz) at Lime Kiln, calculated using ECHO 2021 baseline and slowdown data (dashed, bright lines). The original CDF curves presented in Fig. 4 (as reported by the ECHO Program) are shown in muted tones, and the CDF curves presented in Fig. 6 (where only confounding quiet periods are removed) are shown in solid, bright lines. $L_{50}$ values are circled, $L_5$ values are denoted with an asterisk, and $L_{95}$ values are shown with a plus sign

confounding source of noise that was unrelated to the slowdown were filtered out of the analysis. Keep in mind here that this filtration strategy means that periods that may be noisy due to current speeds or wind speeds exceeding their given thresholds were still included in this specific noise mitigation evaluation scenario.

Here, the resulting $\Delta L_{50}$ of 3.5 dB in this scenario leads to the conclusion that the partially filtered slowdown is quieter than the partially filtered baseline, though this result is only 0.2 dB different than the final reported $\Delta L_{50}$ of 3.7 dB (Fig. 4). In other words, there was a 0.2 dB "cost" of excluding noise correlations with environmental covariates (e.g., wind and current), at least when considering the broadband frequency range, at this sampling site, during this particular year. Every year, samples will comprise a unique mixture of different origins of noise, whose relative contributions will differ year on year. Environmental noise, for example, had a greater individual contribution to the data exclusion in the previous ECHO 2020 evaluation year (Warner et al. 2021).

## Findings If Baseline and Slowdown Are of Equal Duration

The slowdown period in the ECHO 2021 season (122 days) was approximately double that of the baseline period (60 days) (Vancouver Fraser Port Authority

2022b). It has been a concern that this imbalance may skew the results used to evaluate noise mitigation effectiveness. To explore this, the amount of slowdown data (which passed *all* filtration criteria) to be included in the creation of the CDF plots was varied.

First, the longer slowdown period was reduced (via random sub-selection of each data point covering 1 min of acoustic data) so that the number of acoustic data points satisfying all filtration criteria for both baseline and slowdown periods was the same. In other words, the number of criteria-passing data points from the 2-month baseline exactly matches the number of criteria-passing data points from across the 4-month slowdown, all of which fed into the CDF analysis. This approach gave a broadband $\Delta L_{50}$ of 3.7 dB, which is the same as the ECHO 2021-reported broadband $\Delta L_{50}$ of 3.7 dB (Fig. 4).

Second, the longer slowdown period was reduced by instead drawing from the same duration of time as the duration of the slowdown (i.e., 2 months) in sequential minutes of SPL, as opposed to random sub-selection from the ~4-month slowdown, as above. Here, minor differences to the reported $\Delta L_{50}$ of 3.7 dB were found (presented in Fig. 4), with a broadband $\Delta L_{50}$ of 4.04 dB when comparing the baseline to a 2-month slowdown period pool of only July and August (0.34 dB different from the reported $\Delta L_{50}$ result). When this is repeated for different consecutive 2-month periods of slowdown, a broadband $\Delta L_{50}$ of 3.59 dB results when drawing from the 2-month slowdown period pool of only August and September (0.11 dB different), and a broadband $\Delta L_{50}$ of 3.54 dB when drawing from the 2-month slowdown period pool of only September and October (0.16 dB different).

This means that reducing the amount of slowdown data used for comparison with the baseline data had a broadband $\Delta L_{50}$ of anywhere from as little variation as $3.7 \pm 0$ dB to as big of a variation as $3.7 + 0.34$ dB. In other words, the scenarios with equal durations of baseline and slowdown periods had a broadband $\Delta L_{50}$ of 3.7–4.04 dB, compared to the final reported broadband $\Delta L_{50}$ of 3.7 dB when informed by a slowdown that was approximately double that of the baseline. Effectively the same overall broadband $\Delta L_{50}$ result occurs when reducing the amount of slowdown data available to compare with the baseline data and with which to assess any soundscape benefits of the slowdown (Fig. 8).

## Exploring How Much Data Are Needed for CDF-Based Conclusions

As mentioned earlier, only a fraction (~10%) of available acoustic data passes criteria and filtration metrics for inclusion in the assessment of the noise mitigation effectiveness. The logical next step to exploring this further is to reduce this amount even more and examine how much the overall conclusion, expressed at the difference between baseline and slowdown $L_{50}$ values, changes. In other words, how much data is needed for comparisons between baseline and slowdown periods to be justifiable? To do this, all acoustic data collected at times when criteria for inclusion were met were randomly subsampled to either 10% or 2% of this. These are extremely small amounts of time, with a 10% subsample drawn from a period where only ~10% of the data passed criteria for inclusion equivalent to a total of

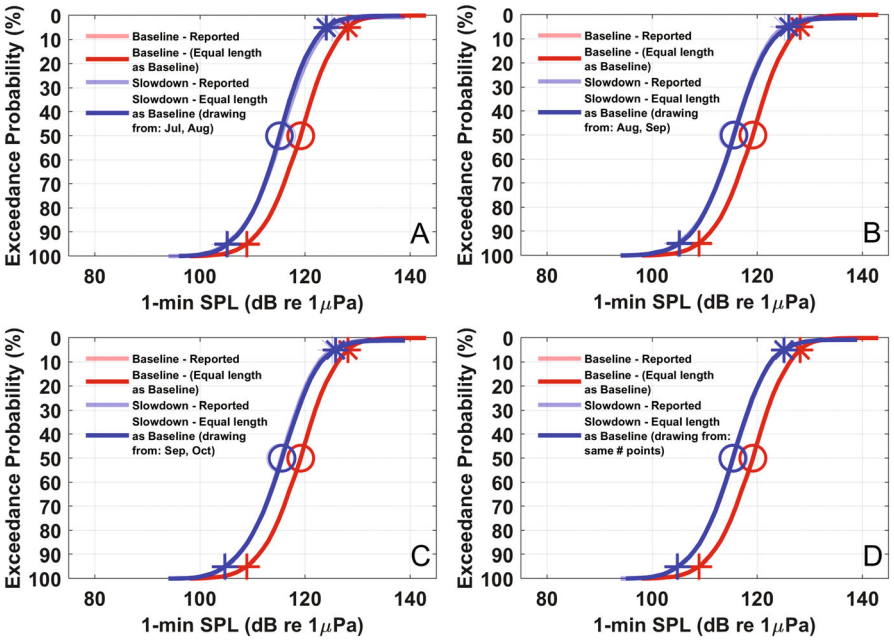



**Fig. 8** CDF plot when all filtration criteria listed in section "Methods" are applied, but the slowdown period is shortened to equal that of the baseline period, whereby in (**a**) the slowdown period is reduced to a 2-month duration from July and August, resulting in a change in $\Delta L_{50}$ value of 0.34 dB; in (**b**) the slowdown period is reduced to 2-month duration from August and September, resulting in a change in $\Delta L_{50}$ value of 0.11 dB; in (**c**) the slowdown period is reduced to 2-month duration from September and October, resulting in a change in $\Delta L_{50}$ value of 0.16 dB; and in (**d**) the slowdown period is reduced so that the total number of criteria-passing data points is randomly drawn and equals the number of criteria-passing points from the baseline, resulting in a change in $\Delta L_{50}$ value of 0 dB. Minor differences exist between the 4 sub-selection scenarios. The cumulative distribution function for broadband acoustic data (10 Hz–100 kHz) at Lime Kiln, calculated using ECHO 2021 baseline and slowdown data (bright lines). The original CDF curves presented in Fig. 4 (as reported by the ECHO Program) are shown directly underneath in muted tones. $L_{50}$ values are circled, $L_5$ values are denoted with an asterisk, and $L_{95}$ values are shown with a plus sign

only ~7.2 h per month of data feeding into the analysis. The differences in the $\Delta L_{50}$ values changed by only <0.3 dB for either case (Fig. 9). However, the non-smooth pattern of the cumulative distribution function when only 2% of criteria-passing data are included suggests that this CDF plot has been created with too few data.

Note that the limit on minimum amount of acoustic data that can be used for evaluating a noise mitigation measure depends on how much the filtration criteria have already subsampled the data and by how representative those acoustic data are of that period. The purpose of showing this is *not* to demonstrate that only a few days of data are sufficient to draw from to evaluate a noise mitigation measure. Instead, the purpose here is to show that it is acceptable to not use all available data and to show that it is fine to be strict with exclusion criteria.

A final way to explore the limits of how much data is needed between treatment (slowdown) and control (baseline) conditions is to reduce the number of consecutive

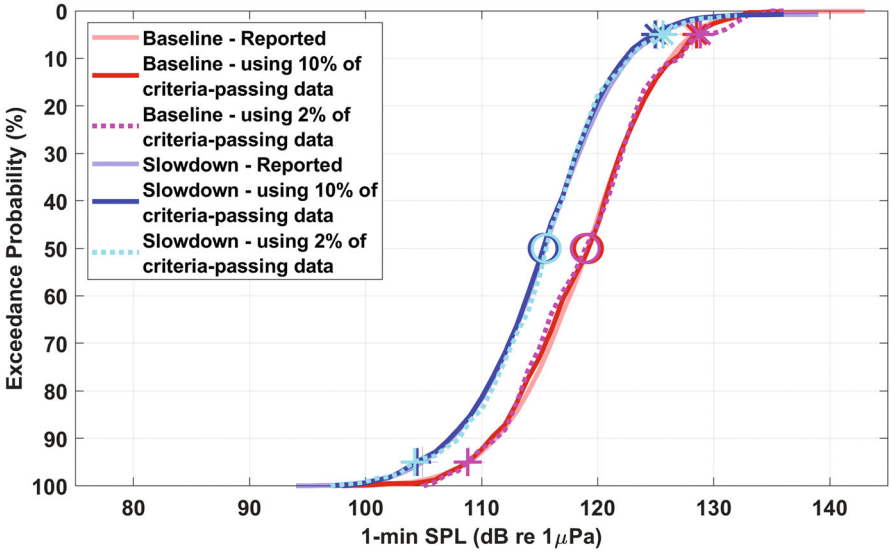



**Fig. 9** CDF plot when all filtration criteria listed in section "Methods" are applied, but the amount of acoustic data has been further reduced to either 10% (solid lines) or 2% (dashed lines) of the criteria-passing data available for both baseline and slowdown periods. The cumulative distribution function for broadband acoustic data (10 Hz–100 kHz) at Lime Kiln was calculated using ECHO 2021 baseline and slowdown data (bright lines, solid, and dashed). The original CDF curves presented in Fig. 4 (as reported by the ECHO Program) are shown directly underneath in muted tones. $L_{50}$ values are circled, $L_5$ values are denoted with an asterisk, and $L_{95}$ values are shown with a plus sign

days of available data from which to draw from can be modified. Below, another way to visualize how much data is enough – from which to draw conclusions on the effectiveness of a noise mitigation effort – is shown. Specifically, the difference in the $\Delta L_{50}$ values that would arise, depending on how much (or how little) consecutive data feeds into the analysis, is shown.

Figure 10 shows that only drawing from the first 10 consecutive days from the 60-day baseline of the ECHO 2021 dataset (May 3–Jul 1, 2021) produces a broadband $\Delta L_{50}$ result that was ~0.8 dB different than if drawing from the full 60 days of available baseline data. This dB difference diminishes with more consecutive days of available baseline. For example, when ~35 consecutive days of baseline are fed into the analysis, the difference in the broadband $\Delta L_{50}$ (of drawing from 35 consecutive days vs. 60 consecutive days) drops to ~0.3 dB.

The green lines on Fig. 10 shows a similar exercise, but here, data were only drawn from the baseline month of June (instead of a baseline of 2 months, from both May and June, as before). Here, it can be seen that only drawing from the first 10 consecutive days of a shorter, 30-day baseline period produces a broadband $\Delta L_{50}$ value that is now 1.2 dB different than if drawing from a 60-day baseline.

Such simulations are helpful when considering and informing what a sufficient amount of data would be for a given site. So, it would be possible to use only a

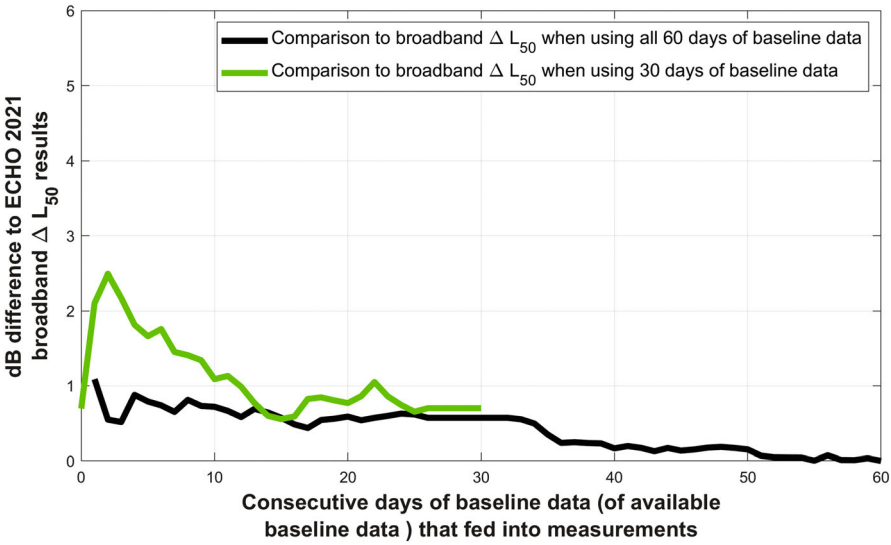



**Fig. 10** Exploration into how many consecutive days of baseline is necessary, via comparison to the broadband $\Delta L_{50}$ results reported for ECHO 2021. The difference in the $\Delta L_{50}$ value for broadband noise (10 Hz–100 kHz) that arose when comparing the baseline period with the slowdown period is shown as a function of a sub-selection on consecutive baseline days, from the entire 60-day baseline period (black, May and June) and from a reduced baseline period of only 30 days (green, only June). Differences in results remain low even with minimal consecutive day sub-selection. The first day is May 3, 2021 (i.e., the first 10 consecutive days of the baseline period correspond to May 3–13, 2021)

30-day baseline (instead of a 60-day baseline) alongside the 122-day slowdown to evaluate the difference in noise levels during the two periods; however, unless the additional baseline period exists, it is impossible to quantify the degree to which excluding that period impacts the resulting broadband $\Delta L_{50}$. In other words, more data are better, as the effect that a reduced-duration baseline period will have on the overall broadband $\Delta L_{50}$ result (showing the difference in broadband noise levels in baseline and slowdown conditions) would be unknown.

## Discussion

Using the ECHO Program's 2021 dataset, an overview has been provided here of the value and weight of different filtration metrics for including and excluding acoustic data in evaluations of noise mitigation evaluations. At Lime Kiln, AIS data were used to determine the range of vessels potentially participating in the slowdown to the acoustic recording station, the presence of small boats was automatically acoustically detected, and wind speed and current speed were measured near the hydrophone. Together, these confounding factors were monitored and filtered out to assess the noise mitigation effectiveness of the ECHO slowdown initiative. It is hoped that these considerations of removing periods with confounding noisy and quiet periods

are adopted by others trying to quantify the impact of a vessel slowdown period, or of any other noise mitigation initiative.

However, different locations have different soundscapes. Appropriate thresholds for different and/or additional variables (e.g., rainfall, sea state, etc.) that have a correlation with noise levels will need to be explored and defined for each site where acoustic recordings are taken. Nevertheless, understanding the variation in ambient noise levels at a given site is often lacking, and this can limit the effectiveness of evaluating noise mitigation initiatives (Vagle et al. 2021). The soundscapes at different sites can vary due to differences in sound speed profiles, bathymetry, bottom substrates, and sea surface conditions, for example (e.g., Eickmeier et al. 2021; Vagle et al. 2021). These variations may also be impacted seasonally (e.g., changes in sound speed profiles). At Lime Kiln, monthly CTD casts are taken to help inform the appropriate selection of baseline periods according to when the slowdown is in effect, as seasonal oceanographic processes in the Salish Sea can change the sound speed profile and therefore the recorded soundscape. Specifically, in Haro Strait, where the slowdown is in effect during the summer months and into the autumn, selecting a baseline period when the CTD data show the presence of surface ducts is avoided, as they are characteristic of winter conditions at this site. In other words, efforts are made to select a baseline period with a similar measured sound speed profile as that of the slowdown period. For the "quiet sound" vessel slowdown initiative into the Port of Seattle, the slowdown occurs during the late autumn and into the winter months, and so here too a period with a similar sound speed profile, as confirmed with CTD casts, should be selected as the baseline period.

Another variation in soundscape may arise based on how confined a body of water is. Both Lime Kiln in Haro Strait and Boundary Pass are quite constrained areas (Fig. 1) where measured soundscapes are less influenced by distant vessels. In a constrained area, more acoustic shadowing could be present (e.g., Cominelli et al. 2020). Furthermore, in a narrower channel or more confined body of water, it is more likely that vessel transits occur at similar ranges to a stationary hydrophone or at least that there is low variability in the range at the closest point of approach. If the soundscape is measured from a wider channel or an open ocean site, on the other hand, more variation in the range and the closest point of approach between passing vessels and a stationary hydrophone may be expected. However, this does not necessarily translate to more variation in received SPLs due to the logarithmic nature of transmission loss. It is therefore recognized that in open waters, data filtration methods suggested here might need further adaptation.

It is important to also note that even the covariates that are correlated with noise levels may vary from site to site. For example, colleagues applying a similar approach to data collected at a different site noted differences in noise levels depending on whether the wind was moving with the tide or current, versus when wind was moving against the tide or current (R. Burnham, pers. comm.). On a similar vein, the "dB cost" of forgetting to consider a covariate that has a correlation with underwater noise levels will vary from site to site.

It is also important that environmental sensors (measuring wind, current, etc.) are located as close as possible to the acoustic recording station (Eickmeier et al. 2021), and the temporal resolution of your environmental measurements needs to closely

match the temporal resolution of the acoustic analysis. If not, the available data could be filtered, but not in a way that necessarily serves the analysis, and more akin to a random sub-selection. For example, at Lime Kiln, wind data were sampled every minute near the hydrophone, current data were sampled at 10 Hz next to the hydrophone and averaged across 1-minute intervals, and acoustic data were sampled at 250 kHz with SPLs averaged across 1-minute intervals.

The soundscape at a given site can also vary with seasonal prevalence of biological noise. For example, cumulative distribution function plots created with the same acoustic data at the same site vary depending on whether periods with detections of animal sounds are included or excluded (Eickmeier et al. 2021). Therefore, keeping track of the presence of sonorous species is advised. Aural review of recordings, or a sub-selection of recordings, is also advised as this can pick up unexpected sounds (e.g., the repeated clinking of a chain anchoring equipment is readily aurally detected, and such instances should be excluded from noise mitigation evaluations). This and other potential confounds can be identified by appropriate plotting and quality assurance steps with the data; for the acoustic data, this could include reviewing daily/weekly/monthly time series of SPLs in different frequency bands, long-term spectral averages, power spectral density plots, and overviews of third octave levels for a given time period, all prior to any noise mitigation evaluation. The importance of this process cannot be over stressed, especially as such a review of the data can identify unexpected sound sources or draw attention to equipment deficiencies, etc.

In conclusion, comparative acoustic analyses to evaluate the effectiveness of a noise mitigation measure should be aware of different factors that may be correlated to noise levels but be unrelated to the noise mitigation measure. These covariates should be measured in similar temporal and spatial scales as the acoustic data, and how much they matter may vary from site to site and from year to year. Such analyses should consider filtering out these confounding noise covariates, and, as the above explorations with the ECHO 2021 dataset show, it both is acceptable to not use all available data and fine to be strict with exclusion criteria. Analysts should be mindful of the sound speed profiles of treatment and control conditions and appropriately select a baseline during a time where the sound speed profiles resemble one another. One should also be mindful that soundscapes can change over small regional scales and that interannual variability in the contributions of different noise confounds does occur. While the approaches suggested here may need further adaptation in different environments, it is hoped that the considerations presented here help to guide future comparisons evaluating the effectiveness of noise-related conservation measures.

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# Acoustic Measurements During a Seismic Airgun Survey: The Case of the Cascadia Subduction Zone

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Kaustubha Raghukumar, Katherine Heal, Frank Spada, Grace Chang, Sarah K. Henkel, Taylor K. Chapple, and Scott A. Heppell

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**Abstract**

Results are presented from a recently concluded field effort to understand the effect of acoustic particle motion levels from a geophysical seismic survey on rockfish, lingcod, and Dungeness crab located within the Redfish Rocks Marine Reserve near Port Orford, Oregon. The study was conducted in summer 2021 during a

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40-day geophysical seismic survey along the Cascadia Subduction Zone offshore of Oregon and Washington. The study collected measurements of acoustic pressure and particle motion, accompanied by in situ tagging of the aforementioned species. Presented in this chapter are a characterization of the acoustic fields that can potentially drive observed behavioral responses of the tagged species.

The noise floor during the study period was strongly dominated by wind noise, with occasional contributions from small boat traffic. During the seismic surveys, received peak pressure levels were found to exceed 160 dB re 1 μPa (corresponding to peak measured particle velocity levels of 95 dB re 1 nm/s) when the seismic sound source was at a distance of 30 km from the vector sensor array. In contrast, band-averaged ambient pressure levels were in the 100–120 dB re 1 μPa range (corresponding to particle velocity levels that were in the 60 dB re 1 nm/s range).

**Keywords**

Particle motion · Seismic survey · Vector sensor · Fishes

## Introduction

Fishes and many invertebrates use sound, primarily particle motion, not only to acquire information about their environment (i.e., an approaching predator or the presence of a potential mate), but also for communication or navigation. Popper and Hawkins (2018), in their review of the importance of particle motion to fishes and invertebrates, outline the long history of research in this field and point out that regulatory action has emphasized sound pressure (Popper et al. 2014), while disregarding the more important component for fishes and invertebrates – particle motion. They also note that fishes and invertebrates are not solely exposed to particle motion in water; these stimuli may be transmitted through the substrate and at the interface between the water and the substrate (Popper and Hawkins 2018).

Understanding the effects of particle motion on fishes and invertebrates is relevant to regulatory concerns. Adverse effects such as increased mortality, injury (and reduced survivorship or sensory capabilities), and physiological and behavior alterations (including but not exclusively reproductive) are important to understand and to inform the preparation of any environmental impact assessment. Among the issues raised by Popper and Hawkins (2018) requiring further investigation is the need for robust and concurrent measurements of particle motion and the associated behavioral response of fishes and invertebrates.

A seismic survey of the Cascadia Subduction Zone occurred offshore Oregon, Washington, and British Columbia during the summer of 2021 using an array of 36 airguns. This airgun activity presented a unique acoustic source and opportunity to examine the impact of loud anthropogenic sound on the behavior of reef fish and invertebrates. A study was conducted to measure in situ behavioral responses of reef fishes and invertebrates to sound from the seismic airgun survey (Borland et al. 2023). Most studies of the effects of seismic survey sounds in open ocean and shallow-water environments focus on the pressure component of sound (e.g., Cato

1976; Dahl et al. 2007; Martin and Popper 2016; Haxel et al. 2019), but few focus on the associated acoustic particle motion to which many animals have sensitivity (Nedelec et al. 2016; Popper and Hawkins 2018). Thus, the first objective of the study was to simultaneously measure the particle motion and pressure components of the seismic survey activity including the signals from airgun explosions and ensuing reverberation between pulses (Guerra et al. 2011). The second objective was to measure behavioral responses of rockfishes (genus Sebastes), lingcod, and Dungeness crab to these seismic pulses using acoustic telemetry. Described in this chapter are the acoustic measurements made on an acoustic vector sensor array (VSA), the NoiseSpotter® (Raghukumar et al. 2020), while an accompanying chapter ("Responses of Oregon demersal species to seismic survey noise: evaluating behavior and movement," Borland et al. 2013) describes the behavioral responses of rockfishes, lingcod, and Dungeness crabs to the seismic pulses.

## Methods

The R/V Langseth conducted a seismic airgun survey of the Cascadia subduction zone from May 31 to July 11, 2021. The survey commenced near Seattle, proceeded southward until near the Oregon-California border on June 17, and then continued northward until offshore of British Columbia on July 8 before concluding the survey on July 11, 2021 (Fig. 1).

Measurements of acoustic pressure and particle velocity before, during, and after the survey were made using an acoustic vector sensor array which was deployed within the Redfish Rocks Marine Reserve (RRMR) on May 15, 2021, and recovered on July 21, 2021. Two equipment turnarounds occurred within this deployment period.

The vector sensor array, NoiseSpotter® (Fig. 2), was deployed in approximately 36 m deep water using three sensors mounted 35 cm, 50 cm, and 75 cm above the seabed. Sensors are separated by 1 m in the horizontal. Particle motion measurements are comprised of three-dimensional particle velocity vectors (u, v, w), measured in a sensor reference frame. The sensor at 50 cm contains an integrated inertial motion unit (IMU), whose measurements are used to translate the particle velocity vectors into a true earth-referenced frame. The frequency sensitivity of the vector sensors spans the 50 Hz–3 kHz range. Omnidirectional pressure measurements have a flat frequency response, while the particle velocity measurements have a peak sensitivity at 1 kHz. The 50 Hz-3 kHz frequency range is suited to measurement of anthropogenic sounds such as those from seismic surveys and ships (frequency range 0–500 Hz), and biological sounds such as those from fishes, invertebrates, and baleen whales whose hearing sensitivities fall within this frequency range (Nedwell 2004). The vector sensors are capable of measuring sound pressure levels up to 240 dB re 1 μPa at a distance of 100 m from the sensor.

Strong hydrodynamic flow from waves and currents can induce nonacoustic pressure fluctuations that lead to contamination of acoustic signals. Flow noise contamination can be particularly acute with vector sensors due to saturation of the built-in accelerometer signal by energetic flows. To facilitate deployments in

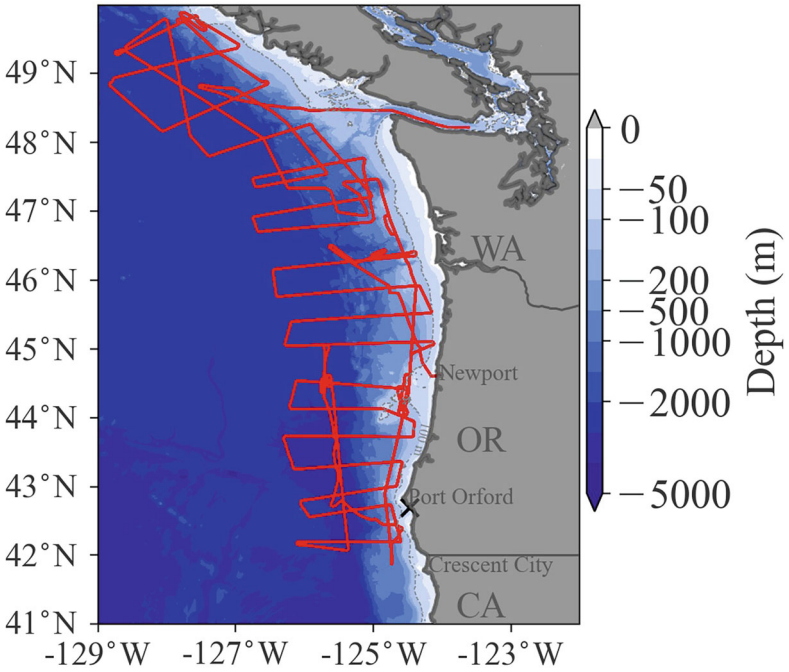



**Fig. 1** Map showing R/V Langseth transects and study location (black cross near Port Orford). Bathymetric data from https://www.gebco.net/

energetic environments, the VSA included flow noise suppression shields around each vector sensor. The flow noise suppression shields have been demonstrated to be effective in reducing flow noise effects, thereby improving data quality and signal detectability (Raghukumar et al. 2019).

## Results

For the purposes of analysis, the three deployment periods are categorized as "pre-survey" (May 18–29), "during" (June 8–16), and "post-survey" (July 3–11, at which point the R/V Langseth was over 500 km away from RRMR). Cumulative Sound Exposure Levels (cSELs, Southall et al. 2007) were computed on hour-long data segments for sound pressure measured at the sensor 70 cm above the seabed (Fig. 3). Also shown in Fig. 3 are wind speeds measured during the deployment periods, measured at the National Data Buoy Center Station PORO3 meteorological buoy, located roughly 3 km north of RRMR. During all three analysis periods, cSELs are seen to be strongly influenced by wind speeds, with a general increase in cSELs during the postsurvey period when wind speeds were highest. A diel modulation of cSELs due to diel variability in wind speeds is observed, most prominently in the

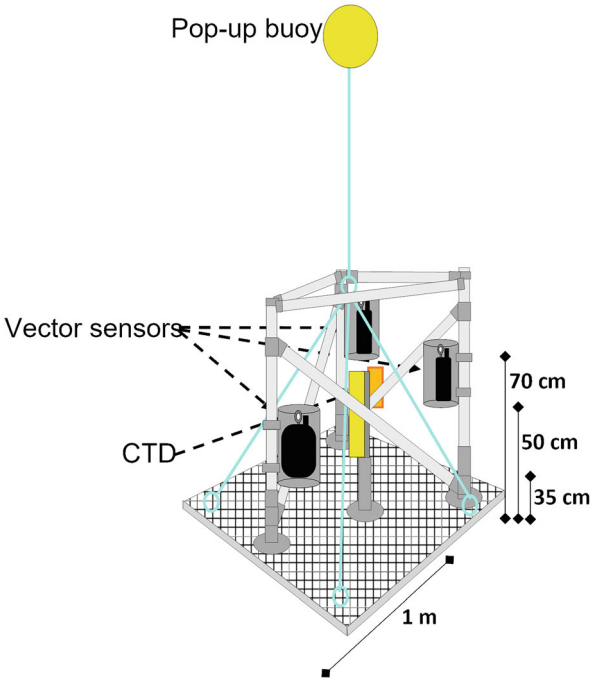



**Fig. 2** NoiseSpotter® vector sensor array consisting of three vector sensors and a conductivity-temperature-depth sensor

"before" period. cSELs are highest (approximately 145 dB re 1 $\mu Pa^2s$) on June 11, at which time the R/V Langseth was at its second closest point of approach (30 km). Wind speeds were also highest on June 11, suggesting that the high cSELs measured are a combination of sound from seismic airgun pulses and high wind-generated ambient noise.

A long-term spectral average (Fig. 4) of the acoustic pressure shows periods of elevated acoustic energy between 0 and 500 Hz associated with the seismic surveys between June 11 and 13, 2021. Similar to Fig. 3, a temporal banding of acoustic energy associated with diel changes in wind speeds is seen, with an increase in acoustic energy levels observed toward the end of May, 2021, likely due to increased wind speeds. Occasional boat passages (short-duration broadband spectral lines) are seen during all three data periods, with the most frequent boat passages occurring during the presurvey period (upper panel, Fig. 4). The postsurvey period shows higher acoustic energy across the 0–1000 Hz frequency band considered, with the exception of a dip in acoustic energy on July 7, 2021, that coincides with a drop in wind speeds during that period.

The contribution of seismic airgun pulses to the particle velocity soundscape is discerned in statistics of the frequency spectra (Fig. 5, x-component of particle velocity) for each of the deployment periods. The mean, maximum, and 99-percentile curves show an elevation in particle velocity levels due to the presence of seismic airgun pulses during these periods. In contrast, the remaining percentile

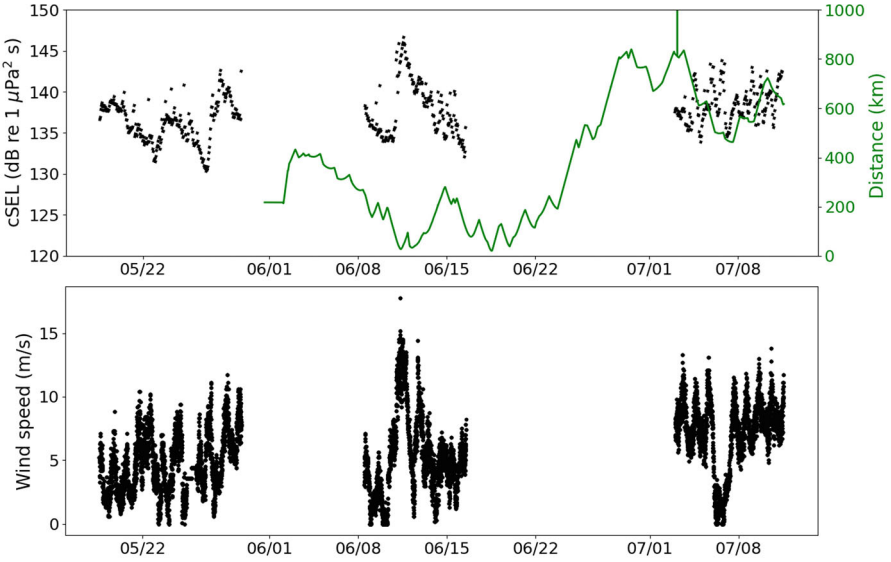



**Fig. 3** (Top) Cumulative Sound Exposure Level (cSEL) over the experiment duration. Also shown is the distance between the R/V Langseth and the NoiseSpotter®. (Bottom) Wind speeds over the experiment period, measured at the Port Orford meteorological buoy (https://www.ndbc.noaa.gov/station_page.php?station=poro3)

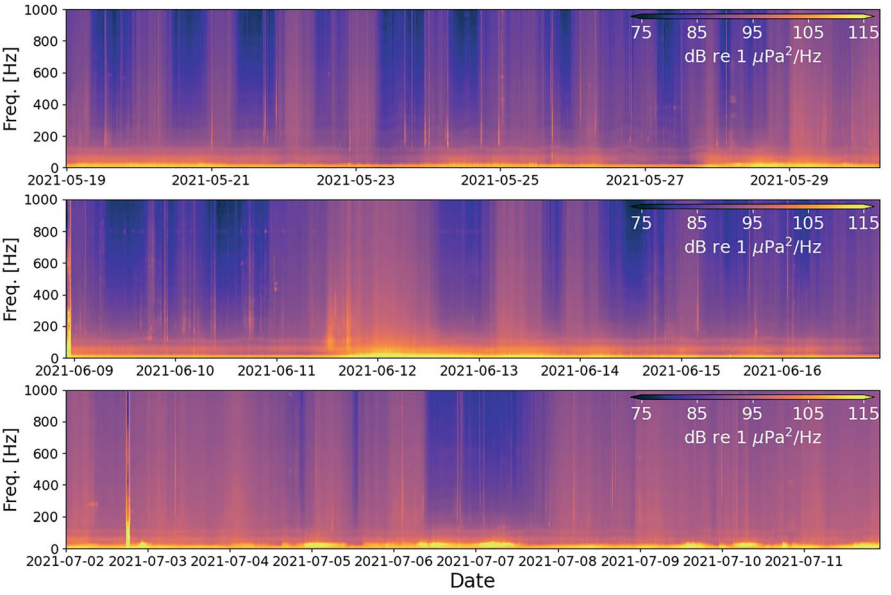



**Fig. 4** Long-term spectral average of acoustic pressure measured 70 cm above the seabed before (top), during (middle), and after (bottom) the seismic airgun survey

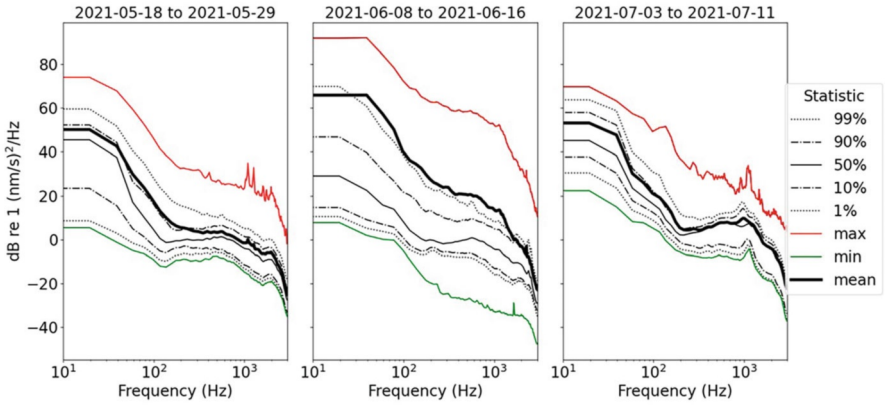



**Fig. 5** Power spectral densities for x-particle velocity before (left), during (middle), and after (right) the seismic airgun survey

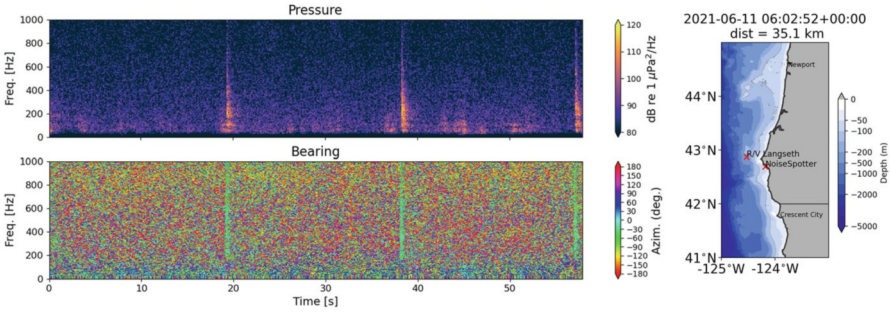



**Fig. 6** (Left) Example of acoustic pressure and azimuthal bearing estimated from particle velocity. (Right) Locations of R/V Langseth and NoiseSpotter® for data period shown above. Bathymetric data from https://www.gebco.net/

curves (minimum, 1%, 10%, 50%, and 90%) show the highest levels during the postsurvey period, likely due to sustained higher wind speeds during this period (Fig. 1).

An example of directional processing applied to the particle velocity data is shown in Fig. 6, for a minute-long data segment during which seismic airgun pulses were measured at a distance of 35.1 km from the R/V Langseth. The true bearing of the R/V Langseth relative to the NoiseSpotter® was −55° with respect to true north (i.e., to the northwest). Frequency-specific bearings were calculated to be in the range of −55–50° with respect to true north for the primary arrivals associated with the seismic pulses. While the pressure data show additional arrivals associated with multipath propagation, the signal-to-noise ratio was insufficient to sufficiently resolve the associated bearings for these multipath arrivals.

## Conclusion

In this chapter, acoustic particle motion was measured before, during, and after a seismic airgun survey of the Cascadia Subduction Zone. Measurements were made inside the Redfish Rocks Marine Reserve in southern Oregon and were accompanied by in situ tagging of rockfish, lingcod, and Dungeness crabs. The acoustic environment was found to be dominated by wind noise, with occasional contributions from boats and seismic airgun pulses from the R/V Langseth during the seismic surveys that exceeded 160 dB re 1 μPa (pressure) or 95 dB re 1 nm/s (particle velocity). The ambient noise floor (100–120 dB re 1 μPa range for pressure, 60 dB re 1 nm/s range for particle velocity) appeared to scale with wind speeds, with the highest ambient noise levels occurring during the postsurvey period. Episodic contributions from the seismic airgun pulses were most evident in the mean, maximum, and 99 percentile levels. The highest measured wind speeds during the study period coincided with the closest point of approach of the R/V Langseth; thus, it is likely that the high cumulative sound exposure levels during this period contain contributions from both wind noise and the airgun pulses. It will therefore be necessary in subsequent analyses to separate the contribution to the acoustic environment from each source. Efforts are ongoing to implement automated detectors for a better quantitative characterization of sound levels from the seismic pulses to inform a more granular understanding of behavioral responses to acute periods of elevated sound levels.

The extraction of periods when seismic pulses were present can also likely be conducted using directional processing of the particle velocity data. A good agreement of estimated azimuthal bearing relative to true location of the R/V Langseth lends confidence that a directional masking of data can shed additional light on sound pressure and particle velocity levels during the seismic surveys.

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# Alignment and Incentivization of Underwater Noise Notations for Quiet Ships

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Krista B. Trounce, Michael A. Ainslie, David E. Hannay, and Justin M. Eickmeier

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**Abstract**

The approaches for measurement and analysis of underwater vessel radiated noise are complex and evolving. As underwater noise gains increased attention internationally, ship owners and operators are faced with the challenge of understanding the noise emissions of their fleet. An owner seeking to determine if an existing or future vessel design may qualify for a quiet notation must carefully evaluate the available certification options and consider the cost. Differences in terminology,

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methodology, and approach from international ship classification societies, coupled with the lack of a standardized approach for vessel measurements in shallow water, can make certification daunting to the owner. Recognizing this challenge, the Enhancing Cetacean Habitat and Observation (ECHO) Program led by the Vancouver Fraser Port Authority, supported by Transport Canada, initiated a project to work with ship classification societies and technical experts toward the alignment of quiet ship notations, through a series of three annual workshops. In 2017, Vancouver became the first port in the world to offer financial incentives to ships with quiet notations, and through the alignment project, strives to increase the number of ships achieving notations and being rewarded for this practice – at the Port of Vancouver and around the world.

**Keywords**

Quiet ship · Metrics · Source levels · Classification society · Underwater radiated noise

## Introduction

The Enhancing Cetacean Habitat and Observation (ECHO) Program, led by the Vancouver Fraser Port Authority, is a research and management program that seeks to better understand and reduce the cumulative effects of shipping, particularly ship-generated underwater noise, on endangered whales. The Southern Resident Killer Whale (SRKW), with a population of 73 individuals, is a key focus of the program. The ECHO Program coordinates seasonal underwater noise reduction initiatives, including vessel slowdowns and route alterations, while also spearheading research and educational efforts on underwater noise. Additionally, the ECHO Program works internationally to promote cleaner and quieter commercial ships.

With a goal to assist port customers in understanding and achieving certification as a quiet vessel, the ECHO Program, supported by Transport Canada, initiated a project for improved alignment of quiet ship notations from classification societies.

The objective of the project is to improve the alignment of measurement, analysis, and reporting techniques among the various class societies' quiet ship notations. Such alignment would enable customers to compare the notations to each other and determine which may be the most appropriate quiet ship certification for their vessel. A key requirement of the recommended procedures is to identify an analytical method and reporting metric that would result in the same underwater radiated noise (URN) result for a given ship measured under the same operating conditions anywhere in the world. Notably, there is a requirement for appropriate formulae to convert the sound pressure level (SPL) received at the hydrophone to a URN level for the vessel that provides consistent results regardless of water depth, as there is currently no international standard for vessel

measurement in shallow water (Ainslie et al. 2022). The end goal is to increase the number of vessels achieving quiet notations and thus receiving incentives at the Port of Vancouver and internationally. In 2023, the Port of Vancouver offers vessels with a quiet ship notation from a ship classification society a platinum level (75% rate reduction) on harbor dues. To this end, the Port of Vancouver and other ports need to know that these quiet notations recognize those customers who are going above and beyond to support quieter oceans.

## Project Methodology

To achieve the project objective to improve the alignment of measurement, analysis, and reporting techniques between the various class societies, JASCO Applied Sciences Ltd., a project partner on vessel source level measurement since 2015, was retained to work with the ECHO Program on this project which included the following steps:

- Comparison of the ECHO Program vessel source level measurement database to existing quiet notations, noting the challenges in doing so given the different measurement and analysis methods.
- Preparation of a draft document proposing options for alignment.
- Engaging with the class societies on review and discussion of the draft document through in-person or online workshops.
- Repeating steps 2 and 3, refining the document based on input received.
- Potential amendments to existing quiet ship notations to reflect the results of this work in 2023.

Vessel noise measurement performed by the ECHO Program began in 2015 and work on the project (step 1 above) began in 2018. The first draft document (step 2) was distributed to participants in August 2020, followed by the first workshop in October 2020 (step 3). Based on information received from the 2020 workshop, the document was refined and re-distributed in 2021 and again in 2022. The third and final workshop was held in October 2022.

The final *Recommended Procedures for Measuring Underwater Radiated Noise Emissions of Ships, for Quiet Ship Certification*, was published in 2023 (Hannay et al. 2023).

## Quiet Ship Notations

Ship classification societies provides services to the maritime industry and regulatory bodies, using technical expertise and experience to support the development of standards and regulations, with a focus on maritime safety and pollution prevention. Classification societies develop their own rules that pertain to design, construction, maintenance, and survey of essential parts of ships and other marine

facilities. Through the development and application of their own rules, and by verifying compliance with international and/or national statutory regulations, classification societies play a key role in self-regulation within the maritime industry (IACS 2022).

Although there are over 50 organizations worldwide offering maritime classification services, there are currently only seven classification societies presently offering notations for quiet ships. These include:

- American Bureau of Shipping (ABS): Guide for the Classification Notation: Underwater Noise and External Airborne Noise (2021).
- Bureau Veritas (BV): Underwater Radiated Noise (URN). Rule Note NR 614 DT R02 E (2018).
- China Classification Society (CCS): Guidelines for Underwater Radiated Noise of Ships 2018. Release date 2019-12-26 (2018).
- Det Norske Veritas (DNV): Class Guideline. Measurement procedures for noise emission. DNVGL-CG-0313 (2021b). Rules for Classification. Ships. Part 6 Chapter 7. Environmental protection and pollution control (2021a).
- Korean Register (KR): Guidance for Underwater Radiated Noise (2021).
- Lloyd's Register (LR): ShipRight Design and Construction: Additional Design Procedures. Additional Design and Construction Procedure for the Determination of a Vessel's Underwater Radiated Noise (2018).
- Registro Italiano Navale (RINA): Rules for the Classification of Ships: Part F, Chapter 13, Section 25. Dolphin Quiet Ship and Dolphin Transit Ship (2021).

All the notations are well designed and generally follow established guidance from the International Organization for Standardization (ISO 2016, 2019) or the American National Standards Institute and Acoustical Society of America (ANSI and ASA 2009). The classification societies, however, utilize differing measurement and analysis procedures and, most importantly, differing reporting metrics for each quiet ship notation.

As the Vancouver Fraser Port Authority began the process to incentivize and reward customers whose ships are certified with a quiet notation, it was important to evaluate how the quiet notations compared to the vessel noise emissions collected while on approach to the Port of Vancouver. Unfortunately, the lack of harmonization among notations precluded direct comparison of notations to each other or to the vessel source level measurements collected by systems deployed on behalf of the ECHO Program and Transport Canada (Hannay 2018; Hannay et al. 2019). A key example of this challenge is that five of the seven classification societies' procedures are based on the radiated noise level (RNL) metric (ISO 2019), whereas two are based on source level (SL) (ISO 2019).

Attempting to undertake the first step of the project, the comparison of quiet ship notations to existing vessel measurements highlighted the importance of proceeding with subsequent steps of the project to align the measurement, analysis, and reporting of these quiet ship notations.

## Drafting Recommended Procedures

The process of drafting recommended procedures began with the identification of the four main aspects of each quiet ship notation:

1. Preparation and measurement – acquiring the raw data. This aspect includes considerations for site selection, oceanographic and weather conditions, deployment configuration of appropriately calibrated equipment, and definition of the measurement path(s).
2. Initial processing – calculating sound pressure levels (SPL) from raw data. This aspect must consider how the SPLs are calculated in specified frequency bands over specified time intervals, including corrections for high background noise.
3. Post-processing – converting sound pressure levels to an underwater radiated noise (URN) metric. The post-processing step is where the conversion factors and consideration of metrics (RNL or SL) come into play, and where the differences among the class society approaches pose the greatest challenge for alignment. The conversion of measured SPL in the data window to URN includes applying distance correction factors and averaging over multiple hydrophones and transits.
4. Thresholding for certification – comparing the calculated URN against previously established URN levels for certification as a quiet ship. This aspect must also consider the magnitude and number of exceedances (if any) of the threshold that may be considered acceptable.

The first draft technical document with proposed procedures for alignment, under the aspects listed above, was distributed to the project stakeholders in August of 2020. At the time of that document, five ship classification societies: ABS, Bureau Veritas, DNV, Lloyd's Register, and RINA had quiet ship notations known and available for review by the project team. The proposed procedures were shared with representatives from the five listed classification societies, as well as with representatives from the International Association of Classification Societies (IACS), and technical experts from the ISO working group on the development of requirements for measurements in shallow water for review and input.

## Stakeholder Engagement

The ECHO Program is built on a collaborative and successful model that brings interested and impacted parties together to advance initiatives. The project to advance alignment of the quiet ship notations followed this collaborative model, ensuring that the affected parties, the classification societies, played an important role in crafting the recommended guidelines. Another hallmark of the ECHO Program is ensuring that decisions and recommendations are based in science. To this end, the project team sought to include emerging research and technical experts in the drafting of procedures.

As noted above, the engagement process included hosting three annual workshops (2020, 2021, 2022) with representatives from the classification societies and technical experts. Prior to each workshop, a technical memorandum with proposed alignment parameters was distributed to the participants for review and feedback. Each workshop sought to refine the proposed parameters based on the input provided by the participants and discussed at each workshop.

The three-year timeframe of the project also allowed for incorporation of emerging research pertaining to the measurement of vessels in shallow water, and the opportunity to share this information with the project stakeholders, who may otherwise be unaware. Of particular relevance is the project sponsored by Transport Canada *Towards a Standard for Vessel URN Measurement in Shallow Water* (MacGillivray et al. 2022, 2023) which investigated several metrics for potential use in shallow water. Sharing the learnings from this project with the international classification society representatives allowed participants to fully understand and seek clarification on data and methodology underlying the recommended metric.

Feedback from the project stakeholders received through review of the technical memorandum and discussions at the annual workshops were integrated into a final document *Recommended Procedures for Measuring Underwater Radiated Noise Emissions of Ships, for Quiet Ship Certification* (Hannay et al. 2023).

## Recommended Procedures

The *Recommended Procedures for Measuring Underwater Radiated Noise Emissions of Ships, for Quiet Ship Certification* (Hannay et al. 2023) is structured to provide clear and concise guidance for classification societies (or others) seeking to accurately measure and report on URN measurements in varying water depths, locations, and oceanographic conditions. The document provides an introduction, background, and discussion of existing URN metrics, followed by detailed recommendations for the following:

- Follow the acoustical terminology of ISO (2017)
- Site selection and water depth
- Limiting weather conditions
- Consideration of tides and currents
- Selection of instruments (in particular, hydrophones and calibration)
- Measurement geometry (distance from ship track, hydrophone depths and measurement angles, number of transits)
- Ship speed and draught
- Initial data processing (data window period, SPL calculation time windows, frequency bands, correction for ambient noise)
- Calculating the recommended URN metric (using an acoustic model or the recommended seabed critical angle (SCA) method derived from MacGillivray et al. (2022, 2023))

The final step of a quiet ship notation entails specifying the decidecade frequency band level limits or thresholds, against which the measured URN values are compared for certification. A decidecade is approximately equal to one-third of an octave. This step is intentionally not included in the recommended procedures, but statistical information (percentiles) of URN vessel measurements collected by the ECHO Program were provided to the project partners. It has been left to the discretion of each classification society to select appropriate limits for their certification.

## Incentives

For nearly 15 years, the Vancouver Fraser Port Authority has been rewarding ships operating in more environmentally responsible ways in and around the Port of Vancouver. By taking voluntary actions that go above and beyond environmental regulations, shipping companies can qualify for up to 75% off their harbor dues fees through the EcoAction program. In 2017, Vancouver became the first port in the world to offer financial incentives to ships for reducing underwater noise. The EcoAction program provides financial incentives for positive environmental practices including the use of renewable energy or cleaner/alternative fuels, maintaining vessel technologies that reduce underwater noise, or participating in voluntary ship environmental certification programs.

In 2023, the Port of Vancouver is offering vessels with a quiet ship notation from a ship classification society a platinum level (75% rate reduction) on harbor dues. The alignment project strives to help ship owners and operators better understand and compare among the different class society notations, with a goal to increase the number of vessels achieving notations and being rewarded for this practice – at the Port of Vancouver and around the world. To this end, the Port of Vancouver and other ports need to know that these quiet notations are comparable, consistent, and recognize those customers who are going above and beyond to support quieter oceans.

## Summary

The ECHO Program and the Government of Canada continually work to reduce vessel underwater noise regionally and nationally, to support the recovery of several endangered marine mammal species in our waters. However, shipping is an international business, and to truly instigate change, work must be progressed on an international scale. The project to align quiet vessel notations from ship classification societies recognizes this need for collaboration from all affected organizations regardless of geographic location. The work advanced on this project provides clear and concise guidance on measurement and analysis, as well as a recommended metric that can be used in differing oceanographic conditions and water depths. The

hope is that this guidance will be adopted by ship classification societies, making quiet ship certification more accessible and appealing to vessel builders, owners, and operators. In turn, prompting port authorities, government agencies, or other organizations to encourage and incentivize ship quieting for the benefit of ocean life around the world.

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# Effect of Vessel Noise on Marine Mammals and Measures to Reduce Impact

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**Abstract**

Underwater radiated noise from shipping is the most widespread and pervasive human noise in marine ecosystems. Mounting evidence indicates that this noise source can cause changes to the behavior and physiology of marine mammals, and that cumulative exposures can have detrimental effects on their fitness and survival. Concern for the impacts of shipping noise to marine species has led regulators to propose measures to reduce underwater radiated noise from shipping either through

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reductions in vessel source levels via technological modifications and operational slowdowns, or through measures to reroute vessels away from important marine habitats. In this overview, we provide a brief discussion of the potential impacts of vessel noise to marine mammals as a group of particular concern, and conclude that technological, operational, and routing measures can be used by the shipping industry independently or synergistically to reduce both global cumulative noise levels from commercial vessels, and potential impacts to marine mammals.

**Keywords**

Anthropogenic noise · Noise mitigation · Shipping · Slowdown · Rerouting

## Introduction

Over the last century, global maritime traffic has steadily increased, with ship numbers growing by a factor of 3.5 and gross tonnage by a factor of 10 between World War II and 2008 (Frisk 2012). Today, motorized vessels transport over 80% of international trade (UNCTAD 2021), navigating the world's oceans following shipping lanes which provide the shortest and fastest pathways between maritime ports (Hsu and Hsieh 2007; Jalkanen et al. 2022). The ubiquitous nature of shipping has resulted in ships becoming the most widespread and pervasive human noise source in marine ecosystems (Hildebrand 2009; Jalkanen et al. 2022); projected to increase with the future growth in shipping activity in accordance with economic demand (Kaplan and Solomon 2016).

Sound is an important acoustic cue used by many marine species for critical ecological functions. Mounting evidence shows that the introduction of human-made underwater noises can have detrimental effects on species (Richardson et al. 1995; Williams et al. 2014; Shannon et al. 2016). Noise from both continuous (e.g., shipping) and impulsive (e.g., pile driving, seismic activity, sonar and acoustic deterrents) sound sources has been suggested to affect individuals behavior and physiology, and if prevalent, these effects may culminate in population-level consequences for species (Richardson et al. 1995; National Research Council 2005; Madsen et al. 2006; Kastelein et al. 2013; Slabbekoorn et al. 2016; Harris et al. 2018; Nabe et al. 2018; Erbe et al. 2019).

Concern for the increasing and widespread effects of ship noise, in particular on marine ecosystems, has led regulators to take measures aimed at reducing underwater radiated noise from shipping (European Commission 2008; IMO 2014; OSPAR 2014; Chou et al. 2021; HELCOM 2021). Three general approaches have been proposed: (1) technological modifications to reduce radiated noise levels (Leaper and Renilson 2012; Gassmann et al. 2017); (2) reducing the noise produced through operational solutions such as vessel slowdowns (Joy et al. 2019; MacGillivray et al. 2019; Burnham et al. 2021); and (3) reducing the received noise by implementing alternative routing measures (Laist et al. 2014; Burnham et al. 2021). In this chapter, we provide an overview of the potential effects of vessel noise to marine mammals as a group of particular concern and discuss proposed approaches for reducing underwater radiated noise from commercial vessels.

## Vessel Noise as a Threat for Marine Mammals

Marine soundscapes are composed of both natural and anthropogenic sources. Natural sources can include sounds from waves, rainfall, and earthquakes, as well as a variety of biological sounds from animals. Anthropogenic noise is generated by human activities ranging from oil and gas exploration and naval operations, to commercial shipping, fishing, recreational boating, and offshore infrastructure (platforms, offshore renewables, etc.) (Hildebrand 2009). Sound travels with little attenuation in the aquatic environment, particularly at lower frequencies (Urick 1983), and many marine species have evolved to exploit these advantageous sound propagation characteristics. Marine mammals in particular rely on acoustic cues for active and passive communication, and as a means of perceiving their environment, using acoustic cues for navigation, biosonar-based foraging, and/or detection and localization of predators (Tyack and Clark 2000). Marine mammals have adapted to carry out these important biological functions in fluctuating natural noise conditions; for example, rain can produce noise as loud as a passing vessel (Wisniewska et al. 2018). However, the rapid growth in human activities at sea has raised concerns as to whether the addition of anthropogenic noise sources to marine environments may negatively affect the auditory scene analysis, behavior, and/or physiology of marine mammals (Richardson et al. 1995; Nowacek et al. 2007; Weilgart 2007; Tyack 2008).

Vessel noise in deep water peaks at low frequencies, which co-occur with the frequencies at which baleen whales (presumably) hear best and produce sound (Fig. 1) (Erbe et al. 2019; Southall et al. 2019). Initial concerns for the effects of shipping noise on marine mammals centered on the potential for low-frequency vessel noise to mask the 20 Hz calls produced by fin whales (*Balaenoptera physalus*) (Payne and Webb 1971). Since this hypothesis, studies on a wide range of baleen whale responses to vessel noise have reported a suite of reactions, including changes in behavior (Sprogis et al. 2020; Arranz et al. 2021), foraging (Blair et al. 2016), physiological stress levels (Rolland et al. 2012), and communication signal characteristics (Dahlheim and Castellote 2016). More recent studies have also reported substantial vessel noise at higher frequencies (Hermannsen et al. 2014; Veirs et al. 2016) (Fig. 1), illustrating that shipping noise is not only audible for, and hence affect, baleen whales, but also for other marine mammal groups, such as pinnipeds, sirenians, and toothed whales (Fig. 1).

Underwater noise from vessels has the potential to mask or alter the communication of pinnipeds (Terhune et al. 1979; Southall et al. 2000; Southall et al. 2003), and affect both their haul-out (Jansen et al. 2010; Andersen et al. 2012; Blundell and Pendleton 2015) and diving behavior (Mikkelsen et al. 2019). Both manatees (*Trichechus* spp.) and dugongs (*Dugong* spp.) also demonstrate the ability to detect and respond to approaching boats by changing orientation, diving behavior and behavioral state (Hodgson and Marsh 2007; Miksis-Olds et al. 2007; Rycyk et al. 2018). In the case of toothed whales, reported responses to vessel noise have included changes in echolocation and communication signal parameters (Kragh et al. 2019), behavior (Nowacek et al. 2001; Lusseau et al. 2009), social interactions (Marley et al. 2017), physiological stress responses (Lyamin et al. 2015), reductions

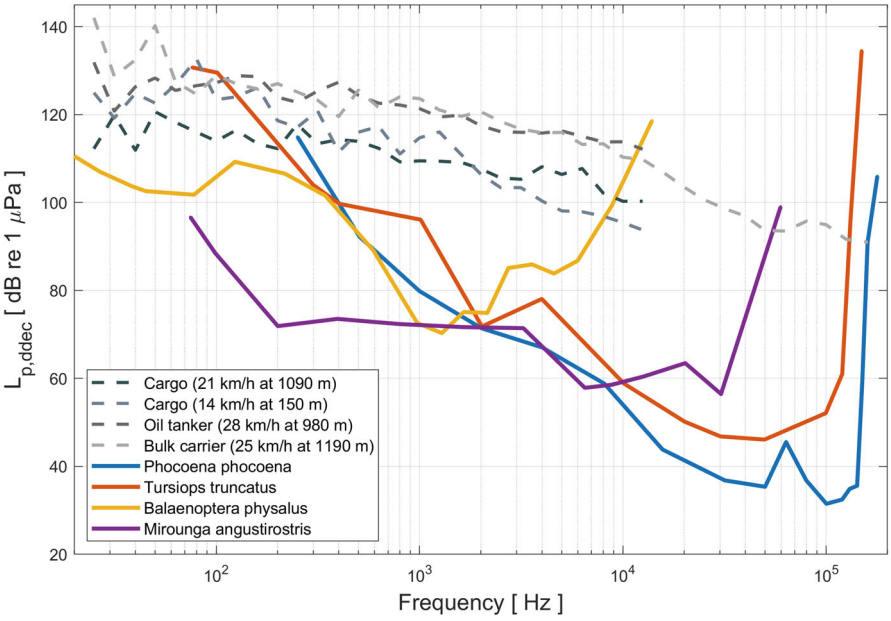



**Fig. 1** Audiogram of different species of marine mammals in relation to four vessel noise recordings (received decidecade levels root-mean-square, $L_{p,ddec}$ RMS). The numbers in brackets in the legend indicate the speed and distance of the vessels to the recording hydrophone. Vessel data taken from Hermannsen et al. (2014). Note that some examples of vessel noise recordings were limited to 20 kHz. Audiograms are presented for harbor porpoise (*Phocoena phocoena*) (Kastelein et al. 2002); bottlenose dolphin (*Tursiops truncatus*) (Johnson 1967); northern elephant seal (*Mirounga angustirostris*) (Kastak and Schusterman 1998); and a predicted audiogram of a fin whale calf (*Balaenoptera physalus*) (Cranford and Krysl 2015)

in foraging effort (Wisniewska et al. 2018), and even habitat displacement (Bejder et al. 2006; Rako et al. 2013). Nonetheless, while numerous studies have focused on these short-term adverse effects, there remains a paucity of quantitative data on how these noise-induced physiological and behavioral changes impact the fitness and survival of the exposed individuals, and in turn also the long-term viability of populations of animals.

While lethal threats from vessel activity, such as ship strikes, can have a direct impact on the survival of marine mammals (Silber et al. 2012; Laist et al. 2014), little is known about the potential for non-lethal threats, such as exposure to vessel noise, to lead to fitness consequences for individuals. In a terrestrial context, chronic exposure to road noise has been shown to reduce the breeding productivity of Eastern Bluebirds (*Sialia sialis*) (Kight et al. 2012). Although the mechanisms responsible for these effects are unclear, the breeding depression experienced by this species indicates that anthropogenic noise have a negative effect on individual fitness either directly, as a physiological stressor, or indirectly, by masking important vocalizations or leading to behavioral changes (Kight et al. 2012). Animals have

evolved to react to generalized threat stimuli, and according to the risk-disturbance hypothesis, perceive human disturbance as non-lethal predation stimuli by exhibiting comparable responses: displaying an aversive response and diverting time and energy from other fitness-enhancing activities such as feeding, parental care, or mating displays (Frid and Dill 2002). Substantial alterations in behavior in response to noise may therefore offset the time- and energy budgets of individuals. Reductions in the time spent foraging, socializing, or resting, or increases in energetic costs through initiation of fleeing responses could lead to an energetic imbalance between energy intake and energy expenditure (Costa and Williams 1999). However, the effect of vessel noise on the energy budget and individual vital rates greatly depends on how often and severely an animal is disrupted, how fast animals return to normal behaviors, and whether individuals are able to compensate for these interruptions (New et al. 2014; Christiansen and Lusseau 2015). Individuals repeatedly disturbed and unable to compensate for these disturbances will suffer a sustained imbalance between energy intake and expenditure that can have repercussions for the growth, reproduction and survival of individuals (Bejder 2005; Lusseau et al. 2006; Fortuna 2007). Importantly, depending on the proportion of the population affected, these cumulative negative impacts on individuals' vital rates can translate into population-level consequences.

To understand the processes by which nonlethal anthropogenic disturbances affects populations, the Population Consequence of Disturbance (PCoD) framework for marine mammals was developed (National Research Council 2005; New et al. 2014). Since its development, this framework has become a useful method for understanding the potential population-level consequences of disturbances (New et al. 2014; Villegas-Amtmann et al. 2017; Nabe et al. 2018). However, despite the extensive and growing literature documenting effects of human disturbance on wildlife, there are still glaring absences of empirical evidence on how short-term behavioral alterations translate into long-term biologically significant impacts on individual vital rates (McHuron et al. 2022). Regardless of the mechanistic link between the vessel noise disturbance and individual and population fitness consequences, exposures to loud vessels has the potential to cause detrimental effects to marine mammals (see review by Erbe et al. 2019). This is of particular concern in highly industrialized and/or densely trafficked areas overlapping with important habitats for marine mammals where animals are likely to be repeatedly or chronically exposed. Thus, to reduce the widespread impacts of vessel underwater radiated noise to marine wildlife, approaches that reduce the overall noise levels of vessels or that move vessels away from important habitats are needed to reduce the general risk of disturbance.

## Measures to Reduce Underwater Radiated Noise from Commercial Vessels

### Technological Measures

Above the cavitation inception speed of a vessel, rotating propellers create oscillations in water pressure, resulting in water vapor bubbles that produce broadband noise when they collapse (IMO 2014; AQUO 2015). Propeller cavitation is

consequently the most significant source of underwater radiated noise from vessels (Ross 1976; IMO 2014; Wittekind 2014), and modifications to vessel technology (i.e., propellers, hulls, rudders, etc.) are increasingly being trialed in attempts to reduce cavitation noise.

A significant proportion of modifications to reduce cavitation noise have centered on amendments to vessel propellers. These amendments aim to improve the propeller blade surface and coatings, and/or change blade number, diameter, pitch, skew, shape, tips, and weight (Leaper and Renilson 2012; IMO 2014). Spence and Fisher (2017) note that cavitation can be efficiently reduced through propeller design. Evidence from studies of a variety of proprietary propeller and hub cap concepts (e.g., High Skew Propellers; PressurePores™; Contracted and Loaded Tip; Kappel Propellers; New Blade Section Propellers; New Profile Technology Propellers; and Propeller Boss Cap Fins) have reported significant reductions in underwater radiated noise ranging from 3 to 20 dB (Ouchi et al. 1991; Ouchi and Tamashima 1992; Mewis and Hollenbach 2006; Ligtelijn 2007; Andersen et al. 2009; Hansen et al. 2011; Hilliard et al. 2014; Ma et al. 2014; Carchen et al. 2015; Sun et al. 2016; Bagheri et al. 2017; Gassmann et al. 2017; Aktas et al. 2020). However, although amendments to propellers seem to be an effective and supported approach for reducing radiated noise from vessels (IMO 2014), these modifications are complex, as they must address an unfavorable trade-off between radiated noise emissions and propeller propulsion efficiency (i.e., fuel consumption; Fig. 2) (Leaper and Renilson 2012; Leaper et al. 2014). Successful propeller designs are also not universal and must be developed for the specific wake profile of the vessel in question (Spence and Fischer 2017). Thus, radiated noise and propeller efficiency need to be considered simultaneously during the development and installation processes (Leaper and Renilson 2012).

Reductions in vessel cavitation can also be achieved through modifications to improve wake inflow and propeller/rudder interactions. Presently, the International Maritime Organization (IMO 2014) recommends that the hull form and propeller design are adapted together during ship manufacturing, as the hull form has a direct influence on water inflow to the propeller and resulting underwater noise. Additional devices which alter the wake flow into the propeller have been recommended to reduce noise and fuel consumption, for example, Mewis and Schneekluth ducts, simplified compensative nozzle, and Grothues spoilers (Breslin and Andersen 1994; ITTC 1999; Mewis and Hollenbach 2006; Carlton 2009; Feizi Chekab et al. 2013; Ebrahimi et al. 2019). Further, interactions between the propeller and rudder can be mitigated using twisted rudders and rudder fins developed to increase efficiency (Leaper and Renilson 2012); or Costa Propulsion Bulbs, which have been shown to reduce noise by 5 dB as a result of hydrodynamically integrating the propeller with the rudder, by fitting a bulb to the rudder in line with the propeller shaft (Ligtelijn 2010).

An increasing number of novel technological solutions are being proposed to reduce radiated noise from commercial vessels. For example, air injections through propeller blades were found to reduce noise by 10 to 15 dB in frequency bands from 40 to 400 Hz; and bubble curtains that isolate the propeller and hull from the

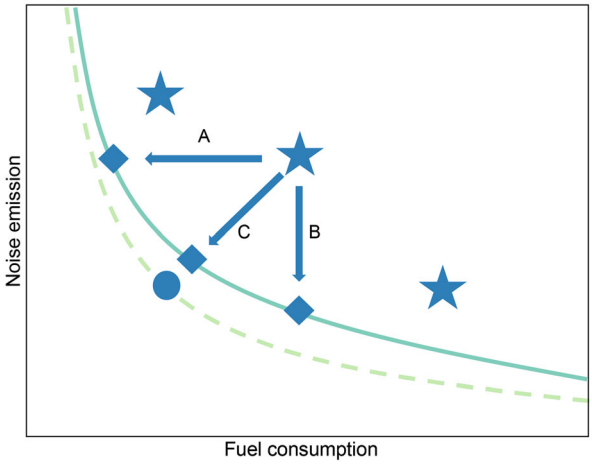



**Fig. 2** Trade-off between fuel efficiency and noise emissions illustrated in a Pareto-diagram for ships operating at their design speed. The solid line, the so-called Pareto front, describes the best possible combinations of fuel consumption and noise emission (for a constant speed). While it is possible to be inefficient in both parameters, i.e., high-fuel consumption and high-noise emission (stars illustrating three different ships poorly designed or maintained), vessels cannot go lower than the Pareto front (by definition). According to the Pareto front, the fuel consumption of a vessel can be reduced without additional noise emissions (as in A), the noise emission can also be lowered without additional fuel consumption (as in B), or any combination of these (as in C). However, once on the curve (diamonds), it is no longer possible to reduce one of the two without increasing the other. The only approach left to push back the Pareto front (broken line) is through innovation and new design approaches

surrounding fluid reduced the radiated noise of a cargo vessel travelling at 14 knots by 3 to 6 dB (AQUO 2015). These novel solutions could be applied to both new and retrofitted vessels but may be less popular than propeller or hull modifications as they are likely to require additional maintenance.

Underwater radiated noise is, to a lesser extent, generated by vibrations of onboard machinery (e.g., engine generators). In these instances, approaches to reduce noise have tended toward changing the engine type (e.g., 2- or 4-stroke diesel, steam/gas, hybrid, electric, etc.), isolating machinery from the vessel hull (e.g., elastic mounting, dampening) (AQUO 2015), or optimizing machinery placement in the hull (IMO 2014). In particular, the use of elastic mountings, where appropriate, can have a considerable effect, reducing noise outputs by up to 10 dB at low-frequencies and 20 dB at high frequencies (AQUO 2015). Such approaches may help to limit vibrational noise and could be used in combination with those that reduce cavitation noise to further reduce overall noise levels from commercial vessels.

Evidence clearly indicates that technological modifications can have a direct impact on underwater radiated noise produced by commercial vessels (Leaper and Renilson 2012; AQUO 2015), and consequently will result in fewer noise impacts to marine mammals. Furthermore, technological measures are advantageous as they

can reduce overall noise levels regardless of vessel speed, giving ship operators greater flexibility in lowering noise pollution while also balancing economic and time constraints placed on shipping. It is therefore anticipated that the shipping industry will adopt many of these ship-quieting technologies for future vessels operating within the global fleet. Even so, the implementation of ship-quieting technologies is expected to be expensive and complex, and less practical beyond the design stage (Spence and Fischer 2017). Consideration will therefore need to be given to the costs of implementing such modifications in existing vessels, as opposed to utilizing alternative (e.g., operational) measures to reduce the source levels of vessels.

## Operational Measures

Operational measures are predominantly viewed by the shipping industry as important techniques to increase energy efficiency (Holsvik and Williksen 2020). However, operational measures such as the maintenance of the hull and propeller, and speed reductions have been shown to also reduce radiated noise (McKenna et al. 2013; MacGillivray et al. 2019). These operational measures are an attractive approach for the industry as they require little effort to implement, few technological amendments to ship structures, and smaller investments compared to technical measures (Faber et al. 2011).

The regular maintenance of the propeller and hull of commercial vessels during the dry-docking process is an operational solution for reducing underwater radiated noise. Regular maintenance will maintain fuel efficiency, reducing overall fuel costs, while also reducing frictional resistance caused by biofouling. This allows vessels to reach greater speeds with less propulsive power and consequently produce less noise (Hilliard et al. 2014; IMO 2014; AQUO 2015). For example, regular maintenance to reduce biofouling was suggested to increase speed by 5% and reduce radiated noise by 1 to 2 dB (AQUO 2015), demonstrating that this is a simple and effective method for reducing noise which can be applied to all ships.

Underwater radiated noise increases with vessel speed, size and load (Ross 1976; Urick 1983; MacGillivray et al. 2019; MacGillivray and de Jong 2021). Extensive empirical observations indicate that radiated noise is proportional to the sixth power of speed for fixed pitch propeller vessels, and this reduction is approximately equal across frequency (Fig. 3) (Ross and Alvarez 1964; Ross 1976; MacGillivray and de Jong 2021). Accordingly, operational slowdowns of fixed pitch propeller vessels can be expected to achieve reductions in vessel source levels across a broad range of frequencies. In the case of controllable pitch propellers, this relationship is inconsistent (IMO 2014; Leaper 2019). Higher revolution rates can result in the emission of loud underwater noise even at low speeds, making operational slowdowns a less effective approach for this propeller type. Regardless, controllable pitch propellers make up only a small proportion of the global fleet (typically tugs, ferries and specialized vessels) (IMO 2014; Leaper 2019) and during higher speed transits, these vessels may perform more like fixed pitched propeller vessels.

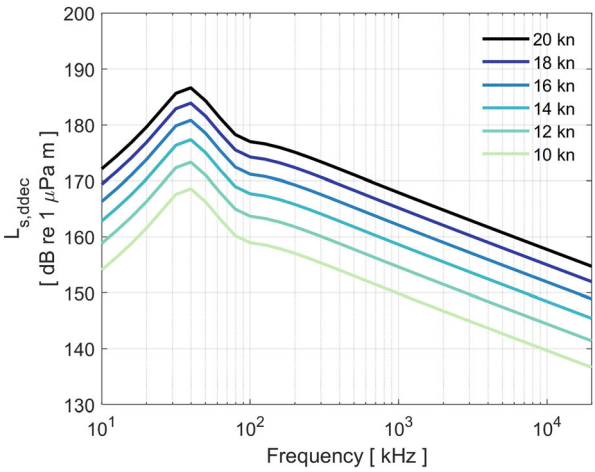



**Fig. 3** Example of the predicted decidecade source levels ($L_{s,ddec}$; dB re 1 µPa m) for a 300 m container ship with a fixed pitch propeller travelling at 20 knots and then 10% to 50% slower. (Source-level predictions are based on the latest vessel reference spectrum model (JOMOPANS-ECHO) (MacGillivray and de Jong 2021) (This figure has been adapted from Findlay et al. 2023)

Voluntary slowdowns to 11 knots as part of the Vancouver Fraser Port Authority Enhancing Cetacean Habitat and Observation (ECHO) program have illustrated that reductions between 11.5 and 5.9 dB can be achieved in the mean broadband monopole source levels of a variety of vessel classes, including container ships, cruise vessels, vehicle carriers, tankers and bulkers (MacGillivray et al. 2019). Further, the voluntary slowdown approach adopted during the ECHO program resulted in a 22% decrease in potential lost foraging time for the Southern Resident Killer Whale (*Orcinus orca*) population (Joy et al. 2019), demonstrating how slowdowns can simultaneously reduce fitness impacts to marine mammals inhabiting areas in the vicinity of high densities of ship traffic. Although the source level relationship with speed varies by and within vessel class due to differences in design and maintenance (Simard et al. 2016; Joy et al. 2019; MacGillivray et al. 2019), it is becoming increasingly clear that running at a lower operational speed is a broadly effective and immediate way to reduce the source levels of all vessels (IMO 2014; MacGillivray and de Jong 2021) and consequently, noise levels experienced by marine wildlife (Findlay et al. 2023).

Speed reductions are already used for economic reasons in commercial shipping when overcapacity in the market exists and when transport costs can be lowered by traveling slower than the vessel design speed (Jalkanen et al. 2018). Faber et al. (2017) suggested that a 10% to 30% reduction is feasible for most ship types, but this may require an increase in fleet capacity to maintain cargo throughput. This approach also has added benefits of reducing fatal ship-strikes (Silber et al. 2012; Laist et al. 2014), greenhouse gas emissions (Leaper 2019) and fuel consumption (Leaper and Renilson 2012; Cullinane and Cullinane 2013; Jalkanen et al. 2018; Leaper 2019), thereby helping the shipping industry to not only reduce underwater radiated noise but also reduce economic costs, meet climate targets set by the IMO (Khan et al. 2012; Cullinane and Cullinane 2013; Faber et al. 2017; IMO 2018;

Leaper 2019), and promote conservation efforts (Laist et al. 2014; Jalkanen et al. 2018; Leaper 2019).

Implementing coordinated slowdown policies to reduce underwater radiated noise from commercial vessels, especially at an international level, presents challenges (Merchant 2019). For example, a review of barriers to operational slowdowns notes that contractual clauses concerning demurrage (a charge payable by the ship on failure to load/discharge the ship within an agreed time slot) and laycan (delivery period for the cargo) often leads to vessels travelling at high speeds to meet delivery deadlines (Faber et al. 2011; Holsvik and Williksen 2020). Furthermore, complex ship traffic dynamics and geographic conditions may require ships to operate at certain speeds to maintain safety (IMO 2018). As a result, the adoption of slowdowns is likely to require coordination and collaboration between numerous stakeholders (e.g., government officials, the shipping industry, ports, vessel pilots, researchers, etc.) (AQUO 2015). Regardless, reducing the absolute (i.e., mandatory speed limits), or relative speed of vessels (i.e., percentage reduction of design service speed) (Faber et al. 2017), are effective and immediately implementable approaches to lower the amount of noise radiated by fixed pitch propeller vessels and reduce potential noise impacts to marine wildlife. Further, these operational and technological measures could be aided by the introduction of "noise labels" for individual ships (e.g., DNV 2010; Bureau Veritas 2014), able to denote louder ships which could be required to travel at slower speeds in areas known to be of importance to acoustically sensitive marine wildlife, or be targeted for technological modifications to reduce vessel source levels.

## Rerouting Measures to Reduce Vessel Noise Impacts to Marine Mammals

Commercial vessels travel along shipping lanes which provide the shortest and fastest routes between maritime ports (Hsu and Hsieh 2007; Jalkanen et al. 2022). Sea areas in proximity to these routes experience high levels of underwater radiated noise from shipping (Folegot et al. 2016; Farcas et al. 2020; Jalkanen et al. 2022), and changes to these routes have been shown to alter local underwater soundscapes. For example, an opportunistic study by McKenna et al. (2012) documented a rapid decline in noise along the California coast by 5.2 dB and 2.5 dB in the 40 and 90 Hz frequency bands, during the enforcement of a regional air-quality regulation zone (California Air Resource Board; CARB). Additionally, the rerouting of a major shipping lane in the Kattegat between Denmark and Sweden to increase maritime safety also resulted in significant and measurable changes to the regional soundscape in the 125 Hz decidecade frequency band (Tougaard et al. 2023).

The use of maritime spatial planning measures that consider ship traffic density, environmental conditions and local wildlife when establishing routes for commercial shipping could prove to be effective for reducing radiated noise in some areas (AQUO 2015). Shipping activity could, as an example, be concentrated or convoyed

along routes that avoid noise-sensitive areas or periods of high animal density (Laist et al. 2014; Chou et al. 2021; Ménard et al. 2022); or traffic could be diluted to homogenize vessel density and prevent the creation of high intensity noise hotspots (Erbe et al. 2012; Williams et al. 2015). Maritime spatial planning measures could also exploit bathymetric features along proposed shipping lanes, taking advantage of areas where the scattering of sound at the ocean surface and seafloor will be greatest, such as in shallow waters, or where natural features such as underwater canyons or islands will limit sound propagation (Urick 1983; Jensen et al. 2011; Etter 2018).

Studies have reported reductions in both noise impacts and the risk of lethal ship-strike impacts to marine mammals when vessels are rerouted. A voluntary lateral displacement of tugs and barges away from known feeding areas for Southern Resident Killer Whales resulted in an approximate 5 dB reduction in broadband noise, which included frequencies used for calls and echolocation (Burnham et al. 2021). The spatial and temporal rerouting of shipping lanes on the east coast of the United States to areas and periods outside of those known to be important for North Atlantic right whales (*Eubalaena glacialis*), combined with a speed limit of 10 knots, was found to result in reductions in noise levels and lethal ship strikes (Silber et al. 2012; Laist et al. 2014). Thus, increasing the distance between vessels and animals in both time and space, especially when combined with vessel speed reductions, can be expected to have considerable conservation benefits for marine wildlife.

Nevertheless, the efficacy of routing as a noise mitigation strategy is strongly dependent on the local sound propagation conditions (e.g., Tougaard et al. 2023); geographic and bathymetric constraints, especially in restricted waterways (e.g., rivers, channels, etc.); and the degree to which species under protection remain within well-defined areas. Speed reductions may therefore be more effective than routing measures to protect species with a broader or variable distribution, or if a change in routing to protect one species risks increasing the impact on another group of animals, in particular when rerouting vessels from shallow to deeper waters. Routing measures will therefore need to be considered on a case-by-case basis, and if possible, be combined with operational and/or technological modifications to further reduce overall noise levels at the source.

## Conclusions

Mounting evidence shows that vessel noise has detrimental effects on marine mammals that may culminate in population-level consequences. Global reductions in vessel radiated noise are therefore required to reduce the impact of commercial shipping activity to marine mammals and other acoustically sensitive marine species. Here, we highlight that technological, operational, and routing measures can be used by the shipping industry independently or synergistically to reduce both global cumulative noise levels from commercial vessels and potential impacts to marine mammals.

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# Mitigating Underwater Noise from Shipping, Seismic Airgun Surveys, and Pile Driving Using Best Available Technology and Environmental Practices

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Lindy Weilgart

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**Abstract**

Best Available Techniques/Technology (BAT) and Best Environmental Practice (BEP) are required under several international agreements. For shipping noise, this generally includes minimizing cavitation by optimizing the propeller design to the hull and to usual operating conditions, and better maintenance, which often also improve efficiency. Slow steaming, or reducing ship speed, has the advantage that no retrofitting is required, and greenhouse gas emissions are reduced. Quieter seismic airgun alternatives, such as Marine Vibroseis, a controlled sound source that can be tailor-made to the specific environmental conditions, can reduce the unnecessary mid- or high-frequency energy and sharp rise time of airguns. Mitigation measures for airgun surveys should show proof of efficacy, avoid sensitive areas and times, and not proceed in conditions of poor visibility such as at night. Quieting technologies would almost certainly require fewer mitigation measures. Especially many new quieting technologies and alternative low-noise foundations have been developed for pile driving, mainly due to governments like Germany setting noise limits. Application of quieting technologies that reduce sound at source will likely be the most effective way to reduce the environmental impacts of underwater noise, and quieting methods that also reduce greenhouse gas emissions or encourage technological innovation should be especially encouraged.

**Keywords**

Quieting technologies · Underwater noise · Mitigation · Best available technology · Best environmental practice · Seismic airgun surveys · Shipping · Pile driving

## Introduction

The application of Best Available Techniques/Technologies (BAT) and Best Environmental Practice (BEP) to prevent and reduce marine pollution is a requirement under several international agreements and conventions, e.g., under the Convention on Migratory Species (CMS) and the Convention on Biological Diversity (CBD). Regional agreements, such as the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention) and the Convention on the Protection of the Marine Environment of the Baltic Sea Area (Helsinki Convention), as well as species-focused regional agreements, including the Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and Contiguous Atlantic Area (ACCOBAMS) also require BAT and BEP.

In this report, the term "best" is used as in Best Available Technology and Best Environmental Practice, to denote what is best, possible, realistic, and practical now, with the choices we have at present. These recommendations, however, are still just steps toward what is truly "best" and require ongoing adjustment and innovation to achieve more substantial reductions in the harmful effects of noise on the marine environment and species therein. "Best" here should not be misconstrued to mean the ideal for restoring ocean health.

## BAT for Shipping Noise

### Noise Levels

Hildebrand (2009) noted that peak spectral levels for individual commercial ships of around 195 dB re $\mu Pa^2$/Hz at 1 m (for fast-moving, >20 knots, supertankers) occur in the frequency band of 10 to 50 Hz. A 173-m-long cargo vessel traveling 16 kts registered a source level of 192 dB dB re 1 μPa at 1 m over a bandwidth of 40–100 Hz (Hildebrand 2009). Shipping noise is not restricted to only low frequencies, though. Shipping noise can extend into the high kilohertz (kHz) range, particularly at close ranges and in shallow water. Findlay et al. (2023), for instance, note that noise from a container ship extended into the 20 kHz range. Vessel noise from various ship types substantially raised noise levels over the whole recording band from 25 Hz to 160 kHz at ranges between 60 and 1000 m (Hermannsen et al. 2014). The study estimated that such noise levels resulted in a hearing range reduction in the harbor porpoise (*Phocoena phocoena*) of more than 20 dB (at 1 and 10 kHz) from ships passing at distances of 1190 m, and more than a 30 dB reduction (at 125 kHz) from ships at 490 m distance or less (Hermannsen et al. 2014). Such a hearing range reduction (20–30 dB) represents a 100- to 1000-fold reduction in intensity.

## Impacts

Rolland et al. (2012) found that shipping noise is associated with increased stress levels in endangered North Atlantic right whales (*Eubalaena glacialis*). Broadband ship noise caused a significant change in beaked whale movement during foraging over a distance of at least 5.2 km, which could reduce their food intake (Pirotta et al. 2012). Putland et al. (2018) found that routine vessel passages shrank communication space by up to 61.5% for bigeye fish (*Pempheris adspersa*) and 87.4% for Bryde's whales (*Balaenoptera brydei*) and by up to 99% for both species at a large commercial vessel's closest point of approach. Nedelec et al. (2015) exposed larval Atlantic cod (*Gadus morhua*) to shipping noise in the laboratory and discovered they were easier to catch in a predator-avoidance experiment and in worse condition. Shipping noise playbacks increased stress indicators in European perch (*Perca fluviatilis*), common carp (*Cyprinus carpio*), and gudgeon (*Gobio gobio*) (Wysocki et al. 2006), European seabass (*Dicentrarchus labrax*) and gilthead sea bream (*Sparus aurata*) (Buscaino et al. 2010; Celi et al. 2016), juvenile European eels (*Anguilla anguilla*) (Simpson et al. 2015), and shore crabs (*Carcinus maenas*) (Wale et al. 2013). Bluefin tuna (*Thunnus thynnus*) schools became more uncoordinated with shipping noise, something that could compromise their homing accuracy during migration (Sarà et al. 2007).

Excessive underwater shipping noise is mainly the result of (a) poor propeller design or one not optimally matched to the vessel and its usual operating conditions; (b) poor hull design particularly of the aft end of the ship, which causes an uneven water flow into the propeller (poor wake field); or (c) a fouled (dirty) or damaged propeller. When a propeller is especially noisy, it probably signifies a ship operating inefficiently. Existing ships can solve these issues by installing newer, more efficient propellers, maintaining them well by cleaning and repairing damaged ones, using wake flow devices that improve the flow into the propeller, and maintaining the hull well.

## Propeller Cavitation

A major source of shipping noise is propeller cavitation, a hissing noise caused by the formation and collapse of air bubbles on the rotating propeller's surface when the pressure falls below the water vapor pressure. Cavitation is broadband, across a wide range of frequencies, but has narrowband or tonal noise peaks related to the rotation rate (rpm) multiplied by the number of propeller blades and the harmonics thereof. The lowest speed at which cavitation begins is called the cavitation inception speed (CIS). CIS occurs around 10 kts or even lower for many ships (Leaper and Renilson 2012). While some cavitation occurs even with efficient propellers, excessive cavitation from the noisiest ships is a sign of possible inefficient operation, with poor wake flow into the propeller and/or poor propeller design (Leaper et al. 2014). If noise from one noise source is 10 dB louder than other sources of noise, then the quieter noise sources are mostly irrelevant

(McCauley et al. 1996). For the noisiest merchant ships, the noise from propeller cavitation is likely to dominate all other noise sources from that ship (IMO 2013). Cavitation noise also dominates other propeller noise, apart from singing (high-pitched notes), and is louder than all other hydroacoustic noise from a ship (Ligtelijn 2007). Propeller singing is easy to remedy by altering the shape of the trailing edge (Leaper and Renilson 2012).

## Overlap Between Increased Energy Efficiency and Noise Reduction

As explained by Leaper and Renilson (2012), a greater blade area can produce the same thrust but with a smaller difference in pressure between the blade's face (pressure side) and the back (suction side). Since cavitation is caused by the difference in pressure, a greater blade area reduces cavitation. Unfortunately, the torque needed to turn the propeller is larger with this greater blade area. There is an optimum design for efficiency in merchant ships that is a trade-off between cavitation and blade area. Generally, one must tolerate some cavitation to design an optimally efficient propeller, so as to minimize blade area and reduce the necessary torque to turn it. The goal, however, is to reduce *excessive* cavitation that can lessen the thrust and cause erosion on the propeller and, in some cases, even on the rudder (Leaper and Renilson 2012). It is likely, by some estimates, that a 5–10 dB reduction in noise can be achieved before there is a loss in efficiency, though, depending on the specific circumstances. This degree of quieting (5–10 dB) would diminish the acoustic footprint of ships appreciably and substantially improve the quality of the marine environment. The amount of quieting necessary to make a difference in reducing environmental degradation is relatively modest, not on the order of that required of stealthy naval vessels.

Improved wake flow around the hull ahead of the propeller, the other major factor in reducing cavitation, will both reduce noise and improve efficiency. The goal is that the propeller, as it rotates through its full circle, does not encounter much of a difference in flow, i.e., a uniform wake field. A nonuniform wake, in contrast, can diminish propulsive efficiency and cause cavitation to fluctuate through the rotation cycle, producing tonal noise and its harmonics. Hull shape also affects the wake going into the propeller (Chris Waddington, pers. comm.). The bulbous stern is designed to provide clean flow into the propeller, thereby reducing cavitation and the noise from it (Chris Waddington, pers. comm.). Designs for new build ships should be evaluated using appropriate modeling techniques to optimize the propeller and hull configuration in consideration of underwater radiated noise (URN). Similar modeling may also assist with retrofitting existing ships with optimized propellers or wake flow devices.

Ideally, propellers should be clean, free of fouling, well maintained, and polished, with no nicks or imperfections, particularly on the leading edge (Leaper and Renilson 2012). Leaper and Renilson (2012) note that such damage can cause more cavitation, reduce efficiency by around 2%, and cause noise. At each dry docking of a vessel, propellers should be assessed and repaired for any damage that

might make it noisier. Well-built and well-designed propellers can improve efficiency and reduce noise. To produce an optimal wake field, the propeller and hull should be designed as a unit. Designing propellers and hulls to the actual operating conditions, rather than the ideal, would improve propulsive efficiency and reduce noise (Leaper and Renilson 2012).

Leaper and Renilson (2012) state that propellers can generate vortices from their hub that reduce efficiency and are prone to cavitate, as well as cause higher-frequency noise. Efficiency gains and noise reduction can be achieved by well-designed hub caps along with devices that can be affixed to the hub such as Propeller Cap Turbines (Leaper and Renilson 2012).

Wake flow optimization is always most effective if done at the design stage, but there may be scope for minor modifications for existing vessels with less-than-optimal wake flow. To improve the wake going into the propeller, wake inflow devices can be employed, which reduce cavitation, likely increasing efficiency while reducing noise. Schneekluth duct, Mewis duct, and Grothues spoilers are such devices that can be fitted to the hull for this purpose (Leaper and Renilson 2012).

There is certainly overlap between noise reduction, and increased fuel efficiency and reduced greenhouse gas emissions. In 2009, the International Maritime Organization (IMO) recommended that member states should identify which of their commercial vessels would benefit most from efficiency-improving technologies since these would also probably quiet their ships as well (IMO 2009). It would be a missed opportunity not to address noise at the same time as fuel efficiency and greenhouse gas emission reduction. Small changes in propulsive efficiency could dramatically lower noise output (Leaper and Renilson 2012).

## Hull Vibration, Engine, and Machinery Noise

The main treatments to reduce noise and vibration to the hull from onboard machinery are vibration isolation, noise insulation, and damping. Onboard machinery can generate tonal sounds but generally less broadband sound energy than propeller noise.

## Onboard Real-Time Noise Monitoring

Giving ship operators immediate feedback about which operating conditions (e.g., trim) are generating the most noise and cavitation could be useful. This could be achieved through a real-time noise or cavitation monitoring system using onboard sensors for both engine and propeller noise. For real-time onboard monitoring of propeller noise, sensors could be mounted near the propeller. The operator would thus be informed which conditions alter the cavitation inception speed or change the noise output. Studies underway as part of the Saturn project (https://www.saturnh2020.eu/) will provide detailed comparisons of measurements from onboard sensors and underwater radiated noise levels.

## BEP for Shipping Noise

### Slow Steaming to Reduce Noise

The practice of operating transoceanic merchant ships, especially container ships, at considerably slower speeds than maximum, mainly to save fuel, is known as slow steaming. A big advantage of slow steaming is that no retrofitting is required so it can be implemented immediately. Most ships use fixed pitch propellers [controllable pitch propellers were only used in 10% of tankers, 1% of bulk carriers, and 9% of container ships in 2000–2004 (Carlton 2007)], and for these, slowing down reduces the overall noise across all frequencies (Findlay et al. 2023). Even vessels with controllable pitch propellers, such as tugs and ferries, perform like fixed pitch propellers at higher speeds (Findlay et al. 2023). Slow steaming since 2007 reduced average speeds from 15.6 kts (sd = 4.2) in 2007 to 13.8 kts (sd = 3.0) in 2013 in the major eastern Mediterranean shipping routes (Leaper et al. 2014). Leaper et al. (2014) estimate that this 11.5% slowdown probably reduced the overall broadband acoustic footprint from these ships by over 50%. In the Haro Strait (between Seattle and Vancouver), 3 dB of overall noise reduction (i.e., a 50% reduction in sound energy) could be achieved by enforcing a vessel speed limit of 11.8 knots (Veirs et al. 2018). The mean ship speed across all classes was 14 kts; the mean for container ships alone was 19 kts. Slower ships due to COVID-19 resulted in less ocean noise pollution (Dunn et al. 2021). Vessel speed was highly correlated to sound pressure level (SPL) of the noise. A 2-month-long trial (MacGillivray et al. 2019) where ships slowed down to 11 knots showed that slowing down was an effective method for lowering mean broadband noise source levels for 5 categories of piloted commercial vessels: containerships (11.5 dB), cruise vessels (10.5 dB), vehicle carriers (9.3 dB), tankers (6.1 dB), and bulkers (5.9 dB).

Some argue that slower vessels generate overall more noise over time [larger sound exposure levels (SELs)] because they remain in an area for longer. This is incorrect, though, since underwater radiated noise (URN) is correlated with the sixth power of speed (Leaper 2019), a very steep curve, and time is just linear, so slower ships will produce lower SELs in addition to lower sound pressure levels (SPLs). Findlay et al. (2023), using modeling, also conclusively demonstrated that when animals were exposed to slower vessels, not just the maximum received level but also the exposure duration were reduced. Additionally, the rate of increase of noise as ships approached was slower, likely causing ships to be perceived as less alarming by animals.

### Slow Steaming to Reduce Noise and Co-benefits with Greenhouse Gas Emission Reduction

Slow steaming across shipping fleets is also an effective short-term measure to cut greenhouse gas (GHG) emissions. In April 2018, the IMO adopted the goal of reducing the total annual GHG emissions by at least 50% by 2050 relative to

2008. To achieve this, the IMO introduced various carbon dioxide reduction measures that could drive a reduction in vessel speed and, thus, noise reduction. The Energy Efficiency eXisting ship Index (EEXI), applied after January 1, 2023, is a technical design-related measure for existing ships above 400 gross tons. Many vessels could achieve compliance by adopting overridable power limitations, i.e., they will slow down (Chris Waddington, pers. comm.). The Energy Efficiency Design Index (EEDI) provides a new building standard, ensuring that ship designs achieve a particular level of efficiency and decrease carbon emissions. It was introduced for new vessels of 400 gross tons and above, in 2013. Ships must meet the reference level, which is tightened every 5 years, for their ship type. Ship owners can select the efficiency measures they deem as most appropriate, which may include speed reduction (Chris Waddington, pers. comm.). Unlike EEDI and EEXI that are design indices, the Carbon Intensity Indicator (CII) is an operational measure that must be reported annually. It is a nonprescriptive measure of how efficiently a ship transports goods or passengers, given in grams of carbon dioxide emitted per cargo-carrying capacity and nautical mile. From January 1, 2023, onward, it applied to new and existing ships above 5000 gross tons. Whether or how much these measures will reduce overall vessel speeds and consequent underwater noise has yet to be fully assessed. This will partially depend on ongoing discussions within the IMO on ways to reduce GHG emissions. Rutherford et al. (2020) found that most ships are being operated at engine loads that would be unaffected by the technical efficiency standard the EEXI sets, and thus EEXI requirements may not result in substantial speed reductions. However, speed reduction can be a very effective way to reduce emissions, so some ship owners may adopt further speed reductions beyond those possibly adopted for EEXI compliance (Chris Waddington, pers. comm.). It has been further proposed that an overridable power option be available for EEDI compliance (new ships). This is the same overridable power option that can be used for EEXI compliance (existing ships). If adopted, this could bring about further speed reductions of new vessels (Chris Waddington, pers. comm.).

Leaper (2019) reviewed modeling work on GHG emissions and its relationship to underwater noise, ship-whale collision risk, and ship speed. He factored in research which considered that slow steaming would increase the number of ships needed to transport the same volume of goods, the cost of operating those extra vessels, and the increase in ship construction that might be necessary. Faber et al. (2017) examined speed reductions of 10, 20, and 30% compared to "business as usual." They determined that in 2017, 3.5% of container vessels were idle or laid up and estimated that bringing these vessels back into service would allow the container fleet to reduce speeds by 8% (Faber et al. 2017). Speed reductions of greater than 10% would probably require an increase in fleet capacity to meet current demand (Leaper 2019). Following an economic model developed by Lee et al. (2015), the savings in total fuel consumption from slow steaming was usually higher than the cost of operating the extra vessels necessary to transport equivalent goods. Slowing down also had business advantages beyond saving fuel in that it increased delivery time reliability (Lee et al. 2015). Leaper (2019) examined various speed reduction scenarios that would help achieve the GHG reduction targets while simultaneously providing the

additional environmental benefits of reducing noise and the risk of ship strikes on whales. Leaper (2019) concluded that modest, 10%, reductions in speeds across the global fleet could reduce the total sound energy produced by shipping by around 40%.

When slow steaming, the propellers and hull should be redesigned for this operational difference, especially controllable pitch propellers (Leaper and Renilson 2012). Controllable pitch propellers are fairly rare in the long-distance commercial fleet, but considering noise from such propellers may be important in localized situations, for example, where CPPs are fitted to ferries.

While slower speeds with the same fixed pitch propeller will substantially reduce noise levels because cavitation will be reduced, it is more complicated if the propeller is optimized for slower speeds for fuel efficiency, since optimizing for fuel efficiency may involve reduced blade area and accepting a greater amount of cavitation. Optimizing for fuel efficiency is of primary importance, but noise should still be considered so as not to inadvertently increase it by modifications, if at all possible. If there are options that can improve efficiency yet not increase noise, those should be prioritized.

## Vessel Load Condition

Propellers are generally designed for vessels under full load, despite ships not spending all their time in this state (Leaper and Renilson 2012). The ship is never loaded close to its full load condition when in ballast, which means the propeller is closer to the surface. The propeller tip may even be above the waterline. Cavitation on a propeller can be increased in ballast because of the lower water pressure on the blades, despite reduced propeller loading (Paik et al. 2013). Moreover, a ship in ballast is usually trimmed by the stern, which worsens the wake field to the propeller, causing yet more cavitation (Leaper and Renilson 2012). As a result, a tanker or bulk carrier in ballast will often be noisier than one in full load (Leaper and Renilson 2012). Installing air injection to a propeller in ballast conditions or Ship Trim Optimization software could help rectify this problem. The development of onboard sensors (see section "Onboard Real-Time Noise Monitoring") that can give a reliable indication of underwater noise may also assist in adjusting vessel trim.

## Cold Ironing

Using a shoreside electrical power connection when a ship is at berth in port while its main and auxiliary engines are turned off is known as cold ironing. It is also called shore-to-ship power (SSP) or alternative maritime power (AMP). Cold ironing is clearly quieter, with fewer emissions. Cold ironing may also reduce biofouling on ship hulls. Multiple studies have shown faster settlement of mussel larvae (*Mytilus edulis*) or other biofouling organisms with generator or ship engine noise (Wilkens et al. 2012; McDonald et al. 2014; Stanley et al. 2014; Jolivet et al. 2016). Only one

study showed a low-frequency sound inhibiting only very young barnacle larvae (*Balanus amphitrite*) from settling (Branscomb and Rittschof 1984). The US Navy spends about US$1 billion annually and US$56 million per single Navy vessel class on biofouling (McDonald et al. 2014), so reducing biofouling can save money as well as reduce noise (biofouling increases turbulence), increase efficiency, and even avoid the spread of invasive species on hulls. Biofouling on ship hulls can be responsible for at least 75% of the invasive species brought in by vessels (McDonald et al. 2014). A clean ship entering a port infected with invasive species and running a generator could attract pest species from around a 500 m radius (McDonald et al. 2014).

## Acoustic Anti-biofouling Systems

Acoustic antifouling use has increased steeply in the last few years. It is a concern as it is likely very damaging to beaked whales and other cetaceans. This technology has been available since about 2007–2012, but only recently (since around 2022) has it become ubiquitous. Legg et al. (2015), reviewing various acoustic antifouling systems, concluded that around 20 kHz is optimal to reduce the settling of most invertebrates. Most systems operate between 17 and 30 kHz. These devices elevated received levels by 40 dB in the 20–30 kHz range (Martin et al. 2022). Their median energy source level (0.1 s) was 183 dB re 1 $\mu Pa^2s$. High-frequency sensitive cetaceans were estimated to be disturbed at ranges of 3 km, with potential to quickly cause temporary or permanent hearing damage. There was an almost complete listening range reduction for high-frequency echolocation up to 5 km, lasting 25 min for a vessel traveling 14 kts, with a hull-mounted device (Martin et al. 2022). Trickey et al. (2022) found tourist vessels with hull-mounted ultrasonic antifouling devices caused clear avoidance by Cuvier's beaked whales (*Ziphius cavirostris*). These devices started at a frequency of 19 kHz, ranging up to 42 kHz or more, with a transmission range of over 2 km.

## Maintenance

Keeping the hull and propeller clean and repaired can yield cost savings, efficiency gains, and noise reductions. Other onboard machinery and engines will almost certainly be quieter and more efficient when well maintained. Proactive in-water ship hull grooming, where the hull is cleaned before it is badly fouled, can also help with both reducing noise and increasing efficiency (Swain et al. 2022). An analysis of how common fouled vessels are and the additional drag associated with fouling (Munk et al. 2009) showed that around 50% of vessels may need an extra 30% of power to maintain cruising speed. Using the observed relationships between underwater radiated noise (URN) levels and engine power (Leaper 2019), an increase in required power of 30% would likely produce an increase in URN of 2.3 dB. Since

Munk et al. (2009) found that around 17% of vessels had an increased drag of greater than 50%, URN levels would probably increase by more than 3.5 dB. These raised URN levels would greatly increase the number of animals unnecessarily affected. The rate of fouling can vary considerably between areas and the operational pattern of the vessel. The degree of fouling should be regularly assessed so that vessels can be cleaned as soon as fouling is likely to measurably worsen drag.

## Port Incentives

Since January 1, 2023, the Vancouver Fraser Port Authority through its EcoAction Program introduced a discount of up to 75% off harbor dues for ships that are certified as quiet vessels by an international ship classification society, such as American Bureau of Shipping (ABS), Bureau Veritas (BV), Det Norske Veritas Germanischer Lloyd (DNV), Lloyd's Register (LR), and Registro Italiano Navale (RINA). The Port of Prince Rupert has a similar program. Best practice for measuring and certifying noise from ships might be a combination of various methods such as computational predictions validated by onboard measurements over a range of speeds, with perhaps one or two at-sea verifying measurements. Such quieter ship incentives should be expanded to other ports worldwide to create a level playing field.

## Underwater Noise Management Plans

Underwater noise management plans can be developed for individual vessels or for all vessels using a particular area. Transport Canada, for example, has encouraged Canadian vessel operators to have plans to reduce their fleet's overall noise output. Noise management planning represents the development of a plan to reduce URN and achieve URN goals. Plans should include measuring a noise baseline, setting a goal or target, and defining operational, organizational, and technological measures to reduce noise. Measures should repeatedly be evaluated to determine their effectiveness. Such noise management plans should become mandatory, as should the targets.

A 5-grade scale (A–E, similar to the Carbon Intensity Indicator) could be developed for URN, given this approach is familiar to the industry. Depending on the rating received during an annual review, the ship could be required to install design upgrades or conduct substantial operational improvements. Each ship could be evaluated for its URN and guidance given on how to make it quieter. The focus here would not be to meet a particular noise target, but rather to make each ship as quiet as possible within the scope of its operational requirements and size, as per the noise management plan for that vessel. This would allow ships to understand the extent of measures that they should consider, to reduce URN to less adverse levels. The hope is also that it would direct attention to the noisiest ships first.

### Focus on the Noisiest Vessels

It is challenging to set source-level standards for each vessel type. Focusing on the noisiest vessels aims to achieve the most benefit with the least economic impact, as the noisiest vessels will likely be operating inefficiently. Differences of 20–40 dB have been reported between the quietest and noisiest ships of a similar type (Carlton and Dabbs 2009). Leaper and Renilson (2012) estimated that the noisiest 10% of ships (those that are 6.8 dB or more over the average) contribute to 48–88% of the total acoustic footprint (the area over which the ship noise increases the background noise over a certain level). Similarly, Veirs et al. (2018) found that, of 1582 ships measured in the Haro Strait between Seattle and Vancouver, half of the total power radiated by this modern fleet came from just 15% of the ships – those with source levels above 179 dB re 1 μPa at 1 m. More than two-thirds of these worst noise polluters were cargo and container ships (Veirs et al. 2018). About 43% of container ships were worst polluters, by far the highest proportion of any ship class of those studied (Veirs et al. 2018).

## BAT for Seismic Airgun Survey Noise

### Noise Levels

An airgun array has a nominal source level of around 260 dB re 1 μPa at 1 m (SPL), with a bandwidth of 5–300 Hz (Hildebrand, 2009). While the energy from airgun impulses is mostly concentrated in the lower frequencies, there is still substantial energy in the tens of kHz to even over 100 kHz (Goold and Coates 2006).

### Impacts

Fin whales (*Balaenoptera physalus*) were displaced from their habitat at the beginning of a seismic survey, and the displacement continued well beyond the 10-day length of the seismic survey (Castellote et al. 2012). Bowhead whale (*Balaena mysticetus*) calling was repressed within a 50–100 km radius of a seismic survey, representing 8000–30,000 sq. km in area. Bowhead calling was almost entirely absent within 10–40 km of the seismic survey or 300–5000 sq. km (Blackwell et al. 2015). Pirotta et al. (2014) found that the probability of recording a prey capture attempt by harbor porpoise declined by 15% in the 25 km x 25 km area exposed to seismic survey noise. Prey capture probability increased the further away the seismic vessel was. Seismic airgun noise killed zooplankton, especially immatures, with a 2–3-fold increase in dead zooplankton at ranges of up to 1.2 km from the airguns (McCauley et al. 2017). Day et al. (2017) discovered a fivefold increase in scallop (*Pecten fumatus*) mortality when subjected to four passes of an airgun. These effects occurred 4 months after airgun noise ceased. Southern rock lobsters (*Jasus edwardsii*) showed a chronic reduction of immune competency and

impairment of nutritional condition, also 120 days post-airgun exposure (Fitzgibbon et al. 2017). Lobsters also showed significant damage to sensory organs, impairing important reflexes, even a year post-exposure to an airgun (Day et al. 2019). Catch rates for haddock (*Melanogrammus aeglefinus*), cod, and rockfish dropped from 21 to 70% (Engås et al. 1996; Skalski et al. 1992) during or after seismic airgun surveys. Declines in fish abundance were also documented (Slotte et al. 2004; Paxton et al. 2017).

## Lowering Source Levels and Reducing High-Frequency Content

According to Leaper et al. (2015), there are seldom cases where mitigation based on visual observation can achieve a greater risk reduction than would be achieved by a 3 dB reduction in source level throughout the survey. This is because visual observation is of limited value since Marine Mammal Observers (MMOs) miss spotting many marine mammals and turtles as they are cryptic, elusive, and often underwater and because survey activities often take place at night and in other limited-visibility conditions. The use of MMOs therefore only results in a limited risk reduction in all cases (Leaper et al. 2015). Therefore, the most effective mitigation for seismic airgun surveys is probably to (a) separate the surveys from areas or times rich in marine life and sensitive species and (b) to lower the source level (quiet the noise). Seismic operators should be required to develop a detailed plan describing how they will keep sound levels to a minimum, including calculations of the minimum required levels to meet survey objectives, how these will change within the survey area and with different weather conditions, and how the survey equipment will be configured to ensure only the minimum sound level is generated. If a 3 dB reduction in source level makes a great difference to the quality of the seismic data, then more controllable sources [e.g., Marine Vibroseis™ (MV), see below] rather than airguns are essential to adapt real time to different conditions. Otherwise, seismic operators must use the source level that applies to the most difficult conditions expected during the survey, which may be much higher than is needed most of the time. This is wasteful and adds more environmental risk than is necessary.

As previously stated, there is still considerable energy in the tens of kHz from airguns, extending even to over 100 kHz (Goold and Coates 2006), which explains why cetaceans with middle- or higher-frequency sensitivities react to the noise (Goold and Fish 1998). Geophysicists do not make use of, nor even record, any energy over about 200 Hz, however. This wasted energy therefore needlessly impacts marine life, especially animals with mid- or high-frequency hearing. There is currently considerable effort being expended by several companies to develop alternative marine seismic sources that are expected to have a reduced environmental impact while being at least as effective as airgun arrays. Improvements in receivers have also allowed for quieter sources. It is challenging to get more information on technological alternatives to airguns, as much is proprietary and still under development.

Several industry players have developed alternative pneumatic marine seismic sources that aim to limit bandwidth, increasing low-frequency content (Brittan et al. 2020) while reducing higher-frequency output. Pneumatic sources have been physically modified to reduce the high-frequency content generally by slowing the release of the air (Coste et al. 2014; Supawala et al. 2017; Tellier et al. 2021). Placing elements close to one another in a "cluster" can also help to produce a larger low-frequency peak output (Hopperstad et al. 2016). These various methods can result in substantial reductions in the signal strength at high frequencies (Brittan et al. 2020; Li and Bailey 2017; Tellier et al. 2021).

Teledyne Marine's eSource™ airgun, developed by Bolt Technology Corporation and WesternGeco and Sercel's Bluepulse™, release air more gradually than the conventional airgun, reducing or attenuating the higher frequencies. The advantage of this alternative is that it does not require any retrofitting of the seismic vessels, as MV does, and can be used as a conventional airgun. The disadvantage is that the approach may be too piecemeal and not comprehensive enough, as other potentially damaging characteristics of airgun pulses (e.g., higher source level and sharper rise time than MV) remain, though rise time is slowed and the SEL, energy over time, is lower. They are also not controlled sources.

The SERCEL TPS (Tuned Pulse Source) is an impulsive pneumatic source with a slower rise time than a conventional airgun. The spectral output is mostly at frequencies under 10 Hz. TPS is 13x quieter in maximum SPL than a conventional airgun array. The frequency at which the source signal gets lost in the background noise is at 140 Hz for TPS vs. 1 kHz for airguns, at 15 km distance. TPS can work as a stand-alone survey even without being supplemented by higher-frequency sources, unlike very low-frequency (1–2 Hz imaging) sources such as Wolfspar™. TPS has been used commercially (Ronen 2022).

## Marine Vibroseis™ (MV)

Another principle is to replace the short, high amplitude, wide frequency-bandwidth signal produced by an airgun array with a much longer duration, lower-amplitude signal, with the same acoustic energy in the frequency band required for the seismic survey (below 200 Hz and sometimes below 120 Hz). To be useful, the signal would have the same energy, just spread over a longer duration, allowing for a lower source level and less wasted energy at frequencies that are not used. The effectiveness of a signal for seismic imaging is solely determined by the signal's energy and bandwidth, so a longer, quieter signal should be just as effective as a shorter, louder one provided they have the same energy and cover the necessary frequencies. The quieter signal should reduce the risk of damage to an animal's hearing at short range.

Marine Vibrator™ (MV) systems are technically not commercially widely available yet and thus could not truly be called a Best Available Technology. This is despite the fact that MV has been in development since 2008 and yet has made slow progress. Still, this technology remains promising. If regulators were to insist on the use of quieter alternatives to airguns, these would be available very quickly, but

regulators feel they cannot require a technology that is not available yet, so it becomes a Catch-22. Moreover, seismic operators do not feel like regulations to pursue quieter technologies are incentivized and are even penalized. Seismic vibrators have been used successfully in land-based seismic exploration for many years. In 2009, when the Okeanos Foundation held a workshop on more environmentally friendly seismic airgun alternatives, the 16 participants (geophysical scientists, seismologists, biologists, and regulators) concluded that controlled sources such as marine vibrators probably offer the best chance at eventually replacing airguns (Weilgart 2010).

Tenghamn (2006) introduced a completely new electromechanical MV concept, using frequencies from 6 to 100 Hz. MV is a scalable source, and output level can be adjusted in real time to environmental and operational conditions (Pramik 2013). MV output can be altered by changing (a) the number of vibrators used in the array (more difficult with airguns due to undesirable acoustic side effects), (b) the output level, and (c) the length of the sweep (Pramik 2013). The emitted frequency band, phase, amplitude (loudness – SPL), or energy over time (SEL) can be selected, something largely impossible for airgun surveys. The controllable nature of MV could also bring advantages in signal processing. Phase and amplitude control could allow for more effective reduction in residual noise and allow sources to improve the reconstruction of the wavefield between source lines with less shots acquired and faster acquisition. Additionally, MV produces no cavitation. The initial capital cost of the MV array is large, but the operational cost is similar to airguns, with geophysical and operational benefits (Jenkerson 2022). MV works best from 5 to 100 Hz.

TotalEnergies, ExxonMobil, and Shell have sponsored the Marine Vibrator Joint Industry Project (MV JIP) since 2011, supporting the development of three separate marine vibrator technologies. Currently, the MV JIP is working exclusively with APS to develop and commercialize the Integrated Projector Node (IPN) powered by an electromagnetic system.

The only MV technology to have acquired seismic data as part of a pilot project is the HUP104 unit developed by Mobil, Total, and Geco in 1996 at Schooner Bayou, Louisiana, and the SAE developed Aquavib™ Marine Vibrator, which acquired a series of 2D lines in Louisiana (Pramik et al. 2015) and east offshore Malaysia in 2019. The Aquavib™ technology is currently in production targeting transition zones. It works better than airguns in water depths less than 5 m. BASS (Broadband Acoustic Seismic Source) MV is a hydraulic-based system developed by Shearwater, supported by the Norwegian Research Council and Equinor.

Most airgun arrays have an effective source level of 255 dB (0-p) re 1 μPa at 1 m in the downward direction, compared with a MV array of about 223 dB rms re 1 μPa at 1 m (Bird 2003) or 226 dB (0-p). Since the decibel scale is logarithmic, MV has a peak pressure that is a factor of 28 lower than that of an airgun array. LGL and MAI (2011), through modeling, estimated that a MV survey would expose only about 1–20% of whales and dolphins to $\geq$180 dB re 1 $\mu Pa_{rms}$ when compared to those exposed to an airgun survey. Matthews et al.'s (2021) modeling comparison of potential impacts on marine mammals from MV vs. airgun noise also concluded

that injury was less likely from MV arrays. High peak pressure and sharp rise time or onset (sounds quickly increasing in amplitude), both of which describe airgun emissions, are two characteristics of sound thought to be particularly injurious to living tissues (Southall et al. 2007). Southall et al. (2007) believed a non-impulsive sound such as MV would have to be 12–17 dB louder than an airgun-like impulse to be as injurious, due to the damage inflicted by the sharp rise time. Duncan et al. (2017) modeled sound levels from a realistic MV array and airgun array with similar downward energy at frequencies <100 Hz and compared the two under various scenarios. MV was 20 dB lower in peak-to-peak sound pressure level vs. the airgun array at 100 m range, decreasing to 12 dB lower at 5 km, the maximum modeled range for peak levels (Duncan et al. 2017). MV also produced 8 dB lower SELs than the airgun array at 100 km range because of MV's reduced bandwidth (Duncan et al. 2017). There are thus benefits to MV even at long ranges and even for animals with good low-frequency hearing. Duncan et al. (2017) also found that changing the layout of the MV array's higher-frequency sources reduced SELs by 4 dB. MV's lower sound exposure level advantage is most obvious in shallow water, as SELs drop off more rapidly in these waters. Moreover, shallow waters are often the most productive and biologically rich, in need of more protection.

In summary, Duncan et al. (2017) listed the main benefits of MV over airguns as:

- Lowering peak pressure (sound level) over short ranges
- Eliminating sharp rise time
- Eliminating unnecessary middle and high frequencies
- Lowering sound exposure levels for distances of over 10 km
- Allowing for greater control and tailoring of the signal (amplitude, frequency, duration, etc.) in real time
- Operationally superior in shallow water and transition zones

Therefore, MV shows potential in providing an environmentally safer alternative to airguns without compromising effectiveness for seismic exploration. MV surveys were predicted to cause less impact (behavioral, physiological, auditory) than airgun surveys in all habitats and environments regardless of water depth or environmental conditions (LGL and MAI 2011). The roughly 20 dB reduction in short-range peak-to-peak SPLs decreases the safety or exclusion zone radius by about a factor of 10, which converts to a reduction in safety zone area of around a factor of 100, exposing far fewer animals to potentially injurious noise.

The greatest disadvantage of MV compared with airguns is the greater potential for masking, since the MV signal is of longer duration (seconds vs. tens of milliseconds for an airgun pulse), and MV will probably have a higher duty cycle (percentage of time it is "on"). MV signal duration estimates range from 5 to 12 s (LGL and MAI 2011). This would impact mainly low-frequency hearing specialists such as baleen whales and some fish. Slight masking effects could extend to a few tens of kilometers from the MV source. Using narrowband FM sweeps would likely lower the potential for masking (LGL and MAI 2011). Moreover, airgun pulses are also not always as short in duration as they appear, if heard over larger distances

from the source. Reverberation and multipaths "stretch" the signal from its original 10 ms to sometimes seconds at long ranges (Guerra et al. 2011). Noise levels often do not have a chance to return to ambient in the 10 s between airgun shots, since there is still reverberation from the previous shot (Guerra et al. 2011). A similar situation can occur with MV signals, but such "stretching" would be proportionally less than for airgun pulses, since MV signals are longer in duration initially, close to the source (LGL and MAI 2011). Most importantly, MV signals would likely fade more quickly into the background ambient noise levels.

MV should certainly be field-tested for impacts on a wide range of sensitive marine taxa, something that should ideally happen in tandem with operationally testing various MV designs. As with other noise reduction measures from seismic surveys, the development of MV could be greatly expedited with incentives, encouragement, and pressure from regulatory governmental agencies (Duncan et al. 2017).

### Monitoring Technology

Monitoring, especially ahead of time, of the proposed survey areas should be conducted with fixed acoustic detectors (buoys, bottom recorders, etc.) or mobile gliders to assess the population density, abundance, and distribution of marine life before, during, and after seismic surveys. Gliders can be used for both vocal marine mammals and fish species.

Infrared (IR) or thermal imaging is promising in detecting warm-blooded marine life, such as whales and dolphins, which can assist with nighttime monitoring, especially of baleen whales (Zitterbart et al. 2013). It is meant to supplement, not replace Marine Mammal Observers, by alerting them to possible whale blows (exhalations). IR imaging also does not function well in some conditions, such as fog, or with species that do not remain long at the surface or with obscure blows, such as smaller whales, even the size of minke whales (Zitterbart et al. 2013). It is also very expensive. It appears to work best in polar regions.

Passive Acoustic Monitoring (PAM) should be used anytime there are vocal species in the area, daytime or nighttime. Towed arrays or other suitable technologies with enough bandwidth to be sensitive to the whole frequency range of animals expected in the area should be used to improve detection capabilities. PAM should be mandatory for night operations or when visibility is scarce. However, PAM may still be inadequate mitigation for night or poor-visibility operations if species in the area are not vocal or easily heard.

## BEP for Seismic Airgun Survey Noise

As previously mentioned, the most effective mitigation for seismic airgun surveys is to (a) separate seismic surveys from areas rich in marine life and sensitive species and (b) to lower the source level (quiet the noise). However, to separate seismic surveys from marine life, good, current knowledge of the abundance and distribution

of that life must be available. Therefore, baseline studies of biological abundance and distribution should occur at least a year, preferably two, in advance of seismic surveys. These should be of sufficient quality and statistical power to detect changes in abundance and distribution of marine life over natural variation. Sensitive and important habitats and seasons (spawning, breeding, feeding, etc.) should be avoided. Turtles, fish, and invertebrates should be included in mitigation and monitoring, wherever possible. Acoustic refuges of still quiet habitat should be established, and marine protected areas should be managed for noise and include acoustic buffer zones around them, considering the possible impact of long-range noise propagation. Seismic airgun surveys should avoid these acoustic refuges and marine protected areas.

The ACCOBAMS (Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and contiguous Atlantic area) Resolution 7.13, Guidelines to Address the Impact of Anthropogenic Noise on Cetaceans in the ACCOBAMS Area, are very close to BEP for seismic airgun survey noise. Where new mitigation measures not contained in the Resolution have been added, these are marked "NEW." ACCOBAMS is only focused on cetaceans, whereas this document is concerned with all marine species and overall ecosystem health, so sometimes more mitigations, or the present mitigations slightly modified, are considered necessary. Other helpful mitigation measures are contained in the Resolution that are not included here in the interest of brevity. The Resolution also contains useful guidelines to address noise from shipping and from coastal and offshore construction, such as pile driving, but only some of the "guidelines for seismic surveys and airgun uses" are included here, as these are particularly detailed, current, and extensive:

(a) Seismic survey planning to avoid key habitat and areas of density of marine life, ensuring that entire habitats or migration paths are not blocked and that cumulative seismic noise is limited within any area (e.g., special regulation or prohibition of multiple vessels operating in the same or nearby areas at the same time).
(b) Some proof of efficacy of the mitigation measures used for all sensitive species (NEW).
(c) Transparent, public notification of when and where seismic surveys will occur as soon as this is known by the operators (months in advance) (NEW).
(d) Use of the lowest practicable source power and (NEW:) have this verified by independent evaluators.
(e) Limit horizontal propagation by adopting suitable array configurations and pulse synchronization.
(f) Elimination of unnecessary high frequencies.
(g) No operation of airguns for any reason outside the permitted project area (NEW).
(h) Adapt the sequencing of seismic lines to account for any predictable movements of animals across the survey area and avoid blocking escape routes.
(i) Model the generated sound field in relation with oceanographic features (depth/temperature profile, water depth, seafloor characteristics) to dynamically set the

safety or exclusion zone (EZ). Verify models of the EZ in the field. (NEW:) EZ should be at minimum 500 m but may be larger depending on the propagation.

(j) Continuous visual and passive acoustic monitoring (PAM) by a specialized team of Marine Mammal Observers (MMOs) and PAM operators to (NEW) reduce the risk that animals are in the exclusion zone before turning on the acoustic sources and while sources are active.

(k) Prohibit airgun surveys at night, during other periods of low visibility, and during significant surface-ducting conditions since mitigation tools are likely inadequate to detect and localize sensitive marine life. Prohibit seismic surveys over Beaufort Wind Speed of 3 because of the impact of adverse weather conditions on the visual detection of animals. (NEW:) Only if passive acoustic monitoring (PAM) alone is shown as effective in detecting sensitive marine life as the combination of PAM together with MMOs should seismic surveys in poor visibility and at night be allowed.

(l) Passive acoustic monitoring (towed array technology or other suitable technologies with enough bandwidth to be sensitive to the whole frequency range of sensitive marine life expected in the area) should be used to improve detection capabilities. PAM may be inadequate mitigation if animals in the area are not vocal or easily heard.

(m) At least two dedicated Marine Mammal Observers (MMOs) should be on watch at one time on every operative ship; shifts should be organized to allow enough rotation and resting periods for MMOs. At least one PAM operator should be on watch and shifts should be organized to allow 24/24 h operation, unless automatic detection/alerting systems are proven to be as effective as PAM operators. Standardized tests (written and in the field) for MMOs and PAM operators, used worldwide, should be developed to ensure MMOs and PAM operators pass standard qualifications.

(n) Before beginning any sound transmission, there should be a dedicated watch of at least 30 min to reduce the risk that animals are within the EZ.

(o) Establish a minimum pre-clearance zone (i.e., pre-ramp-up watch zone) that extends 500–1000 m from the outer perimeter of the airgun array(s). (NEW).

(p) Extra mitigation measures should be applied in deepwater areas if beaked whales are expected or if habitats suitable for beaked whales are approached: In such cases, the watch should be at least 120 min to increase the probability that deep-diving species are detected.

(q) Every time sources are turned on, there should be a slow increase of acoustic power (ramp-up or soft start) to increase the chances that animals might leave the ensonified area (the effectiveness of this procedure is still debatable).

(r) Delay sound transmissions if sensitive species are observed within the exclusion zone (EZ) or approaching it. Ramp-up may not begin until 30 min after the animals are seen to leave the EZ or 30 min after they are last seen (120 min in case of beaked whales).

(s) Shutdown of source(s) whenever a sensitive species is seen to enter the EZ and whenever aggregations of vulnerable species (such as beaked whales) are detected anywhere within the monitoring area.

(t) If more than one seismic survey vessel is operating in the same area, they should maintain a minimum separation distance (dependent on propagation) to allow escape routes between sound fields.
(u) Encourage data sharing among seismic surveyors to minimize duplicate surveying. Also, encourage use of old seismic data if these can be usefully reanalyzed using new signal processing or analysis techniques. (NEW:) Duplicated surveys need to be justified.
(v) Conduct a quantitative analysis of cumulative and synergistic impacts not just of noise but of all anthropogenic threats over time as part of thorough Environmental Impact Assessments (EIAs) following the CMS Family Guidelines on EIAs for marine noise-generating activities, including consideration of historical impacts from other activities (shipping, military, industrial, other seismic) in the specific survey area and nearby region. Develop databases and noise registries to allow such analyses. (Addition of CMS Family Guidelines and synergistic impacts is NEW.)
(w) Include monitoring and reporting protocols to provide information on the implemented procedures, on their effectiveness, and to (NEW:) improve data on biological abundance and distribution, as well as examine impacts from seismic survey noise. Monitoring should be proven to be statistically powerful enough to detect subtle impacts, strandings, fish kills, etc. Before During After (BDA) or Before After Control Impact (BACI) studies to examine impacts must also contain power analyses to show whether possible impacts would be detectable or not. Impact and biological baseline studies should include more fish, turtles, and invertebrates. All biological and impact data collected for mitigation should be publicly available.
(x) Standardize MMO and PAM reporting so that data can be harmonized across all seismic surveys worldwide for maximum statistical power. (NEW).
(y) Conduct a biological survey after the seismic survey is completed to verify if changes in the abundance or distribution of species or anomalous deaths occurred.
(z) In the case of strandings, deaths at sea, or abnormal behavior possibly related with the operations, any sound transmission should be stopped, and maximum effort devoted to understanding the causes of the deaths or abnormal behavior.

## BAT for Pile-Driving Noise

### Noise Levels

Impact pile driving is the most common method of foundation installation for offshore windfarms (OWF) and other structures such as piers and bridges. Pile driving using a hydraulic hammer with an energy of 1000 kJ results in sound levels around 237 dB re 1 μPa at 1 m. The noise generated has a predominant bandwidth of 100–1000 Hz (Hildebrand 2009), but it also extends into the tens of kHz. Currently

used hydraulic hammers have a rated energy range of up to 4000 kJ, which increases noise emissions further, also with increasing pile size.

## Impacts

A broad range of marine life can be affected by piling because of the wide frequency range of the noise generated underwater. The harbor porpoise avoids pile driving out to a mean distance of 17.8 km. At 22 km, this avoidance was no longer apparent. Porpoise activity and possibly abundance were reduced over the entire 5-month windfarm construction period (Brandt et al. 2011). Once piling noise ceases, it can take up to 2 days until harbor porpoise behavior returns to pre-piling conditions (Brandt et al. 2016). Blue mussels (Spiga et al. 2016; Roberts et al. 2015) and seabream (Bruintjes et al. 2017) showed indicators of stress from pile driving. In addition, swimming and schooling behavior was impacted by piling in cod and sole (*Solea solea*) (Mueller-Blenkle et al. 2010), sprat (*Sprattus sprattus*) and mackerel (*Scomber scombrus*) (Hawkins et al. 2014), and juvenile seabass (Herbert-Read et al. 2017).

## Underwater Noise Limits

Largely due to the German government setting an action-forcing standard for noise, major progress in quieting technology has been made for pile driving. In 2004, The German Federal Maritime and Hydrographic Agency introduced noise guidance values for single strikes of 160 dB re 1 $\mu Pa^2$ s (SEL) and maximum 190 dB re 1 μPa (peak) at 750 m in the licenses of offshore wind farms within the German EEZ. In 2008, these became mandatory and were successfully applied in 2013, reaching state-of-the-art reliable compliance despite increasing pile diameters and water depths through 2018. No offshore windfarm in German waters has since been constructed without complying with these underwater noise limits. In 2013, the German Federal Ministry for the Environment also published its Sound Protection Concept. Along with technical mitigation measures, pile-driving companies purposely use lower piling/hammer energies to stay under the German noise limits. A growing number of countries across Europe, Asia, and North America are following Germany's lead and now also impose underwater piling noise restrictions during offshore windfarm (OWF) construction.

## Noise Mitigation and Abatement Systems

Noise mitigation systems (NMS) work to avoid or reduce the noise inputs into the water, whereas noise abatement systems (NAS) function in reducing the impact of existing pile-driving noise already in the water. Koschinski and Lüdemann (2020) detail technical noise mitigation measures for pile driving as well as alternative

low-noise foundation concepts and analyze their applicability. Bellmann et al. (2020) also provide a comprehensive overview of currently available noise abatement systems as well as a thorough explanation of underwater piling noise transmission and the factors that influence it. The report provides estimated noise mitigation levels for almost all NAS available on the market using empirical data sets from the North and Baltic Seas. Bellmann et al. (2020) conclude that the IHC Noise Mitigation Screen (IHC-NMS) is the most effective near-pile mitigation system available, estimating reductions in SELs of up to 17 dB. Quieting further improves up to 22 dB in depths of 40 m when combined with an optimized double big bubble curtain (DBBC). The report addresses a broad range of site-specific influencing factors such as soil parameters, water depth, bathymetry, sound propagation, and measuring details.

Verfuss et al. (2019) reviewed NAS for OWF construction noise and how applicable these were for Scottish waters. The main findings of Verfuss et al. (2019) were that:

- Big bubble curtains (BBCs), the IHC noise mitigation system (IHC-NMS), the hydro-sound damper (HSD), and vibrohammers (VH) have all been commercially deployed as NAS in OWF projects.
- With either the BBC, IHC-NMS, or HSD, broadband sound levels can be reduced by at least 10 dB.
- The NAS are generally more effective at reducing the risk of noise impact on marine mammals and fish sensitive to higher frequencies than on those that are only sensitive to frequencies below 100 Hz.
- VH is an NAS that has been applied in industrial projects in water depths prevailing in potential future Scottish OWF sites (up to 77 m).
- BBC, VH, HSD, and NMS are NAS that have been commercially deployed in OWF projects in water depths up to 45 m.
- The AdBm-noise abatement system (AdBm-NAS) completed its full-scale test in 2018 and deployed commercially in an OWF project in 2019.

Those abatement systems that can be considered Best Available Technology under certain circumstances and some of their limitations are described below.

Recently, piles with diameters of 9 m have been installed, compared with initial piles around 2.5 m. Even 12-m-diameter piles are proposed for the future. Different substrates and water depths require different types of foundations. Driven piles are typically used in sandy soils, such as areas in the North Sea, whereas drilled piles require a higher substrate strength, though substrate strength could be high enough when the drilling gap is filled with a specific filling material. Site-specific calculations are necessary to determine the method of installation.

During pile driving, the acoustic pressure wave can enter the substrate and re-emerge into the water column at a greater distance from the pile ("ground-coupling effect"). Therefore, mitigating the noise emitted through the water near the pile (e.g., by deploying an HSD, AdBm, or IHC-NMS system) may not be sufficient. Big bubble curtains (BBCs) can thus be highly effective, as they can

mitigate noise that reenters the water column at distances up to 150–160 m from the pile. NAS can be categorized as near-pile or far-field systems. It may be necessary to deploy both a near-pile and a far-field system in certain situations for sufficient mitigation.

Primary noise reduction, occurring at the source, has the advantage of solving this substrate transmission problem. Secondary noise reduction occurs once the sound has already been transmitted into the water or substrate. The most effective way to reduce noise at the source is to use a foundation that does not require impact pile driving such as gravity-based, bucket, drilled, or floating foundations.

## Vibropiling

Vibration pile driving, or vibropiling, avoids impulsive sound and thus could be a promising alternative to conventional pile driving. The advantages include lower (but continuous) noise levels, faster installation (and therefore less exposure time), material saved on the monopile, and less mitigation for noise (based on current regulations). There is also considerable offshore experience using vibropiling. First full-scale installations have taken place in the OWF Kaskasi 2 in the German Bight (https://ocean-energyresources.com/2022/04/28/cape-hollands-vlt-completed-work-at-kaskasi-ii-owf-video/), though this installation method is still somewhat experimental. Vibration piling is 10–20 dB lower in peak levels vs. mitigated pile driving. Levels fall to 140 to 145 dB in 8 km for pile driving compared to 1.5–3 km for vibropiling. The area over which these levels are exceeded is thus 7–28 sq. km for vibropiling compared to 201 sq. km for pile driving. Vibropiling, however, produces very low frequencies, so further mitigation using a bubble curtain would not reduce the noise substantially. The noise peaks arise from rattling from the loose connections of the vibrohead. While vibropiling may reduce peak noise levels, the continuous noise still likely causes some impact to marine life, though probably less.

### BLUE Piling

The BLUE Piling hammer replaces the typical steel ram weight of an impact hammer with a large water mass. The resulting blow is considerably longer in duration than a conventional impact hammer, which reduces underwater noise and material fatigue. This could be a cheaper alternative, given the lower material fatigue and less need for costly noise mitigation systems. Whether one can operate the hammer on a large enough scale to handle large monopiles remains to be validated.

## Adaptation of Hydraulic Hammers

By incorporating an adjustable cushion between the ram weight and anvil of either steel or water, the impact force can be prolonged, and the amplitude reduced, resulting in a 4 to 6 dB lower SEL (Koschinski and Lüdemann 2020), while also achieving up to a 60% reduction in material fatigue and stress on the equipment.

Reducing the amplitude of the peak force can, however, result in premature refusal, i.e., when five or more blows of an adequate hydraulic impact hammer will not budge the pile, as the peak driving forces may no longer be sufficient to overcome the soil resistance. Continued operation without this add-on to the hammer or use of a higher energy might then be required which, in both cases, would increase the radiated underwater noise.

The Menck Noise Reduction Unit (MNRU) incorporates a steel cushion block. The IHC-PULSE, currently being developed by IHC, uses a cushioning chamber filled with water. If premature refusal occurs due to insufficient peak forces, the water in the IHC-PULSE system can be drained within minutes and the system operated as a conventional hydraulic hammer to achieve greater peak forces. MNRU can be combined with other noise mitigation methods since the piling cushion is inside the hydraulic hammer.

## Smart Pile Driving

With sufficient engineering preparation and the assistance of online noise monitoring, a piling approach can be followed to minimize noise levels. By optimizing hammer energy and blow frequency in line with real-time noise measurements, noise levels can be reduced while ensuring the pile continues to penetrate at an acceptable rate.

## Bucket Foundations

Suction Caisson/Bucket Foundations are used for low substrate strength (sandy soils, clay, or combinations thereof) and, preferably, a relatively flat seabed. The structure sinks into the soil by its self-weight, and noise barely exceeds background levels. There are installation challenges in very shallow water (water depths $<$20 m). A lower length-to-diameter ratio (and thus a larger "footprint") compared to their deep-water use allows applications for OWFs in shallower water ($<$100 m).

There are two types of bucket foundations: Suction Bucket Jackets (SBJs) and Monobucket foundations. In SBJs, several buckets are connected rigidly to a welded framework structure, whereas the monobucket foundation is a monopile foundation with a suction bucket-seabed interface. SBJ can be used in many substrate conditions, but not very hard soils, soft soils, or rocks. Also, high seabed mobility and loss of substrate could compromise the stability. There is an impact on soil and benthic biotypes. Between 2014 and 2020, 32 three-legged SBJs were successfully installed in depths from 23 to 29 m. A monobucket was also installed successfully in Denmark (Koschinski and Lüdemann 2020).

## Gravity-Based Foundations

Gravity-based foundations, most suitable for depths of up to 50 m, are reinforced concrete or steel/concrete hybrid structures whose stability is achieved by the

submerged weight of the structure, supplemented by additional ballast (e.g., sand). They can also be designed for deeper waters and have been used extensively by the oil and gas industry in depths of up to 300 m. They may have a relatively larger impact on benthic life, however, since at least some types remove the upper layers of the seabed, and their footprint area is larger than that of a monopile.

Crane-free gravity foundations are an example of a noiseless foundation. It is proven technology and is inexpensive. The foundations are self-floating so they do not require cranes or large installation vessels. They are more cost-effective at larger depths and bigger turbines compared to other foundation types. There is no sound emission from the subsea installation process and no deep penetration of seabed. In current prototypes, the base diameter is 31–34 m. Two tow vessels (tugs) pull the vertical pile through the water, and they can be installed in seas up to 2 m. Installation is estimated to take 4 h. Ballast is used so the foundation can withstand high turbine and wave loads. Gravity-based foundations can be designed for life spans of 50 years or more. They require a minimum water depth of 10 m.

## Drilling

There are several offshore foundation drilling techniques for various substrate conditions.

1. BAUER MIDOS-Pile combines mixing and drilling technology to install a structural pile. The drilling and mixing tool is full of grout. This can be used in mainly sandy substrates but also clay and rock. The substrate is mixed with cement and creates a slurry that is injected during drilling. XXL monopoles are too big for this technology, however. The noise is much lower than piling and the structural capacity is better. The substrate must be mixable, e.g., sand with some clay.
2. BAUER Dive Drill Technology is used for the installation of drilled and grouted piles. Drilling occurs inside a casing and is replaced with the pile. Dive Drill Technology installs piles in fully cased boreholes and is suitable for all soil conditions including hard rock. It makes pile driving in marginal soil unnecessary.
3. BAUER BSD 3000 is for drilling piles in rock. The pile is installed and grouted afterward. In 200–300 m, the noise is under background noise (125 dB rms).

## Push-In and Helical Piles

Push-in and helical piles are two concepts for silently driven piles. Both concepts can serve as an alternative for jacket foundation piles and are therefore suitable for deepwater wind turbine foundations. Both have been proven onshore. The push-in pile foundation uses a static force to drive piles into the seabed, and the helical pile foundation uses a rotating motion to drive piles fitted with several helical blades into

the soil. Both concepts are fully silent but will require special tools and, in the case of the helical pile, an interface with the installation vessel using dynamic positioning.

## Floating Wind Turbines

Many waters are too deep for non-floating structures. Floating turbines can be used in all different sediment types. The anchors are fully retrievable, and no effect on marine life has been observed in a prototype wind farm, WindFloat, off the Portuguese coast. The effects of anchors, however, may depend on anchor type. Different anchor types such as gravity anchors, suction buckets, or drilled or driven piles can be used to hold the floating substructure and the wind turbine on top in place. Suction buckets are used most often as anchors. Drag anchors impact the seabed, though they are quiet. Moreover, there are some concerns regarding the high number of vertical mooring lines in important habitats for baleen whales, which could cause harmful collisions.

There are three types of floating wind turbine platforms:

1. Float-stabilized structures or barges
2. Tension leg platforms (TLPs)
3. SPAR buoys

For 1), float-stabilized structures (semi-submersibles), the platform is floating and anchored to the seabed by cables which, in combination with ballast water, provide stability. Semi-submersible floating wind turbines have been deployed in some of the roughest seas of the Atlantic where they survived 17 m waves. Just one tug is needed to place the turbine, and it can be towed up to 500 km.

For 2), TLPs, the wind turbine rests on a platform supported by several cables anchored to the seabed. The cables are tensioned to stabilize the platform by counteracting its buoyancy and to maintain position of the turbine under any type of load. A TLP emits minimal noise. It is best used in $>$20 m water depth. To gain stability, TLPs require a fixed connection to the seabed, which is under tension due to the buoyancy of the floater. Mooring cables come in various types (taut leg, tension leg mooring, etc.). It is a one-step installation, only small tugboats are needed, and little assembly time or seabed preparation is necessary.

For 3), SPAR buoys, used for deepwater applications, the wind turbine is supported on a long concrete or cylindrical steel column, completely submerged, which is ballasted at the bottom to stand upright and to provide stability, withstanding loads produced by the wind and waves.

## Secondary Noise Mitigation

There are currently multiple near-pile NAS on the market compared with far-field systems. Examples of secondary noise reduction include:

1. A big bubble curtain (BBC) consists of a large, perforated pipe positioned around the construction zone. Air is pumped through the hose from both sides using several compressors and is released through the perforations delivering a continuous flow of bubbles around the periphery of the construction zone. From industry experience, BBCs can reliably provide an enclosed ring up to a length of 1000 m (approx. 160 m radius if deployed in a circular layout). BBCs are not limited to the size of the pile being installed since they are deployed at a large distance from the pile. BBCs are extremely effective as they can attenuate acoustic energy that reenters the water column from the seabed, a common drawback of all near-pile systems. Two BBC hoses can be deployed in concentric circles spaced at least the water depth apart, forming what is known as a double big bubble curtain (DBBC) for additional mitigation. The use of (D)BBCs is independent of foundation design and installation vessel. The noise reduction depends on the air supply, water depth, subsea soil conditions, and current/direction/shape. Each deployment therefore needs to take project-specific considerations into account. With an air volume stream of 0.3 $m^3$/min*m (comparatively low), the reduction in SEL for a single bubble curtain was 11–15 dB at 25 m water depth, but only 8–14 dB around 30 m and 7–11 dB at about 40 m (Koschinski and Lüdemann 2020), since the bubble size becomes too small under the pressure of greater depths. Such a loss in mitigation can be overcome by increasing the amount of air or combining it with near-pile mitigation systems like the IHC-NMS, HSD, or AdBm (see below).
2. The IHC Noise Mitigation Screen (IHC-NMS) is a proven near-pile NAS previously applied on piles under 8 m in diameter (though it is being considered for 10-m-diameter piles) and under 40 m water depth. It has been used in hundreds of monopile installations. This system is a double-walled steel pipe with an air gap between the two layers. A multilayered bubble curtain is also used in the center around the pile. Ground-coupling effects are not mitigated, however. Noise reduction is independent of water depth. Noise reductions up to 17 dB SEL have been achieved in water depths up to 40 m (Bellmann et al. 2020).
3. Hydro-sound dampers (HSD), used in hundreds of monopile installations, consist of foam resonators affixed to ballasted nets. HSD baskets, consisting of a net sleeve, container, and ballast, are dropped down into water around the pile, which collapse back up once the pile is installed and the basket is returned. Noise reduction is independent of water depth, and attenuation is predominantly in the low-frequency ranges. HSD are customizable, allowing tuning of the frequency bands in which noise reduction needs to take place by choice of resonator sizes. If the balloons are appropriately sized, their resonant frequencies can align with the peak frequencies of piling noise or with frequency bands of best hearing in sensitive animals for optimum damping effects. Data from multiple OWF projects in Germany suggest the HSD nets reliably achieve reductions up to 12 dB SEL (Bellmann et al. 2020). Overall, the system works for water depths up to 40–60 m and pile diameters up to 8–13 m. It is easily adaptable, weighing very little, and is not affected by water currents. A disadvantage is still the ground-coupling effects.

4. The AdBm noise mitigation system, ready for offshore use, uses rugged Helmholtz resonators whose acoustic properties can also be modified or "tuned" to optimally reduce noise. The system consists of several layers of inverted air-filled chambers, whereby the arrays of each layer are sized according to the depth at which they will be situated during piling. These resonators simply need to surround the sound source, and once they are in place, the resonators will passively absorb the noise. The principle of this system is like that of the HSD nets and also most effective in the low-frequency ranges. The system can theoretically work in water depths beyond 70 m (Verfuss et al. 2019). Although the AdBm noise mitigation system has been used for a full-scale OWF project, very limited data on its acoustic performance have been published.

## BEP for Pile-Driving Noise

Quieting technologies (see the above section on BAT) represent most of the mitigation for pile-driving noise, having great potential to dramatically reduce the area and the number of animals impacted by the noise. Additional BEP can consist of some operational procedures related to the behavior of animals such as deterrence by using acoustic deterrent devices (ADDs) or a soft start for those animals remaining at risk despite noise reduction. There is some debate, however, whether or how much marine life should be purposely displaced at the start of pile driving or over the effectiveness of ADDs or soft starts. Not all animals respond to them as intended, as they may be motivated to remain in an area, despite harassment by noise, because of access to food or mates. FaunaGuard is one device that has been used since research, mostly carried out on harbor porpoises, showed pinger and seal scarers produced more displacement than was necessary. Mitigated pile-driving noise itself can be used but initially at lower energy and/or repetition rate (ramp-up or soft start) to give marine life a chance to flee the area before more harmful levels of noise are generated. As with seismic surveys, the use of MMOs and PAM operators can reduce the risk of exposing marine life to dangerous noise levels. Visual and acoustic monitoring should be used in combination 24 h a day to maximize the probability of detection of wildlife during periods of poor visibility. If this monitoring is deemed insufficiently effective, pile driving should be prohibited during nighttime and periods of poor visibility.

Some examples of best practices for pile driving that have been developed in the United States for the highly endangered North Atlantic right whale are listed below. The full document is available at https://www.nrdc.org/sites/default/files/best-management-practices-north-atlantic-right-whales-during-offshore-wind-energy-construction-operations-along-us-east-coast-20190301.pdf.

- Construction activities with noise levels that could cause injury or harassment in marine mammals must not occur during periods of highest risk for priority species.

- During construction, developers should commit to minimizing impacts of underwater noise on priority species to the full extent feasible through (i) the consideration and use of foundation types and installation methods that eliminate or reduce noise and (ii) the use of technically and commercially feasible and effective noise reduction and attenuation measures, including the use of the lowest practicable source level.
- Developers should commit to carrying out scientific research and long-term monitoring in lease areas to advance understanding of the effects of offshore wind development on marine and coastal resources and the effectiveness of mitigation technologies (e.g., noise attenuation).

As mentioned above, the German Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (BMU) adopted the *Sound Protection Concept.* In it, in addition to the technical noise reduction systems required, the following are also mandatory:

- Modeling of sound level emission for each specific wind farm project
- Restrictions regarding the maximum duration of a piling operation for a single pile
- Restrictions regarding the maximum energy used to drive the piles
- Application of deterrents and ramp-up procedure to avoid injury close to the piling operation
- Measurement and documentation of SELs
- Monitoring of harbor porpoise activity in the vicinity of construction sites
- Requirements regarding a maximum percentage of area that is allowed to be affected, also with due consideration of protected areas or areas and seasons of biological significance

## Conclusions

One of the difficulties in responsibly managing ocean noise pollution is the challenge in detecting the ecosystem and population consequences of underwater noise. There is sufficient evidence that impacts are occurring in at least 150 marine species (around 100 fish and invertebrate species alone – Weilgart 2018), but being able to ascertain exactly to what degree, in which contexts, for which species, and at what sound types and levels these impacts occur remains imprecise. Because of the large natural variability in ocean systems (e.g., in currents, prey availability, chemistry), detecting human-caused changes in ecosystems and populations at all is a daunting task. The ocean is not a controlled laboratory. On top of that, isolating changes that are solely due to ocean noise pollution and not other human-caused stressors such as climate change, overfishing, and toxins is formidable (Weilgart 2023).

As a result, a more precautionary approach is warranted, one of simply turning down the volume of ocean noise pollution, especially in cases where there are ancillary benefits of quieting, such as reducing GHG emissions by finding the

overlap between greater efficiency and less underwater radiated noise in shipping. In this respect, it must be noted that the great variety of quieting technologies and noise abatement systems for pile driving is in stark contrast to the lack of or slow innovation that is occurring for quieter alternatives to the seismic airgun, though more have emerged recently. This may be due to offshore windfarms being a relatively new development compared with seismic airgun surveys, but it does raise questions. Certainly, having governments that are prepared to regulate the offshore windfarm construction for noise, mainly due to the noise-sensitive and protected harbor porpoise, helps, as do European laws, but it is high time that regulators insist on quieter alternatives to airguns, something that seems well within technological capabilities and which would drive innovation. After all, explosions on land to search for hydrocarbons were replaced with Vibroseis™ because explosions were no longer acceptable to humans. Keeping more fossil fuels in the ground would also reduce our need for seismic surveys and cut greenhouse gases. With humans, we do not find the precise point where noise is just tolerable to newborns in neonatal intensive care units, and we do not fund countless studies on exactly how stressed and disturbed they must be to take remedial action – we simply try and quiet the noise, wherever possible and safe to do so. If we value our life-sustaining oceans, we should provide them with the same care and protection.

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# Influencing Parameters on Impact Pile-Driving and Noise Mitigation Measures

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Michael A. Bellmann, Adrian May, Stephan Gerlach, and Patrick Remmers

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**Abstract**

Currently, the impact pile-driving procedure is the most commonly used installation method for foundation structures in offshore wind farms (OWF). Thereby, the foundation structures are driven into the sediment (seabed) by using a hydraulic (impact) hammer. The pile-driving works result in noise immissions (pile-driving noise) in the water body, which can potentially be harmful to marine life, especially to the noise-sensitive species like harbor porpoises (e.g., Lucke, Siebert, Lepper, Blanchet, J Acoust Soc Am 125:4060–4070, 2009; Southall, Finneran, Reichmuth, Nachtigall, Ketten, Bowles, Ellison, Nowacek, Tyack, Aquat Mamm 45:125–232, 2019). However, since 2011, any impact pile-driving activity shall comply with the German noise mitigation values of 160 dB for the 5% exceedance level of the single strike sound exposure level $SEL_{05}$ and 190 dB for the zero-to-peak sound pressure level $L_{p,pk}$ at a distance of 750 m to the source. Up to the end of 2019, more than 2400 piles were installed in Germany (North Sea and Baltic Sea) by using impact piling with and without noise mitigation measures. Most of the underwater noise measurement data and the lessons learnt by using different noise mitigation systems (NMS) and/or noise abatement systems (NAS) were not published or are not publicly available. Within the funded R&D-project NavES (Bellmann, Brinkmann, May, Wendt,

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Gerlach, Remmers, Underwater noise during the impulse pile-driving procedure: Influencing factors on pile-driving noise and technical possibilities to comply with noise mitigation values. Supported by the Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (Bundesministerium für Umwelt, Naturschutz und nukleare Sicherheit (BMU)), FKZ UM16 881500. Commissioned and managed by the Federal Maritime and Hydrographic Agency (Bundesamt für Seeschifffahrt und Hydrographie (BSH)), Order No. 10036866. Edited by the itap GmbH. https://www.itap.de/media/experience_report_underwater_era-report.pdf, 2020), the most influencing project- and site-specific parameters for impact pile-driving noise were identified by a cross-project analysis, based on the German noise registry database MarinEARS (https://marinears.bsh.de). Furthermore, the lessons learnt with applied NMS as well as NAS were summarized. The most influencing parameters on unmitigated pile-driving noise are: (i) pile diameter, (ii) blow energy, (iii) foundation type, and (iv) bathymetry/transmission loss. Currently, only two close-to-pile NAS (Hydro Sound Damper – HSD and noise mitigation screen/cofferdam) as well as one far-from-pile NAS, the single or double Big Bubble Curtain (BBC/DBBC), are offshore reliable and were used in serial applications. With one NAS, a broadband noise reduction between 10 and 16 dB can be achieved. By applying two independent NAS, up to 20 dB for the SEL can be obtained. Slightly higher noise reductions are possible for the $L_{p,pk}$. The effectiveness of all NAS significantly depends on frequency: increasing frequency results in increased noise reduction. But also, other parameters, like water depth or current, partly have effects on the achieved noise reduction. At present, only one reliable and robust NMS – the noise-optimized piling procedure – can be used, which leads to an overall noise reduction of 2–3 dB by halving the used blow energy.

**Keywords**

Percussive pile-driving noise · Noise mitigation · Noise abatement system

## Introduction

The installation of renewable energy sources like offshore wind farms (OWF) has increased rapidly in the last decade. In Germany, more than 20 OWFs were in operation in 2020 (~8.1 GW) and it is planned to reach 30 GW in 2030; therefore, several OWFs are planned to be installed in the years to come. The main goal of all these renewable energies shall go hand in hand with the awareness of sustainable issues and protecting nature and environment. Most offshore foundations are anchored in the seabed by the impact pile-driving procedure, and this leads to impulsive noise emissions, which could harm or will cause a reaction to pile-driving noise by marine life (e.g., Lucke et al. 2009; Southall et al. 2019).

In 2008, the German regulation authority BSH, as first country worldwide, set a dual noise mitigation value criterion to avoid temporal threshold shifts (TTS) for marine mammals, in particular harbor porpoises: (i) The 5% exceedance level of the single strike sound exposure level ($SEL_{05}$ or $L_E$) shall not exceed 160 $dB_{SEL}$ (re 1 μ $Pa^2s$) and (ii) the zero-to-peak sound pressure level ($L_{p,\ pk}$) shall not exceed 190 $dB_{Lp,pk}$ (re 1 μ Pa) at a distance of 750 m to piling. Based on experiences gained from the test OWF *alpha ventus* (2009), the application of noise abatement as well as noise mitigation systems for compliance with the noise mitigation values mentioned above is mandatory in German waters for any pile-driving activity. Furthermore, sufficient monitoring by, e.g., underwater noise measurements, according to the guidelines of the BSH (2011), shall be applied.

The performed underwater noise measurements during unmitigated impact pile-driving, without applying NAS or NMS, have shown values at a distance of 750 m of (i) up to 183 dB for the sound exposure level (SEL resp. $L_E$) and (ii) > 200 dB for the zero-to-peak sound pressure level ($L_{p,\ pk}$); Fig. 1. This will lead to a required overall noise reduction of up to 23 dB for the SEL.

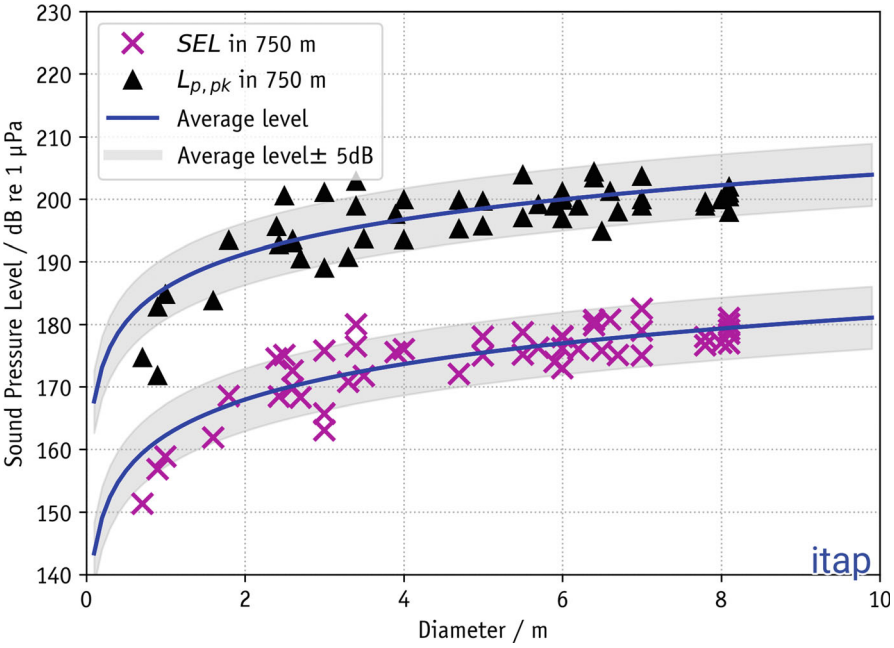



**Fig. 1** Examples of unmitigated pile-driving activities (zero-to-peak sound pressure level – $L_{p,\ pk}$ in dB re 1 μ Pa and broadband 5% exceedance sound exposure levels – $SEL_{05}$ in dB re 1 μ$Pa^2s$) measured by itap GmbH at a distance of 750 m as function of the pile diameter in different European OWFs

Furthermore, during the last decade, guidelines for measuring underwater noise from pile-driving as well as the evaluation of the effectiveness of NAS and NMS were defined on a national level (BSH 2011, 2013; DIN SPEC 45653 2017), but also internationally (ISO 18406 2017), to ensure that empirical data from different OWFs can be compared. The background to this is that all pile-driving activities since 2011 shall be monitored in a comparable manner and integrated into the underwater noise specialist system MarinEARS in Germany.

Most of the underwater noise data form German OWFs as well as the experiences with different NAS/NMS were not published or are not publicly available. Therefore, the funded R&D-project NavES (phase 3), managed by the BSH, was set up in 2016 to perform a cross-project analysis of all available measured underwater noise data in German waters. The aim of this project was to identify the most relevant influencing site- and project-specific parameters on impact pile-driving noise. Furthermore, the experiences with the application of different NAS including the effectiveness of these systems shall be summarized. Based on this cross-project analysis, minimum requirements for the applications of different NAS and combinations of NAS were compiled. In Bellmann et al. (2020), extensive experiences and data from the application of noise mitigation measures during pile-driving procedures in Germany are summarized and made publicly available. The benefit is that the main lessons learnt regarding mitigated and unmitigated pile-driving becomes available in a condensed manner which can be used not only for authorities during the approval process but also support the industry to plan upcoming OWF projects.

## Underwater Noise Specialist Information System MarinEARS

Based on the mandatory extensive underwater noise monitoring during impact pile-driving activities, the BSH developed a specialist information system in 2016 for underwater noise data: the MarinEARS. This database consists of raw data (time recordings) and standardized, post-processed result files of the underwater noise measurements. Additionally, relevant project-specific and technical-constructive information (metadata) of each single OWF construction project are linked to the underwater noise data in a standardized manner for all projects since 2012 in the German EEZ. Such metadata consists of, e.g., geo-reference, pile-driving protocols and description of the used noise mitigation measures, and support an evaluation of the underwater noise data. Table 1 gives an overview of the used measured and post-processed data sets, which were used for the cross-project analysis (Bellmann et al. 2020).

## Site- and Project-Specific Influencing Parameters on Pile-Driving Noise

Based on the cross-project analysis, the site- and project-specific influencing parameters on unmitigated pile-driving noise were determined (Table 2).

**Table 1** Overview of the installed OWFs in the German Baltic Sea and North Sea. For the cross-project analysis (Bellmann et al. 2020), a subsample of all available empirical data was used, since the OWFs erected between 2008 and 2012 in the German EEZ have not yet been implemented in the MarinEARS. Additionally, the OWF projects within the 12-sea-mile-zone were also excluded since these data were not included in the MarinEARS in 2020 either

| Parameters | Value range | Comments |
|---|---|---|
| Offshore wind farms | 21[1] | Baltic Sea: 3 pcs<br>North Sea: 18 pcs |
| Grid connection and other platforms | 28 | Including substations, converter platforms, met masts, and research platforms |
| Water/soil conditions | EEZ of the German North Sea and Baltic Sea | *North Sea*: sands with different densities and thicknesses<br>*Baltic Sea*: sand-, till-, and chalk layers |
| Current | < 0.75 m/s | |
| Water depths | 22–41 m LAT | |
| Number of foundations | 1458 | ~ 80% of the measured underwater noise data were included in the MarinEARS database 2020 and were used for this study |
| Number of piles | 2464 | |
| Pile diameters | 1.829–8.0 m | |
| Type of installation vessel | Floating vessel or jack-up platform | |

**Table 2** Site- and project-specific influencing parameters on pile-driving noise (Bellmann et al. 2020)

| Site-specific parameters | Project-specific parameters |
|---|---|
| Soil conditions | Foundation type |
| Water depth | Pile design/pile diameter |
| Transmission loss, bathymetry, current, and sound velocity | Blow energy |
| Distance between piling and measurement position | Inclination of pile |
| Position in the water column | Hammer type |

## Offshore-Reliable and Market-Ready Noise Mitigation Measures

In case the source level of the impact pile-driving noise can be reduced, such techniques will be named noise mitigation systems (NMS): the only offshore-tested and reliable NMS is currently the "noise-optimized pile-driving procedure" (high blow rate and low blow energy). Besides that, alternative foundation structures and methods like vibro-piling and suction buckets can be used to avoid noise-intensive construction phases, but these methods and foundations mostly produce continuous noise instead of impulsive noise according to the definition of MSFD (2008). Typically, continuous noise will not produce any injury but can cause behavioral effects on marine life (Southall et al. 2019).

Noise abatement systems (NAS) try to reduce the resulting impact pile-driving noise in water to the greatest extent. Until 2014, the German government supported a total of 18 pcs R&D-projects in the field of developing and enhancing NAS and NMS with a total funding of 27 M€ (Verfuss 2014). Additional money was spent by the industry (developer as well as construction companies) to accelerate the development and enhancement of these new technologies and to test them under real offshore conditions.

A general technical overview of technical NAS and NMS as well as possible alternative low-noise foundation structures and installation methods were summarized in 2011 and updated in 2013 and 2019 on behalf of the Federal Agency for Nature Conservation (BfN) (Koschinski and Lüdemann 2011, 2013, 2019). Furthermore, the effectiveness and the costs of an application of NAS and NMS were evaluated by questionnaires and published on behalf of the Scottish Natural Heritage (Verfuss et al. 2019). In Table 3, only the offshore-reliable NAS, which were used in serial application, are summarized with a short description including advantages and disadvantages. However, all NAS currently require a project-specific design.

The achieved broadband insertion losses (overall broadband noise reduction) of an applied single NAS as well as an applied combination of a close-to-pile NAS and Big Bubble Curtain are summarized in Table 4. Therefore, only the offshore-reliable NAS from Table 3 were listed without any technical malfunctions and with its optimized system configuration since the effectiveness of the listed NAS significantly depends on the system configuration, e.g., the amount of compressed air for a BBC. However, also the water depth of the applications of the NAS is listed since the achievable noise reduction significantly depends on the water depth in some cases, especially for the BBC. Due to the fact that the effectiveness of NAS or combinations of NAS are based on a cross-project evaluation, with slightly varying site-specific and project-specific conditions a statistical representation of the minimum, averaged (mean) and maximum achieved noise reduction is summarized in Table 4. The variance of the overall noise reductions achieved by each applied NAS or combination of NAS reflects the importance of the project-specific adaptation and the influence of the optimized system configuration of each system since otherwise the measured variance was much higher and the achieved noise reductions much lower than described in Table 4. Furthermore, the analysis did not explicitly consider the type of the impact hammer. In this respect, the values shown here are for orientation purposes only.

Table 4 also shows the effect of used compressed air on the achievable noise reduction of a single or double BBC: a larger airflow leads to an increased noise reduction. However, Table 4 also indicates that the noise reduction of any kind of bubble curtain also significantly depends on water depth: the deeper the water, the less overall noise reduction. In Bellmann et al. (2020), a specification list for single and double BBC system configurations regarding the state-of-the-art is summarized.

Nevertheless, the achieved overall broadband noise reductions summarized in Table 4 for NAS and combinations of NAS are highly dependent on frequency. The lower the frequency, the lower the achieved overall noise reduction in the specific 1/3-octave band for all existing NAS; Fig. 2. In principle, the most acoustic energy is

**Table 3** Overview of offshore-robust and reliable noise abatement systems that were used in serial application within German waters, including a brief description with advantages and disadvantages

| Secondary, technical noise abatement system | Description/experiences | Advantages/disadvantages |
|---|---|---|
| **Big Bubble Curtain (BBC)** | Impedance differences due to air-water-mixture<br>Single or double BBC as far-from-pile system<br>Used until 40 m water depth and 9 m pile diameter | Pros:<br>Applicable for all foundation types including UXO clearance<br>Independent from installation vessel<br>Cons:<br>Noise reduction depends on water depth, air amount, and system configuration<br>Requires project-specific adaptation<br>Vessel + compressors + experienced crew required |
| **Noise Mitigation Screen (IHC-NMS)** | Shell-in-shell-system near-to-pile<br>Used up to 40 m water depth and 8 m pile diameter<br>Currently applicable only for monopiles | Pros:<br>Noise reduction independent from water depth, current, and bathymetry<br>Including pile-guiding tool<br>Cons:<br>Self-weight and size<br>Ground couplings possible<br>Application in different water depths challenging |
| **Hydro Sound Damper (HSD)** | Resonator principle based on HSD elements close-to-pile<br>Used up to 40 m water depth and 9 m pile diameter<br>Currently applicable only for monopiles | Pros:<br>Noise reduction independent from water depth, current, and bathymetry<br>Lightweight system<br>HSD elements are tunable (preferably for low frequency)<br>Cons:<br>Ground couplings possible<br>Limited lifetime of HSD elements<br>Vessel-specific lighting tool required |

emitted by pile-driving in the frequency range between 63 and 250 Hz, depending on the site-specific conditions and the used pile design: the bigger the pile diameter, the lower the unmitigated pile-driving spectrum.

Examples of the outputs of the MarinEARS system and the effectiveness of a "noise-optimized piling procedure" are depictured in Figs. 3 and 4. Figure 3 shows an example of the OWF *Butendiek* in the North Sea (year of construction 2014). All sound exposure levels ($SEL_{05}$) measured at a distance of 750 m to the monopiles are summarized in chronological order. For the first monopile installations, only one noise abatement system was used. A second noise abatement system was applied in parallel starting with monopile installation #9 due to the permanent noncompliance with the German noise mitigation values. Thereafter, the sound exposure level complies with the noise mitigation values as 1st OWF

**Table 4** Summary of the achieved overall noise reduction of single noise abatement systems and combinations of noise abatement systems based on a cross-project analysis (Bellmann et al. 2020). Only offshore-reliable NAS and combinations of NAS are listed in its optimized system configuration and applied water depths since the effectiveness depends on these project- and site-specific parameters, especially for any application of BBCs

| No. | Noise abatement system resp. combination of noise abatement systems (applied air volume for the (D)BBC; water depth) | Insertion loss ΔSEL [dB] (minimum/average/maximum) | Number of foundations |
|---|---|---|---|
| 1 | Noise Mitigation Screen (IHC-NMS) (different designs) (water depth up to 40 m) | $13 \leq 15 \leq 17$<br>Latest generation<br>$15 \leq 16 \leq 17$ | $> 450$<br>$> 65$ |
| 2 | Hydro Sound Damper (HSD) (water depth up to 40 m) | $10 \leq 11 \leq 12$ | $> 340$ |
| 3 | Optimized double BBC[a] ($> 0.5$ m$^3$/(min m), water depth ~ 40 m) | 15–16 | 1 |
| 4 | Combination IHC-NMS + optimized BBC[a] ($> 0.3$ m$^3$/(min m), water depth $< 25$ m) | $17 \leq 19 \leq 23$ | $> 100$ |
| 5 | Combination IHC-NMS + optimized BBC[a] ($> 0.4$ m$^3$/(min m), water depth ~ 40 m) | 17–18 | $> 10$ |
| 6 | Combination IHC-NMS + optimized DBBC[a] ($> 0.5$ m$^3$/(min m), water depth ~ 40 m) | $19 \leq 21 \leq 22$ | $> 65$ |
| 7 | Combination HSD + optimized BBC[a] ($> 0.4$ m$^3$/(min m), water depth ~ 30 m) | $15 \leq 16 \leq 20$ | $> 30$ |
| 8 | Combination HSD + optimized DBBC[a] ($> 0.5$ m$^3$/(min m), water depth ~ 40 m) | 18–19 | $> 30$ |
| 9 | "Noise-optimized" pile-driving procedure (additional additive NMS) | ~ 2–3 dB per halving of blow energy | |

[a] (D)BBC applications were performed by current $\leq$0.75 m/s to avoid unwanted drifting effects

project in German water except for a few installations with technical problems with one of the applied NAS.

Figure 4 shows an example of an OWF construction project in 2016 which used a combination of a near-to-pile and a far-from-pile noise abatement system and a noise-optimized pile-driving procedure. This method includes active feedback between the measured pile-driving noise at 750 m with real-time techniques and the applied blow energy for each single strike by the hammer operator. By combining different mitigation measures, the German noise mitigation value can be complied with and the construction time can be optimized as well, as shown in Fig. 4.

## Discussion

Figure 5 summarizes a typical mitigated and unmitigated impact pile-driving noise of a monopile installation in shallow water ($< 40$ m), measured at a distance of 750 m to piling. Additionally, the measured permanent background noise at the same

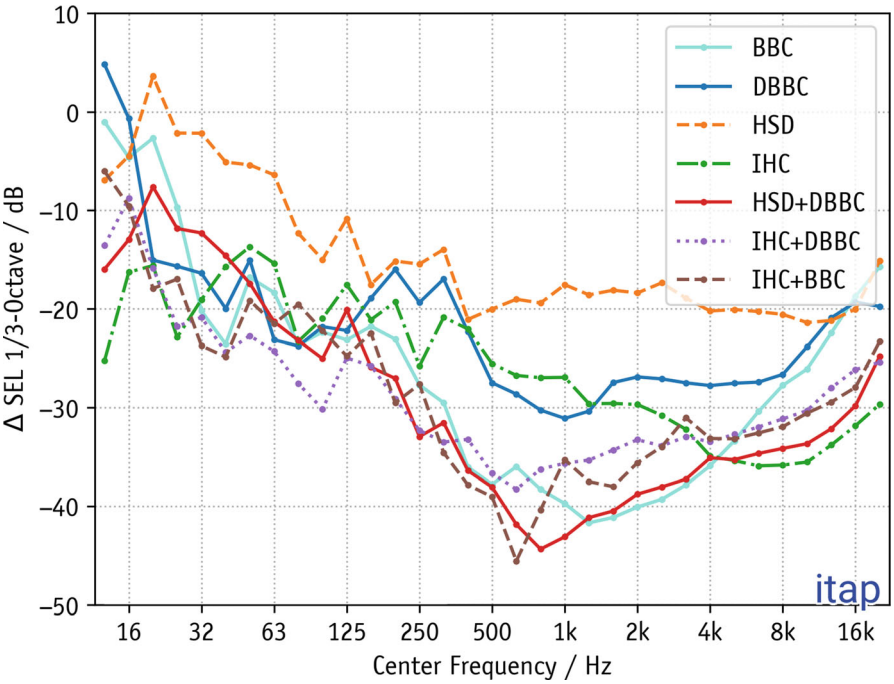



**Fig. 2** Resulting insertion loss of the offshore-reliable noise abatement systems tested in German OWF projects (Bellmann et al. 2020). Note: The differences of mitigated subtracted by the unmitigated pile driving activities are shown which indicates a higher noise reduction for lower dB values on the y-axis

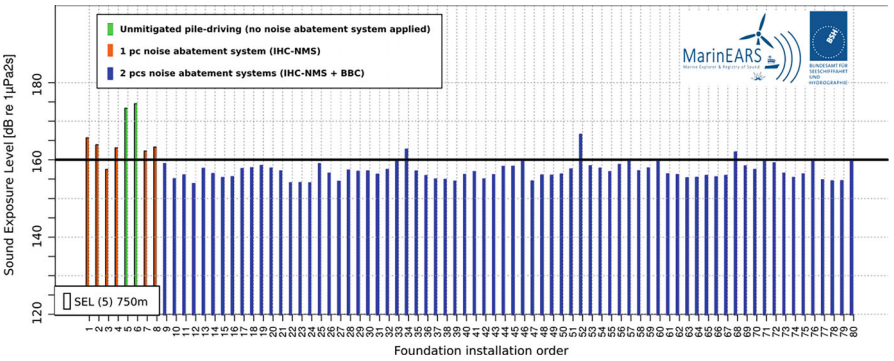



**Fig. 3** Sound exposure level resp. exceedance level ($SEL_{05}$) measured at a distance of 750 m to the respective monopile of the OWF Butendiek in the North Sea in chronological order. Due to the exceedance of the German noise mitigation values, a second NAS was applied starting with monopile installation #9. Additionally, two unmitigated monopile installations were performed to evaluate the effectiveness of the applied NAS combination (Source: https://marinears.bsh.de)

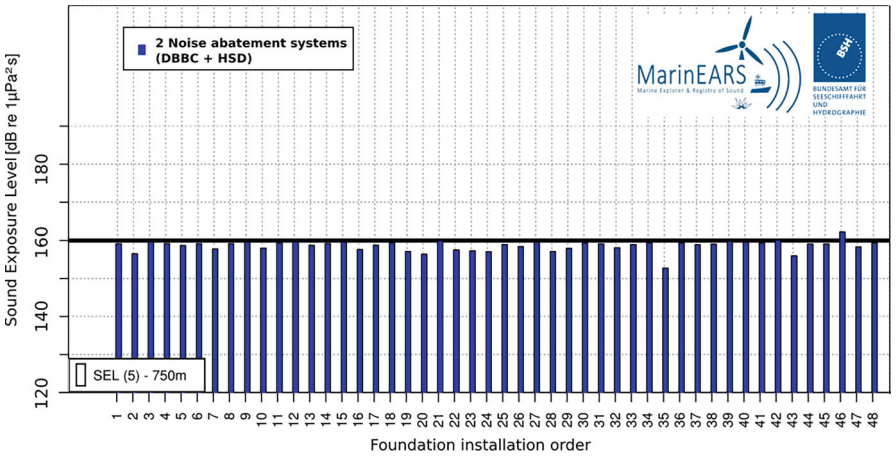



**Fig. 4** An example of another OWF project within 2016 in the North Sea is summarized in chronological order with an application of two NAS and a noise-optimized pile-driving procedure (reduced blow energy and real time underwater noise monitoring) in comparison to the OWF in Fig. 3 (Source: https://marinears.bsh.de)

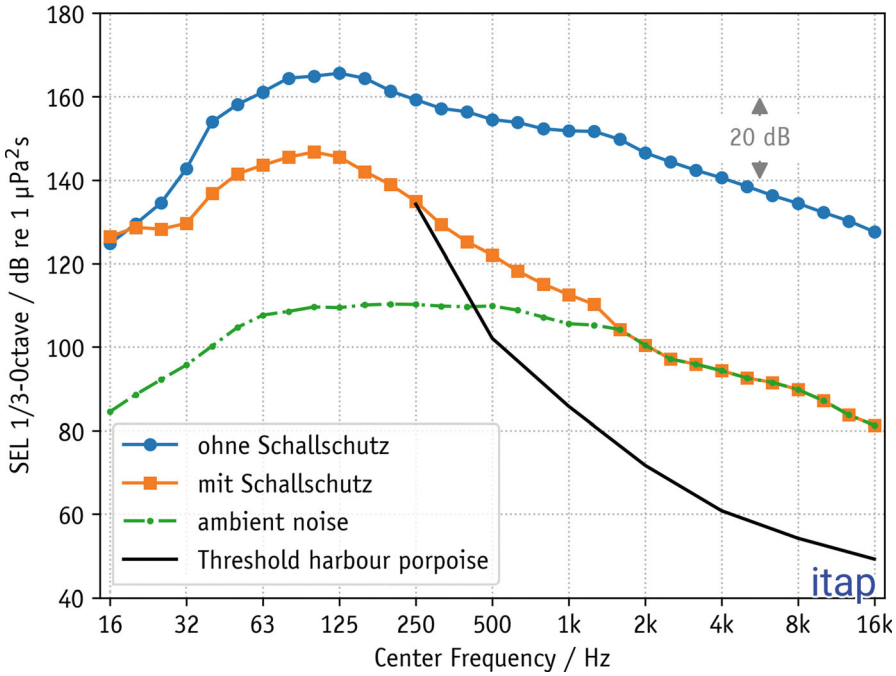



**Fig. 5** Mitigated and unmitigated pile-driving noise 1/3-octave spectrum, measured at a distance of 750 m to a large monopile installation. Moreover, the measured permanent background noise at the same positions as well as the absolute auditory threshold of the harbor porpoise (Lucke et al. 2009) is shown (Source: Bellmann et al. 2020)

measuring position prior to piling as well as the absolute auditory threshold of a harbor porpoise (Lucke et al. 2009) is shown.

Figure 5 shows, as expected, that the noise reduction achieved by applying a combination of two NAS significantly depends on frequency: the higher the frequency, the higher the insertion loss. The signal-to-noise ratio between impact pile-driving and the ambient noise decreases significantly with increasing frequency. This also explains why the determined insertion loss of all NAS and combinations of NAS (Fig. 2) decreases towards a few kHz and a larger variance in the determined insertion loss curves at several kHz occurs. This means the insertion loss of NAS in Fig. 2 might be biased by the ambient noise in the high frequency range (greater than several kHz).

However, studies like Southall et al. (2019) indicate that the individual hearing ability of any marine species should be taken into account by an evaluation of the influence of noise entry into water regarding permanent/temporal threshold shifts as well as reactions of marine species to underwater noise. This might lead to significantly different evaluations of the tested NAS and combinations of NAS with partly much higher overall achieved noise reduction, depending on the observed species, since the mitigated pile-driving noise might not be harmful or audible for some species even at a distance of 750 m to pile-driving.

However, new noise mitigation systems like the new hammer technologies Menck Noise Reduction Unit (MNRU) and Piling Under Limited Stress Equivalent of IQIP b. v. (PULSE) are coming onto the market, which lead to a prolongation of the impact duration and a decrease of the peak force (pulse lengthening techniques). Such new technologies might increase the overall achievable noise reductions of currently available NAS or combinations of NAS. First tests of the new techniques showed that in principle, a reduction of the source level by several decibels is likely. First empirical evaluations of the achieved overall noise reduction will be available in 2023.

Furthermore, new NAS as well as enhanced offshore-reliable NAS are currently under development and will hopefully be tested within the next years, especially in Germany, since the turbine sizes of OWTGs will increase significantly. Larger turbine sizes lead to larger foundation structures in deeper water, so that a compliance with the German noise mitigation values might become more challenging for the upcoming OWF projects.

**Acknowledgments** The project NavES (Phase 3) was funded by the German Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (BMU) and commissioned by the Federal Maritime and Hydrographic Agency (BSH). Special thanks go to all itap seafarers as well as to all involved developers, transport and installation contractors, and noise abatement/system suppliers for the good and proactive support.

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# Acoustic Disturbance Risk Estimates and Mitigation Strategies: An Animal-Centric Approach

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Rianna E. Burnham

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**Abstract**

Sounds of anthropogenic origins are becoming increasingly prevalent in aquatic soundscapes. Acoustic disturbance of individuals, populations, and species is, therefore, increasingly a threat to their success and survival. To better understand the level of risk and implications of elevated sound exposure, the application of a more animal-centric approach is described. Indeed, it is believed that the design of mitigation measures could also follow this approach to diminish the effects. The key to taking the species-centered approach is to first consider the sound frequencies that are most relevant to their acoustics use, including those used for

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conspecific communication, auto-communication such as echo- or electro-location, and acoustic cues interpreted to understand their surroundings. Then, determine how the anthropogenic sound input masks these ecologically important signals, effectively reducing the individuals' active space. Metrics of exposure and the application of hearing thresholds and audiogram data are also explored. Further, how these can be used to estimate the potential for behavioral modifications, physiological stress, or injury is considered. A change in the interpretation of human-made sound sources in the environment is suggested, whereby anthropogenic acoustic additions are considered more than simply a polluter to an individual's niche but as a competitor for acoustic space, habitat, and prey.

**Keywords**

Acoustic disturbance · Noise implications · Behavioral changes · Threshold shifts · Species driven · Active space

## Introduction

The introduction of anthropogenic sound sources into aquatic and marine habitats is increasing. These human-made sounds have the capacity to shape soundscapes and elevate ambient sound levels over a variety of spatial and temporal scales. Although the additions from vessels to sound fields has been well described, there are a large variety of on-water and coastal activities that elevate ambient sound levels. These include, but are not limited to, resource extraction, such as offshore drilling (Blackwell et al. 2004; Wyatt 2008), pile driving (Bailey et al. 2010), dredging (Reine et al. 2014), seismic exploration (Nieukirk et al. 2004; Madsen et al. 2006; Turner et al. 2006; Hildebrand 2009), and sonar use (Watts 2003; Ainslie 2010). Coastal construction and airborne sources have also been characterized, including airplanes and helicopters (Wyatt 2008) or float planes (Burnham and Duffus 2019; Moore et al. 2022).

The acoustic sense is the principal means by which aquatic species send and receive information about their surroundings. The use of sound by animals that contains information regarding an individual's position and surroundings, or physical and emotional state has been well described (Seyfarth and Cheney 2003; Townsend and Manser 2013; Filippi 2016), especially for cetacean species (Au and Lammers 2007; Charles 2011; Filatova et al. 2013; Rendall and Owren 2013; Sayigh 2013). Evidence is also accruing for a similar use of acoustic signaling by fish species (Bass and McKibben 2003; Amorim 2006; Ladich 2014, 2019). This chapter is primarily focused on cetaceans, as the available literature predominantly has focused on the effects of changing soundscapes on whale, dolphin, and porpoise species (Burnham 2023; Burnham and Duffus 2023). However, there are significant implications for other aquatic species. The study of these species could add much to the analyses and management of noise, and so examples are added to the chapter where appropriate.

Marine animals use both passive (Deecke et al. 2002; Gannon et al. 2005; Allen 2013; Cure et al. 2013) and active sensory processes to extract information from the

environment. This adds to the body of information that guides navigation and prey detection; location of conspecifics and identification of the intrusion of another species; defense or territory definition; social hierarchy (e.g., fish species, Amorim 2006); and the search and competition for a mate. Calling can also help maintain group cohesion (e.g., in fish species, van Oosterom et al. 2016) and conspecific contact. Specifically, calls carry messages of warning or alarm, for example, to relay the presence of a predator to conspecifics (e.g., gobies *Asterropteryx semipunctatus,* McCormick and Larson 2008), hierarchy or aggression (e.g., humpback whales *Megaptera novaeangliae*, Seger et al. 2016), habitat suitability or prey abundance (e.g., bottlenose dolphin *Tursiops truncatus*, King and Janik 2015; humpback whales, Stimpert et al. 2011a, b; Parks et al. 2014; killer whales *Orcinus orca,* Ugarte et al. 2006, Deecke et al. 2011), or appeasement or social-sexual activities (e.g., gray whales *Eschrichtius robustus,* Youngson and Darling 2016).

Species have their own distinct call repertoires, with call types or parameters adapted to deliver information in different applications. Call patterning, and timing of pulses, project meaning and is used as part of mating displays (e.g., fin *Balaenoptera physalus*, Croll et al. 2002; Širović et al. 2013; bowhead *Balaena mysticetus,* Johnson et al. 2015; humpback whales*,* Payne and McVay 1971; Darling et al. 2006). Social calls guide courtship or agonistic behaviors (e.g., dolphin species, Dudzinski 1996; Herzing 1996; Díaz López 2011; gray whales, Youngson and Darling 2016). Vocalizations represent caller context, physiological state, or behavioral or social state (Dunlop et al. 2022), and identify group members (e.g., dialects for killer whales, Ford 1987, 1989, 1991; codas for sperm whales *Physeter macrocephalus,* Weilgart and Whitehead 1997) or individuals' identity (e.g., dolphin species, Caldwell and Caldwell 1965; Janik and Sayigh 2013). Those, in turn, are employed for group reunion or, for example, using "motherese" calls between mothers and their young (e.g., pinnipeds, Insley et al. 2003; Charrier et al. 2010; gray whales, Ollervides 2001; Charles 2011; Burnham et al., 2018; Burnham and Duffus 2020, 2022). The transfer of information from a signaling animal to a receiving animal can influence the current or future behavior of one or both individuals. In addition, for echolocating species, the echoes from their own acoustic projections returns information including target type, distance, orientation, direction, and speed of movement (Culik 2011).

However, the reliance on the acoustic sensory modality leaves aquatic species vulnerable to changes in their surrounding soundscape. Any interference with their ability to detect sounds has the potential to generate adverse effects (Popper et al. 2020). Human-generated acoustic additions from our activities in aquatic ecosystems are increasingly being recognized as potential disturbance that may injure or even lead to the death of an animal. To fully appreciate the implications of changes in the soundscape for aquatic species, taking a more "animal-centric" approach is advocated for (Burnham 2017; Popper et al. 2020). Species-driven examples will be used to illustrate suggested approaches and metrics to assess the potential risk and subsequent cost of acoustic disturbance. Much of the work done to date has focused on the noise implications for marine mammals, and so the examples will predominantly come from these species. Increasingly, work is being done to characterize the impacts on fish and invertebrate species, and these will be indicated where possible.

Interest in these species has been taken to understand the effects on populations in their own right and as prey species for upper-trophic species. Following this, research priorities and regulatory actions that reduce the effects of anthropogenic noise, while taking the perspective of the animal or ecosystem, will be discussed.

## Anthropogenic Noise as a Pollutant in the Soundscape

The term "soundscape" describes the acoustic landscape of an area. It was first used by Schafer (1977), who recognized that locations had their own distinct acoustic "sense of place" created by unique tones and "soundmarks." The natural components of the sound field are the non-biological, geophonic components and the biophonies. Geophonies include the acoustic additions from wind, water turbulence, tides, currents, waves, and precipitation. These account for great variation in the background ambient sound conditions (Richardson et al. 1995; Wysocki et al. 2007; Coers et al. 2008; Lugli 2010; Ladich 2013). The biophonies include invertebrate, fish, or marine mammal vocalizations, or acoustic byproducts of their behaviors, which at times dominate certain frequency bands (Zelick and Mann 1999; Cato and McCauley 2002; Myrberg and Fuiman 2002). Anthropogenic sources of underwater sound result from human use of waterways and their surrounding coastlines. The introduction of the mechanically driven vessels and the reliance on oceanic shipping has increased the anthrophonic component of the soundscape and raised the ambient sound level substantially. Pervasive noise from vessel traffic as well as coastal and pelagic construction and resource harvesting can propagate to areas far removed from the source (Jasny 2005; NRC 2005; Hildebrand 2009; Frisk 2012; Gillespie 2016; Miksis-Olds and Nichols 2016).

Vessel noise additions can propagate over great distances but are focused around shipping routes and in shallow coastal waters around cities with major ports and marine-based tourism (Andrew et al. 2002; Malakoff 2010; Frisk 2012). Distant commercial shipping noise dominates the low frequencies, below 100 Hz. However, propeller cavitation and propulsion noise can radiate sounds up to 100 kHz (Ross 1976; Gray and Greeley 1980; Richardson et al. 1995; Arveson and Vendittis 2000; Veirs et al. 2016). In the mid to high frequencies, recreational vessels dominate in the additions to sound spectra, as well as emissions from construction or resource exploration, extraction, and production, which adds sounds over a wide range of frequencies at a variety of intensities (Schoemann et al. 2022). The addition of anthropogenic noise to sound fields has been likened to a pollutant, causing habitat degradation and changes to species diversity, distribution, and habitat use.

## Animal Response to Human-Derived Noise

The impacts of elevated ambient levels or introduced human-derived sound on aquatic species can range from benign to fatal: startle and avoidance behaviors, stress responses, call masking, hearing threshold changes, and tissue damage have

been noted. In extreme cases, fatality may ensue (Casper et al. 2012; Halvorsen et al. 2012; Hawkins and Popper 2018). Therefore, the implications of anthropogenic additions to the sound field can range from sublethal to those that have short-term implications for energy budgets, long-term physiological stress, or permanent hearing damage (Southall et al. 2007, 2018). Growing concern around the impacts of human-made sound additions has increased the research and marine environmental management efforts in this area. Acoustic disturbance is now listed as a threat to the success and survival of several threatened and endangered species (e.g., killer whales, NMFS 2005; DFO 2021; belugas *Delphinapterus leucas*, DFO 2012; right whales *Eubalaena glacialis*, Oleson et al. 2020).

## Masking

Masking is a sublethal effect where an individual is unable to detect a signal of interest in the presence of elevated sound levels and/or interfering signals or noise. In this case, an animal is subject to sound levels that interrupt calling or the reception of biologically significant sounds (Clark et al. 2009; Dooling and Blumenrath 2014; Erbe et al. 2016; Dooling and Leek 2018). The extent to which the frequencies of the human-made sound additions overlap with the vocalization frequencies of aquatic animals provides an indication of the potential for disturbance. The masking effect is also defined by the level of exposure and the proximity to the sound source (Clark et al. 2009; Erbe et al. 2016; Holt et al. 2017). Audibility of conspecific calls is a function of distance between both the interfering sound source to the caller and the source to the receiver, as well as the distance between the two conspecifics (see the "maskogram" by Erbe et al. (2016)).

## Behavioral Responses

The addition of anthropogenic sounds whereby they become a "pollutant" can induce modified swimming behaviors, including altered speed, direction, depth, altered diving patterns, either on the dive and/or on the surface, and evasive or avoidance behaviors away from the sound sources (Norris 1994; Gordon and Moscrop 1996; Frankel and Clark 1998; Croll et al. 2001; NRC 2003; 2005; Weilgart 2007; Erbe et al. 2019). Startle and aversion behaviors have also been noted, with the abandonment of the area around the source observed for short to prolonged time periods (e.g., dolphin species, Lusseau et al. 2006; fish species, Popper and Hastings 2009; Hawkins and Popper 2018; gray whales, Malme et al. 1983, 1984; killer whales, Morton and Symonds 2002; zooplankton species, Slotte et al. 2004).

The impact can be spatially far-reaching depending on the frequency of the sound source and temporally dependent on the duty cycle, if any, of the source. For example, sounds exceeding 120 dB re: 1 μPa initiated startle and avoidance responses in a continuous construction-based sound exposure experiment (gray whales, Malme et al. 1983, 1984, 1986, 1988; Richardson et al. 1995). Substantial area avoidance was noted when the sound level was increased to 160–170 dB re:

1μPa (Malme et al. 1983, 1984, 1986, 1988). These levels have become accepted thresholds for marine mammal behavioral responses. Displacement from migration routes and critical breeding, young-rearing or feeding areas, and the cessation of these behaviors have also been noted as a result of anthrophonic acoustic disturbance (Malme et al. 1983, 1984; Morton and Symonds 2002). However, a behavioral response is not only predicted by the received sound level. The conditions, timing and duration of exposures, animal activity, and its habituation to the sound all help determine the response. The animal's experience, motivation, and conditioning based on previous exposure to the sound may also play a role (Southall et al. 2018).

An acoustic response has also been noted in several species to anthropogenic additions to the soundscape. Calling behaviors are altered to help signal projection overcome increased sound levels. Animals may adjust their vocalizations, displaying Lombard-effect responses by calling louder, longer, or more repetitively, or by changing the energy or frequency of the vocalization to overcome the additions (Nowacek et al. 2007; Weilgart 2007; Tyack 2008; Holt et al. 2009; Rolland et al. 2012; Janik 2014). Increased call amplitude can match sound level increases (e.g., beluga whales, Scheifele et al. 2005; humpback whales, Dunlop et al. 2014; killer whales, Holt et al. 2009, 2011, 2015, 2021). Changes in call structure, including altering the frequency content of the call to minimize spectral overlap with the human-derived sound signal, have also been noted for several species (e.g., right whale, Parks et al. 2007; beluga whales, Lesage et al. 1998; common dolphins *Delphinus delphis*, Ansmann et al. 2007). Repetition of calls or call components also increases in elevated ambient sound levels and particularly with a masking sound present (e.g., beluga whales, Lesage et al. 1998; bottlenose dolphins, Buckstaff 2004; gray whales, Dahlheim and Castellote 2016; killer whales, Foote et al. 2004; Holt et al. 2009; harp seals *Pagophilus groenlandicus,* Serrano and Terhune 2001; humpback whales, Noad et al. 2017). In humpback whales, Dunlop et al. (2010) also noted that whales demonstrated more surface activity and breaching, pectoral slapping, and tail lobbing, which would result in acoustic but nonvocal signals in the presence of human-made sounds. This was a different response to that seen for wind-dependent ambient sound increases, where song source level was increased (Dunlop et al. 2014).

## Physiological Changes and Hearing Threshold Shifts

Discomfort, pain, stress, or injury can result for aquatic life from elevated sound exposure. The sound level may be enough to alter sensitivity in hearing acuity and cause a threshold shift. This acoustic trauma can be mild, where an individual suffers pain and recoverable hearing loss, also frequently referred to as a temporary threshold shift (TTS). The onset of TTS in hearing is the short-term elevation of hearing thresholds and can be described as auditory fatigue, with levels set for pulsive tones, non-impulsive and octave-band sound levels (Southall et al. 2017). Sounds causing TTS can affect the sensitivity of sensory hair cells in the inner ear, modifying the chemical environment or blood flow to these cells; displace or overstimulate inner

ear membranes; decrease sensory neural inputs; or induce metabolic exhaustion (Kryter 1994; Ward 1997; Nordmann et al. 2000; Hastie et al. 2003; Southall et al. 2007).

Moderate hearing loss results from damage in the middle ear, and finally severe damage to the hearing apparatus in the inner ear is seen for great acoustic traumas where permanent threshold shifts (PTS) would occur (Ketten 1995, 2002; Southall et al. 2007, 2018). The onset of PTS could be considered to be the point where tissue damage results from sound exposure (Southall et al. 2017). Permanent injury may be experienced as irreparable damage to sensory hair cells of the inner or outer cochlea, or swelling of the hair cells, neural connections, and vascular system of the cochlea leading to anatomical changes. In more extreme cases, loss of differentiation within the cochlea and degeneration of the auditory nerve may occur (Hastie et al. 2003). This is often followed by neuronal losses and persistent chemical and metabolic cochlear abnormalities (Saunders et al. 1991; Ward 1997; Yost 2000; Southall et al. 2007).

Much of what is known about TTS and PTS is from observation and study of short-term responses from acute exposure. Ketten (2002) considers the onset of PTS to be more general and less species-dependent than TTS, with signal rise time and duration of peak pressure being the more significant factors. PTS is also estimated from TTS-onset, whereby permanent loss is commonly thought to result from longer duration exposure or sudden, intense onset of a broadband source, particularly in the higher frequencies (Ketten 2002). However, the loss of hearing and progressive PTS in individuals is likely a complex nonlinear process, with little known about the physiological response of individuals or populations to long term or chronic elevation in ambient sound levels.

A stress response to elevated sound exposure has also been demonstrated in aquatic organisms. It has been observed through direct testing in the presence of a sound stimuli (e.g., humpback whales, Mingramm et al. 2020) and inferred from testing in periods when sound levels have been reduced. Rolland et al. (2012) demonstrated a reduction of the stress response in north Atlantic right whales with the decrease of commercial traffic by testing stress hormone levels opportunistically after September 11, 2011, when ocean commerce was completely halted. Similar responses have been noted during the COVID-19 pandemic "anthropause" (Rutz et al. 2020), where notable reductions in oceanic soundscape levels in some areas have been recorded (Gabriele et al. 2021; Notteboom et al. 2021; Ryan et al. 2021). During the pandemic, both regional and international movements of people and goods were altered (Baldwin and Tomiura 2020), with maritime trade reduced by approximately 7–10% (Baldwin and Tomiura 2020; Bates et al. 2020; Rutz et al. 2020). Recreational and tourism-based vessels were also reduced (Gabriele et al. 2021).

## Animal-Centric Risk Assessments

To better determine the potential impact of anthropogenic sound sources on individuals or species and, therefore, the most effective means of mitigation, it is not sufficient to consider the acuity or severity of the exposure in terms of changing

sound levels over time. More species-specific metrics should be considered and take a more animal-centered approach (see Popper et al. 2020). The focus under this ideal is first to determine the presence and level of adverse effects that an individual, group, or species will be subject to if they experience a certain sound source at a particular level over a given time. The risk of auditory disturbance and adverse impacts from human-made sound sources to an individual species would be determined by the characteristics of the sound and the level of exposure. This would include a need to measure the animals' presence and residency time in the area with elevated sound levels. The implications of this disturbance to the individual also needs to include the consideration of the importance of that area to the individual or species, and whether it is critical to important life history events like breeding or feeding.

## Auditory Sensitivity

Identifying areas and level of risk of sound exposure includes first distinguishing sound sources that might elicit a response, and then assessing the exposure and dose-response of individuals or groups to the sound (Popper et al. 2020). The audibility of a sound depends on both its amplitude and frequency spectrum. Defining the hearing abilities of an animal and its vocalization and hearing range is, therefore, the first means of describing the disturbance risk (Ellison et al. 2012). The hearing range of a species provides a contextual setting, where an individual is more likely to respond to a sound in that frequency range, along with the assumption that the higher the level above the described sensation level, the greater the potential response.

Auditory thresholds and audiograms have been estimated using behavioral or electrophysiological techniques, showing variation in functional frequency and absolute sensitivity on the species and individual level (Fay 1988; Richardson et al. 1995; Szymanski et al. 1999; Wartzok and Ketten 1999; Ketten 2002; Houser et al. 2008; Supin et al. 2003, 2004; Parks et al. 2007; Tubelli et al. 2012, 2018; Branstetter et al. 2017, 2021). However, this has been measured for only a few aquatic species (Rice 1998; Erbe et al. 2016; Southall et al. 2017). Work has so far focused on toothed whale and pinniped species (Erbe et al. 2016). However, even for these species, testing is still lacking in over 20 delphinid species, 20 beaked whale species, and 18 phocid species. Testing is lacking for all baleen whale species, and little knowledge of hearing sensitivity curves or anatomy and physiology is available for these species (Erbe et al. 2016). Where data for a species does not exist, data from surrogate species within the same functional hearing group are used to create an estimate (Hastie et al. 2003; NOAA 2013). Failing this, 1/3 octave bands are commonly used as an estimate of the acoustic energy within a signal (Payne and Webb 1971; Clark et al. 2009; Hildebrand 2009; Jensen et al. 2009). Giving priority to the frequency range that species-specific calls are made can also guide the animal-centric perspective in assessing disturbance risk and is especially relevant for evaluating the potential for masking (Clark et al. 2009; Hatch et al. 2012; Burnham et al. 2023).

Frequency weighting for aquatic animals based on audiograms has been suggested to tie the sound exposure back to the animal. Differences in hearing capabilities between species are accounted for through a weighting function based on their audiogram. A generalized frequency-weighting for marine mammals, called "M-weighting," for example, has been developed with refinements for mysticetes, odontocetes as mid- or high-frequency species, and pinnipeds listening in air or water. However, these nominal audiograms may not predict well how species will specifically be able to hear certain sounds or indeed their reaction to them. Additional audiometric methods can help determine the frequency discrimination, intensity discrimination, frequency selectivity, duration discrimination, stimulus generalization, and directional hearing of individuals, or an average of individuals to represent a species (McFadden et al. 2022). Also, the critical ratio, as the minimum signal-to-noise ratio for detecting a tone against the background ambient, represents the minimum level a sound needs to be present to just be detectable and so is the lowest level of an individual's hearing sensitivity.

## Active Space

The "active space" of an aquatic organism describes the furthest extent or area of its auditory detection and discrimination. This could be communication signals between conspecifics, where the acoustic range is defined by the area over which a signal may be projected from one animal, and be received and enacted on as intended by another. This may also be described as the communication range or distance (Clark et al. 2009). Active space can also be defined by auto-communication processes, described as the area over which an animal can send a signal and then receive its own echo as part of an echo- or electro-location process (Maynard-Smith and Harper, 2003). In the context of cetacean bio-sonar, this is the echolocation range (Au et al. 2004). However, the active space should also be considered as the area over which an animal is able to send and receive signals in a functional way; this may be the active use of acoustic signals through communication, or interpretation of echoes, but can also include "soundmarks" (Schoemann et al. 2022) and cues from the environment or conspecifics received passively or through "eavesdropping" which supports orientation, navigation, and habitat choices (Slabbekoorn and Bouton 2008; Allen 2013).

Active spaces are dynamic and reflect the variations in signals produced and the sound field they are projected into. The extent is defined by many variables. For example, signal production, transmission, and properties of the signal itself, including the frequency, amplitude, length, and modulation. As well, the signal modifying properties of the environment, and complexity of the acoustic surroundings are significant influences (Bradbury and Vehrencamp 2011).

An appreciation of the active space of an individual, and its dynamic nature, provides insight into how an animal associates with its habitat (Burnham 2017, 2020). Species sensitive to sounds in the lower frequencies have a greater active space due to the long-range propagation of these signals compared to those of higher

frequencies. It considers how an animal makes sense of its surroundings and is fundamental to taking a more animal-centric approach to its role and place in an ecosystem. This may be conceived of as a part of the "operational niche" (Parker and Turner 1961) of an organism. This is synonymous to the "realized niche" described by Hutchinson (1957), but with the influence of "operational environments" introduced by Mason and Langenheim (1957), where there is a time-space-organism concept of environment. However, here the space is centered around the organism and so active space could equally be its "dynamic spatio-spectral-temporal acoustic habitat" (Clark et al. 2009) or its operational ambisonic niche, describing the three-dimensional acoustic landscape it is actively engaged with and interpreting.

The risk and extent of masking can be estimated using the concept of active space. If signals or a signal return falls below background sound levels, calls or cues will become less discernable, and the active space will be reduced. One of the first descriptions of the potential for acoustic masking of calls by human-made sound sources was presented by Payne and Webb (1971) for fin whales. They demonstrated a hypothetical reduction of signaling range for 20-Hz calls to fall from 900 km in a quiet ocean to 90 km under moderate ambient sound conditions under spherical spreading conditions, and from a maximum range of 13,000 km in the sound fixing and ranging (SOFAR) channel to just over 1 km with the same elevation in ambient sound levels (Payne and Webb 1971). Further examples have been reported in the reduction of communication space for baleen whales (Richardson et al. 1995; Hatch et al. 2012), delphinids (Janik 2000; Williams et al. 2014; Burnham et al. 2023), seals (Southall et al. 2003), and fish species (yellowfin tuna *Thunnus albacares,* Finneran et al. 2000; cod *Gadus* spp. and haddock *Melanogrammus aeglefinus*, Stanley et al. 2017).

## Metrics to Assess Risk of Affect

Identifying areas and level of risk from sound exposure includes distinguishing sound sources that might elicit a response and then assessing the exposure level of the individuals or groups (Popper et al. 2020). Likening the presentation of a masking sound in a soundscape to a pollutant allows exposure-response or dose-response relationships to come to the fore. The relative impact as a function of dose can be defined by considering the sound level, exposure duration, and proximity to the source (Clark et al. 2009; Hatch et al. 2012; Erbe et al. 2016).

Metrics have been developed to quantify the level of exposure to assess the risk of sublethal effects such as masking or behavioral change. These include the root mean squared ($SPL_{rms}$) or peak sound pressure level ($SPL_{peak}$), sound exposure levels (SEL), and kurtosis. Few studies discern between the root mean square or peak SPL, or direct measures of the received sound levels, and none of these metrics account for the duration of exposure. Sound exposure levels (SEL) offer a first approximation of the sound levels experienced by an individual on variable amplitude and/or duration schedules (Southall et al. 2017). This is an "exposure summation" metric, whereby an intermittent sound source is expressed as a single continuous exposure (Finneran

2015a). The sound level is normalized to 1 s periods across the period of exposure, which allows time periods to be compared in terms of the total acoustic energy received. This is important to assess cumulative exposure from multiple sources or the same source repeated several times. The SEL can be adjusted to be a more species-specific metric by using frequency ranges representing functional hearing groups (Martin et al. 2019) or by applying auditory frequency weighting functions prior to summation (Finneran 2015b). When considering the potential for acoustic masking, where possible, the metrics should be considered in the frequencies relevant to the species of interest and, where audiograms are available, the frequency that the species has the most acute hearing (Tougaard and Dähne 2017). If there is little to no overlap in frequency ranges of the signal and the noise, then non-masking effects such as behavioral modifications should be considered (Erbe et al. 2016).

Vocal plasticity allows species to respond to varying sound exposure with Lombard-effect like responses. Masking release strategies may also be employed to overcome some of the risk of sound additions interfering with signals. Strategies that alter the structure and projection of the signal from that of the interfering and potentially masking sound additions might allow the signal to still be discerned. Here, signal alteration allows for a release by creating disparity between that and the soundscape additions. Spatial and temporal release in the location, direction, and timing of signal production can also be used. Spatial release occurs when the level of masking is decreased due to the signal and masking sound sources being located in different locations and/or being projected in different orientations (Erbe et al. 2016), and relies on the receivers' ability to localize sound (Penner et al. 1986; Erbe et al. 2016). Directionality in calls has been demonstrated in several low- to mid-frequency vocal projections (e.g., bowhead whale calls, Blackwell et al. 2012; whistles and stereotyped calls of killer whales, Miller 2002; Lammers and Au 2003) and echolocation signals (e.g., killer whales, Au 1993; Au et al. 1995, 2004; sperm whales, Beedholm and Mohl 2006). Temporal release strategies take advantage of periods of quiet or relief from noncontinuous sound sources (Erbe 2008; Erbe et al. 2016). In addition, co-modulation release describes the variation of the frequency of a signal if the sound additions are present in a predictable modulation pattern across its range of frequencies (Brainsetter and Finneran 2008; Erbe et al. 2016). Adaptation of the signal is used to make it discernable from the patterns in soundscape additions that would otherwise result in masking.

Sound source parameters, and variables defining an individual's sensitivity to varying sound exposure, influence the onset of changes in hearing capabilities. These criteria will differ based on the sound source and if it is a single pulse, multiple pulses, or continuous. Hearing threshold shifts occur when either the cumulative sound exposure or peak sound pressure level set per species is exceeded. However, pressure-based metrics should be applied conservatively in reference to this variation. Yet, at some point, sound levels can reach a point where injury will result from all sounds over all frequencies. Measures of kurtosis describe the abruptness of the onset of the sound source and can be used as one variable to quantify potential impact. Typically, a statistical measure used to describe the shape of a distribution, here a high negative platykurtic score, would indicate rapid onset

and high instantaneous peak pressure that could elicit a stronger behavioral or physiological response (Ketten 2002; Hamernik and Qiu 2003). This indicates the presence of a broadband, atonal, transient impulsive sound with a short rise time (Southall et al. 2007). Mammalian auditory systems are particularly vulnerable to damage from intense impulsive sounds (Finneran et al. 2000, 2002; Southall et al. 2007, 2019). Distance from the sound source can mitigate some of the effects of an impulsive signal. Indeed, Hastie et al. (2019) investigated the propagation of pile driving and seismic survey signals, demonstrating that range from the source was the defining factor in shaping the received signal duration, rise time, and crest factor, which defines the impulsive or non-impulsive nature of the sound signals. Over distances of several kilometers, only a small proportion of sounds emitted from pile driving, seismic surveys, or underwater explosives would be characterized as impulsive after propagation (Finneran et al. 2000; Hastie et al. 2019).

## The Use of TTS and PTS as Metrics of Risk

Exposure thresholds have been suggested as limits to mitigate the negative impact to organisms and constrain TTS and PTS responses. These thresholds could be used to assess the risk of hearing fatigue or physical injury following elevated sound exposure. They can also be used to understand the potential hazards to animals within the range that the TTS-causing sound is present. This can guide mitigation measures (Popper et al. 2020). However, care must be taken when applying these metrics as TTS can accumulate over multiple exposures or show a delayed onset (Erbe et al. 2022). The most comprehensive criteria have been outlined for marine mammals, with several reviews and refinements (NMFS 2005, 2013, 2018). To limit harm, these criteria state that received levels of impulsive sounds should not exceed 190 dB re: 1 μPa for pinnipeds and most odontocete species, and 180 dB re: 1 μPa for mysticetes and sperm whales (NOAA 2013). Continuous sound sources are deemed to elicit avoidance responses before injury, yet an onset level of 230 $dB_{peak}$ re: 1 μPa has been described for PTS as a result of continuous sound sources for all cetacean species. Thus, any sound with amplitude greater than this level would result in permanent hearing damage in all species and functional hearing groups. This absolute threshold is similar to the definition of the asymptotic threshold shift, which is thought to represent the upper bounds of PTS and results from sources of a specific sound pressure level regardless of duration (Mills 1976; Finneran 2015b).

Sound exposure criteria is increasingly being adopted into regulatory approaches aimed at reducing the risk of acoustic disturbance. Estimates of the onset of TTS and PTS are becoming an integral metric in this process. The applications of these thresholds, however, expect that there is a typical response and that it will be consistent across all individuals. While the use of these sound levels provides an approach for precautionary management to reduce the potential for acoustic disturbance or injury, they do not consider the duration or frequency distribution of sound levels, and so may underestimate the true risk of impact (Stober and Thomsen 2019).

## Assessing the Scale of Impact

The spatial scale of the impact of a sound source ostensibly can be determined by its frequency and magnitude, and the time that it is present. The impact can be far-reaching, depending on the spectrum characteristics, and temporally dependent on the duty cycle of the sound additions. However, this is a simplified approach for assessing disturbance to aquatic species. The scale of impact on an individual, population, or species also depends on the context of how the individuals are using the space, their behavioral state (Goldbogen et al. 2013), and their sensitization (Kastelein et al. 2011). Each of these factors refines the dose-response curve, adjusting the likelihood and implications of a response. This creates an exposure-response risk function (Miller et al. 2012; Southall et al. 2019). Considering this with the factors that influence audibility, different sound sources have different "potencies" for different species or indeed for different individuals.

Identifying high use regions where more individuals are exposed to elevated sound allows for prioritization of risk management and mitigation. The risk of disturbance, however, should take into account both the exposure level and the ecological importance of the area to the focal species. To take the animal-centric approach to acoustic impacts, both environmental factors and receiver variables ("enviroscape" and "psychscape," respectively, Job et al. 1999) should be considered. Mapping co-occurrence of species' presence and human activities might be the first step in establishing risk (e.g., gray whales, Silber et al. 2021; humpback whales, Dransfield et al. 2014). Maps like these are a precursor to aggregate sound level models such as those created by Erbe et al. (2012) until data availability make them possible. Using vessel presence data, Erbe et al. (2012) visualized the cumulative underwater acoustic energy from commercial shipping. These maps can be made more species-specific by considering the sound fields for frequency bands specific to a species of interest. Clark et al. (2009) show an example of this for a construction site and vessel transits. The propagation and potential for masking of communication calls by human-derived sound sources of the same energy levels differed spatially when considering representative ranges for fin (18–28 Hz), humpback (224–708 Hz), and right whales (71–224 Hz). This highlights the importance of mapping specific frequencies of interest when taking the animal-centric approach and not solely relying on the emissions focused around the source (Erbe et al. 2012; Aletta and Kang 2015).

If calling behaviors have temporal patterns, disturbance risk will differ based on the proximity of the signal of interest and the sound source in time. Diurnal patterns, for example, or those related to an activity state might be impacted by sound exposure if there was temporal overlap (e.g., New Zealand bigeye *Pemphersis adsperda,* van Oosterom et al. 2016).

Previous work, using either observations and/or models, have typically only considered the direct impact of sound sources on an individual. The level of response and impact is usually discussed using the dose-response curve on the individual level relating to acute exposures. This, however, limits the assessment of impacts significantly; it considers the response from only one exposure from one source and does not consider the impacts of individual exposures in relation to their group or species success.

A framework to understand the cumulative impacts of multiple exposures of the same sound and/or exposures to several sounds sources concurrently is lacking. It cannot be presumed that exposure is additive when determining the response (Ayres et al. 2012), or that a short-term response can be used to infer the longer-term or population wide implications (Tougaard et al. 2015). Also, the shift to link these to longer-term, population level impacts on a broader scale is more challenging.

However, broad scale ecosystem implications may result from the addition of sound sources and/or elevated sound levels, where the impact is felt across trophic layers (Popper et al. 2020). To consider this, the potential for the sound source to change prey distribution and abundance, or species richness and diversity of an area more broadly, should be estimated. Startle and avoidance behaviors have been noted for fish and zooplankton (Popper and Hastings 2009; Hawkins and Popper 2018). Less directly observable effects might also result, such as changes in swarming or aggregating dynamics or a decline in reproductive success, or increased risk of predation due to masking (Slotte et al. 2004).

The population consequences of acoustic disturbance (PCAD) framework was developed to quantify the population level disturbance of sound exposure (e.g., dolphin species, Noren and Holt 2010; elephant seals, Costa et al. 2016). A similar but more general model, the population consequences of disturbance (PCoD) has also been used to determine the implications of disturbance on a population (New et al. 2014; King et al. 2015). This model attempts to address the scaling issue from individuals to population (e.g., gray whales, Villegas-Amtmann et al. 2015, 2017; humpback whales, Dunlop et al., 2021), but additional information is needed on the dose-response curves of species to be able to provide a more definitive assessment of impact. The PCoD framework suggests a means to quantify the impact of multiple stressors. However, this considers that the relationship between stressors, in this case different sound sources, would be additive in nature and exposures would be consecutive. This does not yet allow for synergistic or antagonistic relationships between sound sources, or response to various ambient sound levels, to be quantified. Neither model has been fully parameterized for a species with empirical data. Data interpolation and expert judgment, or informed assessments are used to apply the model to species (Pirotta et al. 2018). These models could, however, with time, be useful to establish if a single or multiple sound sources represent a risk to population success and survival, which may develop more meaningful evidence-based mitigation measures.

## Assessment of the Cost of Acoustic Disturbance

The adverse effects from acoustic disturbance on aquatic animals can be far-reaching. Masking ecologically important signals can drive behavioral changes in vocalization and habitat use. Displacement from areas critical to feeding, breeding, or rearing of young, or cessation of these behaviors have implications on foraging and weaning success, and the energy balance of individuals.

Loss of communication and echolocation space through masking has significant implications to the health and survival of aquatic species (killer whales, Williams et al. 2014; right whales, Clark et al. 2009; Hatch et al. 2012). Also, a loss of "listening space" (Barber et al. 2010; Pine et al. 2018) reduces the ability to detect threats, to locate prey, or find a mate. There may be energetic implications from altered vocalizations, either calling louder, more repetitively or using nonoptimal frequencies (Bradbury and Vehrencamp 1998). However, this is considered one of the least severe outcomes of acoustic disturbance, with the cost to change calling or echolocation signals low to negligible compared to other behaviors that might result from exposure to sound sources (Noren et al. 2009, 2017). There may be limits for species or individuals to compete with the elevated sound levels, with the adaptation of calling possibly limited by physiology. The capacity of the acoustic apparatus can limit the calls that are able to be produced. Perhaps reaching this threshold instigates calling cessation or maybe even area abandonment.

Implications for foraging behaviors have been noted for aquatic animals in the presence of anthropogenic sound sources. For southern resident killer whales, for example, foraging efficiency was reduced by 38–100% in proximity to the source of vessel-based additions to the soundscape (Au et al. 2004; Holt et al. 2009). Elevated ambient sound levels may make prey less available (Sato et al. 2021) and hinder the location, capture, and sharing of prey (Ford and Ellis 2006). The cessation of foraging or spatial displacement can directly result in reduced reproductive success or survival. A behavioral shift from foraging to travelling, for example, (e.g., killer whales, Williams et al. 2006, 2014; Lusseau et al. 2009) reduces both energy intake and increases expenditure. Differing responses have been shown between individuals based on prior experience or exposure, sensitivity, age, gender, and health (Simmonds et al. 2004; Weilgart 2007). Differences between genders, with the female being more sensitive to the exposure (e.g., killer whales, Williams et al. 2014; Holt et al. 2021), suggest greater population level implications from the disturbance. The transition away from social or resting behaviors have also been noted in the presence of vessels (e.g., bottlenose dolphins, Lusseau 2003).

In elevated sound conditions, the altered use of surface-active behaviors has been noted in some species (dolphins, Henderson et al. 2014; humpback whales, Dunlop et al. 2010; Dunlop 2016; killer whales, Noren et al. 2009; right whales, Parks et al. 2019). As well, increased ventilation rate, swimming speed, and path sinuosity may be apparent with the proximity of a vessel as a sound source.

The behavioral context of sound exposure can explain the varying sensitivities between species or even individuals. Periods, like weaning, also represent a critical period of learning and development, shaping future foraging and acoustic behaviors (e.g., gray whales, Wisdom et al. 2001; Burnham and Duffus 2020). Elevated sound levels could hinder vocal learning from their mother and conspecifics, or reunion events after the cow has left the calf to feed (e.g., gray whales, Burnham and Duffus 2020; humpback whales, Indeck et al. 2020; sperm whales, Whitehead 2017).

There currently is no way to determine the relative impact of disturbance based on the behavioral or social context of the area, or the behaviors that are disturbed. The energetic cost of disturbance might be weighted to reflect the importance of these

areas or behaviors to individual or population success, once we can clearly estimate the cost of the stress or changes in vital rates. Stress responses can occur at low sound levels and far removed from the source, and could impact the immune response, reproductive success, and vigilance of an animal. Behavioral responses can be subtle, whereas the stress response may be unobservable and may be confounded by the presence of other stimuli. More nuanced biological relevance may also be established when considering the repetitiveness of the source and how long the presence of the sound persists. Assessing the cost to an animal with no observable change in behaviors can minimize the estimated impact. Observable changes of behavior could be absent due to increased habituation or, indeed, progressive deafness. However, it may also reflect limited dispersal abilities, for example, when foraging on patchy prey resources.

## Noise Pollution as a Competitor to Aquatic Species

To take the appreciation of human-made sound sources as a habitat level pollutant, a step further in an animal-centered approach, and regarding it as a competitor, may be useful. Competition is defined as two or more entities competing for the same resources in an ecosystem. This can be for the same prey resources or living space when these resources are limited. In this case, the presence of a sound source resulting in acoustic disturbance is in an amensalistic relationship with the individual, group, or species. Instead of using the same resources, the sound addition limits the availability or access to resources to produce the one-sided negative effect. This suggests a more active and direct impact than simply dose-response. Considering asymmetric competition traits may help understand how, as the dose or exposure of the sound increases, a decrease in abundance of a species is seen.

The strength of competition depends on the similarity of the traits of individuals; in this application, the similarity in frequency range of the signal and sound source describes first their sensitivity to the disturbance, and then forms part of the assessment of potential masking. Masking, expressed as proportional reduction of range, area, or volume, represents a reduced active space. To form a masking metric, this reduced range is compared to the range or space of acoustic signals under a minimum ambient sound level condition (Clark et al. 2009; Hatch et al. 2012; Burnham et al. 2023). This enables a proportional reduction of communication or echolocation space to be quantified. This metric is a useful comparator of effect over space and time but could also be considered akin to competition coefficients.

The competition between an individual and the masking sound for effective active space highlights the importance of the frequency and amplitude of the sound. Those individuals that utilize spatial or temporal co-modulation masking release mechanisms could be said to be taking advantage of the "vacuum of competitive release" (Van Den Elzen et al. 2017) or opportunistically making use of any declines or reductions in exposure. Sound sources as competitors have, in some instances, been shown to push species past a short-term response into a long-term adaptation in their acoustics use. Seasonal and longer-term changes in whale

calls have been noted, for example, in response to changes in the acoustic properties of the soundscape on both regional and global scales (e.g., blue whales *Balaenoptera musculus,* McDonald et al. 2009; Leroy et al. 2018).

Under the designation of a pollutant, elevated sound levels cause habitat degradation. It may, however, be more in keeping with the animal-centric approach to consider that the sound source has removed habitat from the realized niche of an individual as a result of avoidance behaviors. Anthropogenic sound sources displace aquatic life over a range of spatial and temporal scales. Gray whales changed their migration route when exposed to industrial (Malme et al. 1983, 1984) and vessel sound sources (Moore and Clarke 2002). Longer-term habitat displacement was seen as a result of construction noise in their breeding lagoons, abandoning a site for several years until construction ceased (Mate 1995). Also, humpback whales avoid air gun arrays at greater distances than the protective mitigation zone that industry would have allowed (Dunlop et al. 2015). Acoustic harassment devices deployed to displace seals from salmon fish farms have also halted killer whale use of an area (Morton and Symonds 2002).

For many species, however, the threshold of the introduction of a sound source that instigates habitat abandonment has yet to be established, and so more subtle behavioral responses are assessed. Startle and avoidance responses can change the ecosystem dynamics, making changes to predator-prey relationships (Francis and Barber 2013; Naguib 2013). Fish catches were reduced after exposure to seismic survey sounds (cod, *Gadus morhua*, haddock *Melanogrammus aeglefinus* and rockfish *Sebastes* spp., Skalski et al. 1992; Engås et al. 1996). Prey may become less available due to acoustic competition, by not being able to effectively locate prey (Sato et al. 2021). Prey displacement and altered habitat use of fish and invertebrate species have also been noted (Simpson et al. 2016). Together, this can alter the trophodynamics of a system (see Popper et al. 2020).

## Animal-Centric Mitigation Strategies and Future Actions

Mitigation strategies to lessen acoustic disturbance can be enacted at the source or through more operational and policy measures. Source-based measures are changes to the design or implementation of the activity generating elevated sound levels. Retrofitting the existing commercial fleet, propeller shields, and bubble curtains, for example, lessens the sound levels emitted from a source. The measures themselves are not species-orientated, but the efficacy of the measures can be assessed in a more animal-centric way by evaluating the change in sound levels in the frequencies pertinent to the focal species. An elevation in sound levels can be compared to estimates of hearing sensitivity and audiograms to better understand the relevance and effectiveness of the measures.

Operational measures may, however, be more targeted to populations or species by considering their temporal and/or spatial application. Animal-centric tools to directly mitigate elevated sound levels include “marine mammal safe zones,” during construction or sonar and seismic surveys, and vessel rerouting or slowdowns in

critical habitat. Policies linked to habitat conservation that include the soundscape, stating that acoustic emissions should not "destroy" habitat, are being developed (DFO 2011, 2021). These measures state that species using areas with this type of protection should be able to use signals to communicate and detect and capture prey. However, threshold sound levels that would constitute a violation of this principle have yet to be defined. The masking metric of proportional reduction in communication or echolocation range or space might go some way to quantify a maximum level of allowable acoustic addition or overall ambient sound level that should not be exceeded. Considering the impacts during important life history events or the presence of whales in areas of intense human use can be the first step to highlighting areas in need of more complex mitigation strategies. Soundscape mapping highlights regions that most need mitigation measures (Erbe et al. 2012, 2014, 2021; Schoemann et al. 2022), further refined by considering species-relevant frequencies.

Increased research effort is needed to clarify the sensitivity and response of animals to sound exposure. Hearing sensitivity curves and audiogram development methods should not be based solely on the response of individuals on narrow band artificial tones, as these might not accurately predict the audibility of complex sounds (Erbe and Farmer 1998; Cunningham and Mountain 2014; Erbe et al. 2016). Natural animal- or environment-derived signals and anthropogenic sound sources are usually more broadband than the signals used in the signal-response testing to establish acoustic capabilities. More studies like those conducted by Kastelein et al. (2011, 2012, 2013, 2014), where behavioral auditory testing was applied to more complex human-derived sources like sonars, construction, and pile driving are needed. Reflection, diffraction, and scattering of the sound signals, and potential multipath arrivals formed from the depth, range, and azimuth between the signal and receiver, reflecting a more natural setting, may need to be considered to refine the three-dimensional shape of an individual's active space (Erbe et al. 2016), before looking at the implications of elevated sound conditions or noise sources on it. Similarly, more data on the auditory filters and critical ratios and the integration time or temporal processing of signals by animals is needed for many aquatic species (Erbe et al. 2016). So far determination of these have been limited mostly to odontocetes, including bottlenose dolphin, harbor (*Phocoena phocoena*) and finless (*Neophocaena phocaenoides*) porpoise, false killer whales (*Pseudorca crassidens*) and beluga whales, and seal species including harbor (*Phoca vitulina*), ringed (*Pusa hispida*), northern elephant (*Mirounga angustirostris*), and fur (*Callorhinus ursinus*), as well as the Californian sealion (*Zalophus californianus*), western manatee (*Trichechus senegalensis*), and sea otters (*Enhydra lutris*) (Erbe et al. 2016).

Perceived loudness is a frequently employed metric to describe sound sources, with additions to a soundscape described in decibels (dB re 1 μPa) relative to ambient sound levels. However, perceived loudness is also tied to the signal duration and frequency content. A better predictor of behavioral response is derived when taking all these variables into account. The sound level threshold is attractive for initiatives designed to manage acoustic disturbance because of its simplicity, but it may not capture the complexities of the response (Tougaard and Beedholm 2018).

## Conclusion and Future Directions

Considering anthropogenic sound sources more as a competitor than a pollutant is a subtle mindset shift but highlights the potential impact to both the individual, group, or ecosystem more clearly. It can have greater implications than that of an additive pollutant and instead be seen as limiting resources and diminishing habitat. The concept of being a competitor also highlights the interaction between the animal, its acoustics use, and the various levels of sound exposure. That said, the analogy of noise as a pollutant has been useful in expressing the problem to managers and the lay public, and remains relevant. The term "pollutant" captures the pervasive nature of human-made sound additions to the soundscape. Although some sources of anthropogenic sounds are consistently present over long time periods, to date modeling efforts have not been able to fully capture the more persistent nature of this pollutants. Also, sound of any source is removed immediately as soon as the source is eliminated and does not linger in the habitat after source removal. This differs from other pollutant types that may be more persistent. However, recovery of animal habitat use before the sound source was present may take time (Mate 1995; Morton and Symonds 2002).

Increasingly, acoustic disturbance is being included as a key threat to the success and survival of threatened and endangered species. Also, with the growing concern around the impacts of human-made sound sources, it has become an expanding area of aquatic and marine environmental management (Popper et al. 2020). By taking the animal's perspective, a change in the appreciation of implications of elevated sound levels gives more weighting to their reliance on the acoustic sense for survival, and that interference in the reception of acoustic signals because of changes in the soundscape has the potential to create adverse effects. Concepts such as active space should be incorporated more readily in the assessments of the effects of acoustic disturbance to truly take a more species-specific and animal-focused approach. Also, a weighting towards mitigating the effects of increased sound levels in areas that represent ecologically important areas should be made, as well as targeting those areas where human activity and aquatic species overlap most.

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# Environmental Risk Assessment Framework for Dredging Sounds

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**Abstract**

Underwater sounds produced from anthropogenic sources may pose risks to aquatic life, yet there is much uncertainty regarding the prediction of these impacts. A structured risk-based framework for identifying, assessing, and managing risks associated with underwater sounds has been previously proposed; however, real-world applications of the approach are lacking. Therefore, this

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chapter describes a risk-based framework for evaluating underwater sounds from dredging operations and demonstrates the process using a real-world dredging scenario. The approach described was used to evaluate risks to aquatic biota from a dredging operation in the Port of Alaska, USA. Sound levels were recorded during dredging activities and used to evaluate the relative risk of hearing impairment to the species of concern, including marine mammals and fish. Demonstration of the screening-level approach was beneficial to inform the relative risk of dredging operations, eliminate species of concern from further evaluation, and provide a path forward for future refinements to decrease uncertainties. This approach provides a transparent and repeatable phased methodology to discern ecological risks associated with dredge-induced underwater sounds that is applicable to a wide range of sound sources and site-specific scenarios.

**Keywords**

Underwater sound · Noise · Effects · Fish · Marine mammals

## Introduction

There is growing awareness on the impacts of anthropogenic underwater sounds on aquatic life (UN 2018). Yet, to inform current and future protective guidelines and regulations, there is a need to apply risk-based approaches (Popper et al. 2020). To achieve this, regulators and researchers need to collaboratively develop and implement strategies that bridge the gaps of knowledge so that ecosystems can be protected and the degree of regulation and mitigation is appropriately proportional to the risks (Hawkins et al. 2020; Popper et al. 2020). The importance of applying a risk-based assessment process to improve regulatory and mitigation decision-making for underwater sounds has been previously acknowledged (Faulkner et al. 2018; Merchant et al. 2018; Popper et al. 2020). Therefore, as more scientific understanding of ecological impacts (or risks) are developed, there is a corresponding need to develop risk-based approaches to integrate data (which are often disparate and complex) and utilize multiple lines of evidence to inform management of underwater sounds from anthropogenic sources.

The goal of a risk framework is to provide a widely accepted and applicable process and methodology that enable resource managers to systematically evaluate ecological risks associated with underwater sounds and to guide appropriate management solutions, if necessary. Specific to dredging operations, there has been substantial progress in the past decade documenting underwater sound exposures for various operational scenarios (Suedel et al., 2019), development of high-level risk frameworks specific to dredging (WODA 2013), and expansion of more detailed approaches and application of these risk-based methods (McQueen et al. 2020). Yet, there remains a need to provide pertinent examples from real-world case studies to apply a risk framework in practice. Therefore, the aim of this chapter is to further

define and demonstrate the utility of a risk-assessment process for evaluating ecological risks from dredging-associated underwater sounds. To achieve this aim, the specific objectives were to (1) apply a tiered risk assessment process for assessing underwater sounds produced from dredging, (2) use real-world data from a dredging project to demonstrate and evaluate the risk assessment approach, and (3) document lessons learned and areas of future research.

## Ecological Risk Framework for Dredging Sounds

A detailed risk framework process has been previously defined and discussed in detail (Fig. 1; McQueen et al. 2020). Prior work has acknowledged the benefits of using a risk-based framework for evaluating impacts of anthropogenic underwater sounds to aquatic life. For example, Boyd et al. (2008) proposed a stepwise framework for evaluating underwater sound risks which included hazard identification, characterizing exposure to hazard, characterizing dose-response relationships, risk characterization, and risk management. Additionally, a high-level description of a risk assessment framework was proposed for evaluating dredging sounds (WODA 2013). Yet, in the case for dredging sounds, such assessment processes have not been described in detail or evaluated using real-world data. Therefore, in this chapter, a tiered process including both a screening level and a detailed assessment process is described. Both assessment steps of the approach consist of three components: (1) define the risk problem (i.e., project formulation), (2) assess exposure and effects (i.e., analysis), and (3) characterize risk (Fig. 1).

## Case Study: Screening Assessment for Dredging Sounds

### Project Formulation

The study site is the Port of Alaska, USA, near the confluence of the Knik Arm and Turnagain Arm of Cook Inlet which connects to the Gulf of Alaska (Fig. 2). The Port of Alaska is the largest cargo port in the region, handling 74% of all in-bound nonpetroleum freight that enters south-central Alaska. Approximately five million tons of cargo (all commodities) passed through the port in 2021 which is also designated as a Military Surface Deployment and Distribution Command. Therefore, maintaining safe and navigable waterways for the Port of Alaska is critical to commerce and national security. To achieve this, the harbor basin requires an annual routine maintenance dredging cycle, and up to one million $m^3$ of sediments are annually removed using dredges. However, environmental consequences (risks) of dredging activities are addressed and managed (if necessary) prior to operations. A conceptual site model is a useful tool to describe the potential interactions between dredging sound sources (exposures) and species of concern (responses). In this case, the conceptual site model defines details of the underwater sound sources, sound

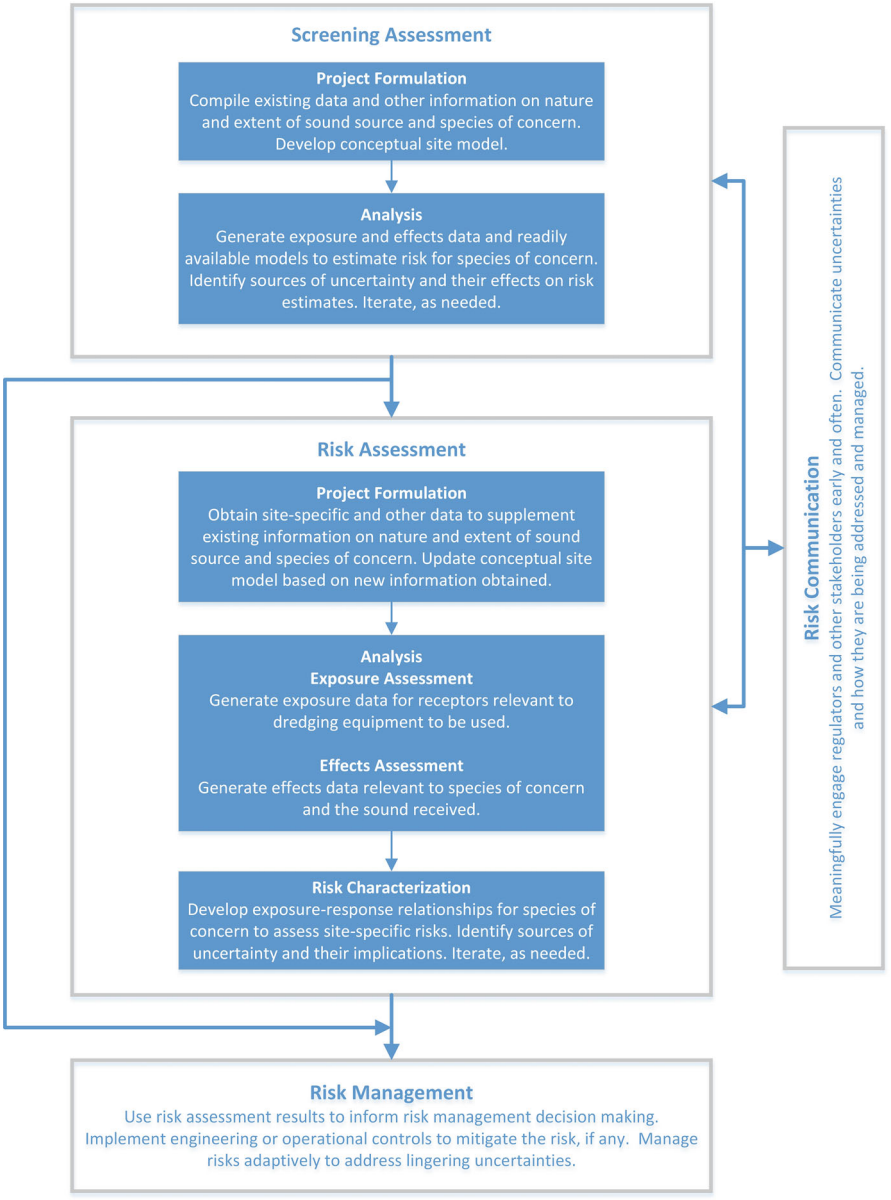



**Fig. 1** Overview of the risk-based approach for assessing and managing risks associated with underwater sound from dredging operations. (From McQueen et al. 2020)

exposure scenario (e.g., spatial and temporal scales, intensity, waveform), exposure metrics (e.g., sound pressure level [SPL], sound exposure level [SEL]), defined species of interest, and specific response measurement endpoints (Fig. 3).

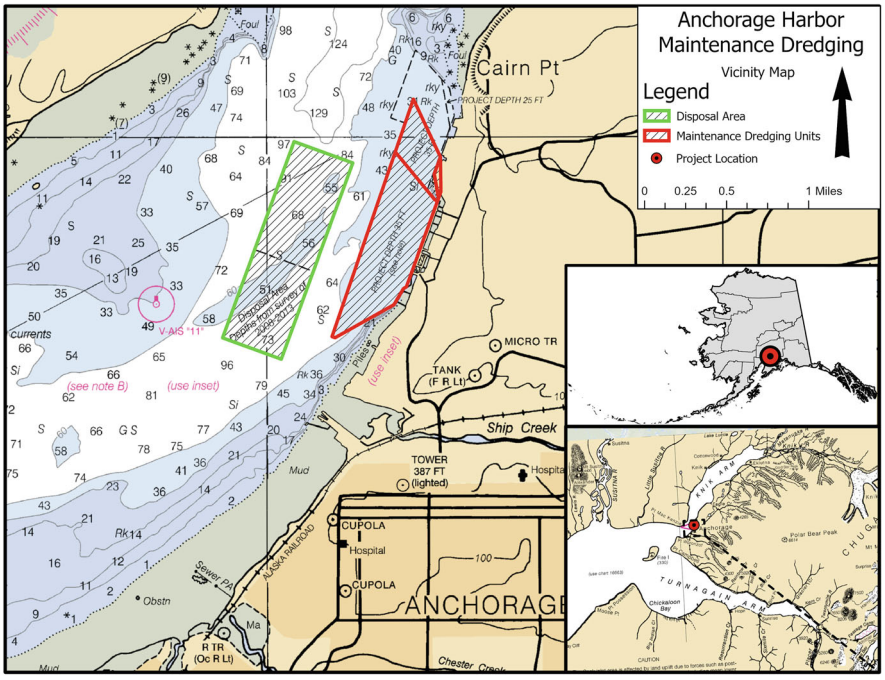



**Fig. 2** Study area located in Anchorage Harbor Alaska, USA

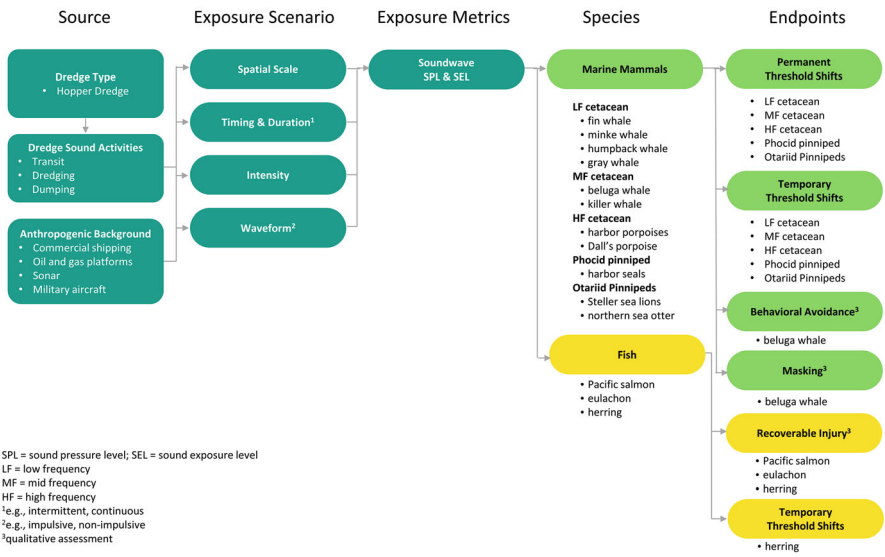



**Fig. 3** Conceptual site model showing sound sources, exposure scenarios, species, and endpoints for a hopper dredge scenario

## Sound Sources

The US Army Corps of Engineers (USACE) Alaska District recorded underwater dredging sounds produced by a hopper dredge with 1500 $m^3$ capacity during active dredging, transportation to disposal site, and placement in the water disposal area. The hydrophone was within the active dredging zone anchored approximately 1 m from the bottom seafloor. To minimize terrestrial sound interferences, closed-cell foam insulation was wrapped around the hydrophone cable between the cable reel and water surface. Unless otherwise noted, sounds were analyzed in one-third octave bands with a 1-sec interval averaging time. In this case study, a simplistic spherical spreading model was used to estimate propagation loss and to inform the sound source level (Eq. 1).

$$\mathrm{PL} = \left(\mathrm{A} + \mathrm{N}\log_{10}\left(\frac{\mathrm{R}}{\mathrm{r_{ref}}}\right)\right)\mathrm{dB} \tag{1}$$

where R is the distance from the measured sound source, $r_{ref} = 1$ m is the reference distance, N is 20 (i.e., spherical spreading), and A is a constant factor.

## Ambient Background

The ambient background soundscape in the Port of Alaska is complex and influenced by both natural and anthropogenic sources. Dominant natural sounds in this region include ice, tides, waves, and precipitation (Blackwell and Greene 2002). Nearby to the dredging project location, there are numerous potential contributors to the anthropogenic underwater soundscape from commercial and military activities the in the Port of Alaska, including shipping vessels, oil and gas platforms, sonar, and aircraft (study area is located near a military airport). Of these anthropogenic "background" sounds, military aircraft overflights were among the loudest measured, with underwater SPLs reaching 150–160 dB re 1 μPa with the peak spectral density levels in the 675–1500 Hertz (Hz) range. Additionally, the maximum SPLs recorded during positioning of cargo ships and associated tugboats at 50 m from the hydrophone were 154 dB re 1 μPa one-third octave band centered at 125 Hz. Assuming spherical spreading (N = 20 and A = 0 in Eq. 1), the source level of the shipping and cargo activity is estimated to reach 188 dB re 1 μPa-m. Yet, these recorded SPLs are intermittent and generally occur within short time periods (i.e., minutes). The most quiescent period recorded during the study period was at an SPL of 128 dB re 1 μPa one-third octave band centered at 125 Hz and represents the mean received ambient background. This background sound level is generally consistent within the range of ambient background sounds levels previously reported in the region (Austin et al. 2016; Denes et al. 2016), which described a mean SPL of 138.8 dB re 1 μPa during a 2016 study, within a range of L95 = 106.8 dB re 1 μPa (level exceeded for 95% of the time and representation of background level without any construction noise present) and a maximum SPL of 164.7 dB re 1 μPa (reached by a single aircraft event) (Austin et al. 2016; Denes et al. 2016).

## Dredging Sounds

Dredging sounds were monitored during operations and included three distinct activities: dredging (sediment removal), transit, and sediment placement on the seafloor. In general, dredging produces continuous, non-impulsive sounds that are low frequency (<1000 Hz) (WODA 2013). Hopper dredging sounds are generally discontinuous, cyclical sounds due to periodic operational changes as the dredging vessel shifts from sediment removal to transiting to the dredging placement site. During the active dredging process, continuous sounds are generated from the draghead movements along the seafloor below the vessel with additional sound sources from the engine, propeller, pumps, and generators. Therefore, the putative primary sources of sounds during hopper dredging activities are from propeller and engines, inboard pumps, and draghead (CEDA 2011; Heinis et al. 2013; McQueen et al. 2019; Suedel et al. 2019; Fig. 4).

In this study, the dredge approached within 28 m of the hydrophone while the dredge pumps were operating, and sediment was being excavated. The mean received SPL during dredging operations was 133 dB re 1 μPa one-third octave band at 500 Hz. Most of the sound energy was in the 160–175 Hz and 575–1500 Hz frequency ranges, with brief exposures around 10 kHz reaching 108 dB re 1 μPa. Therefore, based on a conservative propagation loss of 20 (assuming spherical spreading; Eq. 1), the maximum SPL observed during dredging (sediment removal) was 162 dB re 1 μPa at 1 m (one-third octave band at 500 Hz). The highest SPLs produced by the hopper dredged occurred during transit and maneuvering. The dredge was recorded approaching the harbor from the disposal site. As the tug increased power to swing the stern of the dredge vessel around to parallel the dock face, SPL equaling 139 dB re 1 μPa was recorded 30 m from the dredge, with an estimated source level of 168 dB re 1 μPa-m (N = 20, A = 0) one-third octave band

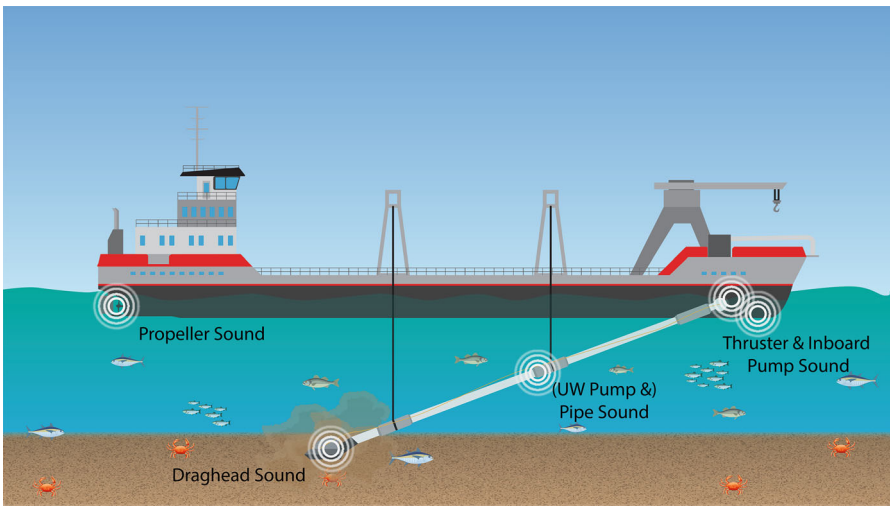



**Fig. 4** Diagram of potential sound sources from a hopper dredge

at 500 Hz. The maneuvering produced powerful sound for about a minute, after which, the dredge continued along parallel to the dock face producing a measured SPL around 128 dB re 1 μPa one-third octave band at 500 Hz as the pumps were turned on and dredging began. Therefore, as a conservative approach, the maximum SPL of 168 dB re 1 μPa was used for this risk assessment with the assumption that 95% of the energy was below 2.5 kHz.

## Species of Concern

The lower Cook Inlet supports a diverse group of marine mammals including whales, beluga (*Delphinapterus leucas*), fin (*Balaenoptera physalus*), humpback (*Megaptera novaeangliae*), minke (*Balaenoptera acutorostrata*), and killer (*Orcinus orca*); porpoises, Dall's porpoise (*Phocoenoides dalli*) and harbor porpoise (*Phocoena phocoena*); harbor seal (*Phoca vitulina*); Steller sea lion (*Eumetopias jubatus*); and northern sea otter (*Enhydra lutris kenyoni*) (Muto et al. 2019). Although these marine mammals have populations in the lower Cook Inlet, there is a lower probability that these species occur near the dredging area in the upper Cook Inlet; nonetheless, for conservative assumptions, they were retained for consideration in this screening-level assessment. The exception to this is the beluga whale, which has a well-documented small, isolated population present in upper Cook Inlet near the dredging area (O'Corry-Crowe et al. 1997; Laidre et al. 2000; Speckman and Piatt 2000; Rugh et al. 2000, 2005). For this assessment, marine mammal hearing impairment in terms of permanent threshold shifts (PTS) and temporary threshold shifts (TTS) were the endpoints of interest. Additionally, a qualitative assessment of behavioral avoidance and masking was conducted for beluga whales.

Beluga whales have been routinely observed near dredging operations in the past years. The population of beluga whales was estimated at 1300 in 1979 but declined by 47% in the 1980s primarily due to unrestrained subsistence hunting (Muto et al. 2020). Since that time, beluga whales were listed and endangered under US Federal Endangered Species Act (73 FR 62919 2008), and critical habitat for the stock was designated in Cook Inlet. Habitat ranges of belugas vary in the inlet based on seasonal patterns and food abundance. During migration of eulachon (*Thaleichthys pacificus*) and salmon, beluga whales occupy regions closer to the large rivers (Moore et al. 2000; Nemeth et al. 2007). During the winter they migrate toward deeper waters in lower Cook Inlet and shallow areas along the western Cook Inlet shoreline (76 FR 20179 2011). Additionally, sea otters and Steller sea lions have designated critical habitat in Cook Inlet.

Fish species of concern include five species of Pacific salmon (i.e., chinook (*Oncorhynchus tshawytscha*), chum (*Oncorhynchus keta*), coho (*Oncorhynchus kisutch*), pink (*Oncorhynchus gorbuscha*), sockeye (*Oncorhynchus nerka*)) (Nemeth et al. 2007). Additionally, herring (*Clupea harengus*) and eulachon (*Thaleichthys pacificus*) occupy regions within Cook Inlet and are of commercial and environmental importance (Nemeth et al. 2007). For this risk assessment, the response endpoints included recoverable injury for salmon, herring, and eulachon based on reported qualitative guidelines for shipping and continuous sounds (Popper et al. 2014). Additionally, TTS was considered for herring as they are relatively sensitive to underwater sound (Popper et al. 2014).

## Analysis

### Marine Mammals: PTS and TTS

For marine mammals, PTS and TTS were evaluated in the screening assessment using conservative maximum SPLs measured during the dredging operations. In this case, a maximum source level of 168 dB re 1 μPa-m was used to calculate SEL for "non-impulsive, continuous, mobile sources" using the National Oceanic and Atmospheric Administration (NOAA) National Marine Fisheries Service (NMFS) tool, which is a publicly available exposure model (NMFS 2018a, b). The model uses frequency-weighted exposures for groups of marine mammals for estimating an SEL using the "safe distance" concept (from Sivle et al. 2014) with exposure assumptions including source level, velocity, cycle, and weighting factor adjustment (NMFS 2018b). The model can be used to estimate risk isopleths (distances) from the sound source where risk of PTS- or TTS-onset thresholds is anticipated to occur. In this dredging scenario, the maximum source level was observed during dredging transit where vessel speeds were estimated at 1.5 m/s (Table 1). Using these stated assumptions, the estimated marine mammal PTS and TTS risk isopleths were $< 1$ m from the dredging sound source for all the marine mammals of concern (Table 1).

### Beluga Whales: Avoidance and Masking

A potential consequence of dredging sound in the underwater environment is masking (Todd et al. 2015), when loud dredging sound drowns out softer animal communication signals at the same frequency. Masking can impact an animal's communication ability with conspecifics or impair food foraging (Erbe et al. 2016; Pine et al. 2020). Beluga whales in the Anchorage Harbor vicinity generally occur in pods consisting of 2–35 individuals, with a median pod size of 8, according to USACE observation data. Beluga whales in the pod communicate and find prey using echolocation. Beluga whale vocalizations cover a broad range of frequencies; but social communication is generally lower frequency (below 10 kHz), and echolocation is much higher frequency (peaks at 40, 80, and 120 kHz) (Gurevich and Evans 1976). The mean frequency of beluga whale whistles is 2.0–5.9 kHz, while their broadband calls occupy a frequency range of 800 Hz to 9.4 kHz with a mean frequency of 3.4 kHz (Sjare and Smith 1986).

Dredging sound (70–1000 Hz) overlaps the lower end of the broadband call frequency range; however, this is below the frequency range of whistles and mean frequency of calls. Beluga whale fish predation is understood to be primarily enabled by echolocation in the highly turbid waters (~500 nephelometric turbidity units [NTUs]) of upper Cook Inlet. Peak beluga whale echolocation frequencies do not overlap with the frequencies produced by dredging sound, so it is unlikely that dredging sound would cause any direct impacts to beluga whale predation efficiency due to masking. Notably, anecdotal observational data suggests that annual dredging activities at the port are not affecting beluga whale use of Knik Arm as evidenced by the consistency of timing, location, and numbers of belugas (including calves) in the area each year (Prevel et al. 2006; Markowitz and McGuire 2007; Cornick and Kendall 2008). These qualitative data indicate a relatively low risk of behavioral avoidance of beluga whale due to dredging activities.

**Table 1** Estimation of PTS and TTS risks to marine mammals based on the maximum recorded source SPL from hopper dredges

| Parameter/assumption | Mobile dredge[a] Continuous operation | Unit |
|---|---|---|
| **Exposure** | | |
| Maximum source level | 168[b] | dB re 1 μPa-m |
| Duration of sound production | Time independent | – |
| Vessel transit speed | 1.5 | m/s |
| Sound propagation method | 20 log R | dB re 1 m |
| **Species** | | |
| Endpoint | PTS and TTS "safe distance" methodology | – |
| Swim speed/movement | Stationary[c] | – |
| Frequency weighting | | |
| Low-frequency cetaceans | −0.05 | Unitless |
| Mid-frequency cetaceans | −16.83 | Unitless |
| High-frequency cetaceans | −23.50 | Unitless |
| Phocid pinnipeds | −1.29 | Unitless |
| Otariid pinnipeds | −0.60 | Unitless |
| **Weighting function equation[e]** | $W(f) = C + 10\log_{10}\left\{\frac{(f/f_1)^{2a}}{\left[1+(f/f_1)^2\right]^a\left[1+(f/f_2)^2\right]^b}\right\}$ | |
| **PTS threshold isopleth (risk decision)** | | |
| Low-frequency cetaceans | < 1 **(no risk)** | Meters |
| Mid-frequency cetaceans | < 1 **(no risk)** | Meters |
| High-frequency cetaceans | < 1 **(no risk)** | Meters |
| Phocid pinnipeds | < 1 **(no risk)** | Meters |
| Otariid pinnipeds | < 1 **(no risk)** | Meters |
| **TTS threshold isopleth (risk decision)** | | |
| Low-frequency cetaceans | < 1 **(no risk)** | Meters |
| Mid-frequency cetaceans | < 1 **(no risk)** | Meters |
| High-frequency cetaceans | < 1 **(no risk)** | Meters |
| Phocid pinnipeds | < 1 **(no risk)** | Meters |
| Otariid pinnipeds | < 1 **(no risk)** | Meters |

*PTS* permanent threshold shift; *TTS* temporary threshold shift
[a]Constant velocity, constant direction
[b]Broadband; assuming 95% of energy below 2.5 kHz (dredge with propulsion)
[c]No avoidance or attraction to source and no vertical or horizontal movement
[d]Based on broadband sounds with 95% frequency contour profile
[e]See NMFS (2018a, b) and supplemental material for parameterization

## Fish: Recoverable Injury and TTS

Near the dredging operations, potential underwater sound impacts were qualitatively estimated for salmon, herring, and eulachon which occupy regions within Cook Inlet and are of commercial and environmental importance (Nemeth et al. 2007). There are very limited exposure response data or quantitative sound exposure guidelines available for fish species relative to low-frequency, intermittent sounds like dredging

(Popper et al. 2014, 2022). Fish sensitivity to underwater sound is dependent on the anatomical makeup (e.g., inner ear connectivity to tissues or organs), and therefore sensitivities vary greatly among species (Popper and Hawkins 2018; Popper et al. 2022). To date, sound exposure guidelines published by Popper et al. (2014) have subsequently taken into consideration these differences and have proposed qualitative guidelines for species with either no swim bladder or if the swim bladder is not involved in hearing. Though it is thought that metrics based on particle motion may be more predictive to inform responses to fish (Popper et al. 2022), quantitative guidelines for particle motion are still lacking. At sufficient distance from dredger, water surface, and seabed, sound pressure and sound particle velocity are proportional.

Salmoniformes (i.e., Pacific salmon) have swim bladders that are distant from the ear (Popper et al. 2022). Therefore, in this assessment, salmon were compared to qualitative fish guidelines from Popper et al. (2014) for non-impulsive sounds ("shipping and continuous sounds") where swim bladders are not involved in hearing. These comparisons indicate salmon have a "low" risks of mortality and recoverably injury (e.g., injuries, hair cell damage, hematoma, etc.) to dredging sounds.

Herring (*Clupea harengus*) have gas bubbles in the ear as well as swim bladders involved in hearing which are physiological adaptations making them more sensitive to sound (Popper et al. 2022). Therefore, herring were compared to quantitative risk thresholds available for "recoverable injury" (170 dB re 1 μPa for 48 h) and TTS (158 dB re 1 μPa for 12 h) developed for fish species which swim bladders are involved in hearing (Popper et al. 2014). Risk of herring "recoverable injury" was considered de minimis since the maximum source level (168 dB re 1 μPa-m) was below the "recoverable injury" threshold. For TTS, the maximum source level and propagation loss (Eq. 1) indicate that SPLs would be above the threshold at a distance of $<3$ m from the dredge. These distances were considered minimal and therefore were not considered as a significant source of risk. It should be noted that this screening-level risk assessment has conservative assumptions and fish avoidance was not considered. Fish avoidance to various underwater sound sources have been well-documented (e.g., Engås and Løkkeborg 2002) and therefore would be an important aspect of estimating exposures (e.g., duration, periodicity) if more detailed assessments were warranted.

## Risk Management

Based on the screening assessment, the risks of hearing impairment from underwater sound from dredging were determined to be de minimis. As a precautionary principle, a 50 m shutdown radius was established around dredging operations and informed by marine mammal observers which would prevent any auditory impacts to beluga whales or other marine species of concern (Fig. 5). Subsequently, a byproduct of the exclusion zone radius would result in marine mammal SPL exposures below $\sim$134 dB re 1 μPa (based on maximum observed source-level SPLs and spherical spreading assumptions; Eq. 1).

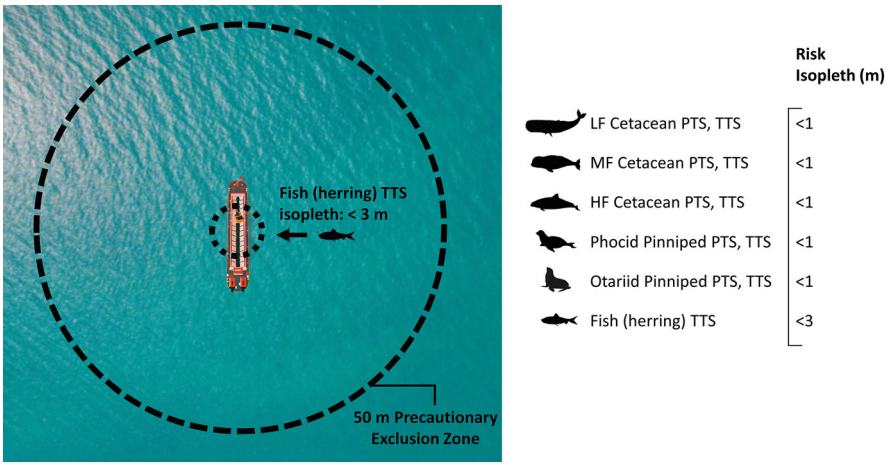



**Fig. 5** Hearing impairment response isopleths for species of concern based on maximum sound levels (168 dB re 1 μPa) from a hopper dredge in the screening assessment. PTS = permanent threshold shift; TTS = temporary threshold shift; LF = low frequency; MF = mid-frequency; HF = high frequency. Risk isopleths based on assumptions from NMFS (2018a,b) and fish risk thresholds based on Popper et al. (2014)

## Uncertainty

The primary sources of uncertainty identified in this case study include lack of site-specific exposure-response data and incomplete model assumptions. In this screening assessment, "worst-case" conservative assumptions were made regarding continuous operation of the dredge vessel and exposures of maximum observed SPLs (100% of time) and 100% site usage of marine mammal and fish. These assumptions therefore likely overestimate risk estimates. However, this assessment assumed a propagation loss of 20 logR (as used in the NMFS tool) which could potentially be less in Cook Inlet due to a variety of location conditions (water depth, sediment characteristics, etc.) and would subsequently underestimate underwater sound ranges.

It is pointed out that in this study only the risk of hearing impairment was quantitatively assessed. Masking was discussed concerning one species. As Todd et al. (2015) point out, at least in theory, dredging noise can lead to chronic impacts due to stress, and there is the possibility that marine mammals and fish are displaced from important habitat due to avoidance reactions. Behavioral reactions to dredging sound have been shown in harbor porpoises (see review by Todd et al. 2015). Yet, for behavioral assessments, there remain significant data gaps that contribute to the uncertainty. Sources of uncertainty for behavioral impacts for both marine mammals are clear exposure-response relationships and also the biological significance of those responses (Erbe et al. 2019). For fish, there remain significant gaps of information (Popper et al. 2020). For example, many fish species only detect particle motion (Popper and Hawkins 2018), and therefore SPLs are likely irrelevant as sound exposure metrics for predicting hearing (PTS, TTS) or other physiological responses (stress hormones) (Popper et al. 2022).

## Lessons Learned and Areas of Future Research

Benefits of this risk assessment approach include the flexibility of the framework to include new data as it becomes available and the adaptability to site-specific scenarios. As the scientific understanding of underwater bioacoustics rapidly evolves, it is particularly beneficial to have an assessment process that can readily adapt to the emergence of new data. The tiered process incorporates an efficient use of time and resources to apply appropriate level of detail to reach informed (actionable) risk management decisions. Future research opportunities include the continued development and refinement of standards and terminology for underwater sound measurements (e.g., ISO 2017). Additionally, lacking exposure-response data as discussed above remains a key data gap in future risk assessments and offers a pathway for identifying future research.

## Conclusions

This chapter describes an existing risk-based framework applied to assess potential biological impacts associated with anthropogenic underwater sounds. This tiered and iterative framework allows a transparent and repeatable process for evaluating and communicating ecological risks in order to efficiently reach actionable risk management decisions. The present case study demonstrated that a screening-level assessment was achievable using publicly available sound exposure modeling tools (i.e., NMFS 2018b), user spreadsheet, and qualitative threshold guidelines (Popper et al. 2014). Results indicated that using conservative exposure assumptions, dredging sounds in the Port of Alaska do not pose an adverse risk to the hearing of marine mammal or fish species of concern. Sounds generated by dredging and related activities are similar in intensity and waveform to those associated with other harbor activities.

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# Footprint of Sound Emissions from Fishing Vessels in the Adriatic Sea

# 123

Federica Pace, Samuel J. Welch, Nicola Ferri, and Aurora Nastasi

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**Abstract**

An underwater sound modeling assessment was performed of fishing boats in the Jabuka/Pomo Pit Fisheries Restricted Area (FRA) in the Adriatic Sea, offshore of Italy and Croatia. Underwater sound fields were modeled for two types of fishing vessels employing either longlines (LLS) or otter-bottom trawls (OTB).

Estimated broadband source levels ranged between 149 and 153 dB re 1 μPa m for LLS vessels and 171 and 175 dB re 1 μPa m for OTB vessels. Sound fields were calculated for sound pressure levels (SPL) to compare with available impact criteria. The SPL thresholds for recoverable barotrauma injury and temporary hearing threshold shift (TTS) pressure-sensitive fish from Popper et al. (2014) of 170 and 158 dB re 1 μPa, respectively, were not reached for any vessel at any location. The modeled levels considered only the sound generated from fishing vessels and did not account for other sound sources. The maximum modeled range to the behavioral response threshold for marine mammals from NOAA (ESA Section 7 consultation tools for marine mammals on the west coast (webpage), 2019) of 120 dB re 1 μPa was 0.91 km. Modeling results indicate that underwater noise produced by LLS and OTB on marine fauna do not represent a threat to biodiversity.

**Keywords**

Sound · Behavior · Impact · Injury · Mediterranean Sea · Bottom trawling fishing · Demersal longline fisheries

## Introduction

This chapter presents an underwater noise modeling assessment of fishing activities in the Jabuka/Pomo Pit Fisheries Restricted Area (FRA) established by the General Fisheries Commission for the Mediterranean Sea (GFCM) in 2017 (GFCM 2021) in the Adriatic Sea offshore of Italy and Croatia (GFCM Geographical Subarea, GSA, 17). The demersal fisheries in this area target valuable species such as the European hake (*Merluccius merluccius*) and the Norway lobster (*Nephrops norvegicus*), among others.

The Adriatic Sea is one of the most productive seas of the Mediterranean. Promoting the recovery of essential fish habitats and ecosystems is a key priority for rebuilding fish stocks and supporting sustainable fisheries. With this aim, GFCM established the FRA in 2017 (Fig. 1). The area was clearly identified as (1) a site of unique physical features influencing the dynamics of water circulation in the whole Mediterranean basin; (2) hosting important essential fish habitats (EFHs) for European hake and other valuable species such as horned octopus (*Eledone cirrhosa*), monkfish (*Lophius budegassa*), and Norway lobster; (3) a key area for cetaceans, sea turtles, and seabirds; and (4) hosting vulnerable marine ecosystems (VMEs) that might be impacted by bottom-trawling fishing.

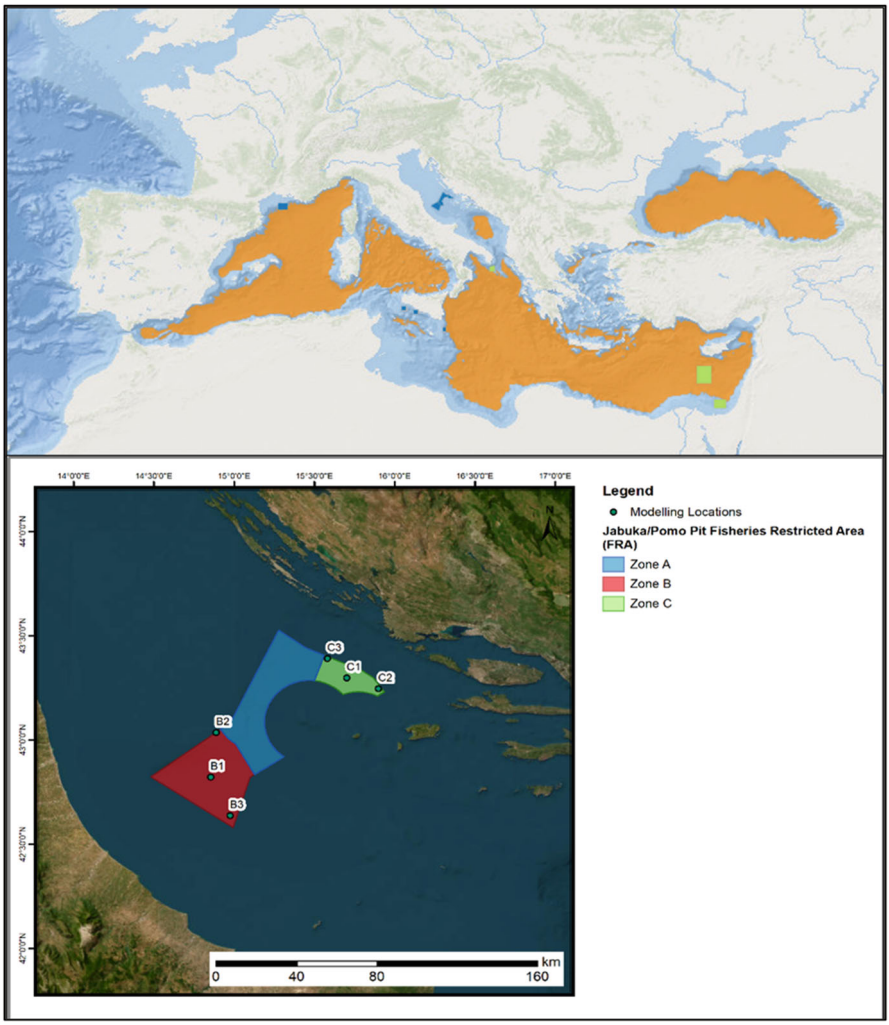



**Fig. 1** The Jabuka/Pomo Pit GFCM Fisheries Restricted Area (FRA) and its zoning (Zone A: No fishing permitted; Zones B and Z: demersal fishing permitted and regulated by GFCM) (top). Selected sites for modeling are depicted by the green dots and labels for each zone (marked by different shading, bottom panel)

The Jabuka/Pomo Pit FRA is divided into three Zones: A, B, and C (Fig. 1). No fishing is permitted in Zone A at any time, while fishing is permitted in Zones B and C subject to restrictions imposed by GFCM.

Modeling of underwater sound emissions from two specific fishing activities (i.e., bottom trawling, longlining) was therefore assessed for the two areas where fishing is permitted (i.e., Zones B and C) to understand whether the underwater noise

generated by these activities could have indirect detrimental effects on those demersal fish and invertebrate stocks that the FRA aims at rebuilding.

The study considered realistic scenarios for the fishing activities, as much as possible, according to the permitted schedule for fishing activity.

## Modeled Scenarios

Three modeled sites shown in Fig. 1 (bottom) were selected in each of Zones B and C corresponding to the geometric center of each zone and two others based on geography and bathymetry. No sources were modeled in Zone A since fishing is prohibited in that area. GFCM provided a spreadsheet containing details of Italian and Croatian fishing vessels, specifically set longlines (LLS) and otter-board bottom trawlers (OTB). Both vessel types were separated by size into small (S), medium (M), and large (L) categories, and each vessel type and size combination was modeled at each modeling location to produce 36 single vessel sound fields.

The likely fishing schedule in each zone was considered when assessing the multiple vessel sound fields. In Zone B, authorized Italian fishing vessels are allowed to operate on any day of the week, with any single vessel entitled to no more than two fishing days per week. In Zone C, authorized Croatian vessels are only allowed to operate on specific days based on fishing gear; LLS vessels are only entitled to fish Monday to Thursday, while OTB vessels are only entitled to fish Saturday and Sunday. Two scenarios were therefore modeled considering sound fields from multiple vessels:

- Croatian OTB vessels operating in Zone C and a mixture of Italian vessels in Zone B, representing Saturday to Sunday
- Croatian LLS vessels operating in Zone C and a mixture of Italian vessels in Zone B, representing Monday to Thursday

These two scenarios were chosen to represent the worst-case scenario for the maximum number of vessels operating in the region simultaneously. Since neither vessel type considered is entitled to fish in Zone C on Fridays, this day is not represented in either scenario.

## Sound Impacts on Fish and Invertebrates

While the impact of ocean noise pollution on marine mammals has long been the focus of scientific studies and research, the impacts of underwater noise on fish have only recently started to be investigated to a greater extent. Studies on invertebrates and noise are still in their infancy (Popper and Hawkins 2012, 2016). Findings suggest that ocean noise may have adverse impacts on fish and invertebrate development. These include the appearance of body malformations, increased mortality of eggs and juveniles, and delays in growth and/or development (Aguilar de Soto et al. 2013;

Nedelec et al. 2014; McCauley et al. 2017). Furthermore, the anatomy (André et al. 2011; Solé et al. 2013b; André et al. 2016) and the physiology of fish and invertebrates (Day et al. 2017) may be impacted by sound, depending on its level, frequency, and duration of the source.

Research is ongoing into the relationship between sound and its effects on marine invertebrates, including the relevant metrics for both effect and impact. The literature suggests invertebrate hearing happens primarily through particle motion detection via the statocyst (a sensory organ providing orientation cues) (Solé et al. 2013a; Tidau and Briffa 2016).

There remains limited real-world data on absolute levels of sound pressure or particle motion from nonimpulsive sources inducing adverse effects in benthic invertebrates (Hawkins et al. 2015; Popper and Hawkins 2018); hence, there are no current widely accepted criteria for assessing the impact of this type of noise on these animals. Some studies identified short-term behavioral and physiological effects on invertebrates exposed to vessel noise. For example, Solan et al. (2016) found that Norway lobster repressed burying behavior and bioirrigation, and reduced locomotion when exposed to broadband continuous noise between Sound Pressure Levels (Lp) of 135–140 dB re 1 μPa (in tank experiment). Other behavioral effects have been observed in crustaceans exposed to vessel noise; these include disruption in feeding in the common shore crab (*Carcinus maenas*), increase in distance and velocity traveled by groups of European spiny lobster (*Palinurus elephas*), and a rise in resting behavior for individual common prawn (*Palaemon serratus*) (Tidau and Briffa 2016; Weilgart 2018). Furthermore, a meta-analysis study on various taxonomic groups of marine invertebrates (Murchy et al. 2019) showed a statistically significant impact on the behavior and physiology (e.g., reduction in foraging and predator response) of cephalopods and gastropods exposed to shipping noise.

## Fishing Vessels Underwater Sound Emissions

The sound signature of a vessel depends on its size, power output, and the characteristics of its propulsion system. Vessels generate broadband acoustic energy. Most energy is in the low frequencies; sound below 200 Hz from onboard machinery dominates the sound spectrum before cavitation begins (Spence et al. 2007). The underwater sound that radiates from a vessel (Underwater Radiated Noise; URN) is produced mainly by propeller and thruster cavitation (Ross 1976, §8.6), with a smaller fraction of noise produced by sound transmitted through the hull, such as by engines, gearing, and other mechanical systems. Sound levels thus tend to be highest when propulsion systems are used at high power, for example, during dynamic positioning or transiting at high speeds.

URN from fishing vessels has received little attention thus far, likely because a wide variety of techniques and types of vessels are used for these activities. A few studies are available in peer-reviewed literature that characterize fishing vessels as an underwater sound source. Recent studies have focused on small fishing vessels that are used for local fisheries and on describing the sound that is generated during the

transit of such vessels (Amron et al. 2021; Picciulin et al. 2022). The authors are not aware of any studies that characterize the sound emitted by vessels during fishing activities or describe the tools and machinery involved with different fishing methods.

Characterizing the URN from fishing vessels presents similar challenges to those described for dredging activity in that the noise-generating activities not only include the standard sources of transiting vessels but also the machinery and activities of dredging (CEDA 2011; WODA 2013; Thomsen et al. 2016).

Daly and White (2021) recorded two trawlers in two locations and calculated decidecade source levels using propagation loss modeling. There is no size information for one of the vessels measured, but the other is listed as 17–20 m long with a calculated broadband source level between 170 and 173 dB re 1 μPa m. With the exception of the studies by Picciulin et al. (2022) and MacGillivray and de Jong (2021), measurement standards, such as (ISO 17208-1:2016; ANSI and ASA S12.64-2009), were not followed.

## Modeling Methods

A combined range-dependent parabolic equation and Gaussian beam acoustic ray-trace model (the Marine Operations Noise Model MONM-BELLHOP, a proprietary model developed by JASCO Applied Sciences) was used to predict the acoustic field around the vessels for frequencies from 10 Hz to 25 kHz.

The acoustic source spectrum of each vessel was used in the propagation modeling to calculate the $L_p$ field in the modeled area. The modeled $L_p$ only consider the generated noise field from fishing activities and do not account for the existing ambient noise levels in the area.

Each ship's sound field is modeled at a distance of at least 85 km from the source location, with a horizontal spacing of 10 m between receivers. The horizontal angular resolution was 2.5° for a total of 144 radials. Receiver depths each 1 m up to 270 m were modeled to cover the entire water column, with step sizes increasing with depth.

### Source Levels

Based on the list of vessels provided by GFCM, the vessels in each category (OTB and LLS) were split evenly into three subcategories by length overall (LOA): small, medium, and large. The median vessel in each size category was used as a proxy for that category. Where available, draft information was obtained from an online vessel database service (VesselFinder 2021). For the small and large LLS proxy vessels where no information on vessel draft was available, the draft was estimated by multiplying each LOA by the draft to LOA ratio of the medium LLS proxy vessel. Table 1 presents a summary of the proxy vessel properties for each category.

Estimated source level spectra were calculated using the JOMOPANS-ECHO model developed by MacGillivray and de Jong (2021). The model calculates a ship source level spectrum based on vessel size and speed, while also incorporating specific reference speeds and spectrum coefficients based on AIS ship type.

Source spectra for each vessel category were calculated using the vessel lengths as specified in Table 1 and the appropriate parameters for the fishing vessel category. The vessel speed for LLS vessels was set as 6 knots, based on a realistic worst-case speed for this type of vessel as determined by AIS data for a single vessel provided by GFCM. Trawlers typically travel at lower speeds than this when actively trawling, around 3–4 knots. There is evidence to suggest, however, that noise levels are higher when trawling than transiting at similar speeds. De Robertis and Wilson (2006) determined that a vessel pulling a pelagic trawl at 3.3 knots produced similar spectrum levels to the same vessel free running at 11.2 knots. Similarly, Hovem et al. (2015) presented measurements showing that a vessel bottom trawling at 4 knots produced slightly higher noise levels than the same vessel transiting at 12.4 knots. The speed for OTB vessels was therefore approximated as 12 knots to represent the trawling condition. Table 2 and Fig. 2 present the resulting source levels and spectra, respectively, for all vessel types.

Typically, the acoustic source depth of a vessel can be estimated from the vessel draft and the diameter of the propeller (Gray and Greeley 1980); however, in the absence of propeller diameter information, the source depth was approximated as two-thirds of the vessel draft.

Since set longlines are static in the water, the fishing gear for the LLS vessels is unlikely to generate noise at any substantial level compared to the vessel-generated

**Table 1** A summary of proxy vessel properties for the modeled vessel categories

| Vessel category | | *Vessel* | LOA (m) | Draft (m) | Total power (kW) | Source depth (m) | Broadband source level (dB re 1 µPa m) |
|---|---|---|---|---|---|---|---|
| LLS | S | *215-KŽ* | 9.9 | Unknown | 55 | 1.5 | 149.2 |
| | M | *Ariete I* | 12.8 | 3.0 | 81 | 2.0 | 151.4 |
| | L | *Vitoantonia* | 14.8 | Unknown | 169 | 2.3 | 152.6 |
| OTB | S | *Indomita* | 15.9 | 2.3 | 220 | 1.5 | 171.3 |
| | M | *Princeza Grejn* | 20.2 | 3.2 | 276 | 2.1 | 173.4 |
| | L | *Marc* | 24.8 | 4.0 | 445 | 2.7 | 175.2 |

**Table 2** Broadband monopole source levels (MSL; dB re 1 µPa·m) for the modeled vessel categories

| Vessel category | | Broadband MSL (dB re 1 µPa m) |
|---|---|---|
| LLS | S | 149.2 |
| | M | 151.4 |
| | L | 152.6 |
| OTB | S | 171.3 |
| | M | 173.4 |
| | L | 175.2 |

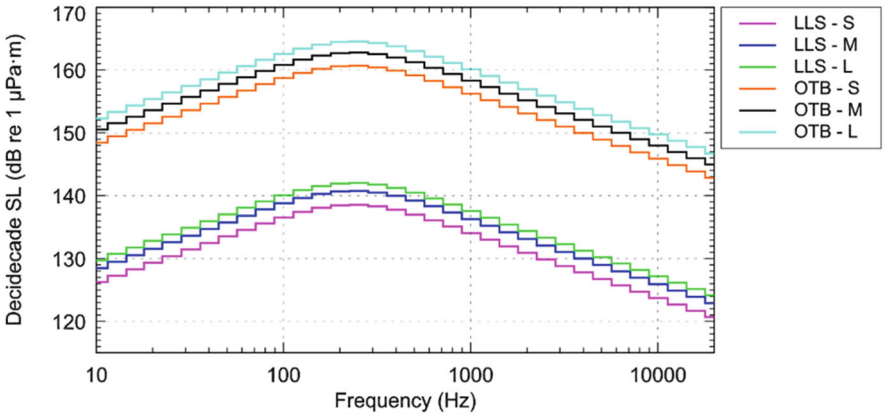



**Fig. 2** Modeled source spectra for the three modeled sizes of OTB and LLS fishing vessels

noise. Conversely, the trawl gear used by the OTB vessels is mobile and hence more likely to generate additional noise on a more consistent basis through humming in trawl cables under tension, rattling chains and shackles, and noise related to the otter boards contacting the seabed (Daly and White 2021). Several studies measured the noise from demersal trawl gear, and in all cases the vessel-generated noise was considered to dominate, particularly in the case of soft sediment (Chapman and Hawkins 1969; Buerkle 1977). For these reasons, the vessel has been considered the primary noise source, and the source depth has been chosen accordingly in this study (Table 1).

## Environmental Parameters

Water depths throughout the modeled area were extracted from the EMODnet European bathymetry grid, a 1/16 arc minute resolution (approximately 115 × 115 m) grid rendered for European sea basins (EMODnet Bathymetry Consortium 2020).

The sound speed profiles (SSP) of the modeled locations were derived from the temperature and salinity profiles of the US Naval Ocean Service's Generalized Digital Environment Model V 3.0 (GDEM; Teague et al. 1990; Carnes 2009) (monthly temporal resolution). The climate profile contains 78 fixed depth points up to a maximum depth of 6800 m; the Mediterranean sea does not reach this depth (in any part). The GDEM temperature-salinity profiles were converted to sound speed profiles according to Coppens (1981). GFCM requested the SSP for June to be modeled. Figure 3 shows the modeled SSP.

In a shallow environment, interactions between the acoustic field and the seabed are important, and accurate geoacoustic profiles are needed for proper acoustic modeling. Since the modeled area is large and geoacoustic information is limited,

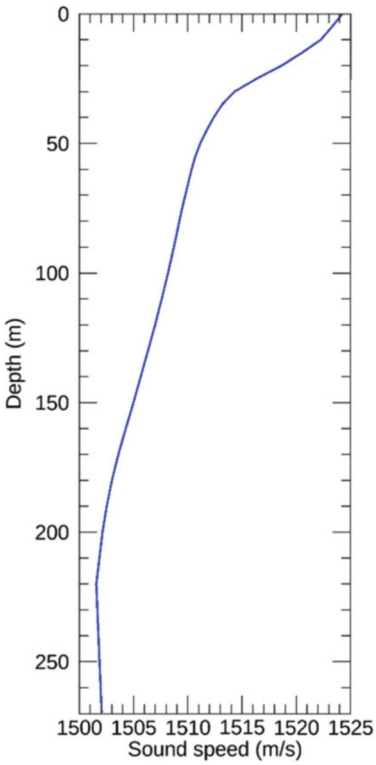



**Fig. 3** Mean monthly sound speed profile for June, derived from data obtained from GDEM V 3.0 (Teague et al. 1990, Carnes 2009)

two simplified geoacoustic profiles were constructed to represent the major features of the sediment at the modeled sites.

The modeled sites were divided into two geoacoustic areas based on the water depth at the source. Sites with a water depth greater than 150 m were categorized as an organic clay-loam substrate (here modeled as calcareous silt-clay), while sites with a water depth shallower than 150 m were categorized as a terrigenous sand-silt-clay, based on information provided by GFCM and the EMODnet Seabed Substrate map (Correggiari et al. in press). The EMODnet Seabed Substrate map is made available by the EMODnet Geology project (http://www.emodnet-geology.eu) funded by the European Commission Directorate General for Maritime Affairs and Fisheries. Well logs available through the *Visibilità dei Dati afferenti all'attività di Esplorazione Petrolifera in Italia* (ViDEPI; Società Geologica Italiana 2021) project indicate that sediment is likely quite thick in the region of interest, and, in the absence of more detailed information, no other subsurface layers were modeled.

The depth-dependent geoacoustic profiles presented in Tables 3 and 4 were calculated from the values of Hamilton (1980) for water depths greater than 150 m and shallower than 150 m, respectively. MONM-BELLHOP only considers shear

**Table 3** Estimated geoacoustic profile for Sites B1, B3, and C3, which have a water depth at source of greater than 150 m. Within each depth range, each parameter varies linearly within the specified range. The compressional (P) wave is the primary wave. The shear (S) wave is the secondary wave

| Depth below seafloor (m) | Material | Density ($g/cm^3$) | Compressional wave | | Shear wave | |
|---|---|---|---|---|---|---|
| | | | Speed (m/s) | Attenuation (dB/λ) | Speed (m/s) | Attenuation (dB/λ) |
| 0.0–62.5 | Silt-clay | 1.40–1.51 | 1510–1610 | 0.17–0.21 | 116 | 2.00 |
| 62.5–125.0 | | 1.51–1.60 | 1610–1710 | 0.21–0.35 | | |
| 125.0–187.5 | | 1.60–1.67 | 1710–1810 | 0.35–0.73 | | |
| 187.5–250.0 | | 1.67–1.72 | 1810–1910 | 0.73–1.07 | | |
| 250.0–312.5 | | 1.72–1.76 | 1910–2010 | 1.07–1.38 | | |
| 312.5–375.5 | | 1.76–1.80 | 2010–2100 | 1.38–1.64 | | |
| 375.0–437.5 | | 1.80–1.83 | 2100–2200 | 1.64–1.50 | | |
| 437.5–500.0 | | 1.83–1.86 | 2200–2270 | 1.50–1.35 | | |

**Table 4** Estimated geoacoustic profile for Sites B2, C1, and C2, which have a water depth at source shallower than 150 m. Within each depth range, each parameter varies linearly within the specified range. The compressional (P) wave is the primary wave. The shear (S) wave is the secondary wave

| Depth below seafloor (m) | Material | Density ($g/cm^3$) | Compressional wave | | Shear wave | |
|---|---|---|---|---|---|---|
| | | | Speed (m/s) | Attenuation (dB/λ) | Speed (m/s) | Attenuation (dB/λ) |
| 0.0–62.5 | Sand-silt-clay | 1.60–1.68 | 1550–1630 | 0.23–0.63 | 250 | 3.65 |
| 62.5–125.0 | | 1.68–1.76 | 1630–1700 | 0.63–1.04 | | |
| 125.0–187.5 | | 1.76–1.83 | 1700–1770 | 1.04–1.32 | | |
| 187.5–250.0 | | 1.83–1.90 | 1770–1830 | 1.32–0.98 | | |
| 250.0–312.5 | | 1.90–1.97 | 1830–1890 | 0.98–0.92 | | |
| 312.5–375.5 | | 1.97–2.03 | 1890–1950 | 0.92–0.91 | | |
| 375.0–437.5 | | 2.03–2.09 | 1950–2000 | 0.91–0.89 | | |
| 437.5–500.0 | | 2.09–2.14 | 2000–2050 | 0.89–0.87 | | |

wave properties of the surficial layer; hence, these values are constant throughout the seabed.

## Estimating Impact Ranges

The noise footprint for multiple vessels operating simultaneously within the FRA zones was analyzed by modeling the $L_p$ for each vessel type at the individual transmission loss modeling sites and by transposing and summing these footprints at various other locations within the FRA zones. This method acceptably reflects large-scale sound propagation features, primarily dependent on water depth, which dominate the multiple source field, and is thus considered to provide a meaningful estimate of the sound field.

Two multiple vessel scenarios were determined based on the local fishing regulations in Zones B and C, set by Italy and Croatia, respectively. Scenario 1 represents a day where only OTB vessels are allowed to fish in Zone C, and Scenario 2 represents a day where only LLS are allowed to fish in Zone C. As a conservative estimate of the number of vessels in both scenarios, it was assumed that all authorized Croatian vessels using the relevant fishing gear type were present in Zone C. Since vessels are only allowed to fish for 2 days in any given week in Zone B, it was assumed that approximately one-quarter of the authorized Italian vessels using each fishing gear type were present. Table 5 presents a breakdown of the number of the vessels modeled in each FRA zone.

Source locations for each scenario were generated by using Geographic Information System (GIS) software to randomly distribute the correct number of source locations in each FRA zone. The minimum point separation parameter of the GIS tool was adjusted to ensure the points were approximately evenly distributed throughout the area. The source locations within each zone were then arbitrarily ordered, and a list randomizer was used to assign vessel categories to each location as appropriate for the zone and scenario. The modeled single vessel sound field within the relevant zone (either B or C) with the closest water depth at source was selected to represent the sound field at each source location.

Sound level contours are based on the underwater sound field predicted by the propagation model sampled at both the seafloor and represent the maxima of all modeled depths above the seafloor for each location. From these contours, predicted distances to specific planes were calculated. Two distances to the sound source are given for each noise level, i.e., (1) $R_{max}$, the maximum range at a specified sound level in all directions; (2) $R_{95,}$ the range at a specified sound level after excluding the 5% of the furthest point.

The $R_{95}$ value is used because sound field footprints are often irregularly shaped. Occasionally, noise level contours may show small ridges or unusual isolated fringes. $R_{95}$ ignores important projection considerations in floor plans. In such

**Table 5** A summary of the vessel types modeled in each FRA zone for each multiple vessel scenario

| FRA zone | Vessel category | | Number of modeled vessels | |
|---|---|---|---|---|
| | | | Scenario 1 | Scenario 2 |
| B | LLS | S | 0 | 0 |
| | | M | 2 | 2 |
| | | L | 3 | 3 |
| | OTB | S | 2 | 2 |
| | | M | 6 | 6 |
| | | L | 6 | 6 |
| C | LLS | S | – | 11 |
| | | M | – | 1 |
| | | L | – | 0 |
| | OTB | S | 26 | – |
| | | M | 13 | – |
| | | L | 12 | – |

cases, $R_{max}$ may better represent the range of motion in a particular direction. Such cases are usually associated with bathymetric topography that affects propagation. The difference between $R_{max}$ and $R_{95}$ depends on the directivity of the sound source and the heterogeneity of the acoustic environment.

## Results

This section presents results for the modeled scenarios involving single vessels and multiple vessels in sections "Single Vessel" and "Multiple Vessel," respectively. The tabulated results present the maximum and 95% distances to specified isopleths and impact criteria. Results are presented at the seafloor and as maximum-over-depth, meaning that the specified threshold is reached at some point in the water column at the distance presented.

## Single Vessel

Table 6 presents received SPLs near the seabed at 750, 1500, and 3000 m from the source for the six vessel types at the six modeled locations. Results are the maximum

**Table 6** Modeled $L_p$ at three distances (750, 1500, and 3000 m) from each modeled source at all modeled locations. Levels are given near the seabed and are the maximum along any radial around the source

| Vessel category | | Distance from source (m) | $L_p$ (dB re 1 μPa) near seabed | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | Site | | | | | |
| | | | B1 | B2 | B3 | C1 | C2 | C3 |
| LLS | S | 750 | 92.9 | 94.0 | 92.5 | 94.4 | 93.7 | 92.1 |
| | | 1500 | 82.6 | 83.9 | 80.6 | 82.5 | 83.9 | 78.6 |
| | | 3000 | 68.3 | 80.9 | 67.4 | 80.7 | 81.2 | 67.1 |
| | M | 750 | 96.0 | 97.7 | 95.7 | 98.1 | 97.5 | 95.5 |
| | | 1500 | 86.4 | 87.3 | 84.3 | 85.8 | 87.4 | 82.5 |
| | | 3000 | 72.1 | 84.8 | 71.4 | 84.6 | 85.1 | 71.3 |
| | L | 750 | 97.6 | 99.7 | 97.3 | 100.0 | 99.4 | 97.2 |
| | | 1500 | 88.4 | 89.1 | 86.3 | 87.5 | 89.2 | 84.6 |
| | | 3000 | 74.1 | 86.8 | 73.6 | 86.6 | 87.0 | 73.5 |
| OTB | S | 750 | 115.0 | 116.1 | 114.6 | 116.6 | 115.9 | 114.3 |
| | | 1500 | 104.8 | 106.0 | 102.7 | 104.6 | 106.1 | 100.8 |
| | | 3000 | 90.5 | 103.1 | 89.5 | 102.9 | 103.4 | 89.3 |
| | M | 750 | 118.1 | 120.0 | 117.9 | 120.4 | 119.8 | 117.7 |
| | | 1500 | 108.7 | 109.6 | 106.6 | 108.0 | 109.6 | 104.8 |
| | | 3000 | 94.4 | 107.1 | 93.8 | 106.9 | 107.4 | 93.6 |
| | L | 750 | 120.5 | 122.9 | 120.3 | 123.2 | 122.7 | 120.2 |
| | | 1500 | 111.8 | 112.2 | 109.8 | 110.6 | 112.3 | 108.1 |
| | | 3000 | 97.5 | 110.1 | 97.2 | 110.0 | 110.3 | 97.1 |

level modeled along any radial around the source at the specified distance, and hence may not be along the same radial for different distances. Tables 7, 8, 9, 10, 11, and 12 list ranges to various $L_p$ isopleths from each of the six vessel types. Table 13 presents ranges to the $L_p$ thresholds for fish from Popper et al. (2014).

Note that, in Tables 7, 8, 9, 10, 11, and 12, an $L_p$ of 120 dB re 1 μPa is the behavioral response threshold for marine mammals (NOAA 2019), and in all results tables, dashes indicate that the stated level was not reached within the minimum horizontal range step of the modeling.

### Multiple Vessels

This section presents results for multiple vessel Scenario 1 (a day where only OTB vessels are allowed to fish in Zone C) and Scenario 2 (a day where only LLS vessels are allowed to fish in Zone C). Figures 4 and 5 show the results. In Fig. 4, it appears as if the modeled sound field passes through some island land masses where this would not happen. This a result of the methodology where sound fields modeled elsewhere were transposed to the new modeling locations and effectively overlapping the land.

As no reference could be found in the public literature for the ambient sound levels in the study area, the results are not presented as exceedance maps; however, levels below SPL of 100 dB re 1 μPa are typically meeting background level conditions in the Mediterranean Sea (Picciulin et al. 2019).

## Discussion and Conclusion

This modeling study predicted noise levels associated with fishing activities in the Jabuka/Pomo Pit FRA. The underwater sound field was modeled at six different locations for two vessel types employing different fishing gear each split into three size categories and for two scenarios involving multiple vessels operating simultaneously based on realistic fishing schedules. A sound speed profile for June was selected considering the fishing behavior peaks and weather conditions and the model also accounted for site-specific bathymetric and local geoacoustic properties.

### Modeled Sound Levels

Modeled broadband MSLs ranged from 149 to 153 dB re 1 μPa m for LLS vessels and 171–175 dB re 1 μPa m for OTB vessels. Source levels were significantly higher for the OTB vessels than the LLS vessels since OTB were larger and more powerful in all modeled size categories (see Table 1). Additionally, the effective speed used in the source model for the OTB vessels was double the speed used for the LLS vessels to account for the increase in source levels during trawl noise. Daly and White (2021) recorded two trawlers in two locations and calculated decidecade source

**Table 7** Small LLS vessel: maximum ($R_{max}$) and 95% ($R_{95\%}$) horizontal distances (in km) to modeled maximum-over-depth $L_p$ isopleths

| | B1 | | B2 | | B3 | | C1 | | C2 | | C3 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| $L_p$ (dB re 1 µPa) | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ |
| 80 | 1.66 | 1.58 | 4.44 | 3.72 | 1.52 | 1.46 | 4.15 | 3.09 | 3.83 | 3.49 | 1.45 | 1.38 |
| 90 | 0.95 | 0.92 | 0.91 | 0.88 | 0.91 | 0.88 | 0.95 | 0.92 | 0.91 | 0.88 | 0.88 | 0.85 |
| 100 | 0.32 | 0.31 | 0.39 | 0.38 | 0.33 | 0.32 | 0.38 | 0.37 | 0.39 | 0.38 | 0.34 | 0.33 |
| 110 | 0.09 | 0.09 | 0.10 | 0.09 | 0.09 | 0.09 | 0.09 | 0.09 | 0.09 | 0.09 | 0.09 | 0.09 |
| 120[†] | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 |
| 130 | – | – | – | – | – | – | – | – | – | – | – | – |

**Table 8** Medium LLS vessel: maximum ($R_{max}$) and 95% ($R_{95\%}$) horizontal distances (in km) maximum-over-depth $L_p$ isopleths

| | B1 | | B2 | | B3 | | C1 | | C2 | | C3 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| $L_p$ (dB re 1 μPa) | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ |
| 80 | 1.83 | 1.74 | 7.39 | 5.66 | 1.83 | 1.64 | 6.65 | 4.98 | 5.99 | 4.99 | 1.79 | 1.73 |
| 90 | 1.24 | 1.19 | 1.79 | 1.66 | 1.16 | 1.12 | 1.77 | 1.65 | 1.81 | 1.65 | 1.12 | 1.07 |
| 100 | 0.48 | 0.47 | 0.62 | 0.60 | 0.48 | 0.47 | 0.61 | 0.59 | 0.62 | 0.60 | 0.48 | 0.46 |
| 110 | 0.13 | 0.13 | 0.14 | 0.14 | 0.13 | 0.13 | 0.14 | 0.13 | 0.14 | 0.13 | 0.13 | 0.13 |
| 120[†] | 0.04 | 0.04 | 0.04 | 0.04 | 0.04 | 0.04 | 0.04 | 0.04 | 0.04 | 0.04 | 0.04 | 0.04 |
| 130 | – | – | – | – | – | – | – | – | – | – | – | – |

**Table 9** Large LLS vessel: maximum ($R_{max}$) and 95% ($R_{95\%}$) horizontal distances (in km) to modeled maximum-over-depth $L_p$ isopleths

| | B1 | | B2 | | B3 | | C1 | | C2 | | C3 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| $L_p$ (dB re 1 µPa) | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ |
| 80 | 2.39 | 2.07 | 8.64 | 7.21 | 2.10 | 2.01 | 7.97 | 7.08 | 7.13 | 6.54 | 2.09 | 1.96 |
| 90 | 1.40 | 1.35 | 2.16 | 1.99 | 1.31 | 1.26 | 2.23 | 2.01 | 2.17 | 2.00 | 1.26 | 1.20 |
| 100 | 0.58 | 0.56 | 0.73 | 0.71 | 0.57 | 0.55 | 0.75 | 0.73 | 0.72 | 0.70 | 0.57 | 0.55 |
| 110 | 0.16 | 0.16 | 0.17 | 0.16 | 0.16 | 0.16 | 0.16 | 0.16 | 0.17 | 0.16 | 0.16 | 0.16 |
| 120[†] | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 |
| 130 | – | – | – | – | – | – | – | – | – | – | – | – |

**Table 10** Small OTB vessel: maximum ($R_{max}$) and 95% ($R_{95\%}$) horizontal distances (in km) to modeled maximum-over-depth $L_p$ isopleths

| | B1 | | B2 | | B3 | | C1 | | C2 | | C3 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| $L_p$ (dB re 1 μPa) | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ |
| 80 | 5.94 | 5.49 | 43.4 | 32.8 | 5.47 | 5.07 | 34.9 | 26.9 | 26.4 | 21.4 | 5.39 | 4.98 |
| 90 | 3.71 | 3.32 | 19.9 | 13.5 | 3.16 | 2.96 | 14.3 | 12.4 | 13.10 | 10.92 | 2.99 | 2.79 |
| 100 | 1.76 | 1.68 | 6.00 | 4.66 | 1.61 | 1.55 | 5.19 | 4.61 | 4.78 | 4.41 | 1.53 | 1.46 |
| 110 | 1.12 | 1.07 | 1.57 | 1.43 | 1.05 | 1.02 | 1.51 | 1.43 | 1.56 | 1.40 | 1.01 | 0.98 |
| 120† | 0.44 | 0.42 | 0.52 | 0.50 | 0.44 | 0.43 | 0.51 | 0.49 | 0.52 | 0.50 | 0.44 | 0.42 |
| 130 | 0.12 | 0.12 | 0.13 | 0.13 | 0.13 | 0.12 | 0.13 | 0.13 | 0.13 | 0.13 | 0.12 | 0.12 |
| 140 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 |
| 150 | – | – | – | – | – | – | – | – | – | – | – | – |

**Table 11** Medium OTB vessel: maximum ($R_{max}$) and 95% ($R_{95\%}$) horizontal distances (in km) to modeled maximum-over-depth $L_p$ isopleths

| | B1 | | B2 | | B3 | | C1 | | C2 | | C3 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| $L_p$ (dB re 1 μPa) | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ |
| 80 | 8.90 | 7.74 | 50.8 | 42.5 | 8.03 | 7.07 | 45.1 | 36.0 | 35.7 | 27.9 | 7.78 | 6.86 |
| 90 | 4.65 | 4.10 | 26.6 | 19.6 | 4.15 | 3.91 | 21.1 | 17.1 | 18.8 | 14.6 | 4.00 | 3.75 |
| 100 | 2.43 | 2.12 | 8.80 | 7.36 | 2.16 | 2.05 | 8.01 | 7.21 | 7.43 | 6.67 | 2.13 | 2.00 |
| 110 | 1.42 | 1.37 | 2.20 | 2.04 | 1.32 | 1.28 | 2.27 | 2.07 | 2.20 | 2.07 | 1.27 | 1.22 |
| 120† | 0.61 | 0.60 | 0.74 | 0.72 | 0.60 | 0.59 | 0.77 | 0.74 | 0.74 | 0.72 | 0.60 | 0.58 |
| 130 | 0.17 | 0.17 | 0.19 | 0.18 | 0.18 | 0.17 | 0.18 | 0.18 | 0.18 | 0.17 | 0.18 | 0.17 |
| 140 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 |
| 150 | – | – | – | – | – | – | – | – | – | – | – | – |

**Table 12** Large OTB vessel: maximum ($R_{max}$) and 95% ($R_{95\%}$) horizontal distances (in km) to modeled maximum-over-depth $L_p$ isopleths

| | B1 | | B2 | | B3 | | C1 | | C2 | | C3 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| $L_p$ (dB re 1 μPa) | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ | $R_{max}$ | $R_{95\%}$ |
| 80 | 11.5 | 10.1 | 60.8 | 50.1 | 10.1 | 9.21 | 48.0 | 41.6 | 45.7 | 34.5 | 10.5 | 9.51 |
| 90 | 5.65 | 5.26 | 38.9 | 26.8 | 4.98 | 4.68 | 30.7 | 22.5 | 23.7 | 18.6 | 4.85 | 4.52 |
| 100 | 3.25 | 2.86 | 12.7 | 9.87 | 2.86 | 2.67 | 11.0 | 9.76 | 10.9 | 8.86 | 2.71 | 2.54 |
| 110 | 1.62 | 1.55 | 3.98 | 2.84 | 1.49 | 1.44 | 3.16 | 2.73 | 3.48 | 2.82 | 1.42 | 1.36 |
| 120† | 0.79 | 0.77 | 0.89 | 0.85 | 0.77 | 0.75 | 0.91 | 0.88 | 0.87 | 0.84 | 0.76 | 0.74 |
| 130 | 0.23 | 0.23 | 0.26 | 0.26 | 0.23 | 0.23 | 0.25 | 0.24 | 0.26 | 0.26 | 0.23 | 0.23 |
| 140 | 0.06 | 0.06 | 0.06 | 0.06 | 0.06 | 0.06 | 0.06 | 0.06 | 0.06 | 0.06 | 0.06 | 0.06 |
| 150 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 | 0.02 |

**Table 13** Modeled distances to impact thresholds for fish specified in Popper et al. (2014) from each modeled source at all modeled locations

| Vessel category | | Criteria for fish: swim bladder involved in hearing | | $R_{max}$ (km) Site | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Impairment | $L_p$ threshold (dB re 1 µPa) | B1 | B2 | B3 | C1 | C2 | C3 |
| LLS | S | TTS | 158 | – | – | – | – | – | – |
| | | Recoverable injury | 170 | – | – | – | – | – | – |
| | M | TTS | 158 | – | – | – | – | – | – |
| | | Recoverable injury | 170 | – | – | – | – | – | – |
| | L | TTS | 158 | – | – | – | – | – | – |
| | | Recoverable injury | 170 | – | – | – | – | – | – |
| OTB | S | TTS | 158 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 |
| | | Recoverable injury | 170 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 |
| | M | TTS | 158 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 |
| | | Recoverable injury | 170 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 |
| | L | TTS | 158 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 |
| | | Recoverable injury | 170 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 | <0.02 |

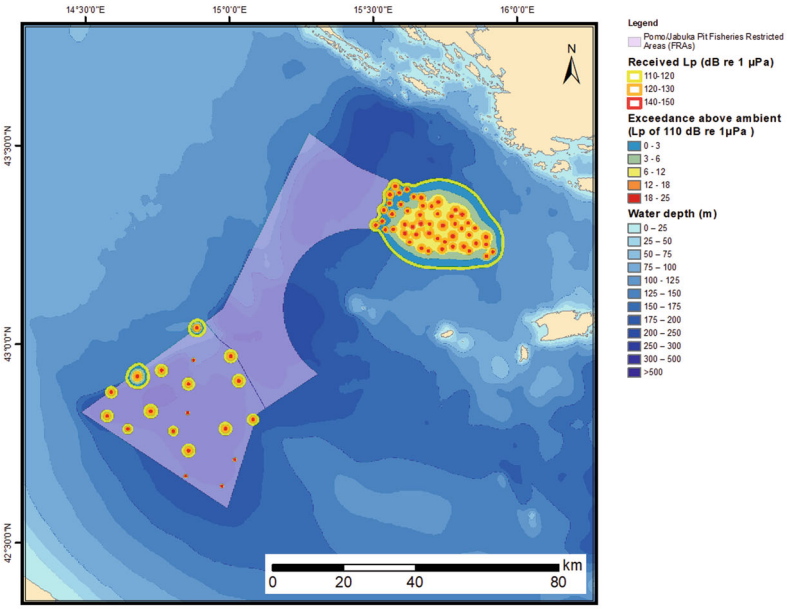



**Fig. 4** Scenario 1, $L_p$: Sound level contour map showing unweighted maximum-over-depth results. Modeled vessels are Croatian OTB vessels operating in Zone C and a mixture of Italian LLS and OTB vessels in Zone B representing Saturday to Sunday. The 120 dB re 1 µPa behavioral response threshold for marine mammals (NOAA 2019) is outlined in gray. (©JASCO Applied Sciences)

levels using propagation loss modeling. There is no size information for one of the vessels measured, but the other is listed as 17–20 m long with a calculated broadband source level between 170 and 173 dB re 1 µPa m. The small OTB vessel modeled in this study was 15.9 m LOA with a broadband source level of 171.3 dB re 1 µPa m, and the medium OTB vessel was 17.1 m LOA with a broadband source level of 173.4 dB re 1 µPa m. While the modeled broadband levels are similar to those measured by Daly and White (2021), the spectrum was different, which is expected when comparing measured data to a generalized source model. It should also be noted that the source levels determined by Daly and White (2021) appear to have been calculated opportunistically rather than according to ISO 17208 (Part 1: 2016, Part 2: 2019).

Predicted $L_p$ near the seabed was extracted at three distances (750, 1500, and 3000 m) from the modeled single-vessel source locations to assess the impact on the demersal fauna of the lowest, and highest predicted levels at all distances were observed for the small LLS and large OTB vessels, respectively. The predicted levels at 750 m ranged from 92 to 123 dB re 1 µPa (at C3 and C1, respectively); levels at 1500 m ranged from 79 to 112 dB re 1 µPa (at C3 and C2, respectively); and levels at 3000 m ranged from 67 to 110 dB re 1 µPa (at C3 and C2, respectively). The SPL thresholds for recoverable injury and TTS in pressure-sensitive fishes from Popper

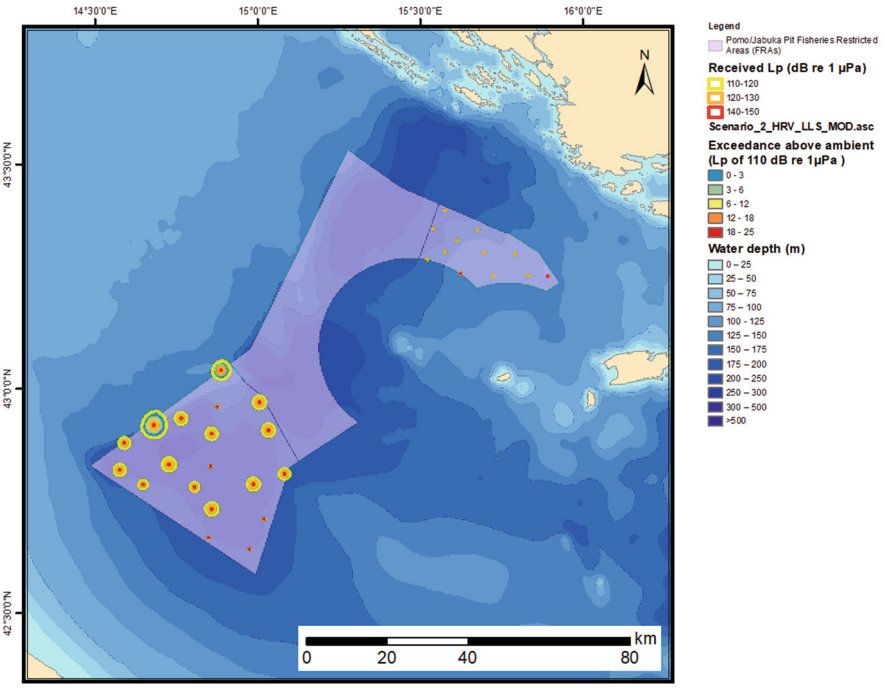



**Fig. 5** Scenario 2, $L_p$: Sound level contour map showing unweighted maximum-over-depth results. Modeled vessels are Croatian LLS vessels operating in Zone C and a mixture of Italian LLS and OTB vessels in Zone B representing Monday to Thursday. The 120 dB re 1 µPa behavioral response threshold for marine mammals (NOAA 2019) is outlined in gray. (©JASCO Applied Sciences)

et al. (2014) of 170 and 158 dB re 1 µPa, respectively, were not reached for any vessel at any location within the resolution of the modeling (20 m), although source levels for all sizes of OTB vessels exceeded these values. The ranges to these thresholds for OTB vessels can therefore be considered to lie at some distance within 20 m of the source (Table 13). Ranges to the behavioral response threshold for marine mammals from NOAA (2019) of 120 dB re 1 µPa varied between 0.03 and 0.05 km for LLS vessels and between 0.44 and 0.91 km for OTB vessels. Ranges to levels below approximately 110 dB re 1 µPa are significantly longer at the shallower sites (B2, C1, and C2) compared to the deeper sites (B1, B3, and C3). This is likely due to the complex interactions of the sound emanating from the source with the sea surface and seabed, which is further exaggerated by the difference in geoacoustics between the different groups of sites; the coarser-grained sediment at the shallower sites is more reflective than the softer sediment in the deeper waters, and hence sound propagates farther.

Results of the multiple vessel scenarios indicated that with vessels operating in close proximity, there is some coalescing of the individual sound fields into a single

larger sound field, which is largely dominated by the noise from OTB vessels (see the difference between Figs. 4 and 5). In Scenario 1 with OTB vessels only operating in Zone C (representing Saturday to Sunday), the entirety of Zone C is ensonified above 110 dB re 1 μPa, and there are some coalesced areas ensonified above 120 dB re 1 μPa where several large and medium OTB vessels are near each other. In Scenario 2 with LLS vessels only operating in Zone C (representing Monday to Thursday), the overall ensonified area is much smaller since the overall number of vessels is lower, and no OTB vessels are operating in Zone C.

**Comparing the modeled levels to the subjective assessment of relative risk for fishes from Popper et al. (2014), there is unlikely to be any risk of mortality for any fish species, eggs, or larvae due to noise from fishing vessels in the Jabuka/Pomo Pit area.** The guidelines also consider there to be a low risk for nonpressure sensitive fish of recoverable barotrauma injury at any distance and a moderate risk of TTS at close range to the source only. These relative risk scores may be reasonable considering the limited coalescing of sound fields at high levels, even when multiple vessels are closely located. **There may be a greater risk of masking and behavioral effects than stated by Popper et al. (2014), considering the large predicted ensonified region caused by multiple vessels, though this does not consider the relative levels caused by the fishing vessels compared to the broader soundscape of the Adriatic Sea** (see section "Modelled Levels in Context"). Caution should be emphasized when comparing to the stated values of relative risk as these are highly subjective and make no reference to source or received levels.

## Modeled Levels in Context

**The modeled levels consider only the noise generated from fishing activities and do not account for the existing ambient noise in the area or other transient sources of noise.** The ambient sound levels that define the ocean soundscape are composed of many natural and anthropogenic sources. The most important sources of environmental noise are wind, precipitation, and sea ice. Wind-generated noise in the ocean has been well documented (e.g., Wenz 1962; Ross 1976), and surf sound is known to be an important component of the coastal noise soundscape (Deane 2000). Precipitation is a common sound source, and its contribution is usually concentrated at frequencies above 500 Hz. Below 100 Hz, earthquakes and other geological phenomena contribute to the soundscape.

Anthropogenic (human-generated) can be a by-product of ship operations, or produced by active acoustic data collection (e.g., seismic surveys, military sonar, and depth sounding). The presence of oil and gas exploration by air guns, offshore piling, and oil and gas production platforms can increase sound levels at distances of 10–1000 km (Bailey et al. 2010; Miksis-Olds and Nichols 2016; Delarue et al. 2018).

**The Adriatic Sea is a busy area for commercial shipping, and small vessels such as fishing vessels do not typically generate a large noise footprint compared to large ships with high source levels.** A modeling study investigating the

change in the soundscape of the northern Adriatic due to the COVID-19 pandemic by **Sertlek (2021) concluded that, while the largest proportion of vessels in the area comprised fishing vessels, the contribution to the overall sound energy density was considerably smaller than that from cargo vessels and tankers.** The Adriatic is also an established area for oil and gas exploration and exploitation (AZU 2021), and the effect of impulsive noise from seismic sources on fish, marine mammals, and (to a lesser extent) marine invertebrates is well studied and documented with established thresholds for injury (Payne et al. 2008; Popper et al. 2014; Day et al. 2016; NMFS 2018; Day et al. 2019; Southall et al. 2019). It is recommended that any further investigation into the effect of noise on sensitive receptors in the area should consider these additional sources.

## Future Work

A knowledge gap currently exists in the characterization of the underwater-radiated sound from fishing vessels. In particular, while a few studies determined the source levels for transiting of these vessels, no dedicated study has looked at the different noise-generating sources and contextualized the sound emissions during the regular operations of these vessels. While vessels transiting at high speed from their base port to a fishing area may be the operation that results in the highest instantaneous sound emissions, the fishing operation should be assessed as it takes up most of the time, with the potential for sound to accumulate over several hours, and it involves not only the movement of the vessel using its engines but actively using tools and equipment onboard that may have implications on the signature of the emitted acoustic signals. For instance, a long liner that is carrying heavy nets that are scraping the seafloor will use more power than an equivalent vessel using a different fishing method that is likely to lead to higher URN. Furthermore, the nets being dragged are likely to generate additional noise that has yet to be described and characterized.

Future work should aim at undertaking detailed measurements of various fishing vessels according to international measurement standards while transiting but also while using the dedicated gear and at the typical fishing speed.

## Regulatory Implications

This chapter indicates that the modeled levels of generated noise from vessels undertaking demersal fishing in the study area do not exceed the auditory injury thresholds for marine fauna. However, the study did not account for other sources of noise that may be present in the area and likely to substantially contribute to the soundscape. Underwater sound from commercial vessels (e.g., cargo ships, tankers) and seismic exploration activity, for example, should be considered, in combination with fishing, in any further investigation into the effect of noise on sensitive

receptors in the area. This would be necessary to decide what regulation is needed and in which sector.

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# Offshore Wind Farms as an Anthropogenic Noise Source

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Michael Stocker

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**Abstract**

The pivot from offshore oil to offshore wind has ushered in the advancement and development of offshore wind farms. While wind energy generation is an environmental improvement with regards to decreasing atmospheric carbon associated with fossil fuel energy generation, it does not come without environmental costs. Siting, construction, installation, operation, and maintenance are all accompanied by noise, including benthic substrate vibration, mechanical impulse propagation of pile driving, and other installation activities, airborne and underwater sound transmission of operational noise. Operational noises include transmission of gearbox noise (on legacy turbines) and propeller noises down the masts into the water and benthic substrate, but also infrasonic noise from propeller and mast interactions creating torsional and mast resonances, and microbarometric noise propagated into the barometric "soundscape." These noises will affect the soundscapes of large areas of our Outer Continental Shelf habitats, both above and below the waterline, and into the benthos with known and unknown impacts on marine and avian biota.

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## Introduction

As the world shifts our energy strategies toward decarbonization, offshore wind may provide one of the more dependable sources of energy conversion due to the near-constant barometric gradient between land and sea. But converting wind energy into electrical energy is not without environmental costs (Bailey et al. 2014). As with any large-scale industrialization efforts, building wind power to scale will rapidly transform large areas of marine habitat that has been open water for millions of years, providing evolutionary context to the current marine biota.

Many of the transformations will be structural – with the introduction of vertical and horizontal structures into what is now open ocean. These structures will serve as new habitat and environment for marine life (Bolser et al. 2020) with unpredictable outcomes (Page et al. 2006).

In addition to the structural and mechanical transformations, the entire process of siting, surveying, locating, constructing, operating, and servicing offshore wind turbines is accompanied by acoustical, barometric, and vibrational artifacts, illustrated herein.

## Site Selection

Wind patterns and variability have been mapped since the beginning of maritime navigation, so areas with the highest density of reliable wind energy are well known (Pike 2018). Intersecting these areas with shore-side needs and distribution infrastructure establishes wind farm site suitability.

Once a site is selected, it needs to be surveyed to determine the most suitable mechanics of the farm in terms of how the turbine mast will be fixed or anchored to the sea floor. Gravity bases, monopiles, "jacket bases," and tripods are all options for shallow water (up to ~60 m in depth). Deeper locations require floating structures tethered to the sea floor (Bhattacharya 2019). In all cases, the sea floor and subsea bottom characteristics need to be determined – predominantly through acoustical probing.

Multibeam and side-scan sonars scan swaths of the sea floor, profiling the topography of the benthic habitat. Frequency selection is correlated to the desired resolution, but ranges from 2 kHz to 3.5 MHz (Smith and Rumohr 2008).

Pingers and Chirpers: The Pinger-type subbottom profiler transmits single-frequency "pings" every few seconds (10–50 m, with vertical resolution of 0.2 m). Chirper systems sweep, giving a more detailed resolution (<100 m, vertical resolution ~0.05 m). Both systems operate typically between 2 and 100 kHz and are generally used to detect substrata geological horizons in upper to mid-range strata depths (Geoscience Australia).

Sparkers and boomers emit seismic pulses reflecting off the subsea bottom geological structure that have bearing on long-term site stability, and substrate variations that have bearing on anchoring and pile driving strategies. Impulse acoustical energy is predominantly <2 kHz. These systems operate with an energy

source of between 50 and 300 J with typical penetration depths of up to 50 m when penetrating fine-grain sediments (e.g. sand) and a penetration depth of up to 25 m when penetrating coarse grain sediments (e.g., gravel).

## Construction

Shallow water turbine base construction – with the exception of gravity bases, and "suction bucket" caissons, require pile driving – using hydraulic hammers on the tops of 1 m to greater than 12 m diameter hollow steel piles. Some larger hammers can deliver up to 4.4 MJ, either as single strikes or as impulse vibration. Noise propagates away from the walls of the pile into the surrounding water, and vibration propagates through the sea floor impacting benthic biota and transmitting noise into the surrounding sea (Tsouvalas 2020).

Common noise mitigation strategies include bubble curtains (which are not without their own noise) that create an acoustical impedance variable in the propagation path of the noise source, and resonator systems, which convert acoustical energy into mechanical energy. Putting this into perspective, 30 wind farm projects are proposed on the Atlantic Outer Continental Shelf – installing hundreds of turbines, thousands of piles, and millions of pile strikes. Regardless of base type, all turbine bases require high concentrations of construction vessels and construction/deployment noise during installation.

## Operation

Once put in service, all turbines introduce a suite of chronic noises into the environment: gearbox noise from some turbines, and torsional and mast resonances transmitting down the mast and into the water (Edwards 2015). (Larger turbines are migrating away from gear box-driven generators.) Turbulence from the blades includes rotating noise of the collapsing tip vortices and blade-mast transect noise, which is largely infrasonic (Edwards 2015), and thus less attenuated by the air/water interface, and likely to propagate far into the surrounding aquatic environment. Permeability of the air/water boundary is correlated with wavelength such that:

$$TL_{\mathrm{dB}} = 20 \log_{10} (k_a h)$$

Transmission loss ($TL$) equals "$k_a$" which is the acoustic wave number, and "$h$" is the height of the sound source. This illustrates how the air/water boundary becomes much more permeable at lower frequencies. At 0.1 Hz, the transmission loss is theoretically −24 dB (nonexistent). Selecting "$h$" for 220 m as the hub height is arbitrary given the wavelengths of concern.

This infrasonic noise also propagates into the above-water environment, interfering with barometric cues that some birds use to navigate (Kreithen and Keeton 1974). This may account for avoidance of wind turbines by procellariiformes

(Dierschke et al. 2016), Anseriformes (Desholm and Kahlert 2005), and Gannets (Peschko et al. 2021).

The evidentiary question remains as to whether mysticetes use low-frequency acoustical energy for navigation cues as well – given that they navigate across vast distances. As underwater visual input is limited by turbidity and marine light conditions, low-frequency acoustical energy remains the best candidate for mysticete navigation cues (Clark and Gagnon 2022). If this is the case, infrasonic noise generated by fields of large wind turbines may interfere with mysticete migration.

## Service and Maintenance

Once a wind farm is in operation, there will be a continuous fleet of maintenance vessels over the course of operation and through decommissioning – tending to the bases, masts, spars, turbines, infrastructure, and instruments deployed around the wind farm structures with their inherent propeller and equipment noises.

## Conclusion

We are advising thorough acoustical baselining before commencing wind-farm development, and acoustical soundscape monitoring throughout the installation and operation – with hopes that we will learn something from the acoustical changes that the energy industry will have on the ocean and its inhabitants. These observations should also be biologically tracked in terms of baselining and monitoring species and population densities of marine organisms.

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# Reporting Impulsive Noise from Underwater Explosions Using Seismic Data

125

Mirko Mustonen and Aleksander Klauson

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**Abstract**

The clearance of mines and other unexploded ordnance is known to pose the utmost danger to marine life, causing injuries and even mortality (Lewis, Effects of underwater explosions on life in the sea. Defence Science and Technology Organization, Canberra, 1996). The EU Marine Strategy Framework Directive requires the spatial distribution, temporal extent of noise levels produced by the explosions do not exceed levels that adversely affect populations of marine animals. For achieving this aim, the data about the explosions have to be collected and reported to the impulsive noise registries, as is the ICES impulsive noise register (ICES, Impulsive noise events registry in support of OSPAR and HELCOM, ICES, 2022). Reporting impulsive noise data can be quite challenging, as data collection methods vary by country and are subject to local legislative restrictions (Merchant et al., Mar Pollut Bull 152:110951, 2020). One way to facilitate reporting is to use publicly available seismic data. This study aims to analyze the feasibility of using national seismic data for this task. The correspondence between the national seismic data and a dataset provided by the authority responsible for the mine clearance operations is studied. Case studies of individual explosions that have been recorded during passive acoustic monitoring are additionally used for further analysis.

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**Keywords**

Explosions · Impulsive noise · Seismic data · Noise monitoring · Reporting explosions

## Introduction

Among the various man-made underwater impulsive sound sources, explosions produce the highest peak level sound that may cause injuries or death to the animals in the vicinity (Lewis 1996; Koschinski 2011). In some cases, the aim for creating the explosion is to remove material on the seabed during offshore construction. More often, explosions are created upon disposal of unexploded ordnances (UXOs). For example, during the two world wars ~175,000 naval mines were laid in the Baltic Sea, of which 10–30% are estimated to remain (Möller 2011). Besides mines, other UXOs like torpedoes, anti-submarine rockets, aircraft bombs, and depth charges are still discovered each year and disposed of.

Due to the potential environmental impact, data about the explosions in the European seas have to be collected. This data has to be reported to the impulsive noise registries, as is the ICES hosted impulsive noise register. Among the data reported, the value code indicating the radiated sound intensity of the explosion is the most difficult to determine. Although before disposal the UXO is usually identified, the explosion's TNT equivalent weight of the energy released by the actual explosion is not usually estimated in situ. Therefore, we investigated the viability of using other available sources for acquiring this information.

The earthquake activity in Estonia and surrounding areas is known to be relatively modest (Grünthal et al. 2009). Regular measurement by seismometers has been ongoing here since 1987 and has been part of the national environmental survey since 2001. The aim for collecting the seismic data has been to register and analyze both the man-made and the natural seismic events in the Estonian and neighboring territories (KESE 2023). The seismic events data used in this study were from the processed recordings by three permanent and eight temporary seismometer stations with additions from the neighboring stations from Finland, Latvia, and Sweden (Soosalu 2022). This data contains time stamps of the events, location coordinates, location error estimates, type of seismic event, and their magnitude estimates. The magnitude listed is the Helsinki University local magnitude $M_{HEL}$ (Uski and Tuppurainen 1996). Of the recorded and located events, the overwhelming majority were man-made and identified principally as signals from routine blasting operations in local mines.

We were interested in the man-made seismic events within this registry that were located in the Baltic Sea. Also for reporting to the ICES registry we considered the events (1) listed as explosions and (2) located within the Estonian EEZ. All the 229 seismic events in the registry from 2017 until 2021 that fit these criteria are shown on a map in Fig. 1 with blue plusses and red crosses.

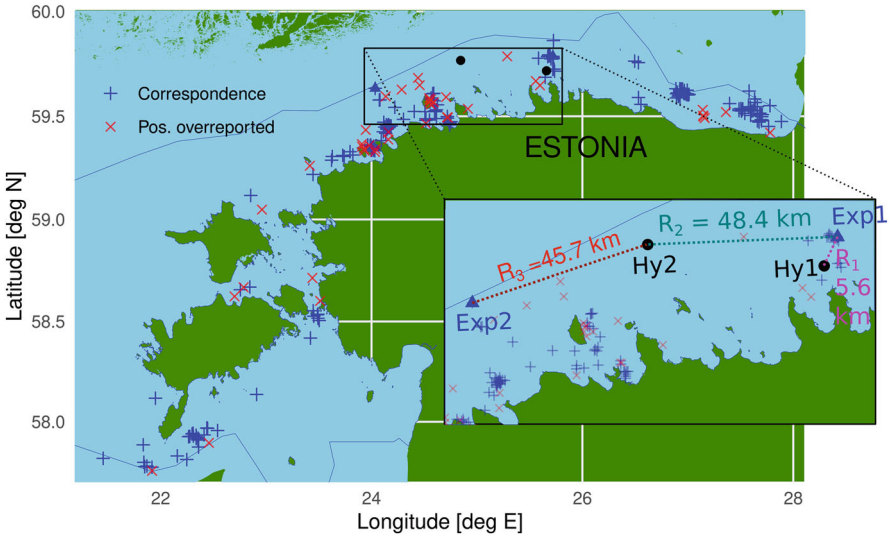



**Fig. 1** Locations of 229 explosions from the national seismic events data from 2017 to 2021, located in the Estonian EEZ. Blue plusses show locations of all the seismic events with corresponding records in the UXO disposal data. The red crosses mark the locations of the seismic events that have no correspondence (possibly overreported). The inset shows the case study locations for the three recordings of two explosions Exp1 and Exp2 (blue triangles) with hydrophones Hy1 and Hy2 (black circles) and ranges $R$ between the explosions and hydrophones

## The Correspondence

The explosions in the Estonian EEZ from the national seismic data were checked against the data acquired from the Estonian Navy about their UXO disposals for correspondence. Among other information, the disposal data listed the geographical coordinates, date of disposal, and type of the UXO. Most of the UXOs were naval mines of various types along with some anti-submarine rockets, torpedoes, depth charges, and aircraft bombs.

A record in the seismic data was considered to correspond to the data in UXO disposal data when there exists at least one record for which two following criteria are satisfied simultaneously:

- Difference in time is less than 2 days: $t_d < 2$ days
- Geographical distance between the points is less than 10 km: $d_g < 10$ km

These criteria were selected to account for the human errors that might occur in manually writing down dates and coordinates during disposal. Altogether, 192 explosions from the seismic data met these criteria and their locations are shown in Fig. 1 with blue pluses. In total, 30 seismic events did not fit the criteria $t_d < 2$ days and

$d_g < 10$ km. As previously only the seismic data has been used for reporting to the ICES registry, these can be considered possibly overreported and are depicted with red crosses in Fig. 1. The UXO disposal data also contained 28 records of explosions with no counterpart in the seismic data that might be considered as possibly underreported. Although there is also a possibility that in these cases the intensity of the blast was so low that they would not have to be reported to the registry. In the end, the overall number of the reported explosions coincided with the number of listed UXO disposals.

## TNT Charge and Magnitude Relation

For assessing the risk proposed by the explosions, their intensity has to be known. The seismic data gives the local seismic magnitude $M_{HEL}$ for every listed event. For reporting explosions, the ICES hosted impulsive noise events registry provides a vocabulary that divides their intensity according to TNT-equivalent charge weights in kilograms. Although some tables for converting between local seismic magnitude and charge weight exist (Mymathtables 2022), the corresponding data from the UXO disposal and seismic data sets enable to give a very rough estimate for their relationship.

The TNT-equivalent charge weights in kilograms of each disposed UXO was considered to be its historic TNT weight listed in online (NavWeaps 2023) or other sources (Коршунов and Дьяконов 1995; A Bureau of Ordnance Publication 1946). The resulting estimate is shown in Fig. 2. Due to uncertainties in the actual amount of explosives within sometimes over 100-year-old ordnance the error estimated for

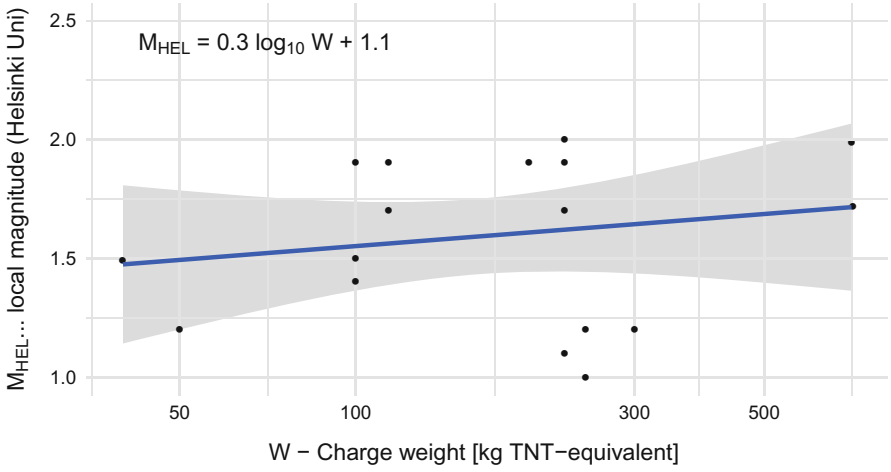



**Fig. 2** Estimate for a relationship between charge weight W extracted through UXO disposal data by finding the UXO's historic weight and its corresponding seismic magnitude $M_{HEL}$ from the national seismic data

the slope and the intercept are quite large. Nevertheless, the result shown in Fig. 2 coincides with a recent result from a controlled experiment (Favretto-Cristini et al. 2022).

## Case Studies

Some of the explosions took place in locations where their radiated sound could also be recorded by long-term monitoring hydrophones. Two of the hydrophones were the SM2Ms by Wildlife Acoustics and one the ST500 by the Ocean Instruments. Here we look at recordings of two explosions that were created when disposing of the same type of naval mines. One explosion was recorded by a single hydrophone and the other by two hydrophones at different ranges. The hydrophone locations Hy1 and Hy2 in relation to the explosions Exp1 and Exp2 are shown in the inset of Fig. 1. The time series of the calculated sound pressure from the recorded received levels are shown in Fig. 3. The lower two time series in Fig. 3 correspond to the disposal of naval mines of the same type with same listed charge weight recorded at relatively similar distances. It can be seen that the recorded peak sound pressures of these two

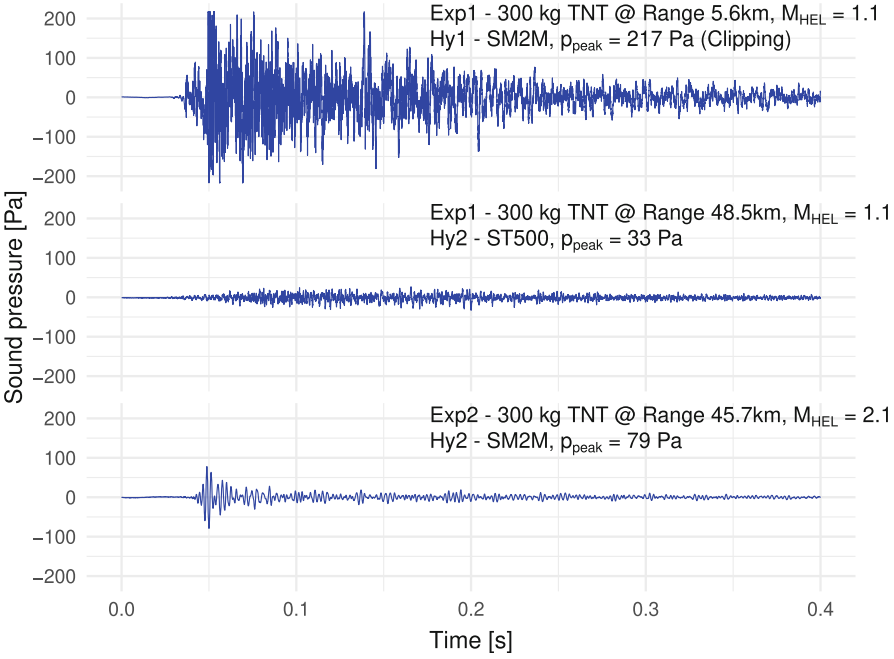



**Fig. 3** Time series of sound pressures calculated from recordings of two separate explosions of same type of mine with TNT equivalent charge weight of 300 kg. One of the explosions denoted Exp1 was recorded by two hydrophones at ranges 5.6 km and 48.5 km. The other explosion Exp2 was recorded with a single hydrophone at range 45.7 km. The seismic magnitude created by the first explosion was 1.1 and second explosion had a listed magnitude of 2.1

explosions $p_{peak}$ were significantly different. Simultaneously, the listed seismic magnitude was also higher for the explosion with higher $p_{peak}$. This indicates that when reporting the explosions to the ICES registry, a good estimate about the actual intensity of the explosion could be acquired from the seismometers' data.

## Conclusions

- In case data from the authority performing the UXO disposals is not available, the seismic data is indeed an invaluable resource for estimating the pressure that underwater explosions pose for the marine animals as the correspondence between the seismic data and the UXO disposals were reasonably satisfactory.
- If records about UXO disposals exist, the seismic magnitude is extremely helpful for determining the actual intensity of the explosion by using the magnitude-charge relation from (Favretto-Cristini et al. 2022).

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# Assessing Risk of Noise Pressure on Marine Life Using Bayes Estimator

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Sérgio M. Jesus, Ricardo Duarte, Giulia Spadoni, Cristiano Soares, and Marc Fernandez

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**Abstract**

The level of noise in the ocean has been increasing in the last decades, putting at risk a wide variety of marine species that rely on sound for their daily life. An important tool for assessing, and eventually mitigating, the potentially harmful effects of ocean noise on marine species is the so-called risk map. Noise risk maps result from the combination of noise pressure-level distribution and species density in the same time-space framework. A known drawback of the existing risk map methodologies is that they do not allow for direct comparison of the

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degree of risk between different periods of time, or locations, or between different species. The present study proposes a Bayesian inference-based technique, as an alternative for determining risk maps that return comparable and quantifiable absolute quantities. A simulation of a shallow water seismic survey along the southwest coast of Portugal is used to illustrate the proposed methodology. The test case considered two periods of the year (winter and summer), using as an example the common dolphin (*Delphinus delphis*) species. The results show that risk maps obtained with the proposed method favorably compare with those obtained with existing methods with, however, the advantage of being based on mean absolute values. These results encourage its use in future studies, targeting different species and/or different areas in order to give some hints for the production of indicators to support ocean protection policies.

**Keywords**

Ocean noise · Risk maps · Bayes estimation · Common dolphin

## Introduction

Ocean noise produced by human activities has been continuously increasing in the past decades due to globalization that uses the ocean as the main route for shipping goods around the world (Redfern et al. 2017; Frisk 2012). This increase of ship traffic is accompanied by the corresponding increase of the global ocean noise level with direct implications in many marine organisms, in particular on cetaceans, as they depend on sound to forage, to interact in their community, to orientate, and to perceive the surrounding environment (Nowacek et al. 2007, 2016).

The potential impact of noise generated by human activities in the ocean drove scientists, governments, and policymakers to design policies, to set thresholds, and to monitor the good environmental status including ocean noise (DIRECTIVE 2008/56/EC 2008). Modeling is a possible way to assess the state of the soundscape that results in spatial- and temporal-level distribution of sound/noise, the so-called sound/noise maps (Soares et al. 2020; Neenan et al. 2016; Maglio et al. 2015; Nicolas et al. 2016). However, sound maps are of little use for policymakers and governments since they do not take into account sensitive species distribution. In fact, policymakers and governments are more interested in the estimation of the level of risk to which a specific species is exposed, as caused by a specific noise level, in a specific area and period of time.

Noise risk maps combine, for the same period of time and area, predicted noise maps with biological distributions. Consequently, noise risk maps may be used to implement and/or support some of the decisions at an institutional level regarding the protection of most exposed areas.

In this specific context, risk can be defined, as proposed by Verling et al. (2021), as the possibility of loss or injury inflicted by ocean noise in marine species described as

$$Risk = Likelihood \times Consequence \quad (1)$$

where "Likelihood" defines the points in space and time where a given sound pressure "overlaps" with the distribution/habitat of a particular species and the "Consequence" defines what exactly might happen in case of some likelihood. Assessing the "Consequence" is challenging and entails several particularities related to biological and physiological aspects of the species. For this chapter, it will be considered that Risk = Likelihood.

In line with this definition, several methods to produce risk maps have been proposed, combining noise prediction maps and species maps (Verling et al. 2021; Erbe et al. 2014; Merchant et al. 2018).

Even though species distribution and noise maps modeling are relatively well accepted and defined, the combination of the two is quite debatable. For example, Erbe et al. (2014) used an example with the Dall's porpoise, where risk maps are obtained through multiplying species density maps by noise maps both provided in normalized scales and re-normalizing the final result. Merchant et al. (2018) follow the same methodology, with slight differences. Nevertheless, a major limitation of these methods is the fact that they preclude comparisons in time, in space, or between different species due to the normalization, so the resulting risk map becomes specific only for the considered area, time period, and species.

The present study proposes an alternative to produce risk maps based on the Bayes estimator which will make it possible to quantify risk areas while overcoming the drawbacks of current methodologies. It is believed that the alternative methodology presented in this chapter will help have a wider perception of the risk resulting from anthropogenic sources of noise and consequently promote the adoption of balanced regulations and legislation to protect marine species.

To illustrate the proposed methodology, a typical test case scenario of a light seismic survey off Setúbal on the southwest coast of Portugal was selected. This test case scenario was focused on two periods of the year, January and June, as representatives of winter and summer seasons. The selected geographic area is known to have a rich biodiversity and to host different species of cetaceans as, for example, the common dolphin that may be consequently affected by such impacting surveys.

This chapter is organized as follows: section "Materials and Methods" describes the materials and methods defining the target area, the target species, the noise source, the modeling inputs (noise and habitat suitability (HS) maps), and the Bayes estimator proposed method. Section "Results and Discussion" presents the results obtained with the proposed method and discusses its comparison with currently established methods. Finally, section "Conclusions" draws conclusions pointing out the main advantages and drawbacks of the proposed method and anticipates future work.

## Materials and Methods

Producing risk maps passes through several well-described stages until obtaining the risk map itself. These steps are summarized in the diagram of Fig. 1. Usually, the process starts by defining the target area and then separating it in two parallel

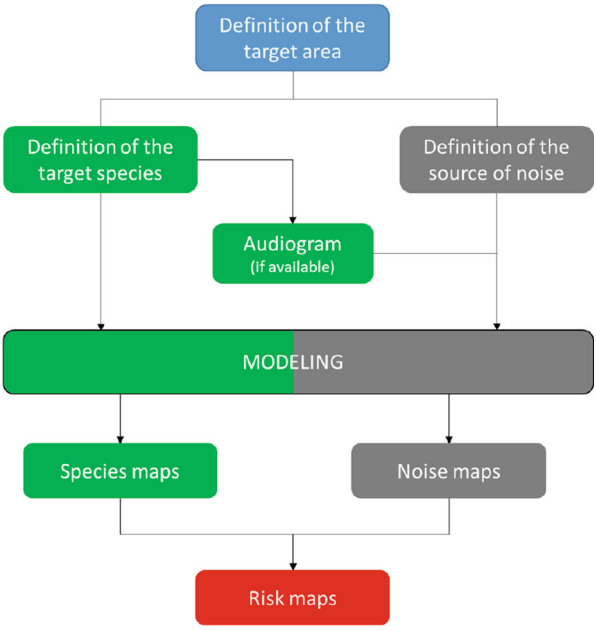



**Fig. 1** Generic noise risk map methodology flowchart

branches: the branch on the left (in green) leading to the habitat suitability (HS) map and the branch on the right (in gray) leading to the noise map. The only link between the two branches is the audiogram that is applied to the perceived noise field and depends on the selected species. Modeling is obviously different for the two branches. On the "biological branch," modelling the temporal and spatial distribution of the species may follow two approaches: through ecological niche models or through species density models depending on the available data (Fernandez et al. 2021). On the "noise branch," modeling the temporal and spatial distribution of noise is usually performed through the combination of several datasets such as bathymetry, sound speed profiles (SSP), Automatic Identification System (AIS) data, and bottom properties that are fed into acoustic propagation models for sound pressure-level (SPL) estimation (Soares et al. 2020; Neenan et al. 2016; Maglio et al. 2015; Nicolas et al. 2016). The last step consists on the combination of the HS map and noise maps to produce the risk map that will be, of course, conditioned on the specific area, the target species, and the source of noise.

To illustrate the proposed methodology, a hypothetical light seismic survey scenario taking place in the southwest coast of Portugal was designed. Shallow water (also called light) seismic surveys aim at superficial bottom sediment structure identification and are typically used in the initial area assessment for wind farms, bridges, breakwaters, wave energy generators, or cabling infrastructure development. This technique involves lower noise levels than classical deep water oil and gas seismic surveys and covers shallower areas when the required bottom

penetration is on the order of only a few tens of meters which consequently involve higher-resolution and relatively lower-energy sources.

The choice of this area was based on the fact that the Portuguese southwest coast is well known for its rich biodiversity and because a few years ago the Portuguese Government had provided concessions for oil and gas exploration in that area (Offshore Magazzine 2007; Almeida and Khalip 2018; Tavares Pires et al. 2019; Castro-Santos et al. 2020; Silva et al. 2018; Salvação and Soares 2017).

Thus, the Portuguese coast is an important spot for marine biodiversity, hosting a large number of fish species, invertebrates, sea turtles, and a variety of cetaceans that visit this coast throughout the year. Species of whales and dolphins are frequently sighted in the central part of the country, in the waters near the region of Setúbal (Correia et al. 2015; Castro 2010). Among them, the common dolphin (*Delphinus delphis*) is notably abundant, and it may, therefore, be at risk due to the plans for extensive offshore seismic surveying.

## Target Area

The area of interest was limited to −9.3° and −8.5° longitude west and 38.2° and 38.6° latitude north, with a spatial resolution of 1 × 1 km, as shown in Fig. 2. The bathymetry shows a slowly varying water depth over the continental platform extending up to approximately 45 km from the coastline, where it reaches the border of the continental shelf at approximately 250 m and then rapidly increases to deeper waters to the west. An interesting feature of this region is the Setúbal canyon, an

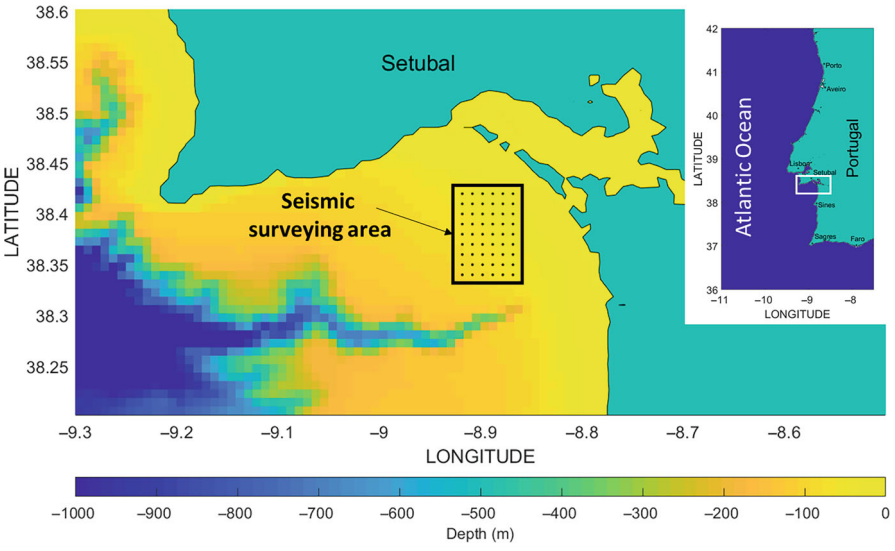



**Fig. 2** Bathymetry of the test case geographical region and the seismic surveying area (black-dotted rectangle)

east-west oriented steep-sided valley at approximately 38.3° latitude north, that entails the platform with depths reaching 1000 m. For simulation purpose, the seismic survey event was centered on a relatively small rectangular area of 50 $km^2$ delimited by the following coordinates, longitude −8.92° and −8.87° and latitude 38.34° and 38.42°, shown as a black-dotted rectangle in Fig. 2. The water depth of the surveying area varies from 30 to 150 m depth which is appropriate for the installation of offshore wind farms (Oliveira et al. 2021).

## Target Species

The selected species for this study (common dolphin) is one of the most abundant species in this region, being in fact one of the species with the highest number of records in dedicated databases. According to the Sociedade Portuguesa para o Estudo das Aves (SPEA), there are more than 3000 records of sightings for this species in 2019. The other reason that drove the selection of this species was the fact that common dolphins may be included in both the small cetacean group and high-frequency cetaceans group which allow future extrapolation regarding the effect of noise in different cetacean groups and gives more relevance for the study (Southall et al. 2019).

## Noise Source

A commonly used apparatus for light seismic surveys is the "sparker." These seismic sources are specially designed for shallow water usage that require less energy and higher bottom resolution. Sparkers operate by sudden discharge (spark) of a high-voltage electrical current between submerged arrays of electrodes. Modern sparker systems use several electrodes, which produce a seismic energy pulse typically between 300 and 20,000 Joules (Trabant 1995). A Geo-Source 200 sparker with two arrays of 100 electrode tips each was modeled as an example. (Developed and commercialized by GEO Marine Survey Systems, the Netherlands.) This system may reach a peak source level of 223 dB at 1 m depth, suitable for water depths from 2 to 500 m.

For the simulation purpose, the sparker source was placed at 1 m depth with a 5 s firing interval considering a ship moving at 5 knot in a traditional lawn mower pattern, which gives rise to a regular set of emission positions along the area (Fig. 2). A total duration of 1 month was considered with a time resolution of 10 min for the estimated SPL field.

## Modeling

As shown in the flowchart of Fig. 1, risk maps require two inputs: species maps and noise maps. Considering that the focus of the present work is to present an alternative methodology for producing risk maps, the modeling aspects will be omitted, and the interested reader is referred to Spadoni et al. (▶ Chap. 127, "Seismic Survey Risk Assessment on Common Dolphins in the South-western Coast of Portugal") for details.

## Species Maps: Common Dolphin Habitat Suitability Maps

Habitat suitability (HS) maps for the *Delphinus delphis* (i.e., common dolphin) population were developed by Spadoni et al. (▶ Chap. 127, "Seismic Survey Risk Assessment on Common Dolphins in the South-western Coast of Portugal") for the months of January and June as representatives of winter and summer seasons (Fig. 3).

HS maps present a higher quality of the habitat in coastal areas, until the bathymetric line of 200 m, especially during the month of June. This observation was expected due to the better feeding conditions in summer than in winter which creates a more suitable habitat for the species. Note that Cabo Espichel and Comporta represent high-quality areas for the common dolphin species in both cases.

## Noise Maps: Seismic Survey

Sound pressure level (SPL) is a representative metric for the noise produced by a seismic source. The propagation of noise resulting from seismic surveys is shown in Fig. 4 (see ▶ Chap. 127, "Seismic Survey Risk Assessment on Common Dolphins in

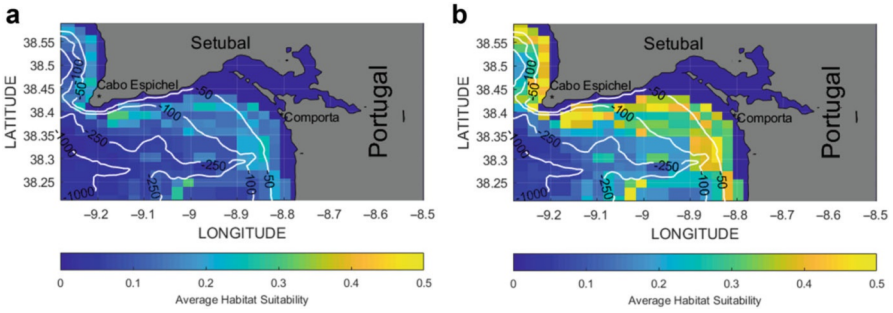



**Fig. 3** Common dolphin habitat suitability for the months of January (**a**) and June (**b**). HS maps were extracted from Spadoni et al. (▶ Chap. 127, "Seismic Survey Risk Assessment on Common Dolphins in the South-western Coast of Portugal")

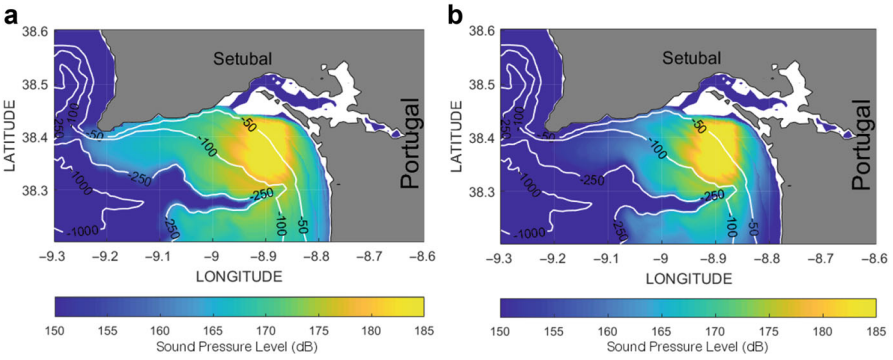



**Fig. 4** Sound pressure-level distribution for the months of January (**a**) and June (**b**). SPL maps were extracted from Spadoni et al. (▶ Chap. 127, "Seismic Survey Risk Assessment on Common Dolphins in the South-western Coast of Portugal")

the South-western Coast of Portugal"). An important influence of the environmental characteristics of the area can be noticed, as, for example, that of the bathymetry, when sound travels to the platform border and of the water column temperature variation, registering higher levels in January than in June. Significant sound pressure levels may be attained in large swaths of the considered area. A larger spread is particularly clear in January. The survey area registered SPL levels in excess of 185 dB re 1$\mu$Pa, for both months. In both cases, it is observed that noise levels may reach exceptionally high values which according to Spadoni et al. (▶ Chap. 127, "Seismic Survey Risk Assessment on Common Dolphins in the South-western Coast of Portugal") may have an impact on the temporary and permanent threshold shifts of species' hearing perception, in a range of about 40 km around the surveying zone.

## Risk Assessment Using the Bayes Estimator

The proposed approach to produce risk maps and estimate risk areas is based on Bayesian inference considering that biological and anthropogenic variables are somehow correlated, if the same location and time period are considered.

To facilitate the explanation throughout this section, one may consider the common dolphin HS as a random variable $A$ and anthropogenic noise denoted by random variable $B$ with an empirical probability density functions (pdf) $p(a)$ and $p(b)$, respectively. Then, if variable $A$ is represented by a prior distribution of the HS ($p(a)$), then the conditional density of $b|a$ (spelled as $b$ when $a$) will be given by the distribution of noise. It follows that the posterior distribution of variable $A$, after noise observation, denoted as the random event $a|b$, will be given by the classical Bayes theorem (2):

$$p(a|b) = \frac{p(b|a)p(a)}{p(b)}, \tag{2}$$

where the denominator is the normalizing factor $p(b) = \int p(b|a)p(a)da$. This normalizing factor fulfils the crucial role of a proper probability density normalization.

Estimation theory indicates that an estimator that minimizes the Bayesian mean square error is given by the conditional mean of the posterior distribution (after observation) or the mathematical expectation $E[A|B]$ written as:

$$\widehat{\mathrm{A}} = E[A|B] = \int ap(a|b)da. \tag{3}$$

Although in many cases the species distribution/habitat suitability is represented by an index which may be assimilated to a probability, in practice, sample sizes may be insufficient (or not available at all) which may prevent us from obtaining a truthful prior pdf estimate $p(a)$ making the usage of (2) and thus of estimator (3) impossible. This limitation may be overcome if one assumes that variables $A$ and $B$ follow

Gaussian distributions which, in fact, are quite acceptable in many applications involving real data (Fisher 2021). If the prior and the observation densities are both Gaussian, then the posterior is also Gaussian, which simply states the fact that the product of two Gaussian functions is also a Gaussian function. So, under the Gaussian assumption (and only under that assumption) if $A : \mathcal{N}\left(m_A, \sigma_A^2\right)$ and $B|A : \mathcal{N}\left(m_{B|A}, \sigma_{B|A}^2\right)$, the posterior density $p(a|b)$ of a random variable $A|B$ is still $\mathcal{N}\left(m_{A|B}, \sigma_{A|B}^2\right)$ in which mean and variance are given by:

$$m_{A|B} = \frac{m_A \sigma_{B|A}^2 + m_{B|A} \sigma_A^2}{\sigma_A^2 + \sigma_{B|A}^2}, \tag{4}$$

and

$$\sigma_{A|B}^2 = \sqrt{\frac{\sigma_A^2 \sigma_{B|A}^2}{\sigma_A^2 + \sigma_{B|A}^2}}, \tag{5}$$

respectively. Therefore, if the prior densities on the right-hand side of (2) are approximately Gaussian and their means and variances are known or can be estimated, the maximum a posterior Bayes estimator (3) can be calculated using expression (4). The variance of the resulting distribution is given by (5), which is not the variance of the estimator but is a measure of performance since it represents the Bayesian mean square error, i.e., the smaller the better.

## Results and Discussion

This section presents first the resulting risk maps obtained using the Bayes estimator, and second, it compares these results with those obtained using a classical method to estimate risk (Erbe et al. 2014). This comparison will be used to validate the results obtained and the assumptions made, for the proposed alternative noise risk map estimation method.

### Bayes Estimator Results

Figure 5 shows the risk distribution maps using the proposed Bayes-based mean estimator for the months of January (a) and June (c) and their variances (b) and (d), respectively. According to the mean estimator which, in fact, represents the simulated risk resulting from the exposure of the common dolphin population to a seismic survey, a higher sensitivity was observed in coastal areas, until the bathymetric line of approximately 200 m, specially near Comporta and Cabo Espichel. The higher- and lower-risk spatial distribution areas are similar in both months, albeit in June the

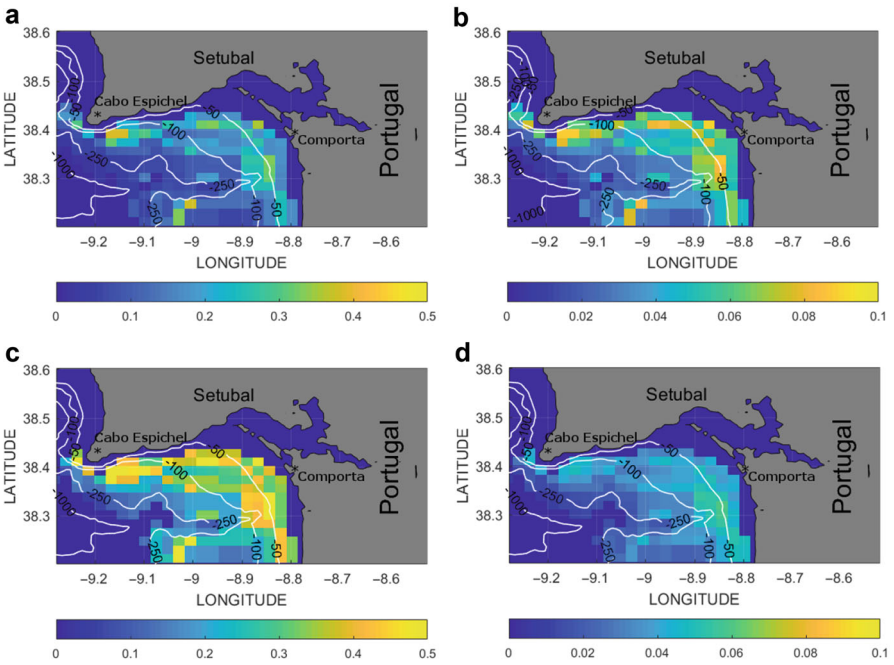



**Fig. 5** Risk maps estimated with the Bayes-based mean estimator for January (**a**) and June (**c**) and their respective variance (**b**) and (**d**), respectively (according to (4) for mean and (5) for variance)

risk levels are, overall, considerably higher than in January. The estimated risk levels show that a seismic survey in June is potentially more risky, for this species, than in January due to the higher species HS and also by the fact that the propagation is more condensed in the areas where the population's HS is higher. These results were expected since, first, the seismic survey was assigned to a coastal area which in fact is the most visited area for this particular species and, second, because the habitat suitability is higher during June than in January, as it was shown in Fig. 3.

A sub-product of the proposed method is that it gives additional information on the variance, providing an indication of the quality of the estimate, since it represents the Bayesian mean square error (Bmse). Smaller variance means a better estimate and a label of higher confidence. In our simulated example, it points out for a better estimate in June than in January, as shown in Fig. 5b, d, respectively.

## Comparison with a Classical Method

As stated in Erbe et al. (2014), normalizing the input maps (noise and species maps) makes it impossible for comparing the resulting risk maps. However, considering the same period of time, same location, and same species, it is possible to perform a spatial distribution comparison of the two methods which will allow to test the

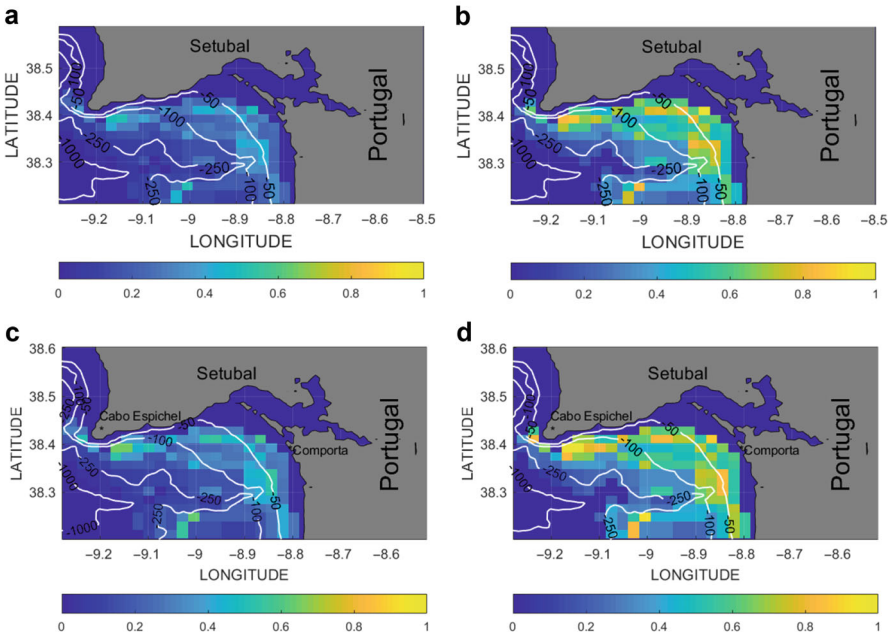



**Fig. 6** Comparison between the classical method of Erbe et al. (2014) (top) and Bayes estimator method (bottom) for January (left) and June (right)

validity of the method proposed in this study (and assert of the assumptions made). To facilitate this comparison, it was decided to set both maps color bars between 0 and 1. Figure 6 shows the risk maps produced using the classical method (top) for the months of January and June (right and left, respectively) and the risk maps produced with the Bayes estimator method for the same months (bottom). Noise risk maps estimated with the proposed method (Fig. 6c, d) look quite coherent with those obtained with the "classical method" (Fig. 6a, b). It can be observed that the spatial distribution of lows and highs follows the same pattern and the overall behavior between the 2 months is also similar. This sustains the validity of this alternative to produce risk maps.

## Conclusions

It is of major importance to assess the impact of anthropogenic noise in the ocean-living organisms, in order to develop effective protection measures. A way to predict the most impacted areas is through the production of the so-called noise risk maps, which combine the biological and anthropogenic distributions with a single time-space map. The current limitation associated with the production of noise risk maps is the fact that the existing methods preclude comparisons between different locations, at different periods of time or between different species. This limitation, at

some point, may prevent the development and the adoption of protection measures for specific areas.

The present work proposes an alternative method to produce/calculate noise risk maps which results in quantifiable maps that can be compared between them. To illustrate this methodology, a case study was designed modeling a seismic survey and its impact in the population of common dolphins in the southwest coast of Portugal. The test case results show that there is a higher seismic survey noise impact if the survey is to be carried in June than in January, specially near coastal areas. This suggests that there are periods of the year that are "better suited" for seismic surveying than others. Although this is only a simulated exercise, it shows that comparability not only in time but also in space and between species is determinant for a previous evaluation of environmental factors of the target area to support environmental management policies or to support political decisions. The contributions of the present work may be summarized as follows: (1) offers a sound theoretical framework to the ad hoc noise risk likelihood estimation used in the literature, allowing for a clear normalization of the result and its absolute comparability; (2) in case the statistic is insufficient for probability density estimation, the proposed method allows to still obtain a data coherent noise risk map estimate in case the statistics are Gaussian distributed, which is often the case whenever substantial real data is involved; and finally (3) the proposed Bayes-based methodology may be adapted to risk assessment maps, other than ocean noise.

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# Seismic Survey Risk Assessment on Common Dolphins in the South-western Coast of Portugal

127

Giulia Spadoni, Ricardo Duarte, Cristiano Soares, Marc Fernandez, and Sérgio M. Jesus

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**Abstract**

While ship traffic is the primary source of anthropogenic underwater noise worldwide, locally, other sources may have a higher impact on marine life, such as seismic surveys. Although the impacts of this practice conducted in deep waters are widely investigated, less is known about the risk to biodiversity when such surveys are undertaken in shallow waters using light seismic techniques, such as for offshore wind development. The objective of this study is to assess at-risk areas by combining habitat suitability maps for the common dolphin (*Delphinus delphis*), selected as the target species, and noise maps from a light seismic survey simulation in the region of Setúbal (Portugal), an area of high marine biodiversity. Noise maps indicate that sound levels may reach up to 190 dB re 1$\mu$Pa, impacting the species' hearing perception in a range of approximately 40 km around the surveying zone. Habitat suitability maps showed high values in low-depth areas of the abovementioned region. Risk maps, obtained by overlapping seismic survey noise and habitat suitability maps, showed the particular sensitivity of coastal areas, especially until the bathymetric line of 250 m and higher levels of risk in a broader area in summer than in winter. The relevance of risk maps as a powerful tool for supporting environmental and marine life management policies is emphasized.

**Keywords**

Light seismic surveys · Sound pressure level · Risk maps · Habitat suitability · Common dolphin · Portugal

## Introduction

The ruthless search for sustainable energy led us to explore the ocean and the wind as potential energy sources (Rocha 2016; Salvação and Soares 2017). Compared to conventional oil and gas, wave and wind energy is cleaner and greener, but it also has significant disadvantages. These consist of the impact of the necessary infrastructures during their installation and operation. Prior to installing infrastructure at sea, the bottom structure must be assessed for platform fixing or anchoring and cable lay down. This is routinely carried out using light seismic surveying, a low power/high resolution version of traditional oil and gas seismic survey (Duarte et al. 2021; Jesus and Miller 2009). Light seismic surveying relies on the emission of high or very high-energy impulsive sound waves that reflect in the bottom and are received on towed streamers of hydrophones. Its effects on marine species, at short term and for those areas where these activities take place, have already been studied (Hastie et al. 2019; Read 1995).

Portugal, as well as other countries with an important seafront, has made plans for building several offshore wind farms along the Portuguese coastline, as referred in

the Portugal's National Strategy for the Sea 2013–2020 (Salvação and Soares 2017; Castro-Santos et al. 2020; Silva et al. 2018; Oliveira et al. 2021).

Besides that, the Portuguese coast is of great relevance in terms of marine biodiversity, partly due to the highly dynamic and complex topographic-oceanographic features of the coast and to the natural upwelling, which favors a great biodiversity richness and a countless number of cetacean occurrences every year (Correia et al. 2015). In particular, the region between Sines and Setúbal, in the SW coast of Portugal, is known to host many different species of cetaceans (Correia et al. 2015; de Castro 2010).

The lack of knowledge on light seismic survey noise emissions and the potential threat it represents for cetaceans and biodiversity led us to design a simulated typical test case scenario for a seismic survey event taking place in the SW coast of Portugal, near the region of Setúbal. The common dolphin (*Delphinus delphis*) was selected as representative species, commonly present in the test area.

A prediction tool was developed to estimate: a) the sound pressure level (SPL) field on the target area due to a typical light seismic survey, b) the common dolphin habitat suitability (HS) through Ecological Niche Modeling (ENM), and c) the potential at-risk areas for this particular species due to a seismic event in two periods of the year (January and June). Habitat can be defined as "an area with a combination of resources and environmental conditions that promotes occupancy by a given species (or population) and allows those individuals to survive and reproduce" (Darracq and Tandy 2019). Suitability indicates the quality of these conditions and resources.

Results on noise modeling showed the importance of bathymetry on the attenuation of the noise field along the continental shelf, while attaining significant sound pressure levels in large swaths of the study area, that also have a high HS for the common dolphin. The deduced potential risk is consistent, in space and time, with the HS distribution and the propagation of noise from the simulated seismic survey. These results give hints to produce indicators and to support management, conservation, and biodiversity monitoring.

This chapter is structured as follows: section "Materials and Methods" gives a description of the physical and biological characteristics of the area as well as the methodology used for seismic survey noise and habitat suitability modeling, and a description of the risk estimation technique. section "Results and Discussion" shows and discusses the obtained results. Finally, Section "Conclusions" points out some conclusions regarding the risk assessment for this study area.

## Materials and Methods

This section describes the environmental and biological properties of the Setúbal test case area used as a hypothetical scenario to estimate risk resulting from seismic survey for the common dolphin population.

## Definition of the Target Area

The study area was limited to −9.3° to −8.5° longitude west and 38.2° to 38.6° latitude north, as shown in Fig. 1. The area reveals a relatively flat continental platform extending up to approximately 45 km from the coastline where the water depth reaches 250 m and then rapidly deepens to the west. In addition, the presence of an east-west oriented steep-sided valley (known as the Setúbal submarine canyon) at approximately 38.2° latitude north, which reaches approximately 1000 m depth, is considered an interesting feature of the area. The seismic surveying area, represented by the black rectangle in Fig. 1, is characterized by a relatively small bathymetric slope and water depths varying between 30 and 100 m, which are suitable for the installation of offshore wind farm structures (pylons, anchors, cables, etc.).

## Definition of the Target Species

The entire coast of Portugal is known to be a rich ecosystem in terms of marine biodiversity, covering a variety of groups from fish to marine invertebrates and sea turtles (Gomes et al. 2018). The cetacean occurrence is one of the most studied aspect as well as one of the highlights of the coast (Correia et al. 2015; de Castro 2010). The common dolphin (*Delphinus delphis*) is one of the most abundant species of cetacean in the Northeast (NE) Atlantic in general and in this region of the Portuguese coast in particular (Murphy et al. 2013). This species has, in fact, the

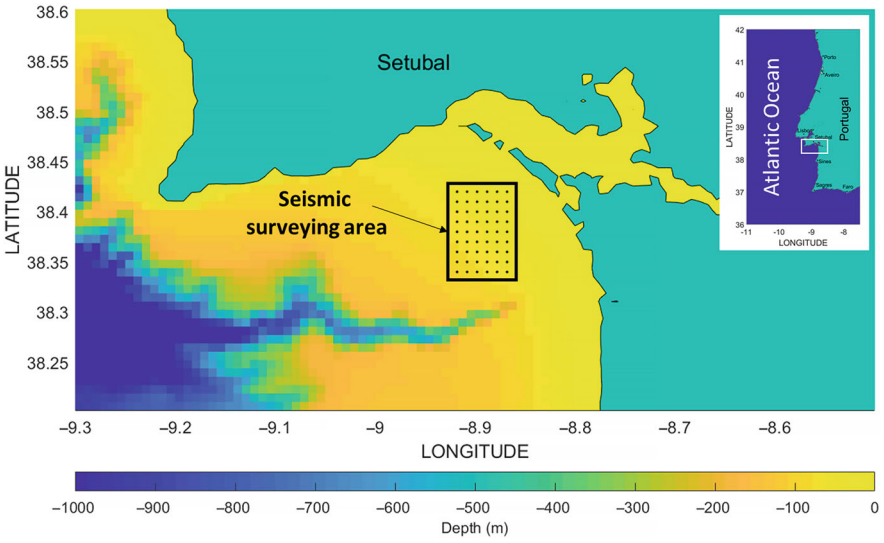



**Fig. 1** Bathymetry of the Setúbal region obtained from GEBCO database (Weatherall et al. 2015). The rectangular box of 50 km$^2$ was defined for seismic surveying simulation (longitude −8.92° and −8.87°, and latitude 38.34° and 38.42°)

highest number of records in dedicated databases (Silva 1999). Additionally, common dolphins belong to both the small cetacean group and to the mid-/high-frequency cetacean group (Southall et al. 2019; Finneran 2016). Therefore, due to their hearing sensitivity, they may be impacted by seismic surveying.

## Noise Source Definition

For modeling purposes, a Geo-Source 200 sparker (developed and commercialized by GEO Marine Survey Systems, The Netherlands geomarinesurveysystems.com) with two arrays of 100 electrode tips each was used as reference for seismic source. This type of seismic source is suitable for water depths from 2 to 500 m with a penetration in the order of dozens of meters below the seabed (depending on seabed type), which is commonly used in wind farms seabed prospecting and is characterized by a peak source level of 223 dB re 1$\mu$Pa.

## Seismic Surveying Modeling

The bathymetric data of the target area was obtained from the General Bathymetric Chart of Oceans (GEBCO, www.gebco.net) database (Weatherall et al. 2015), with a 1 km $\times$ 1 km spatial resolution (see Fig. 1).

In accordance with the description given by (Soares et al. 2015; Maglio et al. 2015) and outlined in Table 1, the bottom and sub-bottom acoustic properties were configured as a two-layer bottom model made up of a fluid sandy sediment layer over a rocky semi-infinite sub-bottom.

Water columns' physical properties (temperature and salinity) of the target area were obtained for the months of January and June 2019 from the Copernicus – CMEMS database (https://marine.copernicus.eu/). Figure 2 shows the superposition of profiles for the total area and for the months of January (upper row) and June 2019 (lower row), for temperature ((a) and (d)), salinity ((b) and (e)), and estimated sound speed ((c) and (f)). For each dataset, the black line represents the mean profile. As expected, temperatures in January are lower than in June. In January, a 50 m thick mixed layer develops, followed by a gradient till the depth of 500 m both in January and in June. A similar behavior is shown by salinity between January (see plot (b))

**Table 1** Assumed seabed parameters. (Adapted from (Soares et al. 2015; Maglio et al. 2015))

| Model Parameter (units) | Value |
|---|---|
| Sediment speed (m/s) | 1650 |
| Sediment density (g/cm$^3$) | 1.9 |
| Sediment attenuation (dB/$\lambda$) | 0.8 |
| Sediment thickness (m) | 10 |
| Sub-bottom speed (m/s) | 1800 |
| Sub-bottom density (g/cm$^3$) | 2.8 |
| Sub-bottom attenuation (dB/$\lambda$) | 0.2 |

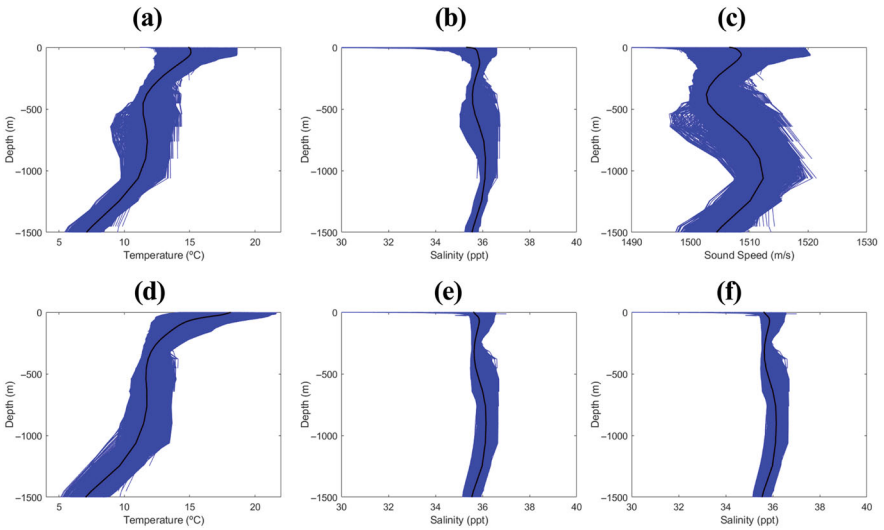



**Fig. 2** January (top row) and June (bottom row) 2019 variations for: temperature (**a** and **d**), salinity (**b** and **e**), and sound speed profiles (**d** and **f**). The mean profile is represented by the black lines. (Source: CMEMS-Copernicus Marine Service)

and June (see plot (d)), with higher variations in the mid water column in January compared to June, with characteristic values ranging from 35 to 37 parts per thousand (ppt). The nine-term Mackenzie approximate equation was used to estimate the sound speed profile: plots (c) resulting from January and plot (f) from June. The sound speed profiles of these 2 months follow, as expected, approximately the same shape as the temperature profiles of the respective months. The sound speed profiles in the deeper area result to have a double minima, which is typical of this area.

For simulation purpose, the seismic survey was centered on a relatively small rectangular area of 50 $km^2$ (see box in Fig. 1) with a spatial resolution of 1 km $\times$ 1 km. The previously described sparker seismic source was set to a source level of 223 dB re 1$\mu$Pa at 1 m, placed at 1 m depth with a 5 s firing interval considering a tow ship moving at 5 knot in a traditional lawn-mower pattern resulting in a regular set of emission positions along the area of Fig. 1. A total duration of 1 month was considered with a time resolution of 10 min.

The range of audible frequencies of cetaceans may extend far beyond 1000 Hz. In order to maintain the noise calculation computationally manageable, in this study only the frequency range 300–1000 Hz in 1/3-octave (10 base) bands was taken into account, which corresponds to the seismic source maximum level frequency band.

The acoustic receivers were placed at 5, 15, 30, 50, 75, and 100 m depth and the final noise map sound pressure level was evaluated as the mean field over depth. The statistical indicator percentile 50 (p50), which describes the percentage of time that a

specific value sound pressure level is exceeded, was adopted for sound map estimates and for the production of risk maps.

The normal mode propagation model Kraken (Porter and Reiss 1984) was used in combination with bathymetry, seafloor parameters, and sound speed profile (SSP) to estimate the transmission loss (TL), and thus the sound pressure level (SPL) was calculated according to Soares et al. (2015).

Since the assumptions for 1/3-octave bands may not be valid when dealing with impulsive noise that is emitted only at given time intervals, the estimation of a correction factor was deemed necessary. Since the seismic exploration implies a succession of periodic pulses at a given time rate, the concept of sound pressure level (SPL) was used in order to include transmission time. In a $T_0 = 5$ s cycle only $T = 1$ s is active. Thus, if one uses the standard definition of SPL as:

$$\mathrm{SPL}_{\mathrm{cycle}} = \mathrm{SPL}_{\mathrm{emitted}} - 10\log_{10}(T/T_0) \tag{1}$$

this gives a correction factor of approximately −7 dB relative to $\mathrm{SPL}_{\mathrm{emitted}}$. $\mathrm{SPL}_{\mathrm{emitted}}$ corresponds to the SPL in 1 s of emission. Nevertheless, this does not consider the time spreading nature of acoustic channels, which may make appear much longer at the receiver the short emitted pulse. Actually, the quantities of interest are the emitted and especially the received energy, rather than the frequency band power based only on 1/3-octave bands, as normally used for SPL definition. To highlight the difference, a test was performed following these steps:

- In the study area, two points, A and B, were selected representing the source and the receiver, spaced 2 km apart. To determine the channel impulse response (CIR), KRAKEN was used calculating modes for all frequencies between 300 and 1000 Hz.
- To obtain the received signal, the input signal was convolved with the CIR.
- A duty cycle of one pulse every 5 s was used to calculate the SPL for a duration of 10 min.
- The obtained SPL was then compared with the one obtained considering 1/3-octave band levels. A 4 dB correction factor was determined to be used in the subsequent simulations.

## Habitat Suitability Modeling

The common dolphin's HS was modeled through an ENM taking into account observation records of the common dolphin and environmental characteristics of the study area using a maximum entropy algorithm. The output of this model are, on one side, habitat suitability maps of the species and on the other side, the percentage of contribution of each environmental predictor to the model (see Fig. 3).

Maxent modeling (Phillips et al. 2006) was chosen among other methods essentially because of the few input data required, the availability of these data for the

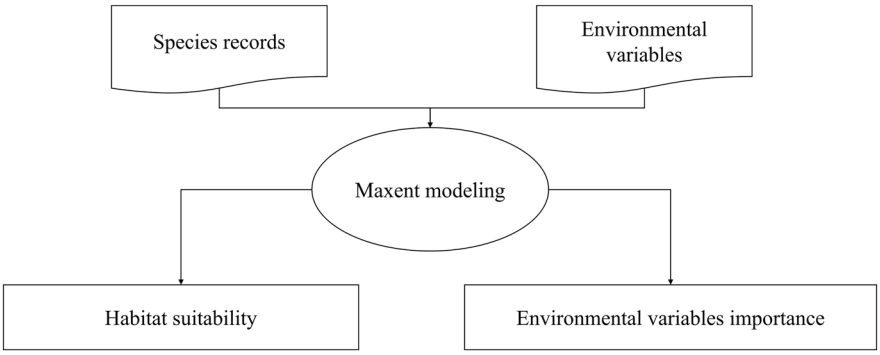



**Fig. 3** Methodological framework of Maxent modeling. (Adapted from (Wang et al. 2020))

**Table 2** Cetacean occurrences' datasets characteristics

| Observations' source | Code | Region | Coordinates lon × lat | Period of time | Background points creation method |
|---|---|---|---|---|---|
| **SPEA** | SPEA | Portuguese coast | [−10.5–7.5] × [36 42.5] | 2005–2020 | Using the transects of boat trips for all the species |
| **SeaEO tours** | SeaEO | Setúbal | [−9.6–8.7] × [38.3 38.8] | 2019–2020 | Minimum Sampled Area (MSA) |
| **MarIlimitado** | MI | Sagres | [−9.3–8.5] × [36.7 37.2] | 2005–2020 | Minimum Sampled Area (MSA) |

present case study and for being proven to give better predictions than other methods for small data amounts, such as those employed in this study (Ng and Jordan 2001). Additionally, following the definition given by Phillips and Dudík (2008), it was supposed that, for one specific species, habitat suitability can be interpreted as an estimate of the probability of species occurrence conditioned on environmental variables as it indicates the quality of the habitat for that single species (Phillips and Dudík 2008).

Cetacean occurrences (top left box of Fig. 3) were obtained over a period of 15 years (from 2005 to 2020) from three different opportunity platforms, as described in Table 2: SPEA, the Portuguese Society for the Study of Birds and two whale-watching companies, SeaEO Tours and MarIlimitado.

SPEA dataset that goes from 2005 to 2020, and SeaEO tours observations, from 2019 to 2020, were collected all year round, while Mar Ilimitado dataset, from 2005 to 2020, was mostly concentrated on the months between April and October.

Five environmental variables (see Table 3), already proven to affect the habitat suitability of small cetaceans and already adopted for the same purpose and same species in the literature (see for example Moura et al. (2012)), were selected as

**Table 3** Selected environmental variables

| Environmental variable | ACR | Unit | Source | Data setup |
|---|---|---|---|---|
| **Depth** | DEPTH | m | NOAA, ETOPO 1 Global Relief Model | Resampled to the selected resolution and extent |
| **Slope** | SLOPE | ° | Calculated on QGIS3.18 | Resampled to the selected resolution and extent |
| **Sea surface temperature** | SST | K | Copernicus Marine System | Scaled to the selected temporal and spatial resolution |
| **Chlorophyll-a** | CHL | $mg.m^{-3}$ | Copernicus Marine System | Scaled to the selected temporal and spatial resolution |
| **Chlorophyll-a standard deviation** | SD_CHL | $mg.m^{-3}$ | Calculated from the CHL layer | Scaled to the selected temporal and spatial resolution |

potential explanatory variables to calibrate HS models for the common dolphin (top right box of Fig. 3). The collinearity among the environmental variables, a challenging issue in ENM (Feng et al. 2019), was evaluated taking into account the variance inflation factor (VIF) approach, revealing no dependency among them.

This data were then pulled together into a temporal resolution of 8 days and 2 × 2 km spatial resolution in the World Geodetic System 1984 zone UTM 29 N projection.

The biggest issue facing presence-only and presence-background ENMs is accounting for sampling bias. The possibility of mapping sampling effort rather than the underlying HS exists when no sampling correction is implemented in the model (Barber et al. 2022). Therefore, in order to avoid this issue, a filtering approach was applied to remove potentially related sightings (Fernandez et al. 2021; Aiello-Lammens et al. 2015; Fourcade et al. 2014; Bystriakova et al. 2012). Moreover, two different target-background methods were applied according to the different nature of the selected dataset:

- **SPEA dataset:** transects between all the points with records for all species of cetaceans were considered over a 8-days-long period, generating the background data and providing a measure of the potential sampled area.
- **Whale Watching datasets:** a Minimum Sampled Area (MSA) technique, as in Fernandez et al. (2017), was used due to the opportunistic character of the sampling. A Minimum Convex Polygon, adding a 1 km buffer, was used to put together all the sightings for all species and for each distinct temporal scale. Grids crossing the polygon were classified as potentially sampled areas and thus considered as background.

The number of sea trips performed during a specific period determined the total amount of effort per temporal unit (8 days). From both datasets, occurrences and the

selected background grids were put together to proceed with the MaxEnt analysis. Random background datasets ($n = 10{,}000$) were generated for each analysis.

The common dolphin's habitat suitability maps along the Portuguese coast were obtained using Maximum Entropy modeling, as described in (Phillips et al. 2006). The HS maps resulting from this modeling, for the selected months of January and June, were then cropped for the detailed study area of Setúbal.

### Risk Maps

The risk maps presented in this chapter were produced taking into account the overlapping between sound pressure level and the biological distribution. In this particular case, this overlapping was performed between the SPL 50th percentile (or median) and the HS maps, for each point in space and time. According to the methodology proposed by Erbe et al. (Erbe et al. 2014), noise maps and habitat maps were normalized between 0 and 1 and then point wise multiplied. The result of this multiplication was then normalized again. As a result of these successive normalization, the resulting risk maps cannot be used for comparison between species, but gives a relatively good assessment of areas at high noise risk for the considered species. The risk maps were produced with a 2 km $\times$ 2 km spatial resolution.

## Results and Discussion

The results obtained for the Setúbal area test case are presented as: a) seismic survey simulation results, b) habitat suitability results, and c) risk assessment results.

### Seismic Survey Simulation Results

The SPL resulting from seismic surveying noise prediction modeling during the months of January and June 2019 is presented in Fig. 4a, b, respectively.

For a matter of comparison between the two cases, the color-coded noise level scale is presented between 150 and 185 dB which means that values lower than 150 are considered in blue and higher levels than 185 dB in yellow. It can be observed that noise propagation is significantly influenced by the bathymetry of the area which is attenuated over the entire platform till it reaches its edge. That's when the signal becomes dispersed over a much wider water column, so the sound level suddenly drops. It is also noticed that the pressure levels computed in January are higher and have a lager spatial spread than those estimated in June. This result can be explained by the sound speed profiles of both months, as described in section "Seismic Surveying Modeling." In both cases, the generated noise widely exceeds the usual mean ambient noise, resulting from the noise produced by continuous

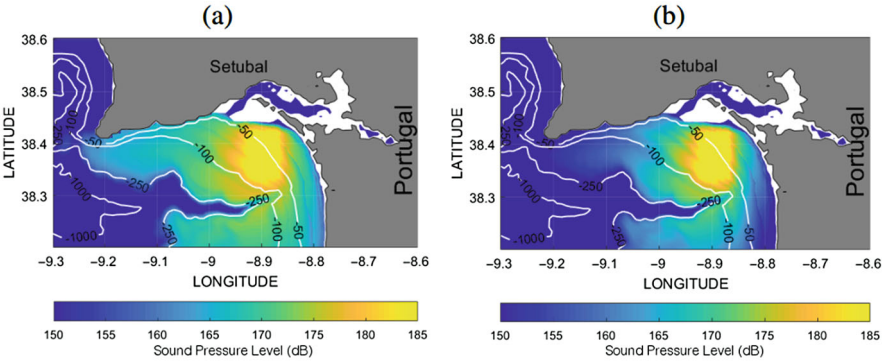



**Fig. 4** Percentile 50 of the Sound Pressure Level due to a seismic survey simulation for the months of January (**a**) and June (**b**)

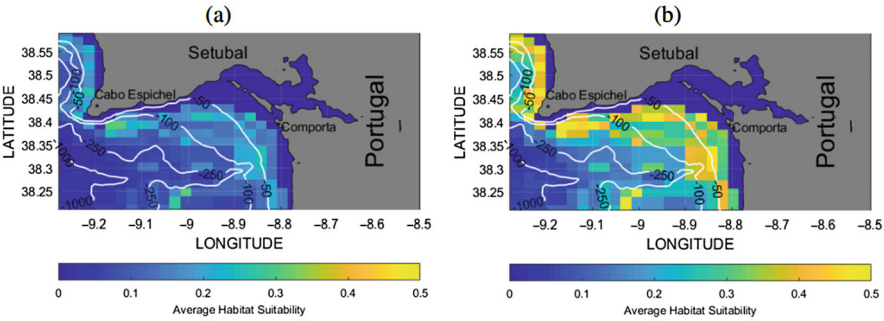



**Fig. 5** Common dolphin habitat suitability for the months of January (**a**) and June (**b**)

sources such as ships (120 dB according to Soares et al. (2015)) and from natural sources such as the wind (65–75 dB). Therefore, noise may represent, on species sensitive in this frequency band, a potential harmful impact.

## Habitat Suitability Results

HS resulting from the ENM for the common dolphin in the area of Setúbal was obtained for the months of January and June as shown in Fig. 5a, b, respectively.

The results are in accordance with a higher habitat quality for the common dolphin in the summer season rather than in winter. Nevertheless, in both seasons, the most favorable conditions are in the region delimited by the bathymetric line of 50 m and in the region delimited by the bathymetric lines between 100 and 200 m near Cabo Espichel, where the platform shortens. The areas where habitat suitability presents the greatest values are those near Cabo Espichel and Comporta.

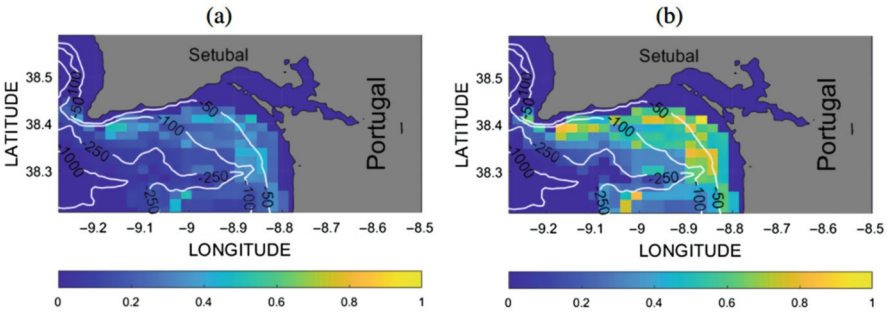



**Fig. 6** Risk maps for: January (**a**) and June (**b**) 2019

## Risk Assessment Results

Figure 6 shows the regions of predicted risk, estimated in the region of Setúbal, for the population of common dolphins for the months of January (a) and June 2019 (b), using the methodology outlined in the previous section.

It can be observed that even though the "affected area" has a much larger extension in January than in June, the former shows lower risk levels than the latter. In both cases, the highest risk areas result to be located near the shore, especially till the 100 m bathymetric line. The region near Comporta shows the highest risk levels, due to the highest HS of common dolphins in this area as described in section "Habitat Suitability Results."

In general, both the risk spatial distribution and its seasonality, reflect the combination of the HS distribution and the noise propagation. The areas with higher risk are those where: (1) HS values were higher (near shore) and (2) noise from the simulated seismic survey reached higher SPL values (near the area where the seismic survey took place). Summer presents higher risk levels because: (1) HS was higher in this season and (2) the highest values of SPL are more concentrated in this season. Additionally, risk maps reveal how regions that showed very high levels of HS, such as the northern coast of Cabo Espichel (between latitude 38.45 and 38.6, and longitude −9.3 and −9.2), but that were protected from the noise dispersion, present levels of risk very close or equal to zero. These findings confirm that risk is consistent with both the habitat suitability of the common dolphin and the propagation of the simulated noise from a seismic survey.

## Conclusions

This chapter presents a risk simulation for the common dolphin population in the region of Setúbal when exposed to a light seismic survey to assess the viability of offshore wind farm structures.

Habitat suitability results show that the highest habitat quality area follows the coastline configuration. This is particularly evident if considering the bathymetric

line of 100 m, which is known to correspond to an area with high productivity (Yen et al. 2004). Furthermore, the lower HS in January may result from a winter decrease in upwelling and subsequently to a decrease of the common dolphins' most important preys such as sardines (Silva 1999; Jefferson et al. 2009). This shows that the seasonality in habitat suitability is mainly related to water productivity and prey biomass (Danil and Chivers 2006).

Seismic surveys inject important quantities of energy into the ocean and consequently put at risk marine species in the surrounding area. Installing offshore wind farms occurs usually in shallow water, between 30 and 50 m depth, even though, as shown in the results, noise levels spread out for many kilometers. However, the coastal bathymetry induces an attenuation of the signal over the entire platform. Since light seismic surveys occur at such low depths which, as previously stated, are known for their high productivity, it is clear that it may potentially impact a higher number of individuals than typical seismic survey that occur in deep water.

The results confirm the important impact that seismic surveys may have on marine species, suggesting that, if they cannot be avoided, then periods and areas at lower risk should be chosen, whenever possible, in order to diminish the impact of these practices on marine species.

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# Categorizing the Effects of Anthropogenic Noise on Aquatic Life

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Sophie L. Nedelec

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**Abstract**

Categorizing the effects of anthropogenic noise on aquatic life according to the mechanisms of effects could help regulators with their decision-making. This chapter proposes categorizing effects into those that likely scale directly with sound level and those that are influenced by many other factors and thus may scale inconsistently with sound level. The first proposed category of mechanisms is physical (direct effects of sound waves on biological tissues or interference of sound during detection). Physical mechanisms could lead to injuries, temporary hearing loss, and masking. The severity of responses with physical mechanisms is likely to scale with received sound level. The second proposed category of mechanisms is perceptual (following sensory experience of sound waves). Perceptual mechanisms, such as stress and distraction, could lead to changes in physiology, cognition, and behavior. The severity of responses with perceptual mechanisms depends on the receiver's external context (e.g., other environmental stressors), current behavioral context (e.g., feeding or breeding), and internal state (e.g., hungry, distressed), as well as the received sound level. Threshold values for noise criteria where effects are driven by perceptual mechanisms will be challenging to establish due to the large number of interacting factors. However,

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regulating noise is an urgent priority as mortality can arise from effects with perceptual mechanisms, for example from a failure to exhibit a normal anti-predator response. Pragmatic policymaking for anthropogenic noise could consider area-based and time-based limits until sufficient information for effective level-based limits is available.

**Keywords**

Sound · Noise · Policy · Physical effects · Perceptual effects

## Introduction

The effects of anthropogenic noise on aquatic life have been categorized according to the proximity to the source and assumed corresponding sound level in relation to the hearing abilities of the animal (Popper and Hawkins 2019). The "Zone of Influence" is considered as the area where the animal can detect a sound and is divided into four sub-zones: Zone of Audibility, Zone of Responsiveness, Zone of Masking, and Zone of Hearing Loss, Discomfort, and Injury (Richardson et al. 1995). Assumed to occur with increasing distance from the source within the Zone of Influence, effects have been categorized by some authors as: death, physical injury and physiological changes, hearing threshold shifts, masking, behavioral responses, and no obvious behavioral responses (Popper and Hawkins 2019). Additionally, a "dose-response" relationship has been assumed between the received noise level and the magnitude of effects within each category. However, several papers have recognized the importance of context to behavioral responses (e.g., (Bruintjes and Radford 2013; Ellison et al. 2012; Harding et al. 2019; Merchant et al. 2022)), the link between physiological responses and behavioral responses (e.g., Mills et al. 2020), and the potential for mortality as a result of behavioral responses (e.g., Bernaldo De Quirós et al. 2019; Simpson et al. 2016). While it is logical that some types of effect are expected to consistently scale with sound level, for other types of effect other factors could influence the degree of impact. This chapter proposes categorization of the effects of noise according to the mechanisms by which effects may occur, in the hope that this could help to target policy decisions to maximize the potential benefits of regulating underwater anthropogenic noise (see Fig. 1). It is hoped that this paper will stimulate discussions about the nature and naming of new categories, and the effects that belong within them.

## Physical Effects

The first proposed category of effects is physical. Physical effects are considered here as effects of a sound wave either directly on the tissues of the body (which could cause injury) or interfering with another sound wave during detection by the animal (which could cause masking). Sound levels for both "continuous" noise (measured by averaging over time, e.g., root-mean-square sound pressure or particle

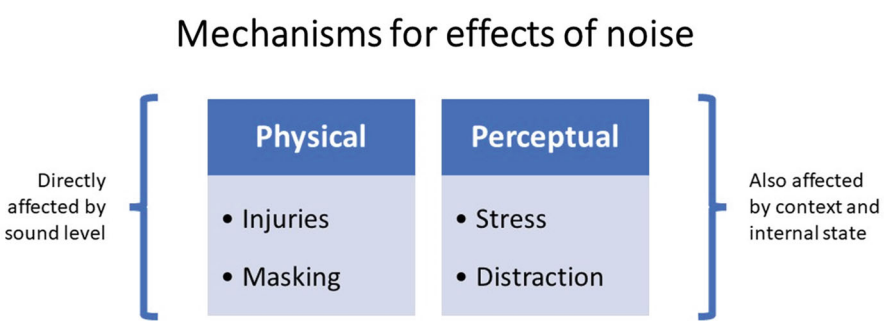



**Fig. 1** A schematic representation of suggested categories for the mechanisms for effects of noise. Any dose-response relationships may be inconsistent for effects with perceptual mechanisms and thus these effects may not necessarily scale directly with distance from the source or sound level

acceleration level) and impulses (e.g., 0-peak sound pressure or particle acceleration level) need to be considered. Due to relatively consistent dose-response relationships, physical effects are expected to be simple to mitigate using measured and predicted sound levels and criteria.

Injuries include barotrauma (tearing of tissues due to extreme fluctuations in pressure of gas bubbles such as swim bladders (Halvorsen et al. 2011; Popper and Hawkins 2019)) and tearing of tissues due to mechanical strain (possibly due to extreme fluctuations in particle displacement (Nedelec et al. 2021)), which may lead to death in the most severe cases, damage to organs, or hearing loss (via damage to auditory hair cells). For example, in a controlled experiment using plane waves in a specialized tank, Chinook salmon (*Oncorhynchus tshawytscha*) exposed to impulsive sounds mimicking pile driving strikes 1–1200 Hz at a range of sound levels experienced a corresponding range of severity of barotrauma injuries; from mild hematomas of fins at the lowest sound exposure levels (960 strikes single strike sound exposure level $SEL_{SS}$ (re 1 μPa) = 171 dB, cumulative sound exposure level $SEL_{cum}$ (re 1 μPa) = 204 dB), to mortal hemorrhage of the heart at the highest sound exposure levels (1920 strikes $SEL_{SS}$ (re 1 μPa) = 187 dB, $SEL_{cum}$ (re 1 μPa) = 220 dB) (Halvorsen et al. 2011). Further work in the same specialized tank established that the number of strikes as well as the $SEL_{ss}$ and the $SEL_{cum}$ influenced the probability of injury in hybrid striped bass (white bass *Morone chrysops* x striped bass *Morone saxatilis*) (Casper et al. 2017). Injuries could impact fitness via immediate mortality, shortening lifespan, or impacting reproduction. Immediate mortality has been observed in several cases, e.g., Halvorsen et al. 2011. Injuries could also impact physiology and behavior, e.g., digestion could be impacted by barotrauma to the gut or swimming could be impacted by hematoma to fins and muscles. The animal does not necessarily need to be able to hear the sound for injuries to occur; therefore, it is relevant to consider frequencies that are beyond the hearing range of the species concerned. Reducing the received sound level of anthropogenic noise (also described as noise abatement (Merchant 2019)) is expected to decrease the severity of injuries in a predictable way.

Masking occurs when the ability to detect or recognize a sound of interest is degraded by the presence of another sound (the masker) (Erbe et al. 2016). Masking is defined by the International Organization for Standardization for humans as a property of the auditory system by which an audio signal cannot be perceived in the presence of another audio signal (ISO/IEC 13818-7:2006). Quantitatively, masking refers to the amount in decibels by which an auditory detection threshold is raised in the presence of a masker (Acoustical Society of America 2013). Masking can occur when noise overlaps signals in the receiving sensory organ or in the central nervous system in time, frequency, and particularly for fishes and invertebrates who detect particle motion vectors, direction (Rogers et al. 2021). Masking is referred to as a physical effect here because it happens as a direct result of the physical properties of the sound present in the medium and inhibits the ability of the animal to detect, recognize, or perceive a signal.

Masking has been experimentally observed in marine mammals (recently reviewed in (Branstetter and Sills 2022)). Models depicting sound pressure levels within the acoustic communication range of marine mammals can be effective at demonstrating risk to marine mammal communication, for example, an assessment of vessel slowdown showed increases in effective listening space for a variety of marine mammals in the western Canadian Arctic (Pine et al. 2018). Masking has also been experimentally observed in fishes (e.g., Hawkins and Chapman 1975; Maiditsch and Ladich 2022); however, generating models that can predict masking in fishes is currently limited by the lack of understanding and measurement of particle motion levels and the potential for directional unmasking (Rogers et al. 2021), also termed spatial release from masking (Erbe et al. 2016).

Particle motion is inherent to sound waves at all frequencies and is detected by all fishes within their hearing range, thus in situations where particle motion is higher than would be predicted from sound pressure measurements (e.g., within the near field of a source), masking could be underestimated if only pressure is considered. Hearing thresholds measured only in sound pressure omit key information about the stimulus and background noise sound particle motion levels that the fish were exposed to during a hearing test, limiting the application of these data to any contexts outside of the tank used for the experiment. Conversely, as particle motion has directionality due to its nature as a vector, fishes are capable of directional unmasking using particle motion when noise is coming from a different direction to the signal (Rogers et al. 2021; Sisneros and Rogers 2016), meaning that masking may be overestimated if only omnidirectional sound pressure is considered. Despite the knowledge gaps and current limitations to our ability to predict masking, the degree of masking relates directly to the levels of noise in the environment and thus noise abatement ought to decrease the impacts of masking in a predictable way.

## Perceptual Effects

The second category of effects could be called “perceptual” (of, relating to, or involving perception, especially in relation to immediate sensory experience;

Merriam-Webster Online Dictionary 2023). Perceptual effects are proposed to be those where anthropogenic noise is detected by the animal, perception of the acoustic signal occurs, and a response arises as a result. The main two mechanisms by which perceptual effects might manifest considered here are stress and distraction. Two observable types of response that may be driven by these mechanisms are physiological and behavioral. Both stress and distraction may occur at the same time or independently. It is not always simple to determine which mechanism is leading to an observed physiological or behavioral response. However, if mechanisms can be established this could help regulators to target more effective mitigation.

Organisms evolve stress responses that are beneficial to fitness under normal environmental variability, for example, stress responses can help animals to survive danger by preparing the body for "fight, flight, or freeze." Stress responses can be maladaptive when the severity or duration of the stressors reaches a point where homeostasis cannot be maintained. If the severity of the stressor is too high the stress response may be inadequate, for example if the temperature in the environment becomes too hot for thermoregulation to effectively reduce an endothermic animal's body temperature to within a safe range. Alternatively, if the intensity of the stressor is overly severe or long-lasting, overactive physiological stress response mechanisms may themselves become harmful to the individual, a state associated with the term "distress" (Wuertz et al. 2023).

Stress responses are mediated by hormones, for example via stimulation of the hypothalamic-pituitary-interrenal or "HPI" axis culminating in the release of corticosteroid hormones in fish (Barton 2002). The observable indicators of a stress response to anthropogenic noise include primary (hormonal), secondary (physiological), and tertiary (behavioral) responses. Experimental evidence of these include: increased plasma cortisol in orangefin anemonefish (*Amphiprion chrysopterus*) in response to motorboat noise playback on coral reefs (Mills et al. 2020); increased heart rate in staghorn anemonefish (*Amblyglyphidodon curacao*) embryos in response to motorboats driving near coral reefs (Jain-Schlaepfer et al. 2018); and increased hiding in threespot dascyllus (*Dascyllus trimaculatus*) in response to motorboat noise playback near anemones (Nedelec et al. 2016). We may infer that stress is at least one of the mechanisms for changes in behavior if changes in plasma hormone levels correlate with changes in behavior, with a temporal pattern that allows for the mechanism to enact. For example, in orangefin anemonefish exposed to playbacks of motorboat noise, an increase in testosterone circulating in blood plasma was followed by an increase in aggressive social interactions (Mills et al. 2020). All aquatic life exists in a multi-stressor environment where the contexts of climate change, over-exploitation of resources and anthropogenic noise must be considered in concert. Stress responses may also scale with perceived risk as well as or instead of the intensity of the stressor measured in the environment. Therefore, where stress is one of the mechanisms for impacts of noise, the sound level in the environment could be a poor predictor of the threat imposed. On the other hand, where stress is a mechanism for impacts, reducing stress via noise mitigation and abatement could increase resilience to other environmental stressors (Nedelec et al. 2022).

Animals need to pay attention to relevant stimuli in order to perform vital life functions such as feeding, finding mates, and avoiding predators but animals may have a limited capacity to attend simultaneously to multiple stimuli (Dukas 2004; Purser and Radford 2011). Distraction is the reallocation of finite attention away from relevant stimuli (Chan et al. 2010; Purser and Radford 2011). The observable impacts of distraction by anthropogenic noise include failure to perform normal behaviors effectively such as feeding (Purser and Radford 2011) and anti-predator responses (Chan et al. 2010; Simpson et al. 2016). Stress could interact with distraction via effects on cognition (Salena et al. 2021). The severity of perceptual effects therefore not only depends on the received level of the sound but also the external context (other stimuli in the environment), current activity (e.g., feeding or avoiding predators), prior opportunity to habituate to the noise, and the internal state of the animal (e.g., impacts of other stressors, level of hunger) (Bruintjes and Radford 2013; Harding et al. 2019; Merchant et al. 2022). Therefore, dose-response relationships for effects with perceptual mechanisms may not scale directly with sound level or may appear inconsistent across contexts. Noise pollution thus presents a regulatory challenge. Experiments where contexts and internal states are tightly controlled are needed to determine dose response relationships for effects with perceptual mechanisms. The relationships revealed may then offer general principles (such as providing evidence that abatement can reduce impacts of noise) but specific sound exposure levels may not necessarily translate to other contexts.

## Considerations for Noise Criteria and Threshold Values

The categorization described here feeds into the need for noise criteria and threshold values for the regulation of anthropogenic noise pollution. Stress and distraction can both be mechanisms for behavioral and physiological effects, behavioral effects can lead to immediate mortality (e.g., via impacts on anti-predator responses (Simpson et al. 2016), or delayed mortality (e.g., via impacts on parental care behavior (Nedelec et al. 2017)), and physiological effects can lead to a state of distress which may be detrimental to fitness. Distress due to physiological effects may be detrimental to fitness without apparent behavioral responses, meaning that the boundary between the zone of audibility and the zone of responsiveness may depend on the animal's internal state, current activity, prior opportunity for habituation, and external context.

The European Union's (EU) Marine Strategy Framework Directive (MSFD) requires member states to develop marine strategies for maintaining Good Environmental Status of European waters. The EU Commission Decision 2017/848 of 17 May 2017 defines two criteria for anthropogenic noise; impulsive noise and low frequency continuous noise. Member States are required to establish threshold values for these two criteria where spatial distribution, temporal extent, and sound levels do not exceed levels that adversely affect populations of marine animals. The Technical Group on Underwater Noise has published advice and guidance for EU Member States developing threshold values (Borsani et al. 2023; Sigray et al. 2023).

The proposed approach is to use the concept of a level of onset of biologically adverse effects ("LOBE") in combination with the spatial distribution exceeding LOBE for an amount of time. The LOBE is a sound level above which an adverse biological effect on an indicator species is expected to occur, i.e., an effect that may affect the comfort, survival, and vital functions of individual animals (Sigray et al. 2023). For impulsive noise, behavioral avoidance is proposed for calculating habitat loss and thus reductions in carrying capacity (Sigray et al. 2023) and for continuous noise, effects on masking, behavior, and physiology are considered as options for defining a LOBE (Borsani et al. 2023). There remain however, significant practical challenges to establishing evidence-based LOBEs for the relevant indicator species.

The strictest noise criteria, particularly in the presence of multiple species and the absence of good data on other factors such as those species' hearing abilities and influence of other stressors, would be any noise level that is detectable among background ambient noise. Avoiding any anthropogenic noise being detectable among background ambient noise is an unrealistic target on a global scale in the near future. Some industrial activity is required for a just transition away from a fossil fuel-based economy and some noise pollution from this activity is unavoidable. However, it may be possible to apply this as a criterion at restricted spatial or temporal scales for where and when there is a high need for protection according to the context, such as breeding fish in vulnerable or recovering populations, or when and where other environmental stressors are severe such as during and the aftermath of a marine heatwave. Any noise criteria that use a sound level above that which is detectable among background ambient noise could involve a compromise as to the extent of the effects of the anthropogenic noise that is considered acceptable. Full consideration of the costs and benefits of such compromises are further complicated by the inability to know what the extent of the effects for a given sound level are. Requiring this information prior to regulating noise could slow down regulatory decision-making. Due to the infinite possible combinations of sound levels, species, contexts, and internal states, and the real-world limitations on the speed of advancement of scientific knowledge and the speed of policymaking and implementation, regulating noise pollution will necessitate compromises, with imperfect knowledge about the risks of noise impacts.

## Conclusions

In summary, the effects of noise on aquatic life could broadly be divided into categories according to their mechanisms: those with physical mechanisms, where impacts could be expected to scale with sound level; and those with perceptual mechanisms, where impacts are influenced by other factors and thus impacts may not always scale directly with sound level. Using noise abatement measures to reduce the area and/or amount of time that anthropogenic noise can be detected among background noise could be the simplest way to minimize all impacts as the benefits that can be achieved by limiting noise to a given sound level cannot be fully defined

rapidly enough to meet regulatory needs in a pragmatic context where time and funding have real-world limits.

In conclusion, there are regulatory challenges for anthropogenic noise. There is a clear need for regulation to prevent harm to aquatic life from human activities, but several factors make regulation complicated. There is a desire for a simple "one-size-fits-all" approach to regulation, where a single sound level can be applied as a limit above which noise is deemed unacceptable. This desire is understandable because it would be relatively simple to apply in terms of mitigation technologies and behavior changes, as well as for monitoring compliance. However, the number of variables means that this issue is more complicated. Considering area-based and time-based limits rather than level-based limits for indicator species who are vulnerable to other stressors or in sensitive contexts (e.g., Atlantic cod (*Gadus morhua*) during a marine heatwave in the breeding season) could be a way of simplifying the regulatory challenge to achieve rapid improvements in the interim while more detailed evidence is gathered.

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# Deflagration: A Method for Lowering Noise from Underwater Unexploded Ordinance (UXO) Clearance Operations

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Paul A. Lepper, Stephen P. Robinson, Sei-Him Cheong, Lian Wang, Jakob Tougaard, Emily T. Griffiths, and John Hartley

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**Abstract**

Significant amounts of historic unexploded ordnance (UXO) are still located in our seas and waterways. Operations to clear waterways of these munitions have increased significantly in the last few decades due to increasing offshore

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activities, for example, the expansion offshore wind energy development. Various methodologies for Explosive Ordnance Disposal (EOD) exist, but a historically commonly used approach is to blast-in-place (BiP). This process involves detonation of a modern explosive charge next to the historic charge initiating a simultaneous high-order detonation in both. These underwater explosive events are highly energetic and have the potential to generate high acoustic and vibrational levels in both the water column and the seabed with the associated potential for both physiological and behavioral response impacts on a wide range of aquatic fauna. An alternative EOD process has been investigated where a smaller shaped charge is used to initiate a burning process, known as deflagration, in the UXO's explosive material. This low-order process has the potential to be significantly quieter and less destructive than equivalent high-order detonation events. This chapter presents results from trials carried out in a flooded quarry in the UK in 2019 comparing the acoustics of detonations and deflagration methodologies on like-for-like munitions. Results are also presented of measurements carried out in genuine historic (WWII) UXO located in the Danish Baltic in 2022. Data from both of these trials demonstrated an around 15–20 dB reduction in the water column acoustic levels and seabed vibration levels comparing low-order with high-order methodologies. Additional factors surrounding the long- and short-term toxicity from in situ and cleared UXO processes and the potential for damage to the seabed are also discussed.

## Introduction

Historic WWI and WWII Unexploded Ordnance (UXO) still exist located in our waterways with an estimated 300,000–500,000 pieces in UK waters alone (DEFRA 2022). These munitions have individual explosive charge sizes ranging from a few tens of grams to greater than 500 kg of TNT equivalence explosive charge mass. These have been largely left undisturbed in situ in the postwar periods. In recent years, however, there has been a significant increase in the number of controlled Explosive Ordnance Disposal (EOD) operations required by a range of offshore activities, for example, offshore wind energy developments and ground fishing to make their areas of operation safe.

A commonly used method of UXO clearance (EOD) involves a *Blow-* or *Blast-in-Place (BiP)* operation carried out by colocation of a modern explosive *"donor"* charge. This is typically around 5–10 kg of TNT equivalence and is used to initiate a simultaneous "high-order" detonation in both the primary donor charge and the secondary often-larger historic charge of the UXO. This process has the potential for generation of very large underwater explosive events with associated high acoustic and vibrational energy levels in both the water column and the seabed. This has led to a growing concern of potential negative impacts on aquatic fauna and their environments related to these detonations. Potential impacts include those generated by high levels of acoustic noise and seabed vibration (Yelverton et al. 1973; Ketten et al. 1993; NOAA 2016; Dahl et al. 2020; Todd et al. 1996; Finneran et al. 2000;

Danil and St. Leger 2011; Sundermeyer et al. 2012; von Benda-Beckmann et al. 2015; Parsons et al. 2000; Salomons et al. 2021; Cottrell and Dupuy 2021; Siebert et al. 2022; Robinson et al. 2022; Jenkins et al. 2022; Smith et al. 2022; Favretto-Cristini et al. 2022a, b), seabed damage (Garlan et al. 2018), chemical toxicity (Strehse et al. 2017; Appel et al. 2018; Maser and Strehse 2020; Schuster et al. 2021; den Otter et al. 2023; Maser et al. 2023), and others.

In 2019, the underwater acoustic characteristics of both traditional high-order methodologies and an alternative low-order EOD methodology known as deflagration were investigated in trials in a flooded quarry in the UK (Robinson et al. 2020). The low-order deflagration process investigated used a small shaped explosive charge to penetrate the outer, typically metallic, shell on a munition and then to initiate a "low-order" burning process of the historic explosive material. In this chapter, data from acoustic measurements of both "high-order" and "low-order" methodologies used on like-for-like surrogate munition shells are presented. These data show a significant reduction in the acoustic levels associated with the low-order methodology of around 20 dB for a 10 kg historic charge disposal. Following on from the quarry experiments, provisional data are presented for open water trials that took place in the Little Belt, Denmark, in January 2022 of both high- and low-order methodologies used on actual historic (WWII) UXOs of charge sizes in the range 200–430 kg. As with the quarry test, significantly lower acoustic noise outputs were observed using a deflagration process on these UXOs compared to the high-order detonation clearances. Potential noise impacts and relative noise levels of both high- and low-order methodologies are discussed as well as other factors such as seabed damage and chemical toxicity in the context of current EOD operations and potential impacts on underwater environments.

## Controlled Quarry Trials of Low- and High-Order EOD Methodologies (2019)

### Methodology

In 2019, a series of acoustic tests were conducted of both high-order (detonation) and low-order (deflagration) EOD methodologies using surrogate shells filled with modern plastic explosives of charge size 5 kg and 10 kg. These were conducted at Limehillock quarry in Aberdeenshire in Scotland. This is a freshwater flooded quarry around 200 m long and 20 m deep operated by Thornton Tomasetti Defence Ltd., UK. Figure 1 shows the experimental layout. Both high-order and low-order test rigs were placed at a depth of approximately 8 m from the surface in around 20 m water. A series of T11 low sensitivity recording systems (Neptune Sonar, UK, nominal charge sensitivity: 0.07 pC/kPa; maximum pressure: 275 MPa) and 138A26 transducers (PCB, nominal voltage sensitivity: 0.29 mV/kPa; maximum pressure: 172 MPa) were used to capture the pressure waveforms at horizontal ranges 8–20 m from the source. Additional measurements were made at the far

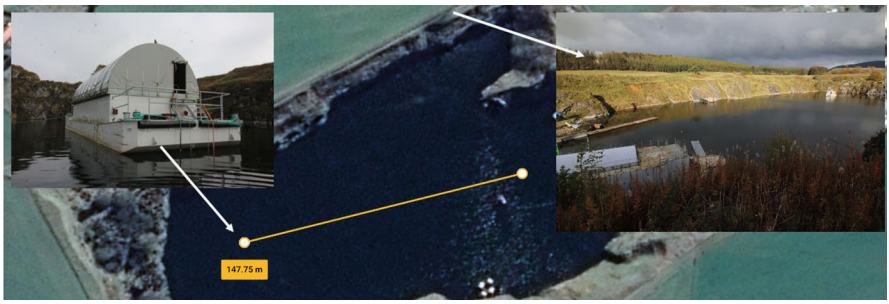



**Fig. 1** Limehillock quarry field site in Aberdeenshire in Scotland (Robinson et al. 2020). (Courtesy of Thornton Tomasetti Defence Ltd)



**Fig. 2** VULCAN deflagration charge placed on a 5–10 kg surrogate shell before deployment (Robinson et al. 2020). (Courtesy of Thornton Tomasetti Defence Ltd)

end of the quarry using a series of hydrophone arrays deployed from a floating barge at a range of 147 m (Robinson et al. 2020).

High-order detonation clearances were achieved by direct placement of a detonator through a threaded endcap into the main charge explosive in surrogate shells filled with either 5 kg or 10 kg of P4 plastic explosive. Low-order clearances were achieved using a deflagration methodology based on a 25 g shaped charge technology known as a VULCAN™ developed by Alfords Technologies, UK (VULCAN 2023), into identical surrogate shells at both 5 kg and 10 kg TNT charge size. The explosive filler was a plastic explosive (PE4) which consisted of 88% RDX plastic explosive (Grade 1 A), 11% plasticizer, and 1% pentaerythritol dioleate. Figure 2 shows a VULCAN deflagration charge positioned on the 10 mm thick steel endcap of a surrogate shell casing before deployment (Robinson et al. 2020).

In total, seven tests were conducted on surrogate shells including 1 × 5 kg (high-order), 2 × 10 kg (high-order), 2 × 5 kg (low-order), and 2 × 10 kg (low-order). In

addition, acoustic analysis was carried on several unconstrained charge masses including 15 g, 25 g, 48 g, 250 g, and 18.7 kg with 17 charges tests in total. Trials were also carried out on the deflagration VULCAN charge (25 g) directly onto a steel plate. All systems were deployed at similar depths and measured using identical methodologies outlined above (Robinson et al. 2020).

## Quarry Trial Results (2019)

All the high-order detonation trials demonstrated apparent disintegration of all the charge mass, the shell, and the detonator. For the total charge sizes 5 kg and 10 kg, significant surface plumes were observed as shown in the example in Fig. 3 for a 5 kg shell high-order event. By comparison, the low-order deflagration events showed no significant surface plume; however, bubbles were observed at the surface within a few seconds of the initialization indicating the deflagration "burning" process happening at a much slower speed than the high-order detonations. Inspection of the remaining shells where deflagration was carried out showed a significant reduction in the original PE4 material from the deflagration process, as shown in two examples in Fig. 4. This reduction in explosive material without an associated detonation was between 47 and 58% in both the two (5 kg & 10 kg) tests considered here.

Figure 5 shows the peak sound pressure level ($L_{p,pk}$) measured from the initial highest amplitude for each of the test charges at ranges of around 15–20 m. Note these arrivals were considered free-field before significant multipath arrivals from the surrounding quarry walls, bottom, and water surface. The data, shown in Fig. 5, shows a strong relationship between charge size and peak sound pressure amplitude, with significant peak pressure amplitudes of around 4 MPa (a $L_{p,pk}$ level of 252 dB re



**Fig. 3** Surface plume from a high-order 5 kg filled surrogate shell detonation. (Courtesy of Thornton Tomasetti Defence Ltd)



**Fig. 4** Examples of two post deflagration (after the endcap has been removed from surrogate shell) events showing the degree of burnout of the original P4 explosive (Robinson et al. 2020). (Courtesy of Thornton Tomasetti Defence Ltd)

1 μPa) being reached for a 10 kg charge size at 20 m and greater than 255 dB re 1 μPa for the 18.7 kg charge size. This measured data relationship (black dashed line) aligns well with midwater detonation models such as those developed and described in various forms by investigators such as (Cole 1948; Arons 1954, and Weston 1960). Here the peak pressure in the initial positive-going shock wave as a function of scaled range is as follows:

$$L_{p,pk} = 52.4.10^{6} \left( \frac{R}{W^{1/3}} \right)^{-1.13}$$

where $L_{p,pk}$ is the peak pressure in Pa, $R$ is the measurement range in meters, and $W$ is the charge weight in kg of equivalent TNT (Robinson et al. 2020, 2022). By comparison, the low-order deflagration trials showed peak sound pressure ($L_{p,pk}$) levels around 20 dB lower at around 232 dB re 1 μPa for clearance of identical 5 kg and 10 kg filled surrogate shells compared to the equivalent high-order events. These levels were almost identical (green dashed line) to the levels measured for initialization of a 25 g deflagration charge on its own (without an associated surrogate shell), suggesting that most of the acoustic output of the deflagration clearance was

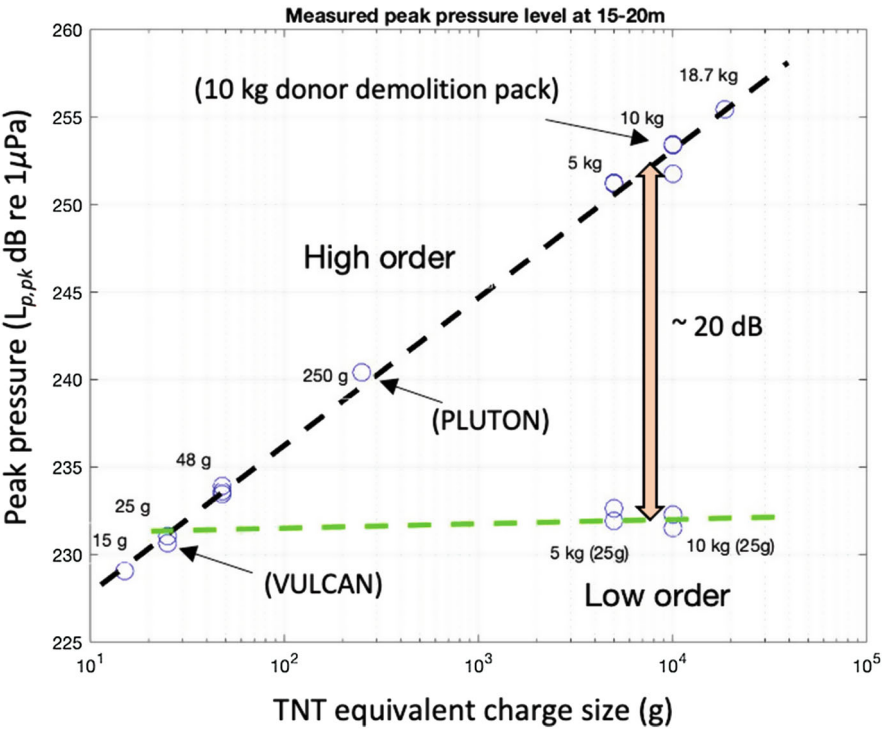



**Fig. 5** Peak pressure ($L_{p,pk}$) measured at ranges 15–20 m for each of the various charge sizes. Note that the low-order charges indicated show either the 5 kg or 10 kg of the original shell size that was deflagrated by the VULCAN charge in this case of 25 g. Also indicated here is the data for the equivalent charge size for the PLUTON deflagration technology and the typical 10 kg donor charge equivalent discussed in section "Comparison of Low- and High-Order EOD Methodologies on In Situ Historic UXO (2022)"

primarily due to the deflagration charge solely. Good replication of acoustic levels was observed for similar charge sizes across all the trials carried out suggesting a high degree of predictability in determining potential acoustic levels using models such as those outlined above in the case of the use of "modern" explosive material and midwater detonations.

## Comparison of Low- and High-Order EOD Methodologies on In Situ Historic UXO (2022)

### Methodology

In January, 2022 acoustic measurements were made of four high-order EOD and four low-order EOD clearances of historic WWII mines (UXO) in the Little Belt area of the Danish Baltic. The EOD operations were carried out by the Royal Danish Navy

on six WWII British marine mines (MK4, MK6) of original UXO-equivalent TNT charge sizes ranging from 340 kg to 430 kg in water depths from 18 to 22 m. Water column acoustic measurements of all these events were made on a range of bottom-deployed autonomous recorder systems at ranges from a few kilometers to greater than 20 km. Figure 6 shows the bathymetry of the trial site, locations of the six UXOs (with the following identifiers: Foxtrot [F], Golf [$G_1$,$G_2$], Mike [M]), November [$N_1$,$N_2$], Juliet [J], & Lima [L]), and the deployment of an array of ten bottom-mounted autonomous recording stations (R0-R9) in water depths of 12–20 m. Additionally vertical hydrophone arrays were also deployed from the *RV Aurora* operated by Aarhus University at ranges varying from 1.54 to 6.08 km as well deployment of a continuously recording GPR™ seabed vibrational system (Sercel, France) at the R0 station (GPR 2023). All recordings were made with sufficient high bandwidth to allow capture of representative time domain waveforms due to rapid rise times typical with explosive sources, and these bandwidths were also sufficient to capture higher frequency components from the acoustic signal to >150 kHz bandwidth (Lepper et al. 2024). Measurements were carried out where possible in line with protocols described in (NPL 2020).

In total, eight clearance events were carried out, listed here chronologically, F(high-order), $G_1$ (low-order), $G_2$ (high-order), M (high-order), $N_1$ (low-order), $N_2$ (high-order), J (low-order), and L (low-order). Both the Golf [G] and November

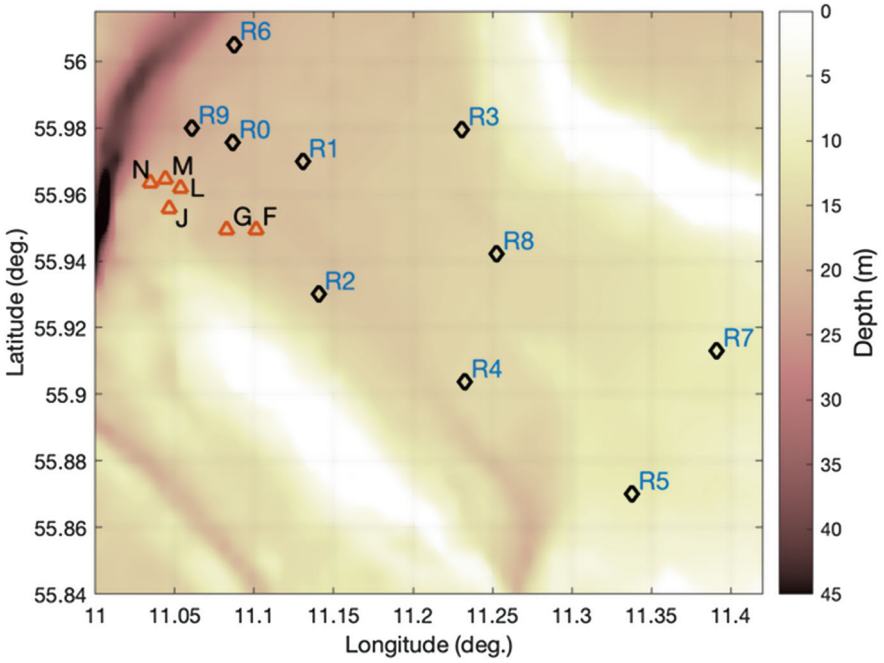



**Fig. 6** Bathymetry of trial site overlaid with UXO positions [F, $G_1$, $G_2$, M, $N_1$, $N_2$, J, & L] (red triangles) and recorder station R0-R9 (black diamonds)

[N] mines were initially cleared using a low-order deflagration methodology [$G_1$ & $N_1$] followed by high-order clearances of the remnants of the low-order clearances described as [$G_2$ & $N_2$]. Diver inspection and sonar scans were also made before and after each event as well as chemical analysis of water and sediment samples before and after each of the clearances. The mines were in initial variable condition on and in the seabed, ranging from severely corroded with the original explosive material exposed to the environment (Fig. 7 shows the mine Golf [G] before clearance), compared to the mine Lima [L] (Fig. 8 shows the mine Lima [L] before clearance) appearing relatively intact with minimal evidence of corrosion. The mines also showed a variable degree of burial ranging from 10% to 80% (Lepper et al. 2024).

The high-order clearances were carried out using a 10 kg (TNT equivalent) demolition pack placed adjacent to the historic UXO. Low-order clearances were carried out using a PLUTON™ deflagration charge from Alford Technologies with 250 g (TNT equivalent) charge size (PLUTON 2023). One of these systems was used on each of the low-order trials, deployed and operated by divers from the Royal Danish Navy. Both the high-order detonation and low-order deflagration events were

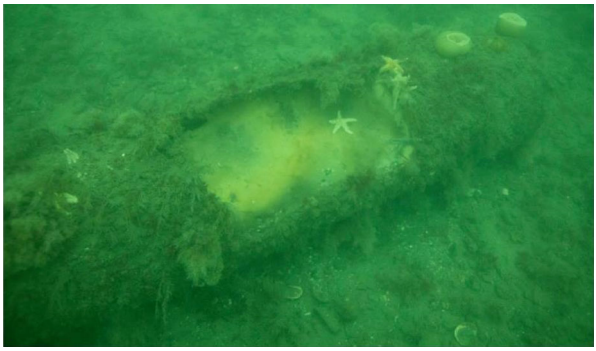


**Fig. 7** Preclearance mine Golf [G], heavily corroded with significant marine growth. Identified as a MK4 British WWII marine mine with equivalent TNT original charge mass of 340 kg (Lepper et al. 2024) (Supplementary courtesy of the Royal Danish Navy and Alfords Technology)



**Fig. 8** Preclearance mine Lima [L], partially buried but appearing relatively intact. Identified as a MK6 British WWII marine mine with equivalent TNT original charge mass of 430 kg (Lepper et al. 2024) (Supplementary courtesy of the Royal Danish Navy and Alfords Technology)



**Fig. 9** Preclerance deployment of Pluton device for low-order deflagration of mine Golf [G] (seen in Fig. 7) (Lepper et al. 2024) (Supplementary courtesy of the Royal Danish Navy and Alfords Technology)

initiated using a shock-cord to a near-by surface vessel. All acoustic systems recorded continuously during each of the trials. Figure 9 shows the deployment of a Pluton deflagration charge on mine Golf [G] before the low-order $G_1$ clearance.

## Explosive Ordnance Disposal (EOD) Results

From an Explosive Ordnance Disposal perspective, the Royal Danish Navy considered each of the eight clearance events (four high-order and four low-order) successful and they considered the munitions safe and free from further risk from uncontrolled detonation and could be approached by EOD divers for postclearance analysis (Royal Danish Navy, personal communication). The four high-order events (Foxtrot [F], Golf [$G_2$], Mike [M], & November [$N_2$]) resulted in generation of significant surface plumes, several tens of meters high, observed from several kilometers away. Post analysis of the seabed by divers and sonar scans showed no visibly detectible fragments of the UXO but often with significant cratering at the UXO site. Figure 10 shows postdetonation sonar scan of the Foxtrot high-order clearance with generation of a seabed crater around 6.4 m diameter and approximately 1–3 m deep. Similar cratering was seen for all the high-order events.

By comparison, no surface plume was observed during any of the low-order clearance events (Golf [$G_1$], November [$N_1$], Juliet [J], & Lima [L]). Postclearance inspection by divers showed in all cases evidence of a deflagration burn with reduction in the original UXO explosive material. In the case of mines N and L which appeared relatively intact in preinspection, there was evidence of the mine splitting or bursting, most likely due to overpressure from generated gasses during the deflagration process. Images of the chronological sequence from (initial condition) – (deflagration preparation) – (post deflagration) of the low-order clearance of mine Golf are shown in Figs. 7, 9, and 11, respectively. Other deflagration events showed similar results with all having some remaining explosive materials either still

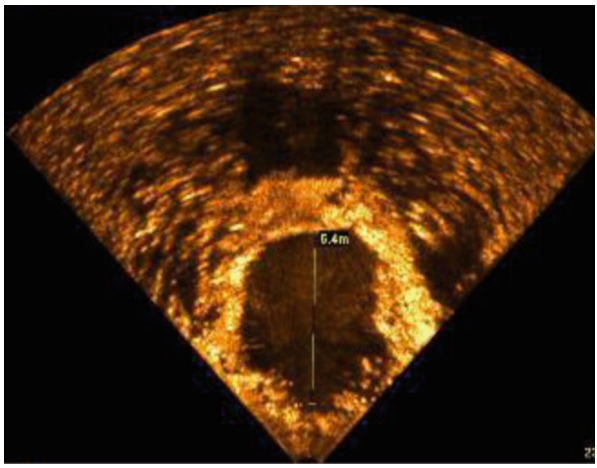



**Fig. 10** Post-high-order clearance of mine Foxtrot. No visually detectable fragments of the UXO observed with significant seabed cratering observed on all of the high-order clearnce events (Lepper et al. 2024). (Supplementary courtesy of the Royal Danish Navy and Alfords Technology)



**Fig. 11** Post-low-order clearance of mine Golf. Note the UXO has been split open and much of the original Amotol explosive charge has either been burnt via the deflagration process or shattered and ejected as the shell casing has split (Lepper et al. 2024). (Supplementary courtesy of the Royal Danish Navy and Alfords Technology)

within parts of the mine shell or nearby. In the case of all the deflagration trials, the debris from the clearance is scattered within a few meters of the original UXO site. For none of the low-order deflagration events was there any evidence of seabed damage of the form seen with the cratering observed on the high-order clearances. Analysis of the debris and acoustic outputs suggest that none of the four low-order deflagration trials carried out here underwent a deflagration-to-detonation transition (DDT).

## Acoustic Results

Analysis of all of the acoustic data from the static array and the *Aurora*-deployed sensors was made for broadband amplitude metrics $L_{p,pk}$ and sound exposure level

(SEL) at ranges from a few kilometers to around 22 km as well as linear and decidecade band spectral analysis as well as time and frequency analysis of seabed vibration data recorded with the sensor on station R0.

Figure 12 shows the acoustic time-domain waveforms measured at longer ranges (18.6–22.4 km) for each of the eight clearance events. Typical levels for peak pressure ($L_{p,pk}$) at these longer ranges for the 4 × high-order events (F, $G_2$, M, & $N_2$) were in the range 172–178 dB re 1 μPa compared to 151–154 dB re 1 μPa for the associated low-order events ($G_1$, $N_1$, L, & J). Direct comparison of both low- and then high-order events on both the Golf and November UXOs allows comparison at identical distances. In these cases, the Golf [$G_1$] low-order event had a 14 dB lower $L_{p,pk}$ than the [$G_2$] high-order event at the same distance of 19.7 km. A 17 dB lower level can be seen for $L_{p,pk}$ between the November low- and high-order events [$N_1$ & $N_2$] at identical ranges of 21.3 km. In both cases, the low-order clearances [$G_1$ & $N_1$] took place first, acting on the original, potentially much larger UXO charge mass of around 340 kg. The secondary high-order clearances [$G_2$ & $N_2$] were, however, carried out on the remnants of the preceding low-order deflagration clearances on a smaller estimated equivalent charge size of around 34 kg each + the 10 kg high-order donor charge used and therefore potentially acting on a significantly lower original + donor = total charge mass. Note the 340 kg stated here is the estimated original manufactured munition charge size of the UXO. However, several of the UXO investigated showed evidence of environmental degradation over time with, in some cases, the explosive material exposed to the marine environment for significant periods. Despite these potential differences in the size of the original explosive

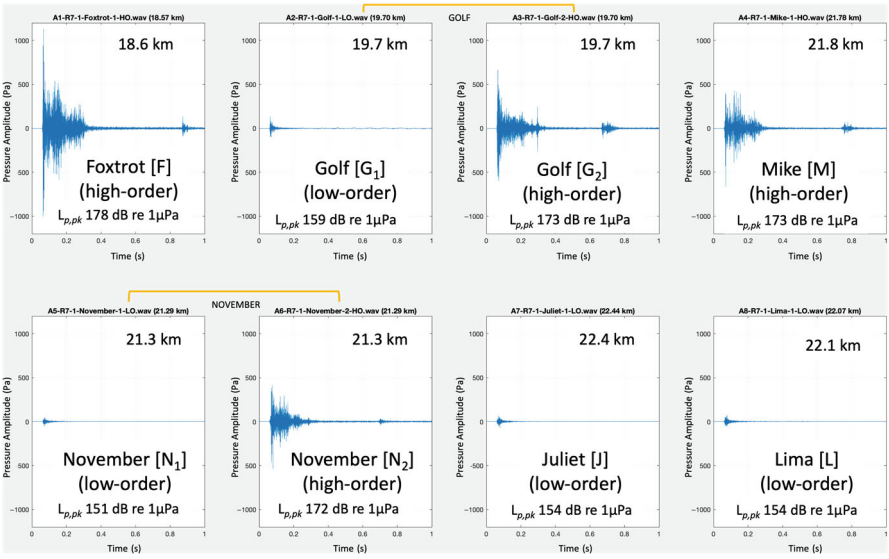



**Fig. 12** Long-range time-domain waveforms for each of the 8 × clearance events Foxtrot [F], Golf [$G_1$,$G_2$], Mike [M], November [$N_1$,$N_2$], Juliet [J], and Lima [L]

material being acted on, the low-order events still showed significantly lower overall acoustic levels. Similar differences of 14–20 dB lower levels were observed at all measured ranges for both $L_{p,pk}$ and SEL between all the high- and low-order events.

Figure 13 shows SEL versus range for all the clearance events. As with the peak pressure metric ($L_{p,pk}$), typical SEL amplitudes were between 15 and 20 dB lower for low order at all the ranges observed. Both the SEL and $L_{p,pk}$ metrics show a wider variation in absolute level for the high-order events compared with the low-order. In the case of the low-order events, the absolute measured level correlated relatively well with controlled measurements of the high-order detonation of a 250 g charge carried out in the quarry (Robinson et al. 2020); see Fig. 5. Combined with the modeled broadband transmission loss profiles for this environment, the low-order data versus range seen in Fig. 13 also aligns relatively well with the suggestion that a majority of the acoustic signal observed in these cases is from the low-order deflagration charge on its own.

By comparison, the levels associated with the high-order events were often close to, or some fraction greater, than would be predicted by a 10 kg donor charge on its own. None of the high-order events in these trials generated levels, based on the same models, that indicated detonation of the historic UXO at its original full

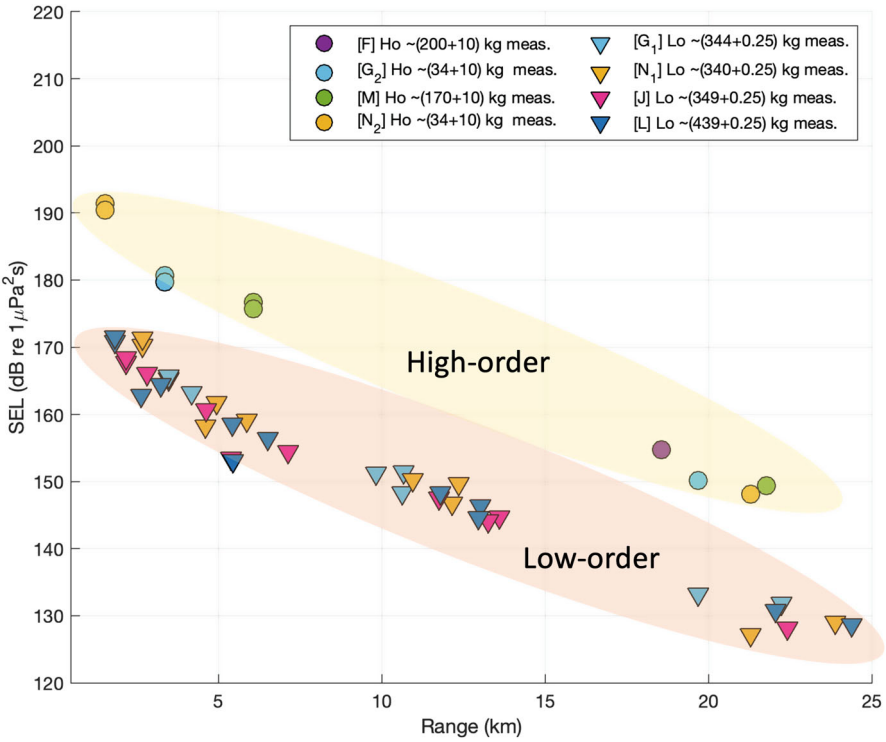



**Fig. 13** SEL level versus range for each of the eight clearance events. Legend gives the clearance methodology used and the estimated charge mass acted on for each EOD operation

potential with many having levels indicative of the donor charge plus some partial contribution from the original UXO full explosive potential. This broadly aligns with observations of 54 high-order clearances of WWII UXO made in 2022 in the North Sea, where most showed lower observed levels than might be expected from the original munition's full explosive potential. In that case, only a couple of examples out of the 54 events analyzed gave levels close to levels that models might predict based on original munition size (Robinson et al. 2022). Observation of reduction in measured versus predicted levels for high-order events have been reported by a number of investigators particularly in shallow water environments (Gaspin et al. 1979; Chapman 1985, 1988; Hannay and Chapman 1999; Soloway and Dahl 2014; Robinson et al. 2020; Salomons et al. 2021; Robinson et al. 2022; Favretto-Cristini et al. 2022a, b), and a number of or combination of potential reasons have been suggested by a number of these studies including degradation over time of the explosive materials, energy lost into the seabed compared to midwater detonation, cavitation shielding by the bubble generation and surface interaction, surface blowout, etc.

Even with these slightly lower than predicted levels, the actual measured values shown in Figs. 12 and 13 show the potential in both SEL and Peak ($L_{p,pk}$) metrics for generation of stimuli that could cause both physiological (for example, permanent threshold shift (PTS) and/or temporary threshold shift (TTS)) as well as behavioral responses at significant ranges from source in a wide range of marine species (Southall et al. 2019; Popper et al. 2014; NOAA 2016).

Provisional analysis of the seabed vibration also indicated significant levels detectable at ranges of up to several kilometers from the clearance events likely due to sediment or seabed interface borne wave propagation. Comparison of relative levels of signals from the seabed-mounted vibration recorder in both the vertical and horizontal planes showed similar relative level reduction in amplitude between the high- and low-order methodologies (Lepper et al. 2024). These propagating signals may be of significant importance in the assessment of potential impacts on a wide range of aquatic fauna particularly sensitive of vibration and particle velocity, particularly with the likely generation of complex pressure and shear wave propagation in the water column, the seabed, and the water-seabed interface from these events (Jenkins et al. 2022).

## Conclusion

Controlled measurement trials conducted in a quarry on like-for-like known munitions showed around a 20 dB reduction in noise levels for a 10 kg charge disposal using deflagration versus traditional high-order methods. The measurements from open water trials in Denmark on historic UXO of original sizes of up to 460 kg TNT equivalent showed similar 15–20 dB lower radiated noise levels at the measured ranges between the high- and low-order methodologies. None of the low-order deflagration clearances observed in these trials appear to have undergone a

deflagration-to-detonation transition (DDT); however, the potential for this is an acknowledged risk throughout the EOD industry. The overall reliability of the low-order deflagration methodology is currently based on a relatively low number of observed operations of underwater deflagration events and therefore remains an important unanswered question warranting further investigation.

In both these trials, the signal output from the deflagration methodology appears dominated by the initiating deflagration charge making it much easier to predict and model. Note that the deflagration charge technology used in these trials is essentially a much-smaller (25–250 g compared to ~10 kg for the high-order donor charge equivalent) high-order detonation and therefore intrinsically quieter. However, the historic UXO high-order clearance levels observed in the trial ranged from close to just the donor charge (around 10 kg) up to a fraction of the full historic charge in addition to the 10 kg donor charge. The potential net reductions between high- and low-order methodologies could therefore be significantly higher for larger charge sizes if the original UXO explosive material was to detonate at its full potential. This potential uncertainty in how powerfully and how much of the historic UXO detonates in high-order events combined with environmental effects, particularly in shallow water, makes assessment of potential levels more difficult with a higher degree of uncertainty for the high-order compared to low-order events.

The deflagration trials carried out did show evidence of a reduction in the original explosive material through the deflagration process but with some residual explosive material remaining either contained within the original UXO shell or dispersed over a relatively small area compared with high-order detonations that have shown significant seabed damage but with little (visually observable) evidence of the original UXO's materials remaining. Chemical analysis has, however, shown increased levels (compared to high-order events) of some explosive material by-products in the immediate vicinity of the UXO site in the case of low-order events (Maser et al. 2023; Lepper et al. 2024) which warrant further investigation.

A low-order deflagration methodology approach to EOD was demonstrated in a real-world environment on historic UXO, and the associated reduction in radiated acoustic levels and seabed damage serves as a significant improvement in the reduction of potential impact on marine life; however, it should be acknowledged that even at these lower levels the absolute levels are still relatively high and/or comparable to other typical marine noise sources (Ainslie et al. 2012). This can be seen in the water column measured data but also potentially in the seabed and water/seabed interface where impact levels are still relatively unknown. These signals in both the water column and seabed may still serve as a source of noise impact over significant ranges (Lepper et al. 2024). Reduction of levels still further may require a combination of mitigation strategies to be carefully considered and warrants further investigation.

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**Note** Parts of this chapter have been reproduced from the following sources: (Robinson et al. 2020) for the quarry trial and (Lepper et al. 2024) for the trials in Denmark by the same authors.

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# Joint Monitoring Program for Ambient Noise in the North Sea

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**Abstract**

A joint transnational monitoring program for ambient noise in the North Sea has been developed by the Jomopans project (Joint Monitoring Program for Ambient Noise in the North Sea). Validated soundscape maps were produced that quantified the ambient sound in the North Sea. From these maps, the underwater noise can be assessed and they are valuable information to develop measures to improve the quality of the environment, required by the European Marine Strategy Framework Directive (MSFD). Jomopans developed soundscape maps for the North Sea based on acoustic modeling of the sound produced by ships and wind. The spatial and temporal exceedance of ship noise over wind noise is assessed through so-called Dominance maps. In total 15 measurement stations around the North Sea were employed and gathered long-term sound data. These data were used to validate the soundscape maps.

Following that approach, it was found that shipping noise dominates the underwater soundscape of the North Sea. In the southern part and along the major shipping routes, the shipping noise exceeds the broadband natural sound by more than 20 dB for more than 50% of the time.

**Keywords**

Monitoring · North Sea · MSFD · Shipping noise · Soundscape maps · Shipping noise

## Introduction

Management of the marine environment must be based on the best available information. This information for most pollutants is obtained by a network of monitoring stations. Pollution by continuous noise consists of various sources with large spatial differences. A network of measurement stations alone cannot deliver reliable

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information to be used for policy and management. Especially for the North Sea, which is almost 600,000 $km^2$, soundscape maps are required for implementing policy measures. Up until now reliable quantitative information on continuous underwater noise is largely lacking.

The Marine Strategy Framework Directive (MSFD, European Commission 2008) is the major legal framework for improvement of the marine environment in the European Union (EU). In the MSFD for the first time underwater noise was addressed as an environmental pollutant and subsequent obligations followed from this for the member states of the EU. In 2017, the regulations from the MSFD were more specified in the so-called Commission Decision (European Commission 2017). Another policy driver affecting underwater noise and its sources is the Directive on Marine Spatial Planning (MSP, European Commission 2014). This directive aims to regulate human activities at sea and the spatial planning of these activities.

Stimulated by the MSFD the regional sea convention OSPAR elaborated on the monitoring and assessment of underwater noise. For continuous noise, this resulted in a monitoring strategy (Snoek et al. 2015) and an assessment framework (Van Oostveen et al. 2020). Also, the Jomopans project (Joint Monitoring Program for Ambient Noise in the North Sea, 2018–2022) was executed to set up an operational monitoring framework. Eleven institutes from seven countries bordering the North Sea worked close together in Jomopans.

The Jomopans project contributed largely to the methodology for assessment continuous noise proposed by the MSFD working group for underwater noise (Sigray et al. 2021). Further development of the methodology by this working group is ongoing.

## Jomopans Approach to Monitoring

To develop a framework for the monitoring of ambient sound, Jomopans built upon the method proposed by the European Commission MSFD Technical Group on Underwater Noise (Dekeling et al. 2014) and the latest knowledge from relevant regional projects (e.g., the EU LIFE+ project Baltic Sea Information on the Acoustic Soundscape (BIAS), see Nikolopoulos et al. 2017). Jomopans developed a combination of numerical modeling and field measurements of ambient sound. A combination of models (creating sound maps) and measurements is regarded the best way to describe levels and their temporal and spatial variations.

The use of numerical models is cost-effective and can be operated in a controlled environment. Maps of the ambient noise levels for the North Sea region were calculated based on information associated with major human activity in the North Sea, namely shipping that contributes the most to continuous underwater noise. Modeling is a powerful tool for better insight in causes and distribution of underwater noise.

Numerical modeling always uses a schematization of reality and is dependent on various input parameters, which also have uncertainties.

Measurements for long-term monitoring of ambient sound in a large area like the North Sea is very challenging. In total 14 measurement stations were deployed around the North Sea (see Fig. 1). These stations were operated by the Jomopans

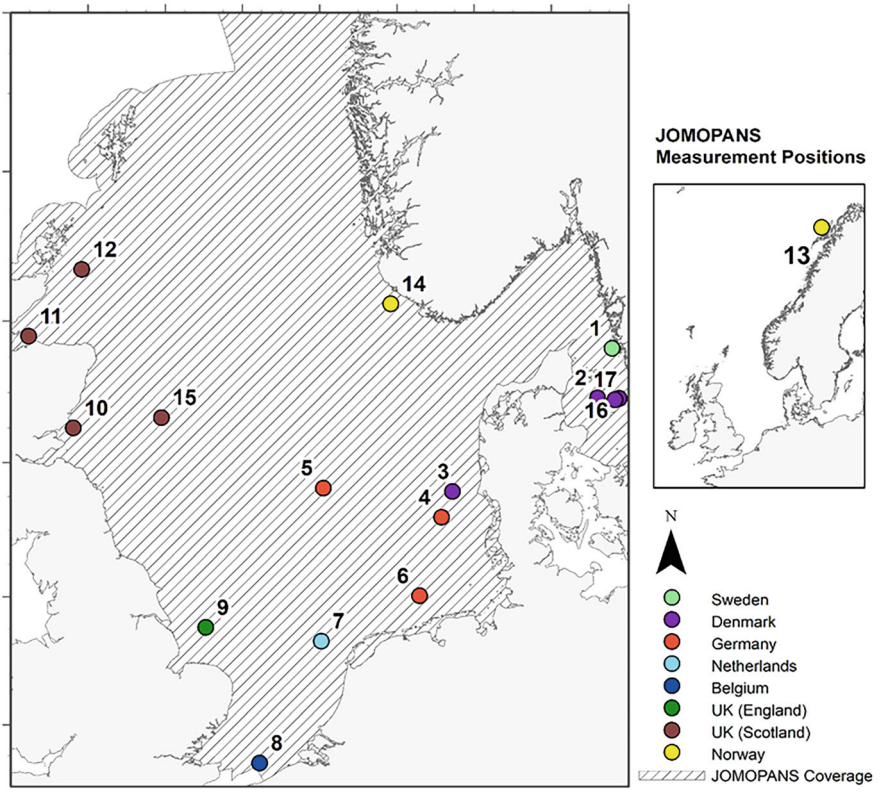



**Fig. 1** Positions of the 18 measurement stations indicated by pie charts depicting the share of ship classes on total traffic per station; contour map in the background exhibits over what percentage of the year 2019 the predicted anthropogenic noise dominates the natural soundscape by 20 dB or more (broadband 10–20 kHz), soundmap produced by TNO as part of the JOMOPANS project

partners and consisted of a range of different instruments. Special effort was made to perform a cross-calibration of the instruments and derive consistent working methods across the various parties.

Measurements are accurate and they provide a direct quantification of the local sound levels from all sources of interest at particular locations. On the other hand, measurements are influenced by sea conditions. In the North Sea, flow noise is a particular problem for measurements. Although measurements deliver a huge amount of data and information, it is not enough to make soundscape maps. Jomopans chose to use these measurements primarily for the validation of the (modeled) soundscape maps.

Assessment of underwater noise should be based on three aspects of the noise: (1) noise level, (2) temporal extent, and (3) spatial extent (TG Noise 2021). The choice of the noise level for assessment is based on the environmental effect that is considered. Jomopans specifically addressed the masking effect, which limits the communication range of various marine species. Therefore, the excess level

(difference between the total sound pressure level [SPL] and the SPL of background sound) was chosen as a metric. Various soundscape maps of both the background SPL (especially caused by wind) and the total SPL (caused by shipping) were modeled. The background SPL was calculated from a model for wind-induced sound (Ainslie 2010). Shipping noise was modeled using data from the AIS (automatic identification system) ship tracking system and the source-level model based on measurements from the ECHO Program (Enhancing Cetacean Habitat and Observation) of the Port of Vancouver (MacGillivray and De Jong 2021), in combination with an efficient sound propagation model (TNO Aquarius 3 is a range-dependent hybrid analytical mode sum + flux integral model). The modeling enabled the generation of a variety of maps. Maps for the months of the year, different ship types, and frequency bands were calculated.

The temporal aspects were addressed from all these soundscape maps by using the percentage of time over which the excess level was above 20 dB. The map giving this information is called the Dominance map. Increased knowledge on the effects of continuous noise on marine animals is needed to determine the level, which is now chosen as 20 dB. These studies were outside the scope of the Jomopans project.

## Results

The quantities that were mapped were (see also Merchant et al. 2018):

- Sound pressure level of background sound (dB re 1 μPa)
- Total sound pressure level (dB re 1 μPa)
- Excess level (dB)
- Dominance

Each of these maps is calculated for six different frequency bands:

- "Broadband," covering the 10 Hz–20 kHz decidecade octave bands
- "Decade 1," covering the 20 Hz–160 Hz decidecade octave bands
- "Decade 2," covering the 200 Hz–1.6 kHz decidecade octave bands
- "Decade 3," covering the 2 kHz–16 kHz decidecade octave bands
- 63 Hz decidecade octave band
- 125 Hz decidecade octave band

The maps covered averages over each month of 2019 and 2020 as well as yearly averages. In total, 12 ship types (fishing ships, cruise/ferry ships, bulkers, container ships, tankers, dredgers, tugs, naval ships, recreational ships, government/research ships, passenger ships, and vehicle carriers) were modeled in separate maps, based on the classification in the AIS data provided.

Here only the major soundscape maps are presented.

In Fig. 2, the median sound pressure level (125 Hz decidecade octave band) of the background sound in 2019 is shown. The background level is almost uniform for the

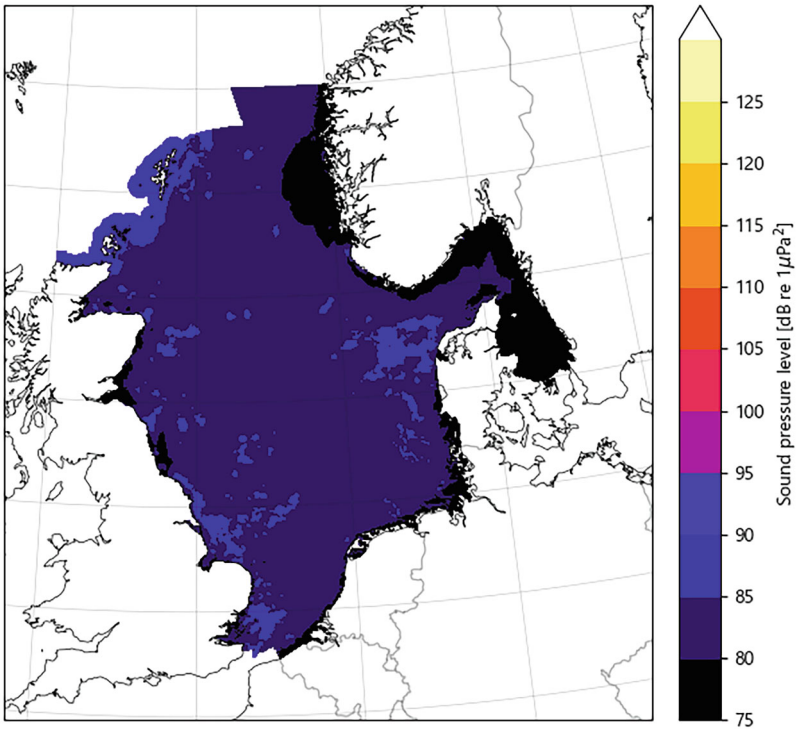



**Fig. 2** Median background sound level 125 Hz band (wind)

entire North Sea. This means that for the North Sea assessing the excess level will give similar results as assessing the SPL.

In Fig. 3, the median SPL (125 Hz decidecade octave band) of the total sound in 2019 is shown. The total SPL is dominated by the intense shipping in large parts of the North Sea. This shipping (and the SPL) shows only small seasonal variations.

Figure 4 shows the excess level (125 Hz decidecade octave band) in 2019. These levels show that for most parts of the North Sea the sound field is dominated by shipping noise. Minor seasonal variations of the excess level are caused by the variations in the background level.

Figure 5 is the Dominance (the percentage of time over which the excess level was above 20 dB) in 2019. This shows that the high excess level in some areas persists most of the time.

The independent validation of the models, using 15 Jomopans field stations, is illustrated in Fig. 6 (see also Putland et al. 2022). Monthly percentiles of the measured and modeled data were compared within decidecade frequency bands from 10–2000 Hz, in decade frequency bands, and as broadband sound levels. The general pattern of uncertainty showed that the model predicted sound levels, which were lower than the measurements at low frequencies (<1 kHz), while the model more closely agreed with the measurements at higher frequencies (>1 kHz). The

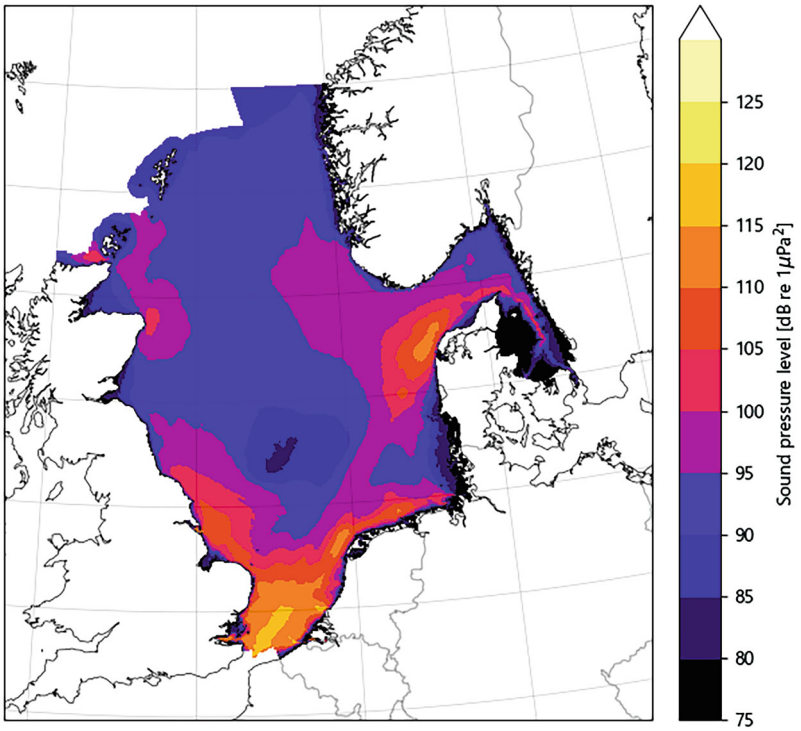



**Fig. 3** Median sound pressure level by shipping, 125 Hz band

model uncertainty is likely due to incomplete input data, because not all ships are captured by AIS and other sound sources other than ships may have contributed to the measured SPL at low frequencies (Fig. 6).

## Discussion

This chapter presents a framework for a joint monitoring program for ambient noise in the North Sea. This framework is a major building block for the assessment framework for continuous noise, which is adopted by the European Commission (Sigray et al. 2021).

Jomopans developed validated soundscape maps for the North Sea based on acoustic modeling of the sound produced by ships and wind. The modeled maps show that the soundscape in the North Sea is dominated by shipping noise. There are large differences across the North Sea. In the southern part toward the English Channel and along the shipping routes to Hamburg and the Baltic Sea, the noise exceeds the natural background noise by more than 20 dB for more than 50% of the time. The central part of the North Sea shows relatively low noise levels.

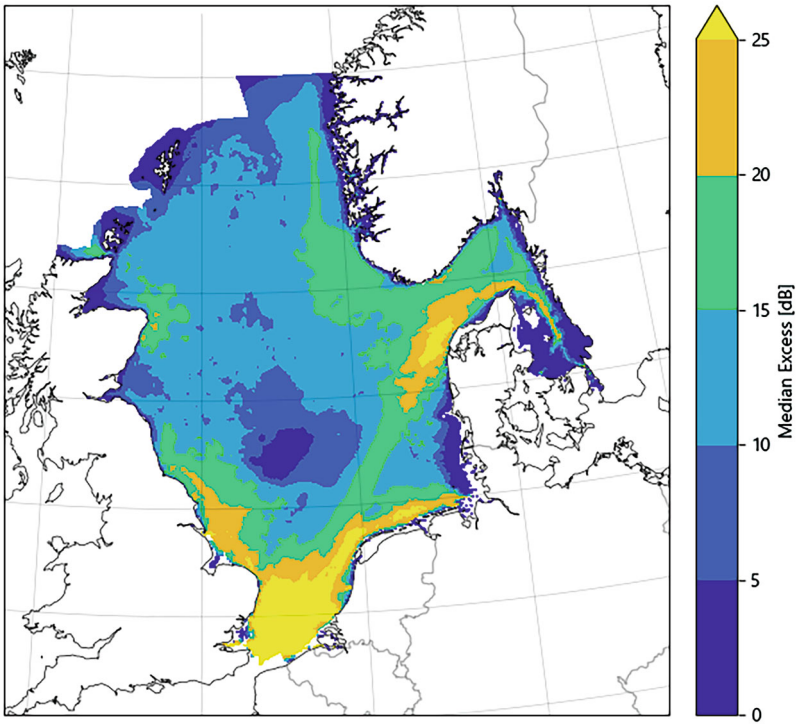



**Fig. 4** Median excess level, 125 Hz band

The Jomopans validation showed that both modeling and measurements have limitations and complement each other. The validation of the maps confirms that for large parts of the North Sea the soundscape is dominated by shipping. The modeling did not include other source types beyond shipping, which especially influenced the maps in low-noise areas. The measurements in the North Sea were in some cases hindered by flow noise. In addition, differences occurred from having different operators and instruments. A cross-calibration proved to give an insight into these differences.

Numerical modeling of soundscape maps provides marine managers with detailed information on the spatial distribution of the sound as well as the relative strength of various sources. Validation of the soundscape maps by field measurements is essential.

In future work, all major sources need to be accounted for. For many sources, a proper description of the source levels is still lacking. The measurements contain much information, which could not be evaluated within the limitations of the Jomopans project. Basan et al. (2023) give an insight of this additional information.

Improving of field measurements under North Sea conditions remains a challenge for the years to come.

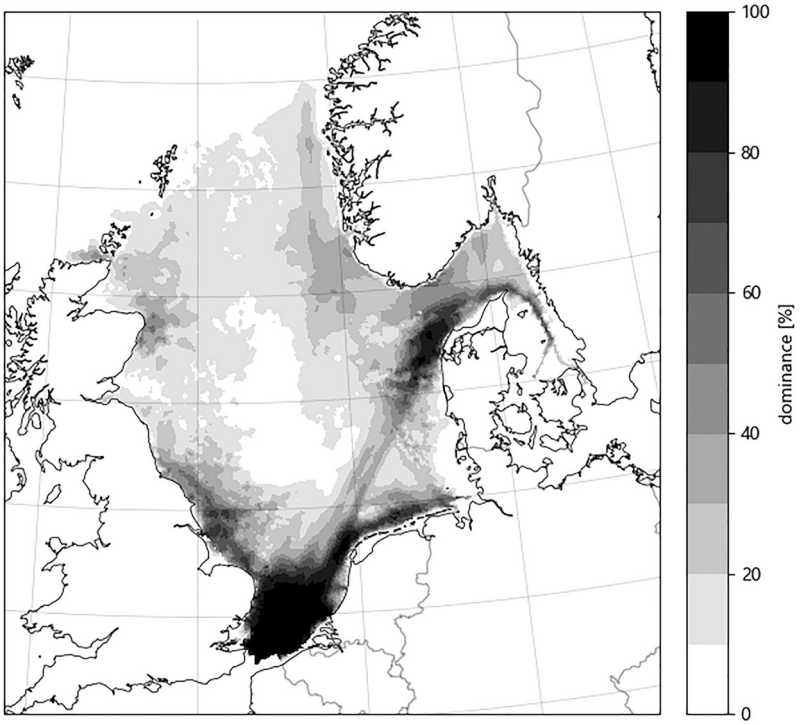



**Fig. 5** Dominance maps for a cutoff level of 20 dB of the excess level

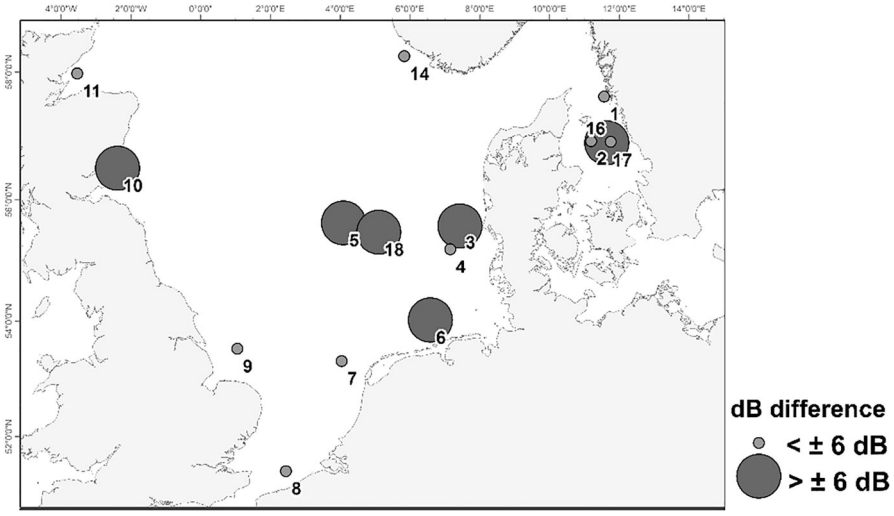



**Fig. 6** dB difference for broadband frequency (20–20,000 Hz) comparing P50 yearly measurement and model data. Possible reasons for uncertainty at individual stations: 3; proximity to an oil rig, 5; generator at platform, 6: offshore wind farm noise, 10; non-AIS fishing vessels, 17; sediment uncertainty in the Kattegat, 18; seismic survey present and proximity to oil rig

## Conclusions

Jomopans was a very successful project that quantified the continuous noise in the North Sea. These results contributed to the development of an assessment framework for continuous noise in the European Union. It was shown that transnational co-operation is the key to success.

**Acknowledgments** This project (Joint Monitoring Program for Ambient Noise in the North Sea, Jomopans) was partly funded by the North Sea Region of the European Regional Development Fund (InterReg) of the European Union. Also, thanks to all the partner institutes of the Jomopans consortium and their associated funding agencies that made this project possible.

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# Section XIV

# Monitoring

# Automated Monitoring Technologies for Real-Time Marine Mammal Detection

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Carolyn M. Binder, Maj Dugald Thomson, and Craig Reesor

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**Abstract**

Active sonar training and testing is necessary to ensure well-trained, combat-ready maritime forces; however, in-water transmission of sonar has the potential to cause harm to marine mammals. Timely information on marine mammal presence in, or near, sonar training areas can be used to minimize the risk of harming these animals. Defence Research and Development Canada (DRDC) has investigated a variety of acoustic technologies to facilitate real-time marine mammal monitoring. A combination of fixed and autonomous systems was considered. Fixed automated monitoring capabilities included a moored surface buoy with a volumetric hydrophone and a network of acoustic volumetric and vertical line arrays. Mobile passive acoustic monitoring (PAM) systems included an underwater glider with a single hydrophone and an uncrewed surface vehicle with a towed line array. Each system is capable of onboard real-time detection and species-level classification, which generates an alert to be telemetered to a shore-based station. Alerts may be used to alter when or where active sonar training and

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testing occurs. Advantages and disadvantages of each type of passive monitoring system considered were weighed to provide recommendations on the effectiveness of each system and to assess which type of system is best suited for a particular monitoring requirement.

**Keywords**

Marine mammal monitoring · Automated detection · Classification · Passive acoustic monitoring

## Introduction

The Canadian Armed Forces (CAF) active sonar operations are critical to the defense of Canada; active sonar is often the only practical way to detect modern submerged targets. Realistic active sonar exercises must be conducted to ensure well-trained, combat-ready maritime forces and ensure operators are proficient with the required skills; however, active sonar can negatively impact marine mammals. Noise-related impacts range along a spectrum from echolocation masking and behavioral disturbance to permanent and temporary hearing damage to death or population-level consequences depending on the frequency of the signal, source level, duration of exposure, habituation, and contextual factors (Frantzis 1998; Southall et al. 2016).

During sonar operations, the CAF implements layered mitigations to reduce harm to marine mammals. The most effective measure to reduce the potential of harm is to confirm that there are no marine mammals in the area around the source where the sound levels remain above the threshold that is likely to cause hearing damage. Visual and passive acoustic monitoring for marine mammal presence during a period before the sonar exercise and throughout the sonar exercise is conducted to ensure there are no marine mammals in this area; however, these methods currently rely on operators onboard the ship or aircraft who must conduct the monitoring in addition to their primary duties. A further disadvantage of relying completely on exercise-based monitoring is that when a marine mammal is sighted, a decision must be made either to wait until the animal leaves the exercise area or to reposition the exercise to a new location and re-start the monitoring period. This reduces the time available to conduct the event and detracts from the primary objective by introducing procedures which are not intended to be used in real-world operations.

The focus of Defence Research and Development Canada's (DRDC) marine mammal mitigation project is to refocus these risk mitigation measures from operator-centric to planning-centric. In part, this is being accomplished by employing acoustic monitoring technologies in typical sonar training areas to gain better insight and understanding to how marine mammals use these areas, such that sonar exercises are planned in the right places at the right times (i.e., when/where marine mammals are less likely to be impacted). With improved information on typical marine mammal habitat usage, exercise planners and sonar operators can make decisions on exercise location and timing based on best available information.

Furthermore, timely information on marine mammal presence in sonar training areas during exercises can be used to minimize the risk of harming these animals while maximizing training opportunities, requiring real-time in situ automated detection and classification of marine mammal calls with detection alerts passed to decision-makers.

DRDC investigated four specific passive acoustic monitoring (PAM) technologies to facilitate real-time marine mammal monitoring, which included a combination of fixed and mobile systems. The ability to efficiently deliver detection results to decision-makers was a critical capability that was assessed. Advantages and disadvantages of each type of PAM system were weighed to provide recommendations on the effectiveness of each system and to assess the type of system that is best suited for a particular monitoring requirement. Additional capabilities, such as archival/long-duration recording (to obtain information on spatiotemporal habitat usage) and directional recording (to assist in localization), were also considered in the system evaluation.

## Monitoring Technologies Assessed

### Fixed Systems

Fixed monitoring systems have several advantages, including that they are:

- Relatively quiet, with mooring designs refined over decades to reduce mooring and flow noise around the sensors.
- Often easily deployed and recovered from ocean-bottom moorings.
- Cost-effective and persistent, because they can be left in place independent of a human operator in extreme or remote locations for relatively long periods of time (Sousa-Lima et al. 2013), on the order of weeks to months.

The main disadvantage of fixed systems is that they can only provide monitoring coverage within a relatively small distance surrounding the system, typically referred to as the detection range. The detection range is often seasonally variable and dependent on such factors as the sound speed profile and the local ambient noise field.

#### SmartBuoy

As part of an increasing effort for DND to establish a baseline for marine mammal presence in active sonar training and testing areas (Thomson and Binder 2021), DRDC is in the process of evaluating a JASCO-developed acoustic system for longer-term real-time and archival marine mammal monitoring. The SmartBuoy is a moored buoy with a large surface expression (see Fig. 1) and four-element volumetric array at 60 m depth. A 15 m length of electromechanical stretch hose from EOM Offshore LLC (EOM Offshore LLC 2023) isolates the subsurface array from movement of the surface buoy to reduce mooring noise in the recordings.



**Fig. 1** Example of a fixed real-time monitoring system. DRDC's SmartBuoy is anchored to the seafloor with a large buoy surface expression that allows detection alerts to be transmitted to a shore station in near real time

Onboard detectors/classifiers for nine whale species provide a real-time detection capability. A satellite communications (SATCOM) package transmits detection alerts and species classification estimates from the JASCO OceanObserver acoustics package. The large surface expression allows for the installation of solar panels which can provide continuous and stable power to achieve up to 1 year endurance with no duty-cycling of the acoustic recordings (sample rate of 30 kHz).

Deployment locations for evaluating these systems were carefully selected with locations prioritized for sites where active sonar testing is likely to occur and marine mammal monitoring has rarely been conducted. These systems will be used to generate real-time alerts to provide situational awareness of marine mammal presence before/during a sonar activity; additionally, these systems provide a baseline of marine mammal habitat usage in the absence of sonar to compare to periods of sonar activity to assess if there is any resulting modification in habitat usage. The first SmartBuoy was deployed in a Royal Canadian Navy (RCN) active sonar training area in the Pacific Ocean in September 2021. A second SmartBuoy was deployed in

an offshore RCN active sonar training area in the Atlantic Ocean in December 2022; unfortunately, the Atlantic SmartBuoy was lost at sea approximately a week after deployment due to the passage of a large storm system with offshore winds. Both deployment locations are in approximately 1500 m water depth.

Archival data was recovered from the first deployment in October 2022; acoustic data quality assurance and preliminary analysis is ongoing. Initial results show concurrent recordings of whale vocalizations and active sonar signals; these recordings combined with baseline ambient noise and marine mammal recordings will be used to assess the impacts of active sonar training activities in the RCN's Pacific active sonar training area. Multi-year data sets collected at fixed locations will allow continuous monitoring of potential impacts of active sonar on marine mammals that reside or transit through these areas. Additionally, these multi-year data sets will be used to develop and refine marine mammal habitat models for these sonar training areas. These habitat models will be used when planning sonar training events to select locations and times of year when marine mammals are least likely to be present in the area, thus reducing the risk of harm.

### Distributed Underwater Sensor Network (DUSN)

The Distributed Underwater Sensor Network (DUSN) is a DRDC-developed system, designed primarily for underwater acoustic surveillance, and thus is optimized for underwater vehicle detection. This network of sensors is made up of individual DUSN nodes that each contain a seven-element planar array and optional nine-element vertical line array (VLA) sampled at 30 kHz; an example node without the VLA is shown in Fig. 2, with the hydrophones of the planar array visible in their white protective casing on the top of the DUSN node. Each DUSN node can run real-time automated detectors onboard the node and perform bearing estimates for each detection. Multiple DUSN nodes can be used for cooperative localization and tracking by sharing detection and bearing estimate information via underwater acoustic modems. Detection and localization results can be transmitted back to a shore station via a cabled connection or surface gateway buoy.

An example of bearing estimate results for data recorded by the DUSN directional array is shown in Fig. 3. This data was recorded during a trial that took place in Bedford Basin, NS, in October 2021. Bedford Basin is a busy harbor with high levels of pleasure craft and commercial shipping activity. During this trial, North Atlantic right whale upcall recordings were retransmitted from a fixed location to assess the utility of the DUSNs for marine mammal monitoring. There are two distinct signal sources present in Fig. 3, the transmitted North Atlantic right whale upcalls from the north-easterly direction (blue) and the continuous tonals generated by a ship from the south-easterly direction (pink).

The example bearing results in Fig. 3 demonstrate how directional data may assist detection of marine mammal calls in noisy data. A display like this one may be used for real-time PAM to help visually separate signals arriving from different directions, making it easier to identify low signal-to-noise ratio (SNR) whale calls. Additionally, detection may be done on beam data; that is, one can step through bearing angles and run a detector on sound arriving solely from a limited range of bearings, thereby



**Fig. 2** Example of a fixed real-time monitoring system under development by DRDC. The image shows one DUSN node with a volumetric hydrophone array (hydrophones in white protective casing on the top part of the DUSN)

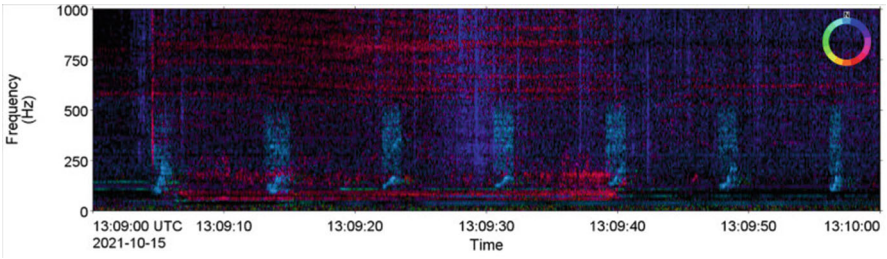



**Fig. 3** Example bearing estimate results from a trial in a busy harbor, with color representing the estimated direction from which the signal was received. The color wheel in the top right corner indicates what each color represents, with blue colors representing sounds coming from the north

rejecting noise or other signals arriving from different bearings. Using the beam data to make detections is a very useful method for improving the detection of low SNR signals, as it reduces directional noise originating from other bearings.

Though originally designed for underwater acoustic surveillance, DUSN is a capable multipurpose system that has proven to be effective for real-time marine mammal monitoring for deployments lasting up to a month. DUSNs have been tested for Southern Resident killer whale and North Atlantic right whale monitoring. Initial results show that the array design is best suited to providing bearing estimates for baleen whales; however, odontocete whistles and the low-frequency portion of some odontocete clicks can be detected within the current capabilities of this system.

Given the DUSN's strengths (real-time detection and localization) and deployment duration limitations (see Table 1), the DUSN system is well-suited for real-time monitoring during noise-generating activities, particularly in constrained waterways. For example, DUSN could be used during active sonar experiments in a bay or narrow straits. In this scenario, a "trip wire" of DUSN nodes may be set up a few kilometers away from the experimental site. If any whales are detected crossing the "trip wire" and moving toward the experimental site, the sonar experimentation would be paused until the whales leave the area where acoustic exposure is likely to cause hearing damage.

## Mobile Systems

Autonomous mobile system technology has advanced rapidly over the past two decades. Mobile systems are now capable of collecting high-quality acoustic data at relatively low cost and use onboard detection and classification algorithms to provide important data on marine mammal presence in real time (Baumgartner and Mussoline 2011; Kowarski et al. 2020). The main advantages of this type of system are a result of their mobility; they are:

- Often very easy to deploy from near-shore locations and can be programmed to transit to an offshore location to conduct their monitoring task.
- Capable of following a pre-programmed track after arriving on station. This track can be updated via satellite communications.
- Able to cover a fairly large area over the course of several days, making these systems effective tools to provide wide-area monitoring coverage.

**Table 1** Comparison of technologies. The * denotes that a capability is not currently implemented, but could be in future versions. The † highlights that ongoing work is actively being conducted to increase deployment duration

| Technology | Fixed vs. mobile | Rapidly deployable | Deployment duration | Sample rate | Directional recording | PAMLab detectors |
|---|---|---|---|---|---|---|
| **SmartBuoy** | Fixed | No | 1 year | 32 kHz | Yes | Yes |
| **DUSN** | Fixed | Yes | 1 month† | 30 kHz | Yes | No* |
| **Glider** | Mobile | Yes | 2 weeks† | 16/512 kHz | No* | Yes |
| **USV** | Mobile | Yes | 4–16 days† | 32/512 kHz | Yes | Yes |

- Capable of transmitting their position, detection/classification, and other relevant summary statistics via SATCOM because they spend at least some portion of their time at the surface.

The primary disadvantage of mobile systems is their relatively short deployment duration relative to fixed systems. The motion of the platform also results in ambient noise in the acoustic recordings which may mask low SNR whale calls. Human operators also need to monitor the mobile systems to ensure that they are staying on track and to update the track if necessary. Local currents can affect mobile system navigation and must be taken into account when planning a mission, as strong currents have the ability to "blow" mobile platforms off track.

### Glider

A Teledyne Webb Research Slocum G1 glider with integrated acoustics package (see Fig. 4) was tested during the international antisubmarine (ASW) exercise CUTLASS FURY 21 (CF21). CF21 took place in the offshore waters south of Nova Scotia in September 2021, as shown in Fig. 5. The intent of the demonstration was to monitor for the presence of marine mammals in the week preceding CF21, during CF21, and for a few days after the exercise had concluded. Acoustic recordings and real-time detection/classifications were conducted with the JASCO OceanObserver (Kowarski et al. 2020) using a Geospectrum M36 hydrophone mounted on the top of the glider in protective housing. Acoustic data were sampled simultaneously at 16 and 512 kHz sampling rate from the same hydrophone.

Initial glider detection/classification is done onboard the glider using the contour detector described in Kowarski et al. (2020). Candidate detections were transmitted via satellite communications and were verified by analysts during CF21. An example candidate detection snapshot is provided in Fig. 6, showing confirmed delphinid whistles (unidentified species) and possible fin whale calls. Candidate detections are transformed from a typical spectrogram to these contours (Kowarski et al. 2020) to save transmission bandwidth, relative to sending a full spectrogram. They contain



**Fig. 4** Example of a mobile uncrewed underwater vehicle (UUV) real-time monitoring system. This Teledyne Webb Research Slocum G1 glider has a hydrophone payload (in black protective casing on the top of the glider) and integrated JASCO OceanObserver acoustics package for real-time recording, signal detection, and marine mammal species-level classification

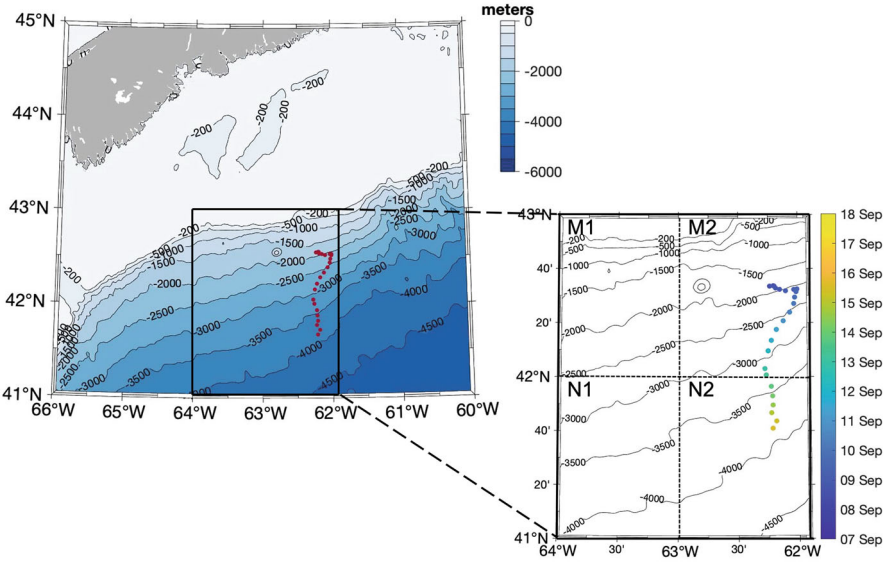



**Fig. 5** The surfacing locations of a glider used for real-time monitoring of marine mammal presence during an ASW training exercise (CUTLASS FURY 21). The zoomed-in map (bottom panel) is divided into squares representing the designated RCN operating areas

sufficient information for a trained analyst to discriminate true detections from false detections and confirm marine mammal species, to varying levels of certainty. Daily detection summary reports of the analyst-confirmed detections were passed to participating ships' commanding officers to provide situational awareness on the presence of marine mammals close to the exercise area.

Post-trial manual analysis of the glider's acoustic data verified recordings of dolphins, pilot whales, fin whales, Cuvier's beaked whales, sperm whales, kogia or porpoise species (high-frequency clicks), and beaked whales (either True's or Gervais'). Annotations made during the post-trial analysis will be used to refine the automated real-time detectors for future deployments.

## Uncrewed Surface Vehicle (USV)

DRDC assessed an Open Ocean Robotics (OOR) uncrewed surface vehicle (USV) with towed array and onboard acoustic processing (JASCO OceanObserver) during a demonstration conducted at the Canadian Forces Maritime Experimental and Test Ranges (CFMETR) in Nanoose Bay, BC, during March 2021. The Data Xplorer USV (Fig. 7) is 3.35 m long, is remotely controlled, and contains batteries and photovoltaics to provide propulsion. It was designed for offshore use, with a self-righting capability that allows it to function in high sea states; on the other hand, its high degree of controllability and shallow draft allow it to effectively operate in shallow and congested coastal waters. The USV is equipped with a configurable payload of oceanographic, environmental, and communications sensors. Standard sensors include a 360° camera, automatic identification system (AIS) receiver, weather station, and

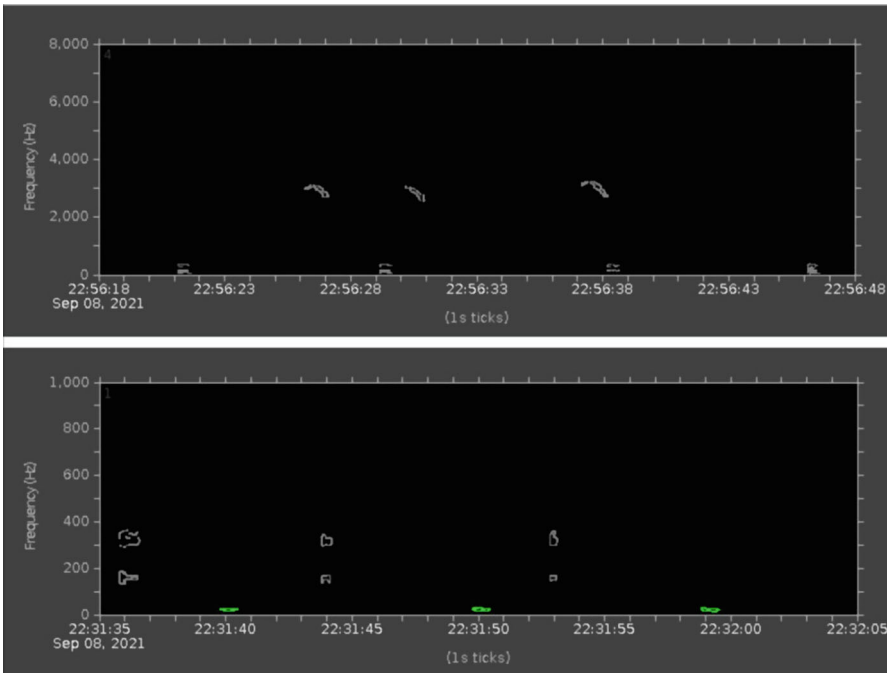



**Fig. 6** Candidate OceanObserver detections, represented as contours around automatically identified features in a spectrogram. Top panel shows confirmed delphinid whistles (unidentified species), and the bottom panel shows possible fin whale calls (highlighted in green)



**Fig. 7** Example of a mobile uncrewed surface vehicle (USV) real-time monitoring system. This USV is capable of towing a thin line array of digital hydrophones and has an integrated JASCO OceanObserver for real-time recording, bearing estimation, signal detection, and marine mammal species-level classification

triducer (for measuring depth, water temperature, and speed over water). Optionally, the USV can tow an acoustic line array. Currently, there are two options for eight-element arrays that were designed for use with the Data Xplorer. Each array has non-linear element spacing to increase the useable frequency range for broadband

beamforming. One array was designed for Southern Resident killer whale monitoring (1.875 m aperture), and the other was designed for North Atlantic right whale monitoring (15 m aperture), with bearing accuracies less than 5° for high SNR (i.e., SNR > 5 dB) signals arriving from angles within ±60° from broadside; bearing accuracy degrades for angles close to the endfire and for low SNR signals.

During the CFMETR evaluation, the USV was supported by a two-person vehicle deployment and recovery team located in situ and a remote team of acoustic analysts in Halifax, NS, while the vehicle was remotely piloted from an operations center in Victoria, BC. The USV towed an eight-element towed digital acoustic array (1.875 m aperture, with 40 m tow cable) designed for Southern Resident killer whale monitoring with detection, classification, and bearing estimate processing done onboard the USV using JASCO's OceanObserver PAM system. The OceanObserver acoustics package was fully integrated with the USV's power and communications modules. Detection/classification, bearing estimates, and ambient noise summaries can be transmitted via a low-bandwidth telemetry system, with the potential to transmit more elaborate data products if higher-bandwidth telemetry options (e.g., cellular network) are available. Similar to the process described for the glider, candidate detections are transmitted as contour plots (example in Fig. 6) that are verified by expert analysts at JASCO before being passed to decision-makers.

Testing of the USV system was done by transmitting simulated killer whale calls from a projector suspended from a drifting support vessel. During the evaluation, the USV maintained a speed of 1.25 kt (2.3 km/h), with hydrophone array at 18–20 m tow depth. Detections were automatically reported and could be viewed in (near) real time. Following the 2-day experiment, a live demonstration was conducted with the USV patrolling the waters in Nanoose Bay for marine mammals. During the live demonstration period, humpback whale call detections were reported and validated in real time.

Solar- and battery-powered USVs have limited endurance and speed, especially when towing acoustic arrays at times of the year when days are short and solar power is less effective. In addition, it is desirable to lower the vehicle speed to reduce the system self-noise (from the vehicle, towline, and array) and to make the system as power efficient as possible. At-sea trials showed that 1.5 knots (2.8 km/h) is the optimal USV speed when towing the acoustic array to achieve low flow/self-noise and maintain the array at an appropriate depth for monitoring. Given the data collected during the sea trials, estimates of deployment duration for the current USV range from 4 to 16 days in mid-latitudes, depending on seasonal solar irradiance, local sea state, wind, and currents. Efforts are ongoing to increase the deployment duration of the USV while towing an acoustic array.

## Comparison of Monitoring Technologies

Table 1 presents a comparison of the capabilities of each technology assessed. Each system with JASCO's PAMLab detectors is capable of onboard real-time detection and species-level classification (Kowarski et al. 2020), which generate alerts telemetered to ship- or shore-based command centers. The intent of these alerts is to provide

situational awareness on marine mammal presence and may be used to alter when or where active sonar training and testing occurs. DUSNs have embedded real-time detectors that have not yet been tested with marine mammal calls; however, PAMLab has been successfully used in post-trial processing of DUSN recordings for automated detection.

In general, fixed systems are best suited to longer-term monitoring scenarios at a single location. The SmartBuoy is a large system that is not rapidly deployable; however, this disadvantage is offset by its long deployment duration, providing baseline information on how marine mammals use an area over all seasons, and can be used to assess behavioral impacts of active sonar training. The DUSN systems on the other hand are small, rapidly deployable systems. They do not provide the same baseline data as the SmartBuoy; however, they could be effectively employed to provide a marine mammal monitoring "trip wire" in a bay or other area constrained by the local land mass.

The mobility of the glider and USV systems provides an advantage for larger area monitoring, though their deployment durations are shorter than for fixed systems. These remotely piloted systems are easy to deploy and control from ship- or shore-based stations and provide real-time detection/classification alerts and bearing estimates (in the case of the USV with towed array). Multiple mobile systems can be used to cover a larger area. These types of systems are best suited to conducting real-time monitoring before and during a sonar exercise to provide situational awareness on marine mammal presence to sonar operators and exercise decision-makers.

In practice, no single system is capable of meeting all the marine mammal monitoring needs of the CAF. These systems should be used in combination; for example, the SmartBuoy may be deployed in a typical sonar operating area to collect data on typical marine mammal presence. Sonar exercise planners could use this information to decide on the best time of year to conduct a sonar exercise (i.e., when marine mammals are least likely to be present). Two weeks prior to conducting the sonar exercise, a glider, USV, or combination of gliders and USVs could be deployed to the planned exercise area to monitor for marine mammals, sending back real-time detection alerts. If a high density of animals (or a sensitive and/or at-risk species) is noted in the area, a decision can be rendered to move the exercise to an alternate area. Continued monitoring with the mobile systems would take place throughout the exercise to alert exercise participants when marine mammals approach the exercise area.

## Conclusions

A review of monitoring technologies assessed by DRDC was presented. DRDC's SmartBuoy is good for area monitoring on a long timescale; it can provide baseline data on habitat usage and real-time detection capability for exercises in a localized area. In general, the DUSN network was found to be good for area monitoring on a relatively short timescale. The DUSN systems are the only technology assessed that does not have JASCO's PAMLab detectors integrated; instead, the nodes currently

only have band-limited energy detectors. Further, DUSNs are best suited to lower-frequency band detection (mysticetes) and are not capable of detecting many odontocete clicks. In contrast, the glider and USV are both good for area clearance before and during exercises. The USV has the potential for longer endurance and greater control in strong currents, but the glider is better suited to detecting highly directional beaked whale clicks since the glider has deep-diving capability.

DRDC continues to work with partners to improve acoustic monitoring technologies. Future improvements include removing the requirement for human validation of all automated detections from the JASCO OceanObserver, increasing the endurance of both the glider and USV, integrating PAMLab detectors with DUSN, and performing detections on beamformed data to improve SNR.

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# Passive Acoustic Monitoring Power Analysis: A Tool for Designing an Acoustic Monitoring Program

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**Abstract**

Monitoring trends of marine mammal populations are notoriously challenging, as density and abundance estimates are frequently characterized by high levels of uncertainty. Statistical power is described as correctly detecting a trend when one is present and is influenced by the uncertainty surrounding the input parameters. The "AVADECAF" tool evaluates the power of a fixed Passive Acoustic Monitoring (PAM) array to detect changes in animal density, via simulation. The simulations can be based on existing surveys or can be used to design monitoring programs. The tool allows for density estimation via distance sampling where ranging of acoustic detections is possible, or Spatial Capture Recapture (SCR), when an animal's sounds can be detected on more than one of the acoustic sensors. The tool provides an opportunity to evaluate how the power to detect population declines might be improved by comparing alternate survey designs. Improving statistical power increases the probability of detecting a change in the population and can reduce the required duration of a monitoring program. The AVADECAF tool was applied to simulated Blainville's beaked whale clicks detected by hydrophones at the US Navy's Atlantic Undersea Test and Evaluation Center (AUTEC), and the potential future applications of this tool are discussed.

**Keywords**

Animal population density estimation · Passive acoustic monitoring · Power analysis · Distance sampling · Spatial capture recapture

## Introduction

Monitoring marine mammal populations is well established to be challenging. Animals can occupy remote regions, exist at low densities over large areas, spend much of their time subsurface, and even at the surface can be difficult to observe. They are often monitored using visual surveys; however, this is a costly process, and such surveys are weather dependent (i.e., requiring periods of good visibility and calm conditions) and restricted to daylight conditions. Due to these challenges, standard visual surveys often result in considerable uncertainty in abundance estimates (Taylor et al. 2007).

Many marine mammals produce sounds underwater as part of navigation, foraging, and communication with conspecifics, making Passive Acoustic Monitoring (PAM) a viable method for monitoring them. Monitoring equipment can be deployed at or near the seabed, making data collection less susceptible to weather conditions, and can be deployed for extended periods prior to retrieval. Therefore, this approach is widely used around the world for long-term monitoring of marine mammals.

The analysis of PAM data has been used for estimating the spatial distribution of cetacean species, species identification, research into behavior and communication, and mitigation of anthropogenic activity (Pirotta et al. 2014; Wisniewska et al. 2016; Curtis et al. 2020; Sarnocińska et al. 2020). In addition, there are now a range of methods by which cetacean density may be estimated using fixed passive acoustic devices including plot sampling, distance sampling (DS), and Spatial Capture Recapture (SCR) (Marques et al. 2009, 2012; Moretti et al. 2010; Barlow et al. 2021; Oedekoven et al. 2022).

Statistical power is the probability of correctly rejecting the null hypothesis, but could also be described as correctly detecting a trend when one is present (Thomas and Juanes 1996). While somewhat arbitrary, a value of 0.8 is considered to be an indication of a statistical test with reasonable power to detect a trend (Authier et al. 2020). In terms of population monitoring, statistical power can be described as the probability of correctly detecting a true change in animal abundance or density. Statistical power is affected by a combination of sample size, effect size and parameter variability, or sampling variance (Thomas and Juanes 1996). Effect size relates to biological significance, and in the context of population monitoring is the quantitative population trend over time, that is, a 50% decline in density over 10 years of monitoring. Considering the potential effect and sample size prior to data collection can help define the survey design and methods required to accurately assess trend and estimate the minimum detectable effect. Conducting a power analysis prior to committing to a monitoring program can help ensure that the survey design is sufficiently robust to detect a specific change in the population. Therefore, to successfully monitor long-term population trends (i.e., to determine if populations are stable, increasing, or decreasing), consideration needs to be given to both statistical power and biological significance, that is, the minimum effect size that is considered to be biologically important. This is crucial for the effective management and conservation of marine species.

Given the challenges of long-term monitoring of cetacean populations, it is unsurprising that between 1995 and 2018, only 20% of the marine mammal stock assessments published from NOAA (National Oceanic and Atmospheric Administration) fisheries were associated with population trend analyses, and even well-studied stocks required annual estimates over at least 10 years to obtain sufficient power to detect current trends (White et al. 2022). Similarly, Taylor et al. (2007) assessed power to detect trends in a range of marine mammal species and reached comparable conclusions. For example, they predicted that from the reviewed stocks, 72% of precipitous declines in large whales would not be detected given current monitoring efforts. Likewise, Moretti (2019) determined that it would require approximately 25–30 years of photo-ID data collection to detect an annual decline of 5% for Blainville's beaked whales (*Mesoplodon densirostris*) on the US Navy range "Atlantic Undersea Test and Evaluation Center" (AUTEC). The photo-ID monitoring program for Cuvier's beaked whale (*Ziphius cavirostris*) on the Southern California Acoustic Range (SOAR) is considered to be a high-intensity survey effort, however these efforts would not allow for detection of a 50% decline over the 11 years of survey effort (Curtis et al. 2020).

As the statistical power to detect a decline in marine mammals is often low, it is logical to consider how it might be increased. The original purpose of the AVADECAF project was to evaluate the utility of fixed PAM arrays for estimating cetacean density over the lifespan of "typical" oil and gas fields. Booth et al. (2017) explored the importance of detection and classification methods, hydrophone array design, and evaluated the sensitivity of the statistical power to the input parameters required to estimate annual density from PAM data. The tool was built with the capacity to estimate power given surveys from two different survey methods, distance sampling and SCR. In this overview, the methods behind the AVADECAF tool are discussed and exemplified through a simulation of distance sampling using PAM of Blainville's beaked whales at AUTEC in the Bahamas (see Booth et al. 2017 for more details of SCR approaches, sensitivity analysis for DS simulations, and the associated available R code and helpfiles). Earlier literature provides thorough explanations of distance sampling and SCR (Buckland et al. 2001, 2015; Marques et al. 2013). This chapter represents a broad overview of these methods, provides initial insight into density estimation from fixed PAM, and discusses the potential of the AVADECAF tool for guiding effective management and conservation activities.

## Methods

### Overview

The AVADECAF tool allows users to estimate statistical power to detect a trend in density using PAM – either using DS or SCR. DS with PAM data uses acoustic detections of animal cues, combined with a distance-based detection function derived from distances between the hydrophone and the cue producing animal to estimate animal density within a study area. In contrast, SCR does not rely on distance information, but requires that a proportion of the cues produced by the animals are detected by more than one hydrophone and that cues can be matched across hydrophones (Oedekoven et al. 2022). For SCR, having auxiliary information for each detected sound, namely received sound level, bearing angles or time of arrival can greatly increase performance (Borchers et al. 2015).

In either case, the tool simulates passive acoustic arrays in a hypothetical study area and detections made by the individual hydrophones. Within the tool, the DS package simulates a passive acoustic array that is suitable for applying distance sampling methods, that is, allows for localization of the detections and hence obtaining distances between the sensors and the localized detections (to generate detection functions). Within the AVADECAF tool, for DS methods it is assumed that the passive acoustic devices represent clusters of hydrophones that are capable of localization (described as p-nodes). The utility of these respective methods relates to the characteristics of the vocalizations of the species of interest.

When designing any survey (using either DS or SCR) the spacing between the PAM sensors needs to be large enough that, with a given budget, sensors can be distributed throughout the study area. For SCR, however, the spacing also needs to

be small enough that a proportion of calls can be detected by two or more detectors (e.g., Oedekoven et al. 2022). Furthermore, SCR requires that individual detections can be matched between detectors. Hence, knowledge about the approximate detection range and characteristics of the calls is necessary. DS methods require that at least for a representative proportion of the detections, distances to the detections can be obtained using the acoustic recorders from which a detection function can be estimated reliably.

Despite being a relatively recent addition to the statistical toolbox, SCR is also considered to be a "standard" abundance and density estimation approach given its genesis from mark-recapture and distance sampling methods (Borchers 2012; Borchers et al. 2015). SCR has also become a frequently used tool for estimating vocalization density or abundance (Stevenson et al. 2015). In contrast to distance sampling, SCR does not require distances to the detections for estimating the detection function or call density. It does require, however, that call detections are matched between different detectors. This "recapturing" of detected calls enables the detection probability of the sounds around the sensors to be estimated. In contrast to standard mark-recapture methods, for which abundance is strictly undefined, SCR allows the estimation of both density and abundance for a given area. Further, because the heterogeneity in detection probability is modelled as a function of the location, SCR methods deal directly with the unmodelled heterogeneity that poses a challenge to conventional CR (e.g., Link 2004). SCR has been successfully applied to both visual (Pirotta et al. 2015) and passive acoustic data (Marques et al. 2012) to estimate the abundance and density of marine mammals.

To determine the appropriate method for density estimation from PAM, it is important to identify the target species for the survey, whereby auxiliary information is required regarding the species of interest. The call frequencies, cue production rates, species behavior, and vocal detection ranges are all relevant considerations when selecting an appropriate method for density estimation using passive acoustic monitoring. This information will influence the array design, hydrophone elements, and sampling frequency. A sensitivity analysis using the original AVADECAF tool has been carried out (Booth et al. 2017) to determine the importance of each of these key input variables, including both the survey design and species-specific parameters.

This paper focuses on Distance Sampling and includes a brief overview of parameter sensitivity, however the original report (Booth et al. 2017) contains a comprehensive discussion of SCR methods and further results and outputs.

## Implementation of Models in the AVADECAF Tool

The simulation workflow within the AVADECAF tool includes the creation of a study region, the generation of a survey design (the number of localizing hydrophones and non-localizing hydrophones), and the simulation of the species of interest derived from population-specific parameters (Fig. 1). The species information is incorporated to generate one animal density surface throughout the study area and a buffer surrounding it per simulation. The parameters defining the survey

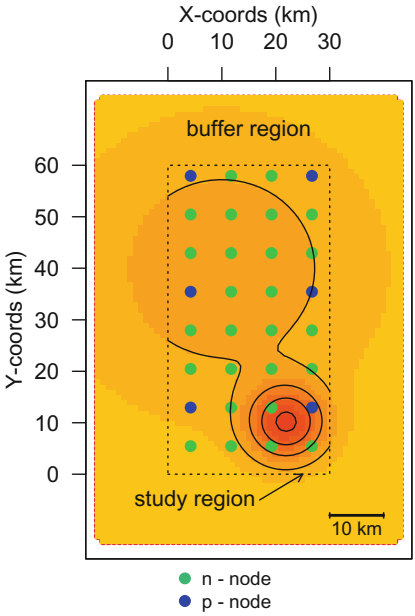



**Fig. 1** A visual representation of how the AVADECAF tool simulates the survey region, hydrophone array, and animal density. The black outline denotes the study area, the red/yellow surface represents a non-uniform animal density surface, green circles indicate a single hydrophone, and blue circles represent a cluster of localizing hydrophones

design are used to generate an array of point transects distributed throughout the study area using a systematic design with a random starting point. Combining the density surface and the parameters describing the vocalization production rate of the animals, vocalizations are generated throughout the study area.

In DS, these vocalizations are then detected by each of the sensors according to a user-defined detection function. This can be either a hazard-rate or half normal detection function, with its parameters describing the decay in detection probability with increasing distance from the detector, and a probability of detection at distance zero. In the distance sampling package, distances from the sensors to the detections are provided; the proportion of these detections depends on the number of localizing sensors that are deployed, and the number of detections that are available (see Booth et al. 2017 for presentation of SCR methods).

The subsequent step requires estimation of animal density using the "observed" data, which is described in more detail below. These steps are undertaken for each survey year and season specified by the user, resulting in a time series of animal densities and associated density estimates. Lastly, a generalized linear model is fit to the estimated animal densities, to determine the power to detect a change. The AVADECAF simulation produces estimates of the power to detect a specified decline at a user-defined prespecified level (of significance, e.g., alpha = 0.1, 0.05 or 0.01). A simulation includes 1000 iterations (or otherwise specified), and the percentage of iterations with p-values below the chosen level of significance estimates the power to detect the specified trend (e.g., a 1% annual decline over 50 years of monitoring).

## Density Estimation Equation (Distance Sampling)

An overall equation for estimating animal density from PAM using distance sampling is as follows

$$D_{animals} = \frac{n_c(1-f)}{k\pi w^2 pTca_{perc}}$$

where a total number of cues are observed ($n_c$) and detected on $k$ hydrophones over a recording time $T$. A proportion ($f$) of the detected cues may be false positives, meaning that the proportion of correctly detected cues is given by 1-$f$. As it is difficult to identify individual animals using acoustic cues, the above estimator includes a "cue production rate" $c$, representing the average number of cues produced per animal per unit time. Estimating animal density requires information on the effective survey area by the $k$ sensors, where $\pi w^2$ represents the area monitored surrounding each sensor. This value is then multiplied by $p$, the average probability of detecting a cue within this region. The directionality of animal cues, combined with the depth and orientation of an individual, can influence the probability of detection by an acoustic sensor, as such a bias parameter ($a_{perc}$) is included to account for the proportion of missed detections. Usually in visual surveys, $g(0) \neq 1$ (i.e., incomplete detection at distance 0 from the sensor, survey platform or trackline) is caused by availability bias. However, in the case of PAM, there are instances where the animal is vocalizing, that is, is technically available to be detected but for the reasons described, the animal is not detected at the sensor. Here, this might be considered a perception bias issue, causing g(0) issues. As such the $a_{perc}$ parameter in this tool represents the proportion of cues that are missed directly over the sensor (i.e., at distance 0). The perception bias here may arise due to the directionality of an animal's beam profile.

To demonstrate the application of density estimation via DS, a case study for Blainville's beaked whales is presented below. Given vocalizations are medium-high in frequency, very directional, and they are often not detected on multiple bottom mounted hydrophones, the general use of SCR is precluded. In this context, SCR might be more suited to omnidirectional sources. SCR approaches are not discussed further here (see Booth et al. 2017 and associated documents for more details).

## Case Study: Blainville's Beaked Whale

As a deep-diving, visually elusive species group that uses echolocation, beaked whale populations highlight the utility of acoustic monitoring for marine mammals. Conservation concern stems from stranding events of beaked whales that have been reported globally (D'Amico et al. 2009; Filadelfo et al. 2009) and have been associated with exposure to naval sonar use (Bernaldo de Quirós et al. 2019; Simonis et al. 2020). However, there are multiple beaked whale populations that are consistently exposed to sonar on navy training ranges, where local abundance is believed to be stable, and strandings have occurred apparently unrelated to sonar exercises

(Curtis et al. 2020; Filadelfo et al. 2009). The uncertainty surrounding the effects of sonar on marine mammals has led to long-term monitoring efforts on US Navy ranges to determine the potential short- and long-term impacts at an individual and population level (Falcone et al. 2009, 2017; Schorr et al. 2017; Jacobson et al. 2022).

The AVADECAF example provided considers distance sampling and is based on input parameters from the Blainville's beaked whale population at AUTEC (Table 1). However, not all the required metrics were available for this specific group of animals, and therefore some of the data were sourced from other Blainville's beaked whale populations. Hence, this simulation provides an illustrative example of the utility of AVADECAF, rather than a true estimation of the power to detect a decline in density from Blainville's beaked whales at AUTEC.

Developing a range of scenarios allows for a better understanding of the potential power to detect a decline. Here, this pertains to increasing the number of monitoring years and simulating a variety of survey designs. In data-rich situations, it may be possible to increase the number of density estimates made per year, improving the power to detect a decline and reducing the number of monitoring years that are required to detect a population level change. In this scenario, doubling the number of estimates per year resulted in a reduction of approximately 7 monitoring years when the population was experiencing 1% annual decline (Fig. 2), on the assumption that these estimates are independent. While this example demonstrates the change in statistical power when the number of density estimates is increased to two per year, providing there is sufficient information regarding detections, and seasonal changes in animal presence and cues, the number of density estimates per year could be increased further, with expected increases in power. However, caution is advised in the implementation of this approach, as density estimates must be independent. Consequently, there will be a practical limit in how many estimates can be derived within a single year.

In addition to varying the number of estimates per year, the model allows for specification of the number of localizing hydrophones within the array. These p-nodes are more expensive to implement (as they require multiple hydrophones per node) but, given multiple p-nodes are implemented throughout the study area, they allow for multiple detection functions to be derived, which can address

**Table 1** The parameters for the simulations shown in Figs. 2 and 3, representing only a subset of the inputs required for the AVADECAF tool

| Parameters | Value | Reference |
|---|---|---|
| Array dimensions | 1500km$^2$ | Matsumoto et al. (2013) |
| Hydrophones | 82 | Marques et al. (2009) |
| Initial population density estimate | 25.3/1000 km$^2$ | Marques et al. (2009) |
| Cue rate (clicks per second) (cv) | 0.407 (0.098) | Marques et al. (2009) |
| False positive rate (cv) | 0.549 (0.0199) | Marques et al. (2009) |
| Perception bias (cv) | 0.07 (0.1) | As used in simulations from Booth et al. (2017) |
| Density estimates per year | 1–2 | Not applicable |
| Annual decline | 1–3% | Not applicable |
| Number of localizing hydrophones (P-nodes) | 3–8 | Not applicable |

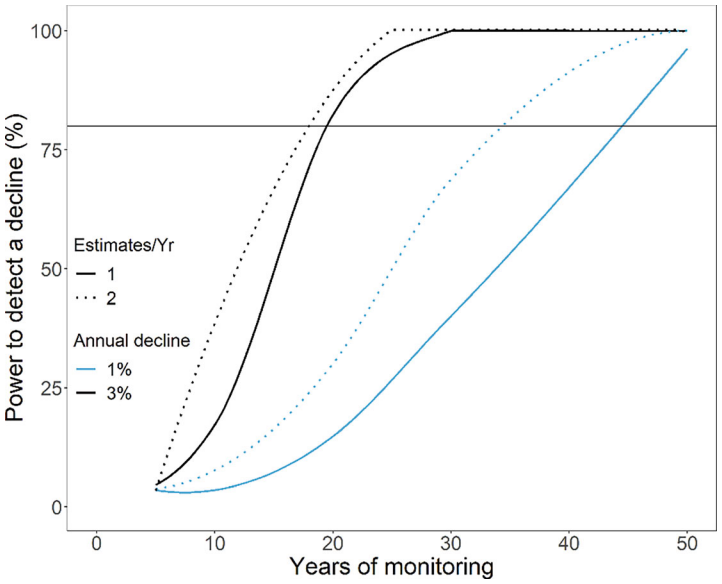



**Fig. 2** The power to detect a decline in Blainville's beaked whale based on input values derived from AUTEC. Simulations based on 1 or 2 density estimates per year, and 1% or 3% annual declines. This simulation was based on an average of 5 localizing hydrophones within the array and alpha $< 0.05$

heterogeneity in the detection probability across the study area. Increasing the number of localizing hydrophones on the array allowed for the threshold for high power (0.8) to be reached 5 years earlier, when the population declined 1% per year (Fig. 3). The reduction in the required number of monitoring years to detect a given trend will differ depending on the input parameters, but also depending on the heterogeneity of the environment. The required monitoring time will be species- and region-specific, and the proportion of localizing hydrophones and the number of estimates per year are illustrative. The impact on power for several other parameters could also be evaluated within the simulations, for example, the influence of cue rate, false positive rates, and how environmental heterogeneity may impact power by increasing the variability of the detection probability. For an example of these explorations, see Booth et al. (2017). Monitoring scenarios that are extended over a longer time series allow us to estimate the minimum number of survey years required to detect a specific decline, and the level of decline that is possible to detect, given the survey effort (noting the caveats and assumptions highlighted above).

## Discussion

In this chapter, the relevance of the AVADECAF tool is discussed in relation to estimating density using PAM. It is critical to have a PAM array and supporting auxiliary information with high precision and ideally unbiased estimates (though

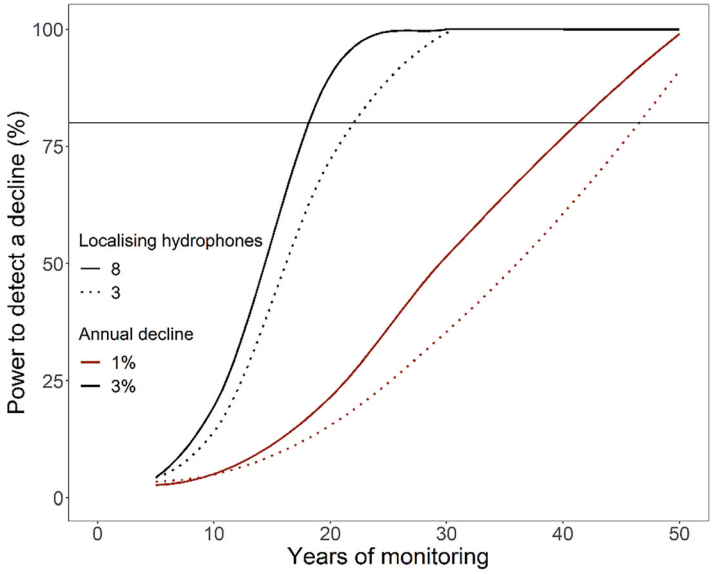



**Fig. 3** The power to detect a decline in Blainville's beaked whale based on input values derived from AUTEC, including a comparison of statistical power between arrays with different numbers of localizing hydrophones. This scenario was based on one density estimate per year, while varying the number of localizing hydrophones and the intensity of the annual decline, alpha $< 0.05$

these are practically hard to achieve). The various sources of error or bias and their impact on density estimates are described more fully in Booth et al. (2017). Key methodological elements (e.g., cue rates, detection and classification capabilities, survey design) are discussed below; these components influence density estimates, but also represent practical considerations that can help improve the utility of PAM for density estimation.

## Cue Production Rate

Booth et al. (2017) outlined the importance of cue production rate for PAM density estimation, where improved precision (and, as with all parameters, having an unbiased estimate of) the cue production rate drastically improved the power to detect a decline. It's important to note that cue production rate is not only species-specific but may vary with region/population, behavioral state, season, age, and sex, and as a result of ambient noise levels (Warren et al. 2017; Thode et al. 2020; Blackwell et al. 2021). As such, integrating cue production rate data that is region-specific and temporally aligned with the acoustic datasets is imperative to ensure an accurate representation of local density over time (e.g., Marques et al. 2013) and is equally important to evaluate the power to detect a decline.

## False Positives

The false positive rate (and variance around it) is another important input for density estimation, representative of detections that have been wrongly associated with the species of interest.

False positive rate is both species-specific and linked to the detection and classification system that is being used to process the acoustic cues; it is also spatially and temporally sensitive as other local sound sources can result in false positive detections. To identify infrequently detected species, the detector/classifier may be programmed to be sensitive (i.e., to not miss many detections), however this will likely lead to more false positives. Where the application is to use PAM for density estimation, there is some tolerance around higher false positive rates (Booth et al. 2017), providing the variance around the estimate is low (i.e., the false positive rate is precise – and unbiased) (Marques et al. 2009).

Booth et al. (2017) determined that when the false positive rate detections were imprecise (i.e., the variance around the estimate of the false positive rate was high), the power to detect a decline was considerably reduced. False positive rate is typically measured by manually checking a sample of the acoustic recording (on the assumption that an automated detector/classifier is used initially). The more often such validation is carried out, the higher the precision on the estimate of false positive rate will be. As such this is under the control of the analyst, so it is feasible to reduce the variance on the estimate.

Another practical challenge is that species with similar call frequencies and patterns co-exist in space and time – which can make detection and classification challenging. For example minke whale boing classifiers in Hawaiian waters have been found to be sensitive to humpback whale song (Ou et al. 2012), while humpback whales and bowhead whales also overlap in at least part of their vocal range (De Vreese et al. 2018). However, as long as the precision is understood and kept low, a poor false positive rate in isolation contributes very little to the final variance (sensu Marques et al. 2009).

## Survey Design

Designing a survey requires careful consideration of logistics and budget available as this directly relates to the ability to detect and estimate ranges to animals (to generate detection functions), a key component of density estimation via distance sampling. In practical terms there will be error around the estimates of the range of animals from the sensor (see Booth et al. 2017 for further details). In AVADECAF, the user is able to specify if the range of animals from the sensor (i.e., the localization of the animal) occurs accurately (which is simplistic) or with error. Error in range estimation increases with distance between hydrophones – therefore considering the species of interest and an appropriate spacing for hydrophones will help improve the estimation of components needed for PAM density estimation. As the frequency and

source level of vocalizations influence propagation distance, the space between hydrophones on the array should be species-specific. Greater distances between hydrophones increase the magnitude of the acoustic time delay of multiple detections of the same sound, which reduces the time delay error. This suggests that greater aperture size (the distance between hydrophones) would be optimal for a localizing hydrophone array. However, smaller distances between an animal and a hydrophone will increase the probability of detection. This is especially true for highly directional vocalizations. Therefore, selecting the array dimensions requires consideration of the species of interest, and an evaluation of these trade-offs.

The number of localizing hydrophones depends on the study area and corresponding array dimensions, but also on the underlying density of animals (unknown), environmental conditions, and local bathymetry. In a larger survey area, there may be more variation in habitat conditions, which will influence the detection function for acoustic cues. In this case, it would be preferable to increase the number of localizing hydrophones, to support the estimation of site-specific detection functions, or at the very least detection functions dependent on site-specific covariates, for example, depth. Increasing the number of localizing hydrophones allows us to account for heterogeneity in the environment, which may improve the precision of the detection function model. However, estimating cetacean density using passive acoustic monitoring can be a costly process if the equipment is not already deployed. Booth et al. (2017) evaluated the potential costs of the acoustic technology required for an array unit capable of localization, estimating ~$20-30 k per unit, extending this array to cover a large survey area would magnify these costs, with the distance between units (and therefore the number of units required) depending on the species of interest and their respective vocalization characteristics. Conversely, initial investment in passive acoustic technology can support long-term monitoring efforts, with acoustic devices being redeployed over several years.

## Applications

The AVADECAF simulation tool presented here provides an opportunity to explore the statistical power of a monitoring program for PAM density estimation before deployment of equipment. For existing hydrophone arrays, it provides a means to assess the utility and scope of monitoring. To our knowledge this tool has been infrequently applied since its development, possibly due to lack of awareness and difficulty accessing the software. However, there is a need for PAM survey designs associated with the fast-growing offshore wind sector where significant uncertainty around occurrence and response of marine mammals to construction noise remains. Baseline density and long-term monitoring at offshore wind farms could be explored using the AVADECAF tool, to determine the potential power of an array to detect cetacean density changes considered to be the result of windfarm development. Booth et al. (2017) stressed the importance of precisely estimating cue production rates (following Marques et al. 2009). Ongoing efforts are improving our knowledge of the factors driving cue rates and their variability for a variety of marine mammals.

As such it is expected that within the next few years these outputs will be highly relevant to the AVADECAF tool by facilitating additional cue production rate parameters, allowing for more realistic simulations.

The potential applications of the AVADECAF tool are numerous, including opportunities to evaluate existing passive acoustic monitoring programs. Distance sampling-based density estimation from fixed PAM has been minimally explored for marine mammals, however these methods have been applied to North Pacific right whales, fin whales, and bowhead whales (Marques et al. 2011; Harris et al. 2013; Oedekoven et al. 2022). Navy training ranges represent regions where cetacean density estimation using PAM is possible using their cabled bottom-mounted hydrophones. In other locations, developing an array for the purpose of cetacean density estimation is also possible, however Booth et al. (2017) suggest that deploying a small-scale acoustic array as a trial would facilitate a better understanding of the required dimensions for detection and localization for a range of species. The results of a carefully designed trial could directly contribute to the planning of acoustic arrays for the purpose of density estimation in the future.

The US Navy has already invested heavily in PAM to evaluate the impacts of sonar on marine mammals at both the individual and the population level. As the current literature demonstrates low power to detect declines from marine mammal visual surveys, statistical power of PAM surveys should also be evaluated to better understand the underlying trends that may be missed. Evaluating statistical power should be a component of designing a monitoring program, facilitating the detection of less dramatic declines, and allowing for earlier management of struggling populations. While a single data source monitoring method may not provide sufficient power to detect a moderate decline, initial power analysis offers a baseline value of statistical power that can be improved upon. Efforts to integrate multiple data streams have successfully demonstrated their capacity to reduce uncertainty in population estimates and improve the power to detect population level changes. Jacobson et al. (2020) determined that by combining multiple data sources, they were able to calculate biologically plausible estimates of population trends for Cook Inlet beluga whales, where this had previously not been possible. Boyd and Punt (2021) simulated data representative of a large whale population and demonstrated that only an integrated population model (IPM) with the maximum combination of data sources (including environmental and demographic data) was capable of differentiating between a stable or heavily declining population. The AVADECAF software can provide an evaluation of statistical power for fixed passive acoustic monitoring, however if power remains low despite a range of monitoring scenarios, the next step could be to consider integrating data sources to improve monitoring power.

## Conclusion

The AVADECAF software tool is freely available online and provides an opportunity to investigate the statistical power of an acoustic monitoring program, where density estimation of sound producing species like cetaceans is the desired outcome.

The software is customizable to tailor the simulation to a region of interest and simulate a population trend for a specific species. The original development of the tool included a parameter sensitivity analysis that demonstrated the importance of precision around input parameters including cue production rate and false positive rate. This tool is most applicable to species that are well-researched and supported with detailed information on vocalizations, notably this information should be population-specific where possible. Given the frequent use of fixed PAM to monitor cetaceans there is scope to apply simulation power analysis tools like AVADECAF to a range of future research projects and species groups to help improve the utility of this field.

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# Acoustic Activity of Harbour Porpoise Around an Offshore Oil and Gas Platform

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Victoria L. G. Todd, Jane C. Warley, Laura D. Williamson, and Ian B. Todd

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**Abstract**

Energy companies have been exploring for (and extracting) oil and gas (O&G) from offshore hydrocarbon fields since the late 1890s. Yet, relatively little research has been conducted into how marine megafauna interact with, and are subsequently impacted by anthropogenic structures in the marine environment. Utilizing passive acoustic monitoring equipment (C-PODs), this study investigated harbor porpoise (*Phocoena phocoena*) occurrence in the vicinity of a new O&G platform, recording click data between 2015 and 2020, at control and impact sites, pre-platform

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installation, during construction, and once the platform was *in situ*. Within five months, porpoise detections per day had returned to pre-platform levels. Drilling-noise measurements of a nearby drilling installation produced a Sound Pressure Level of 120 dB re 1 μPa at 2–1400 Hz. Porpoise forage regularly near offshore O&G installations and are exposed to various noise sources while travelling through the water column, and may hear high-frequency noise from drilling activities up to 70 m from the stem. Findings have important consequences for decommissioning, which does not presently consider impacts of anthropogenic infrastructure on marine megafauna in anything other than a noise-mitigation milieu.

**Keywords**

Harbor porpoise · Echolocation · C-POD · Offshore platform construction · Oil & Gas (O&G) · North Sea · Dogger Bank · Decommissioning · Rigs-to-Reefs · Drilling noise

## Introduction

While many Oil and Gas (O&G) reserves in the North Sea dwindle, some remain productive, and the current Ukrainian-Russian war has seen a resurgence in new creation of shallow-water O&G infrastructure. Consequently, offshore platforms are likely to persist for decades to come (Leporini et al. 2019), so further insight into how they may impact marine fauna in the long term is vital.

Some well-located and older offshore platforms (and their 500-m shipping exclusion zones) form a series of interrelated manmade reefs (McLean et al. 2022), attracting and providing sites of reproduction for fish and invertebrate species (Fujii et al. 2014, 2020; Todd et al. 2018, 2020b, 2021).

As part of routine integrity checks, platforms are assessed regularly for marine growth, but are considered rarely in the context of generated underwater noise and impacts on marine mammals. This is especially important, since cetaceans utilize different frequency ranges for communication, feeding, and other behaviors (Southall et al. 2007; André et al. 2010; Erbe et al. 2018; NMFS 2018; Southall et al. 2019). While mitigation measures are carried out during offshore O&G Exploration and Production (E&P) activities, long-term underwater noise studies are missing from the literature. Noise-measurement campaigns are brief, controlled by operator schedules, often in poor measurement conditions, and data are bound mostly by Non-Disclosure Agreements (NDA). Moreover, while underwater noise monitoring datasets are often included in Comparative Assessments (CA) (Bagstad et al. 2013) and Net Environmental Benefit Analysis (NEBA) procedures (Nicolette et al. 2023) for decommissioning activities, they do not consider marine mammals in anything other than a mitigative context, during, for example, the Plugging and Abandonment (P&A) process (*e.g.* protections from explosions). Consequently, peer-reviewed literature investigating associations between marine megafauna and O&G structures

is limited to a handful of papers (Todd et al. 2009, 2016, 2020a, 2022; Triossi et al. 2013; Delefosse et al. 2017; Clausen et al. 2021).

Research quantifying effects of installing a new O&G structure on harbor porpoise presence and activity is presented here while impacts on potential prey species are published elsewhere (Todd et al. 2020b, 2021). In this study, between 2015 and 2020, passive acoustic click-detectors (C-PODs) were deployed to record harbor porpoise presence around the platform legs and at control sites further afield, pre-platform installation, during construction, and once the platform was *in situ*. Recordings of underwater noise from a nearby operational jack-up-drilling rig were also carried out to elucidate what noise porpoise are exposed to when searching for prey around offshore installations.

## Methods

### Location

C-POD data were collected from the A18 gas-production platform, which is positioned in The Netherlands sector of the Dogger Bank special Area of Conservation (DBSAC) (Fig. 1a). Near-field drilling noise was monitored from the exploration jack-up rig,

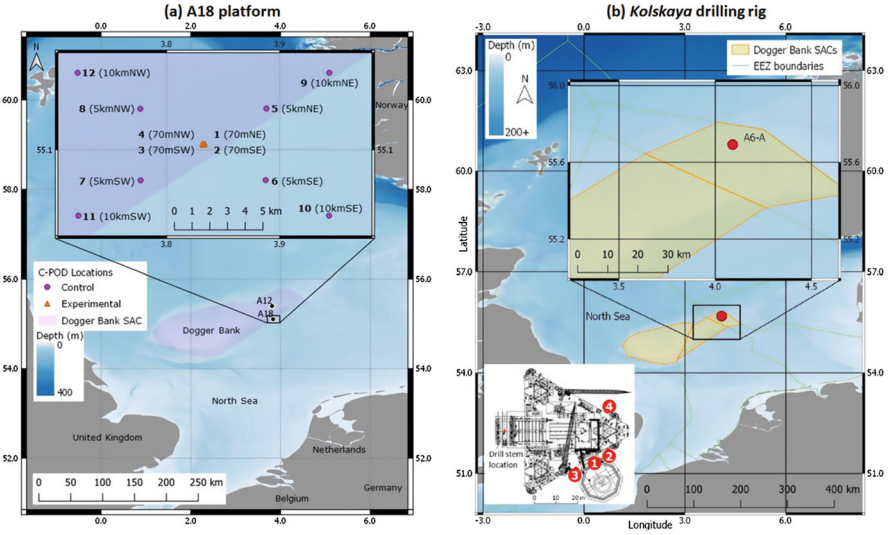



**Fig. 1** (**a**) Position of the A18 platform and control and experimental C-POD deployments in The Netherlands region of the Dogger Bank Special Area of Conservation (DBSAC). (**b**) Position of the exploration jack-up rig, *Noble Kolskaya* (red dot), in the German region of the DBSAC, with the A18 shown for context. Near-field noise measurement locations are shown on the rig plan: (1) Starboard rail, (2) Starboard explosives platform, (3) Container, and (4) Port forward anchor. Black arrow = drill-stem. Coordinates WGS'84 decimal degrees. (Source: Todd et al. (2020c and 2022)

*Noble Kolskaya*, stationed to the north of the A18 in the German sector of the DBSAC (Fig. 1b).

## BACI Measurements and Analysis

Full details of Before After Control Impact (BACI) study design are available in Todd et al. (2022). Twelve acoustic loggers (Version 1 C-PODs, Chelonia Ltd., UK) were deployed in the vicinity of the A18 gas-production platform before, during, and after its construction. Four were deployed in experimental locations 70 m northeast (NE), northwest (NW), southeast (SE), and southwest (SW) of the predicted wellhead location. Eight controls were deployed in the same formation, with four at 5 km and four at 10 km distance away from the wellhead. Default settings were used for all C-POD deployments, and they were deployed using acoustic-release moorings.

Several factors were considered when choosing C-POD deployment locations. Experimental C-PODs were positioned to ensure no interference with routine platform operations, but also to maximize chances of detecting porpoise present around the structure. Beamwidth of porpoise echolocation is narrow and, therefore, porpoises are detected primarily on-axis (Koblitz et al. 2012); therefore, deploying four C-PODs 70 m from the proposed wellhead and sound source increased likelihood of detecting all porpoise sonar around the platform, as per research by Tougaard et al. (2006). While no standard practice exists in the literature for this type of study, control locations were established based on comparable BACI research around wind farms (*e.g.* Diederichs et al. 2008). Control locations were synonymous with experimental locations in regards to topography, water depth, bottom type, fauna, and flora, which was established by consulting surveys of the site and pipeline (Fugro 2013, 2015). C-PODs were also close enough to the A18 to minimize potential overlap with nearby O&G fields. The possibility of simultaneous detections was also eliminated given that C-PODs detect porpoises when they are within 300 m (e.g. see Tougaard et al. 2006; Kyhn et al. 2012; Nuuttila et al. 2018).

In accordance with Dähne et al. (2013), CPOD.exe (Chelonia Ltd.; version 2.044) was used to process C-POD data. The KERNO classifier was run to extract Moderate (Mod) and High (Hi) confidence detections and detection Positive Hours per Day (DPH $d^{-1}$) were derived.

To determine short (<12 months) and long (>12 months) -term impacts of A18 platform construction and operation on porpoise detections, ecological causal assessment was carried out according to Norris et al. (2012) (Norton and Suter 2014; Webb et al. 2016). Additional analysis was also performed on select data sets collected between 2015 and 2020 to determine whether any observed biological effects occurred prior to construction of the A18 (Cormier et al. 2010; Norton and Suter 2014).

Using the '*CausalImpact*' package in R (Brodersen et al. 2015), casual inference was investigated, as per explanations included in Todd et al. (2022). As described by Abadie and Gardeazabal (2003), a synthetic control was determined using detections recorded at control sites to predict the number of detections that would have occurred at experimental locations if no platform had been installed. The synthetic control was

established using a Bayesian-structural-time-series model (BSTS) with DPH $d^{-1}$ from the impact site before A18 construction as the response variable and DPH $d^{-1}$ from control sites over the same timeframe as explanatory variables. Posterior distribution was calculated using Markov Chain Monte Carlo; 10,000 simulations were performed with a burn-in period of 10%. To investigate impact of individual A18 industrial operations, the difference between derived values for the synthetic control and values recorded at experimental locations were examined for each day. To avoid autocorrelation between experimental moorings, a single experimental location was utilized for calculations. It was predicted that any variation between detections at the experimental location and the synthetic control were because of A18. Refer to Todd et al. (2022) for additional description of equations.

Using the '*randtests*' package in R (Mateus and Caeiro 2014), a Cox–Stuart test was carried out to examine whether DPH $d^{-1}$, recorded at the experimental site changed over time. See Todd et al. (2022) for the R code and further information on model choice and validation. A Kruskal-Wallis test was conducted to compare detections at experimental and control sites.

## Drilling-noise Measurements & Analysis

Noise from the jack-up-drilling rig, *Noble Kolskaya,* was recorded using methods outlined by de Jong et al. (2011) and NPL (2014) in Beaufort Sea state $\leq$3; procedures conformed to current ANSI/ASA (2009) and ISO (2016) standards. Near-field measurements were taken from four locations on the platform (as shown in Fig. 1b) and far-field measurements were recorded by researchers onboard the vessel *Northern Seeker*, in an attempt to calculate source levels.

Two hydrophones (Sonar Research and Development; SRD) were deployed simultaneously to obtain noise measurements: a 25 mm HS70 and a 12.5 mm HS150; primary resonance frequencies were 70 kHz and 150 kHz, respectively, and low frequency sensitivity for both was −205 to −211 dB re 1 V/μPa for noise below 10 kHz. Data were transferred to a PC hard disk using a Roland UA30 digital interface (16-bit resolution 48 kHz sampling rate) for low-frequency noise (<24 kHz) and a National Instruments PCMCIA interface (12-bit resolution 320–400 kHz sampling rate) for high-frequency noise (10 Hz – 200 kHz). Further noise recordings of the audio bandwidth (24 kHz) were taken using digital magnetic tape and a Sony TDS-D7 digital DAT recorder (16-bit resolution). Signal-to-noise ratio was enhanced using conditioning preamplifiers, and a 1 Hz high-pass filter was utilized for Alternating Current (AC) coupling of recordings in the audio-bandwidth; high bandwidth recordings were made with a band-passed preamplifier set from 2 to 150 kHz.

Custom-written scripts created in MATLAB v. 2.0 were used for time and frequency domain analysis. Broadband and tonal components associated with rig operations were analyzed and compared with generalized ambient noise spectra published by Wenz (1962). Using methods similar to Alessio (2016), and allowing for hydrophone systems' transfer functions, data were converted to receive Sound Pressure Level, SPL (dB re 1 μPa), or Power Spectral Density, PSD (μPa/√Hz), using

a 0.091 Hz bandwidth for analysis of low-frequency noise (1–110 Hz) and a wider 2.93 Hz bandwidth for analysis of higher frequency noise (90 Hz–24 kHz). For spectral analysis, high resolution, Fast Fourier Transforms (FFTs) were performed (Welch 1967). In order to assess tonal-component-time stability and reduce indistinct noise variance, Welch (1967) averaging techniques were applied to 43.7–87.4 s time windows; ⅓ octave band analysis was carried out for noise in the 25 Hz – 13 kHz frequency band. Data show standard noise characteristics of rig operations over time. Sound-speed profiles obtained from Conductivity Temperature Depth, CTD (Seabird Scientific, Washington, USA), measurement data were used to calculate Transmission Loss (TL) at several frequencies using Bellhop ray tracing (Jensen et al. 2011).

### A18 Platform & Kolskaya Drilling Rig Operations

No O&G activity occurred during the baseline period in summer/autumn 2015, which lasted 61 days. The construction phase lasted five days and took place between October 9th and 13th, 2015; various operations were then on-going until November 2nd, 2015, at which point a jack-up rig joined the production platform for 109 days to drill new wells; the rig left on February 18th, 2016. Following this period, the platform was unmanned.

A detailed record of *Kolskaya* drilling operations was cross-referenced with noise measurements recorded during various rig activities; recordings spanned several frequency bands. Operations were categorized as follows: rig installation with tugs present, tank discharge, preloading, drill preparation, drilling 30″ hole, drilling 24″ hole, coming and going of helicopters, and support vessels. Operations data were sourced from platform and rig logbooks.

## Results

C-PODs were deployed for 997 days between August 2015 and January 2020 at 12 monitoring stations. Of the 132 deployments, 129 C-PODs were recovered and three were lost; recovery rate was 97.7%. A timeline of C-POD deployments is shown in Fig. 2.

Noise measurements were taken directly from the rig's barge around the legs of the *Kolskaya* and from the support vessel *Northern Seeker* in 2004. Results reported here are additional to those previously reported in Todd et al. (2020c).

### Effects of A18 on Porpoise Detections

Differences between expected and observed platform operation-related porpoise detections were significant at all impact locations and the probability of achieving this result by chance was negligible ($p < 0.001$). There was, however, variation over

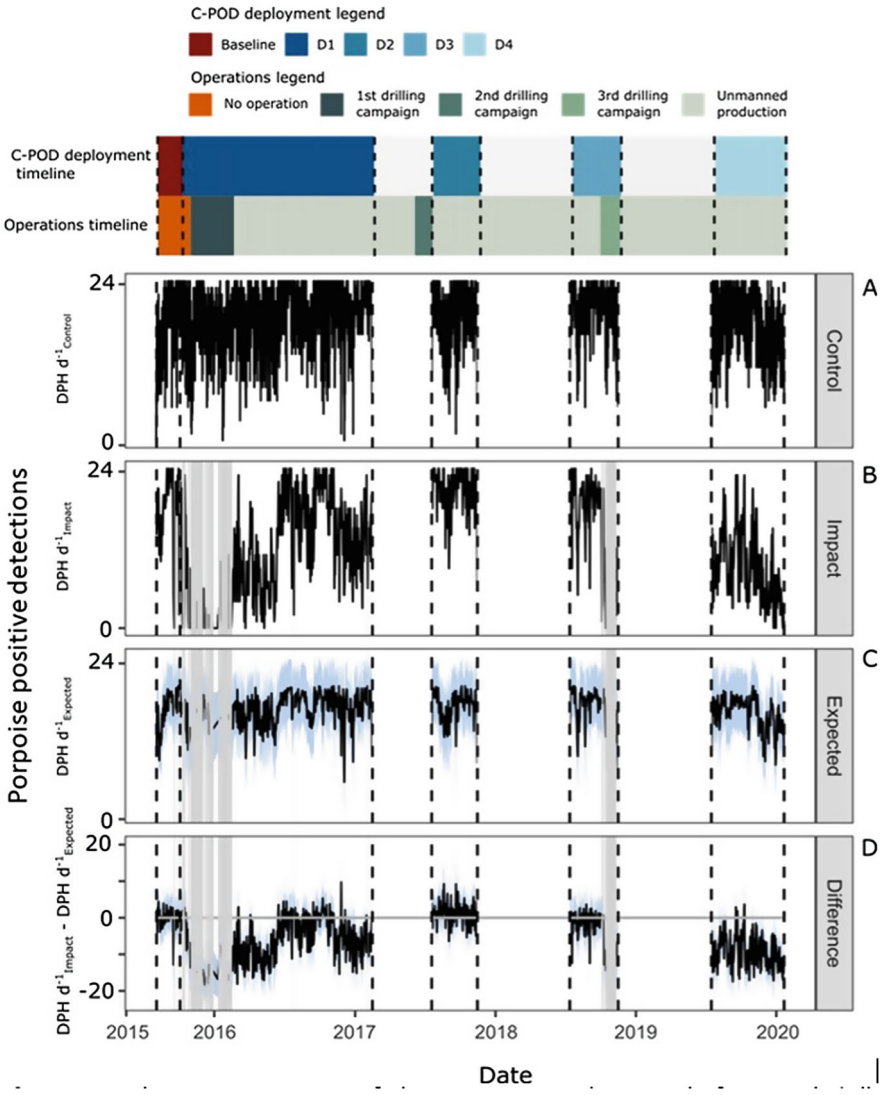



**Fig. 2** Short-term impacts of the A18 gas production platform and drilling rig on porpoise detections. (**a**) Detection positive hours per day (DPH $d^{-1}$) at 5 and 10 km control locations (DPH $d^{-1}_{Control}$), (**b**) DPH $d^{-1}$ 70 m NE of the platform (DPH $d^{-1}_{Impact}$), (**c**) Expected DPH $d^{-1}$ if the platform had not been installed (DPH $d^{-1}_{Expected}$), (**d**) Difference between expected and observed DPH $d^{-1}$ at the 70 m impact location (DPH $d^{-1}_{Impact}$ - DPH $d^{-1}_{Expected}$). No difference is shown by the grey line. A breakdown of C-POD deployments is shown (vertical dashed lines), including the baseline period when no platform was present, the construction period when the A18 was installed, and the three drilling campaigns that took place once the platform was in situ (D1–D4); platform installation is included in the first drilling campaign. Gray bars show where >10% of data were omitted, for example, because of excess noise resulting in C-POD shut-down. 95% credibility interval is shown by the blue ribbon in plots C and D. (Source: Todd et al. (2022))

time; any differences found between expected and observed DPH $d^{-1}$ pre-platform installation (baseline) were not significant (Fig. 2).

As predicted, there were significantly fewer porpoise detections following A18 installation and the first drilling campaign in all locations except controls (Fig. 2); however, between 2nd November 2015 and 18th February 2016, significantly more porpoises were detected at control location 5, 5 km away from the A18, than during the baseline period (Kruskal-Wallis test, $P < 0.001$).

During summer and winter in 2019, DPH $d^{-1}$ were lower than anticipated in all locations (Fig. 2).

For all sites, DPH $d^{-1}$ were explored to determine if short-term changes in porpoise presence following A18 installation persisted or whether presence returned to baseline levels with time. No significant change in porpoise detections was found ($p = 0.2985$), indicating that porpoises were neither attracted to nor displaced by the platform.

## Drilling Noise – Near-Field Measurements

Noise levels recorded during different operational phases on the *Kolskaya* rig are reported by Todd et al. (2020c). Results shown here expand on those noise profiles by investigating how the noise regime changes with depth and time during the drilling phase.

Figure 3 shows depth profiles for low (Fig. 3a) and mid-frequency (Fig. 3b) noise. Strong tonals are evident, for example, at 880 Hz, 1263 Hz, and 1373 Hz, and for sound below 1 kHz, noise level increases with depth.

Drilling noise varied over relatively short time scales. Figure 4a shows prominent 2.8 and 5.6 Hz components present in one recording but absent 30 min later, and this corresponds with a general increase in surrounding noise density level, which is also shown in Fig. 4b for frequencies up to 1.5 kHz; tonal components are evident at

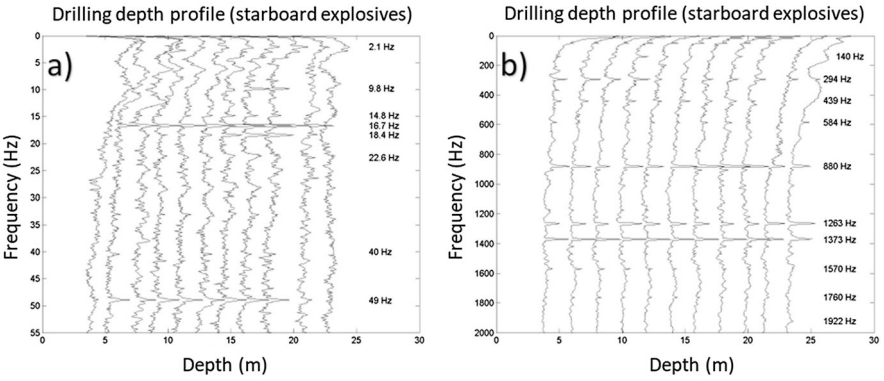



**Fig. 3** Low (**a**) and high (**b**) frequency noise depth profiles taken from the starboard explosives platform of the *Kolskaya* during drilling. (Source: Todd et al. (2007))

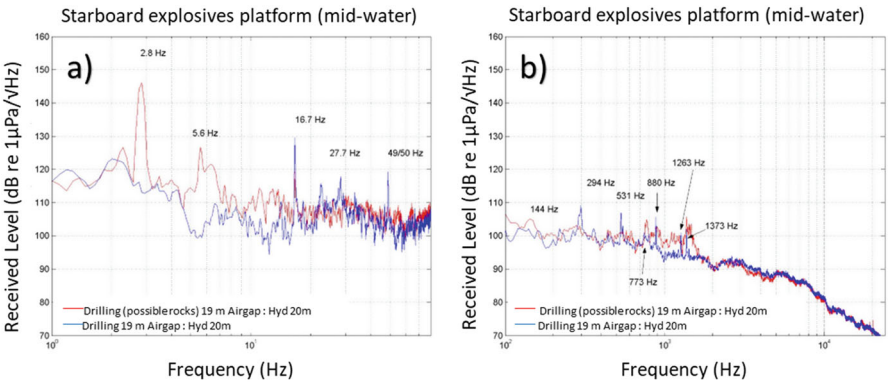



**Fig. 4** Spectral comparison of two mid-water noise recordings made from the starboard explosives platform of the *Kolskaya* drilling rig within 30 min of each other during the drilling phase for (**a**) low-frequency and (**b**) high-frequency noise. (Source: Todd et al. (2007))

294 Hz and 531 Hz. Over 1.5 kHz, noise density level remains stable. These findings are corroborated in Fig. 6 which shows a single time analysis window; Received Level is highest at 210 s and 260 s (Fig. 5).

## Drilling Noise – Far-Field Measurements

Far-field measurements taken 9260 m to the west of the platform from the support vessel, *Northern Seeker*, are shown in Fig. 6. Recording conditions were suboptimal (Beaufort Force 6) and high levels of vessel noise (for safety reasons, engines were switched off only for brief periods, of insufficient duration to obtain high-quality recordings), so attempts to calculate source level are not presented here, as figures are likely highly inaccurate.

At frequencies below 1 kHz, ship noise exceeded noise emitted by the rig while it was drilling. Ship noise was quieter above 400 Hz once main engines were turned off, but with the main engines on, ship noise was louder for frequencies up to 2 kHz.

## Discussion

This chapter investigates porpoise echolocation activity pre, during, and post construction of the A18 gas platform and presents noise measurements taken from the *Kolskaya* exploration jack-up drilling rig and its support vessel, *Northern Seeker*. Knowledge on porpoise/offshore O&G drilling rig/platform interactions is limited and, thus so far, is confined to six peer-reviewed papers (Todd et al. 2009, 2016, 2022; Delefosse et al. 2017, 2020; Clausen et al. 2021). This research, therefore, responds to the requirement for more in-depth studies in this area and builds on results reported by Todd et al. (2020c).

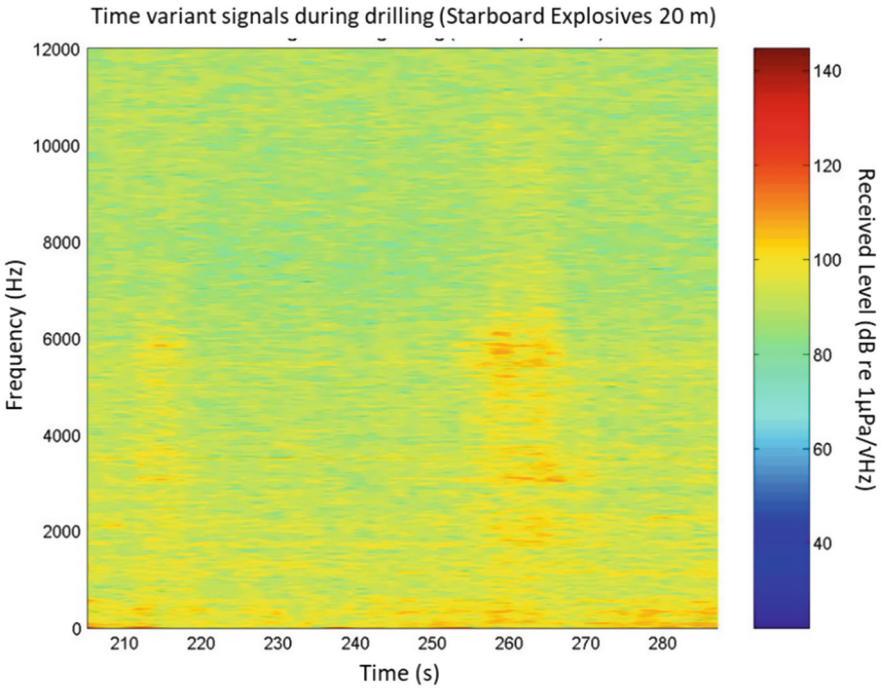



**Fig. 5** Time variant components in noise recorded from the mid-water starboard explosives platform of the *Kolskaya* exploration drilling rig during drilling activities. (Source: Todd et al. (2007))

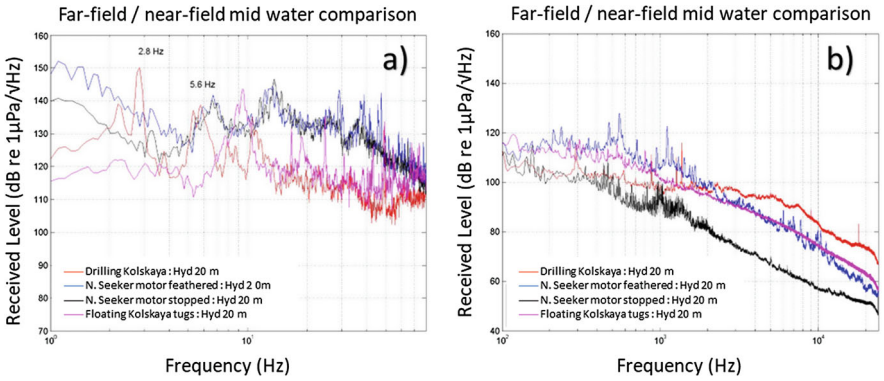



**Fig. 6** Low (**a**) and high (**b**) frequency plots of far-field noise measurements taken from support vessel, *Northern Seeker*, with motor running and switched off. For comparison, noise measurements taken during drilling and during rig positioning when three tugs were present are shown. (Source: Todd et al. (2007))

## Porpoise Presence

Significantly fewer porpoise detections were recorded during A18 platform construction and the first drilling campaign than during the baseline, with the only exception to this being Location 5, 5 km from A18, where more porpoises were detected in the four-month period between November 2015 and February 2016 than during the baseline. A reduction in porpoise presence is in line with studies investigating impact of windfarm construction (e.g., Carstensen et al. 2006; Brandt et al. 2018) and was expected given likely noise levels produced during the construction phase and in subsequent phases prior to start of unmanned production where various preparation activities are taking place. It is, however, important to acknowledge that the absence of detections during the construction phase could be a result of suboptimal recording conditions and, therefore, not a true reflection of porpoise presence. Furthermore, though the increase in porpoise detections at location 5 in 2015/2016 is interesting and potentially linked to platform presence, the result is considered cautiously. While ROV research, reported previously in Todd et al. (2020b), revealed that fish and invertebrates were attracted to the A18 platform just four days post-construction and previous research shows that porpoises are attracted to rigs and platforms generally (Todd et al. 2009, 2016, 2020a; Fernandez-Betelu et al. 2022), few comparisons can be made here with older, more established structures given that the A18 was newly installed and therefore relatively free of marine life (Todd et al. 2020b, 2021). Moreover, Research by Fernandez-Betelu et al. (2022) found that the increase in porpoise presence and foraging behavior around platforms was limited to 200 m; therefore, further research at the A18 is required to determine whether more porpoises are present around the platform than elsewhere and whether numbers have increased over baseline levels.

Considering that, once in place, platforms are operational for 30–40 years, construction/drilling effects were comparably short term, with porpoise acoustic activity returning to pre-platform baseline levels within five months. This is in agreement with porpoise observations during other studies investigating the impact of industrial noise on porpoises, such as piling during the construction phase of windfarm development (Graham et al. 2019) and seismic surveys for the O&G industry (Pirotta et al. 2014).

Over time, porpoise detections remained stable, with no significant increase or decrease, which suggests that, in the long term, porpoises were neither displaced by nor attracted to the platform. Unmanned gas-production platforms like A18 continually emit low-intensity noise, with research suggesting that the higher frequencies, more audible to harbor porpoise, are highest around the platform (Clausen et al. 2021). During unmanned operations in 2017 and 2018, differences between expected and observed DPH $d^{-1}$ were not significantly different, which could suggest that platform operations are not an ideal indicator for porpoise acoustic activity around platforms, though noise data are needed to confirm this hypothesis. Statistically significant differences were, however, found between expected and observed detection rates in the winter of 2016–2017, when there were no *in situ* operations, and during the 2019–2020 monitoring period. Without a further collection of winter data, it is impossible to ascertain what caused the observed decrease in

porpoise detections though, it is likely, given that detections were lower in all locations in 2019–2020, changes are linked to other factors. For example, season, weather, prey abundance, and fishing activity are all likely to have contributed and are the subject of further study. While the majority of data were collected during the summer months, the impact of this bias may be limited given that porpoises are present within the DBSAC throughout the year (Todd et al. 2009, 2016; Gilles et al. 2012), though they are most abundant between July and November (Gilles et al. 2011). Furthermore, Clausen et al. (2021) stated that between July and January, porpoises are most active around O&G platforms, and that is when the majority of these data were collected. Further year-round data are required to investigate this further.

## Drilling Noise

The drilling noise study within 60 m of the drill stem assessed potential noise exposure levels for porpoise near a drilling rig. Results published in Todd et al. (2020c) stated that rig noise had SPLs of 120 dB re 1 lPa in the frequency range of 2–1400 Hz and that, at a depth of 10 m, there is potential for harbor porpoises to detect drilling noise of 880 Hz within a range of 69 m and of 1263 Hz within 66 m from the source horizontally. Further analysis, reported here, demonstrates how drilling noise varies throughout the water column, with a gradual increase in broadband sound level with depth for frequencies below 1 kHz. Strong variations in noise level also occur over relatively short time periods. This is likely due to variations in substrate type with the strong 2.8 Hz tone evident in the sound files present only when boulders or similar relatively hard materials were encountered, as confirmed by drilling logs. This would also explain the general increase in sound level at the time; the 5.6 Hz component is likely a harmonic.

Hearing detection thresholds for harbor porpoise at the low frequencies produced by drilling are difficult to ascertain given that measuring hearing thresholds at those frequencies is impractical – a 1500 m deep acoustic holding tank is required to play a 1 Hz signal to a porpoise. They are most sensitive to sound between 100 and 140 kHz (Kastelein et al. 2002) and they produce dominant Narrow-Band High-Frequency (NBHF) click components within 110–150 kHz (Møhl and Andersen 1973; Verboom and Kastelein 1995; Verboom and Kastelein 1997; Au et al. 1999; Teilmann et al. 2002; Villadsgaard et al. 2007; Kastelein et al. 2017). Given that porpoise peak hearing and vocalizations are five orders of magnitude higher than dominant tonal components of drilling noise, it is unlikely that drilling will impact or mask their echolocation. There is potential for them to detect HF components of drilling noise, and it is evident that noise exposure varies on a spatial and temporal scale as they traverse through the water column. Behavioral effects, whether short or long term, are outside the scope of this study, so no inferences are made here.

To obtain more complete picture of noise emanating from a drilling rig, far-field measurements were taken from a support vessel 9260 m west of the platform. Conditions in the field were not ideal with high winds, swell, and vessel engine noise, and, at

frequencies below 1 kHz, vessel noise far exceeded noise from the rig. To measure far-field noise levels accurately, the support vessel would have had to have been located at least 500 m away from the rig, where, considering (for example, basic spherical spreading applied), noise from the rig would have reduced by 53 dB. This decreases to 26 dB using cylindrical spreading equations, but even then, noise from the vessel would exceed received levels from the rig, so accurate quantification of rig noise was not feasible under the circumstances.

With regard to conservation and impact assessments, this study questions the traditional conservation view that all effects should be temporary and that, post activity, areas need to be returned to pre-construction (pristine) conditions. As an example, common understanding of the European Union Environmental Impact Assessment (EU EIA) Directive is that projects assess and mitigate negative impacts only. While methodologies such as Net Environmental Benefit Analysis Comparative Assessment, NEBA-CA (Nicolette et al. 2023), do balance positive and negative effects of a project, OSPAR's 'clean-seabed policy' is based on the idea that, with the exception of outside of windfarms and O&G fields, the North Sea is pristine. This, however, is arguably not the case: fishing methods cause considerable damage to the seabed and have a larger impact than other industrial activities (Pauly and Zeller 2016; Kroodsma et al. 2018), and, furthermore, the fishing industry, along with decommissioning and other industries, benefit financially from the North Sea's clean seabed policy, therefore, it is essential, when developing decommissioning regulations for end-of-life offshore infrastructure, that the potential of leaving structures *in situ* is considered alongside both benefits and costs involved in implementing policies dictating everything must be removed at end of life (Fowler et al. 2018; Sommer et al. 2019). Consequently, it is vital that NEBA-CAs/EIAs and regulations all consider the science and that both ecological benefits and aversive effects are evaluated.

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# Acoustic Detection of Franciscana Dolphins Near Artisanal Fishing Nets in Argentina

134

Gisela V. Giardino, Mel Cosentino, Giuseppa Buscaino, Ricardo Bastida, and Diego Rodríguez

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**Abstract**

There is interest in the ability to detect the acoustic presence of franciscana dolphins in an area with active bycatch. To meet that objective, two-dolphin click detectors (F-PODs) were placed in two artisanal fishing gillnets in Claromecó (Buenos Aires province, Argentina). Narrowband high-frequency (NBHF) clicks were found near both gillnets, compatible with the description of the franciscana sound production. These preliminary results confirm that passive acoustic is a good tool to assess franciscana presences. Moreover, understanding the echolocation behavior around gillnets can lead to the design of better mitigation tools in order to reduce the bycatch.

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## Introduction

Cetaceans are found in all marine ecosystems as well as some rivers and the biggest threat to their survival is being caught in fishing nets, known as bycatch (e.g., Brownell Jr et al. 2019). One of such cetaceans is the franciscana dolphin (*Pontoporia blainvillei*), whose distribution is restricted to small marine areas in coastal waters off Argentina, Uruguay, and Brazil (Cunha et al. 2014). It is the most affected dolphin species in the Southwestern Atlantic Ocean, with between 2000 and 3000 individuals incidentally caught in gillnets every year. The population along the coasts of Buenos Aires province is estimated at 20,000 (Crespo et al. 2020), and at least 400 dolphins get entangled annually (Negri et al. 2012). For this reason, the International Union for Conservation of Nature (IUCN) and the Mastozoological Society of Argentina (SAREM) have classified franciscanas as vulnerable A3D (Denuncio et al. 2019). In Argentina, the annual levels of bycatch (3.5%–5.6%) far exceed the 2% suggested as the maximum sustainable level by the International Whaling Commission (Crespo et al. 2010). With a potential growth of less than one percent for this species, this bycatch level endangers the population's viability (Cáceres et al. 2020). Additionally, gillnets are killing functional social units that involve mother-offspring pairs along the Argentinean coast (Mendez et al. 2010).

Why franciscanas get entangled is still unknown, but several hypotheses have been raised, including failing to acoustically or visually detect the nets (Bordino et al. 2002, 2013), immature echolocation system (Tellechea and Norbis 2014; Frainer et al. 2015), or because they may be using passive listening for foraging and thus not detect the nets (Tellechea et al. 2017). Several solutions have been developed to decrease franciscana bycatch levels, including acoustic deterrent devices (i.e., pingers) (Bordino et al. 2002, 2004; Paitach et al. 2022) and alternative gear with increased acoustic reflectivity or stiffness (Bordino et al. 2013) with mixed results. The use of bottom longlines would reduce bycatch compared to gillnets but entails an extra effort for fishermen, who are not always willing to endure it (Berninsone et al. 2020).

Although Tellechea et al. (2017) suggests franciscanas forage using passive listening, there have not yet been any studies to support this hypothesis. Studies of sound emissions in franciscana, both in captivity and in the wild (not near fishing nets), show they produce narrowband high-frequency (NBHF, sensu Madsen et al. 2005) sounds with a peak frequency of 139 kHz (Melcón et al. 2012, 2016) and center frequency at 130 kHz (Paitach et al. 2021). Franciscanas have also been recorded producing burst pulses (Barcellos and Santos 2021; Tellechea et al. 2016) and whistles at lower frequencies under stress conditions (Cremer et al. 2017). Calves and juveniles also produce echolocation clicks with low-frequency content

(Melcón et al. 2016). It is therefore thought that franciscana would have the ability to detect gillnets, as suggested for other species with similar echolocation characteristics (Kastelein et al. 2000). Given the franciscanas is a small dolphin, with cryptic and erratic behavior, that lives in small-sized groups, they are difficult to observe in the wild (Bastida et al. 2000; Bastida and Rodriguez 2009; Barcellos and Santos 2021). Therefore, passive acoustic monitoring can potentially be used to investigate the acoustic behavior of franciscanas in close proximity to fishing operations where they are potentially entangled, as it is still unknown (Paitach et al. 2021, 2022).

The main goal of this work was to study the acoustic activity of franciscanas near gillnets in coastal waters of Argentina. Preliminary results of the project are presented.

## Materials and Methods

In November 2021, two artisanal gillnets in Claromecó (Buenos Aires, Argentina) (Fig. 1), 1.4 nm offshore (depth = 15 m), were fitted with automated passive acoustic monitoring instruments (F-PODs).

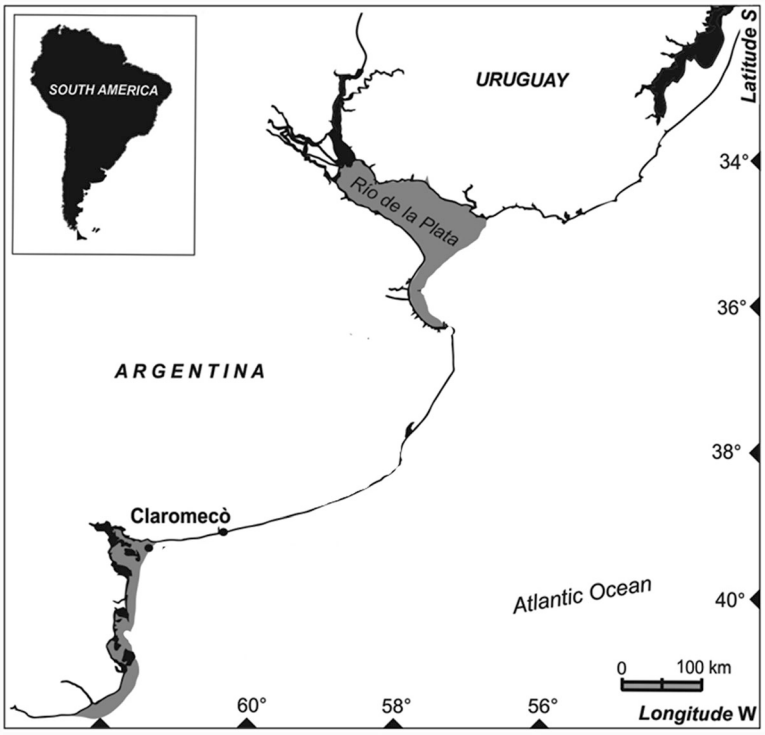



**Fig. 1** Study area. Recording site in two gillnets of Buenos Aires province (Argentina)

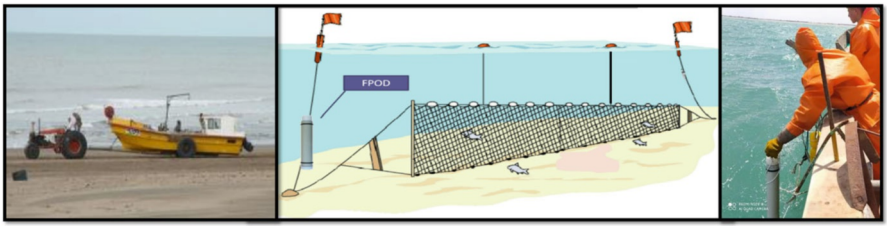



**Fig. 2** Fishermen fiberglass boats from Claromecó artisanal fleet (left), in the middle schematic of the gillnet. (Picture from http://www.miseagrant.umich.edu/nets/largegill.htm and Das et al. 2007) and location of the detector, and on the right fishermen setting up the equipment

The fishermen travel in 8 meter fiberglass boats and use 50-meter-long (Fig. 2), 1.5-meter-high, monofilament gillnets with a mesh size of 10.5 cm as fishing gear. This rigging is composed of 50 m net sections that are generally assembled in 5 or 6 sections (250–300 m). The upper part is equipped with floats or buoys and a footrope with lead weights and monofilament or multifilament nylon threads. These nets are anchored at the bottom, with the main fishery target being sciaenids (*Cynoscion guatucupa* and *Micropogonias furnieri*). Sciaenids are sonorous (Tellechea and Norbis 2012) and are the common prey species of franciscana (Rodriguez et al. 2002; Paso Viola et al. 2014; Denuncio et al. 2017; Tellechea et al. 2017).

Each net was fitted with one F-POD (Chelonia Ltd., UK) placed on the anchor. The nets were deployed by two different fishermen on the 11th and the 14th of November 2021, where the first stayed underwater for 23 hours and the second for 3 days and 21 hours.

F-PODs detect a frequency range from 16 to 220 kHz. According to the manufacturer, F-PODs have a maximum detection range for NBHF cetacean of approximately 400 meters.

However, F-PODs do not record the waveform themselves, but store metadata such as measurements of frequency, amplitude, number of cycles, and bandwidth of the detected sounds. Some full waveforms are also saved. The imported data of F-POD was analyzed with the software provided by the manufacturer (the F-POD app) and the standard KERNO click and click train classifier.

## Results and Discussion

The F-PODs recorded clicks during 20% of the deployment time of net 1 and 17% on the deployment time of net 2. Most of the logged clicks (two-thirds of all clicks), during both deployments, were classified as narrowband high-frequency clicks (NBHF) (Table 1).

For both deployments, the number of clicks and the number of detection positive minutes (DPM) for NBHF signal types were recorded more frequently after sunset (Fig. 3). This is similar to what has been reported for other dolphin (e.g., Caruso et al.

**Table 1** Summary of the F-POD detections during two deployments off the province of Buenos Aires (Argentina). *NBHF* narrowband high-frequency clicks. Other Cet, all other cetaceans – dolphins, beaked whales, pilot whales, etc. Unclassed, clicks not classified into any of the cetacean categories. *DPM* detection positive min

| | Total | Clicks/min | Clicks/h | Clicks/day | DPM/day |
|---|---|---|---|---|---|
| Net 1 | | | | | |
| All | 2813 | 2 | 120 | 2873 | 38.81 |
| NBHF | 1938 | 1.4 | 82 | 1979 | 23.68 |
| Other Cet | 0 | 0 | 0 | 0 | 0 |
| Unclassed | 875 | 0.6 | 37 | 894 | 18.38 |
| Net 2 | | | | | |
| All | 5635 | 1 | 63 | 1503 | 18.67 |
| NBHF | 3808 | 0.7 | 42 | 1015 | 15.75 |
| Other Cet | 96 | 0 | 1.1 | 26 | 0.8 |
| Unclassed | 1731 | 0.3 | 19.2 | 462 | 11.73 |

2017; Gregorietti et al. 2021) as well as porpoise species (Carlström 2005; Todd et al. 2009; Brandt et al. 2014).

The mean peak frequency of the recorded clicks was 126.8 kHz (SD, ± 3.1; range, 114–140 kHz). These numbers are similar to what was previously reported for franciscanas recorded in Brazil (Paitach et al. 2021), whereas in southern Argentina the reported peak frequency was much higher, at 139 kHz (Melcón et al. 2012) (Table 2). It is worth noting that Melcón et al. (2012) used continuous recordings with different equipment (handheld hydrophone connected to the compute) in their study, whereas Paitach et al. (2021) placed the same PODs we used here.

The mean number of cycles (i.e., peaks in the waveform) within each click was 10.8 (SD, ±8.10; range, 3–59). This is consistent with what has been reported for other species that emit NBHF clicks. For example, the number of peaks in harbor porpoise clicks range between 8 and 15 (Møhl and Andersen 1973). This indicates that the detections were in fact of franciscanas and not of other species, as delphinids in the area emit clicks with different characteristics. While NBHF clicks are long in duration, with many cycles, and all the energy concentrated in high frequencies (Fig. 4), dolphin clicks have fewer cycles (only one or two) and are broadband and short duration (see Cosentino et al. 2022, for a comparison).

On the other hand, the detection classified as "other cetacean" may also be a franciscana, as according to Melcón et al. (2016), juveniles can produce sounds with energy in lower frequencies.

The average number of click trains detected per minute was 3.46 (SD, ±2.84; range, 1–12), with a mean of 17.43 (SD, ± 16.37; range, 5–95) clicks per train. The mean interclick interval (ICI) was 24.76 ms (SD, ±17.19; range, 1.54–110.51). This means that the click production rate within a click train varied between 9 and 647 clicks per second. Tellechea et al. (2017) reported between 25 and 58.8 clicks per second during echolocation events, reaching a maximum of 110 clicks per seconds during "burst pulses" for franciscanas. However, these franciscanas were recorded in Uruguay, in estuarine waters, which may explain the difference, although

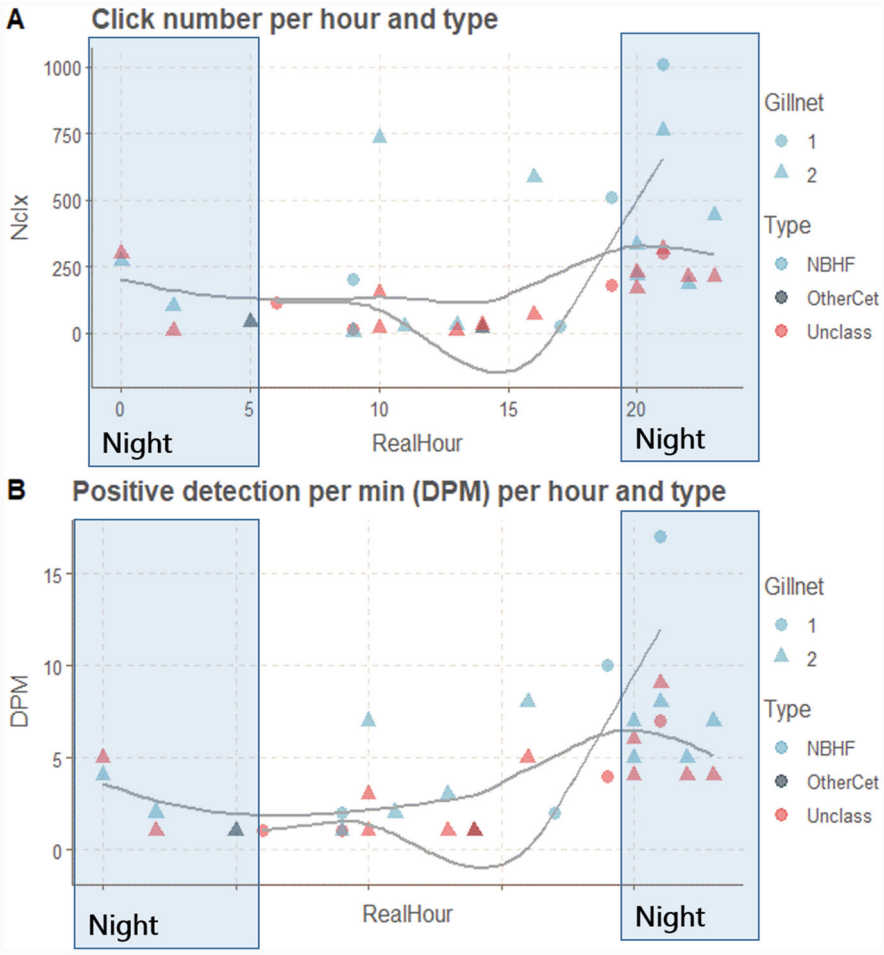



**Fig. 3** Number of click recorded (Nclix) per hour (below) and n. of minutes per hour with detection (DPM) by F-PODs in two fishing gillnets (A and B) in Claromecó (Buenos Aires, Argentina) classified as narrowband high-frequency (NBHF) click other cetacean and unclassified per hour

the equipment was limited to a sampling frequency of 192 kHz. On the other hand, Cremer et al. (2017) reported similar numbers for franciscanas recorded in estuarine waters of Brazil, ranging between 125 and 734 clicks per second. Similarly, Melcón et al. (2012) reported between 23 and 71 clicks per second for echolocation events and between 142 and 250 clicks per seconds for burst pulses.

In other echolocating species, the production rate was found to be related with the behavior of the emitting animal. For example, harbor porpoises typically emit 20–60 clicks per second when moving through the environment, while they reach hundreds of clicks per second when feeding (DeRuiter et al. 2009). Given the existing literature, it is expected that the click production rate differs depending on the behavior of the emitting animal. Further research on this topic is warranted.

**Table 2** Narrowband high-frequency (NBHF) click characteristic (mean ± SD (min-max))

| Author | Year | Habitat | Recording time | Full frequency cover | Sample rate | Train class | Duration (ms) | ICI (ms) | Low frequency (kHz) | High frequency (kHz) | Center frequency (kHz) | Peak frequency (kHz) | Place |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Giardino et al. present work | present work | Open Sea in gillnets | 23hs | Yes | 320 | No distinction | 161 ± 171.72 | 25.50 ± 16.88 (1.4–110) | 114 | 140 | | 127.8 ± 3.3 | Claromecó (Argentina) |
| | | | 3 days 21 hs | Y127es | 320 | No distinction | 238.03 ± 143.44 | 23.45 ± 17.14 (1.8–98.8) | 118 | 138 | | 126.3 ± 32.9 | |
| Paitach et al. | 2021 | Estuarine | 53 days | Yes | 320 | No distinction | 400 ± 535 | 12.6 ± 13.6 | 118 ± 13.9 | 135 ± 5.7 | 129 ± 5 | – | Babitonga bay (Brazil) y |
| | | Open Sea | 65 days | Yes | | No distinction | 399 ± 486 | 12.3 ± 9.7 | 123 ± 4.6 | 134 ± 4.0 | 130 ± 3 | – | Itapiruba |
| Barcellos and Santos | 2021 | Open Sea | 129hs 44 min | No | 288 | Echolocation | 668 ± 624 | 30 ± 43 | 107.09 ± 63.31 | 144.00 ± 0 | 107.53 ± 2.65 | 104.14 ± 3.8 | Achieta island (Brazil) |
| | | | | | | Burst pulse | 153 ± 215 | 6 ± 20 | 98 ± 7 | 144.00 ± 0 | 104 ± 5 | 99 ± 7 | |
| Cremer et al. | 2017 | Estuarine | 46 days | No | 192 | Burst pulse | 75 ± 65 (19–276) | 3 ± 2 | 77.00 ± 3.81 (69.00–82.10) | | | | Babitonga bay (Brazil) |
| Tellechea et al. | 2017 | Estuarine | 24 min | No | 192 | Echolocation | 210 ± 230 (200–230) | 35 ± 2.24 (17–40) | | | | 86.00 ± 1.86 (82.00–93.00) | Santa Lucia del Este (Uruguay) |
| | | | | | | Burst pulse | 150 ± 7 (100–170) | 10 ± 1.02 (9–12) | | | | 41.00 ± 4.69 (32.00–63.0) | |
| Melcon et al. | 2016 | Estuarine | 3 hs | Yes | 500 | No distinction | | | 37.00 ± 3.00 | 160.00 ± 3.00 | | 130.00 ± 3.00 | Rio Negro estuary, Argentina |
| Tellechea and Norbis | 2014 | Captivity | 10hs | No | 192 | Burst pulse | 130 ± 30 (190–1100) | 29 ± 9 (15–48) | | | | 47.44 ± 1.21 (44.00–48.00) | Captivity (Uruguay) |
| | | | | | | Echolocation | 200 ± 100 (170–290) | 10 ± 1 (9–12) | | | | 83.63 ± 2.45 (76.00–90.00 | |

(continued)

**Table 2** (continued)

| Author | Year | Habitat | Recording time | Full frequency cover | Sample rate | Train class | Duration (ms) | ICI (ms) | Low frequency (kHz) | High frequency (kHz) | Center frequency (kHz) | Peak frequency (kHz) | Place |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Melcon et al. | 2012 | Estuarine | 3 days | Yes | 500 | Echolocation | 212 ± 56 | 33 ± 4 (14–43) | 130.00 ± 3.00 | 149.00 ± 3.0 | | 139 | Rio Negro estuary, Argentina |
| Von Fersen et al. | 2000 | Captivity | 6 days | Yes | 1048 | No distinction | 90 sec | | | | | 130 | Mundo Marino (Argentina) |
| Busnel et al. | 1974 | Estuarine | 1 month | No | 60 | No distinction | 900 to 175 | 25.3 ± 5 | 30 | | | | Punta del diablo (Uruguay) |

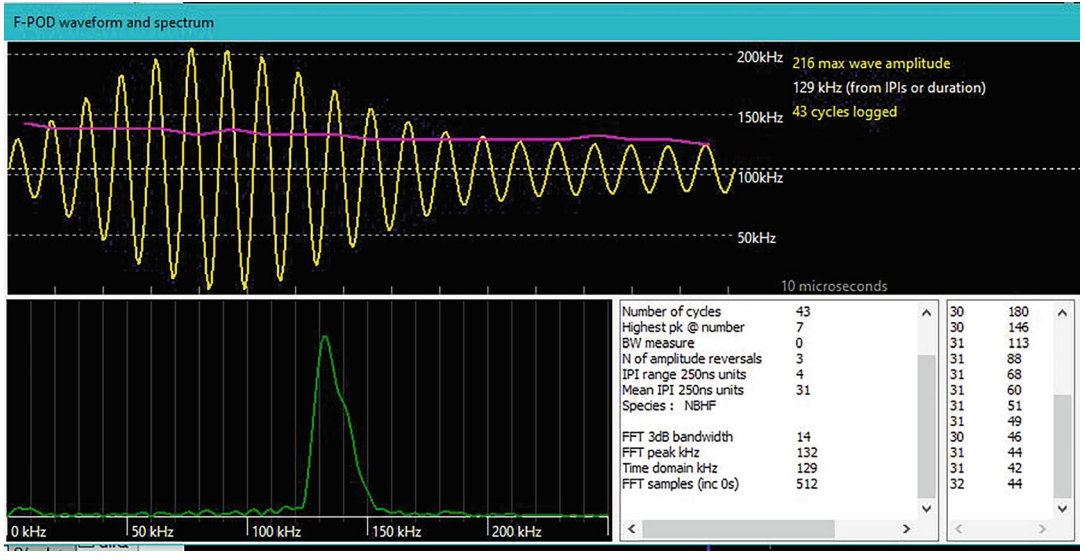



**Fig. 4** Franciscana click. Top left: waveform of a click classified by F-POD as NBHF. Bottom left: power spectrum of the click displayed in the top left. Note the energy is concentrated in frequencies higher than 100 kHz. Bottom right: additional information about the deployment. Figure extracted from the F-POD software

## Conclusions

This is the first time that acoustic detectors have been placed in artisanal fishing nets anywhere within the distribution range of franciscanas. The results show that it is possible to monitor the acoustic activity of this species around fishing nets and confirm that they indeed are acoustically active near gillnets. Furthermore, they seem to emit clicks at high repetition rates, which could indicate foraging.

The information derived from these two fishing days can give preliminary information of occurrence, timing, and movement of franciscanas around fishing gear in the coastal marine waters in southern Buenos Aires province. If the higher echolocation rate during twilight hours is confirmed in future studies, this is key information for further protective measures for this species. For example, if franciscanas are present in the area mostly during the night, fishing activity could be restricted to sunlight hours to reduce incidental entanglement.

Therefore, the information derived from this ongoing project will provide valuable information for the design of new mitigation measures based on the differential acoustic response of dolphins to the existence of nets in the area. This first step is key to understanding what mitigation strategy could be better to implement. These results show the importance of passive acoustics as a complementary tool to improve conservation regulations of this vulnerable dolphin and prevent extinction risk.

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# Passive Sonar Equation-Based Marine Mammal Detection Probability Modeling

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Baptiste Menetrier, Morgane Pommier, and Joanne O'Brien

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**Abstract**

Unknown or variable detection ranges are one of the obstacles to address before being able to derive accurate animal abundance and density estimations from acoustic data alone. Many factors can influence the capability of a device to detect acoustically active animals, and assessing the ability of detectors to receive and identify acoustic cues should therefore be an essential step prior to any passive

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acoustic study, especially as it may influence the reliability of results and interpretation of the data. This chapter describes a passive sonar equation-based detection probability model. This passive sonar equation-based model uses state-of-the-art propagation model BELLHOP to simulate the propagation of a specie-specific cue around a given detector. The model, handled by a standalone graphical user interface, allows users to specify environment parameters and source properties. Other environment features, bathymetry and ocean properties, are, respectively, queried from GEBCO_2021 global grid and online Copernicus models. The set of parameters is then used to estimate a detection probability map. The overall objective of the model is to provide any researcher using passive acoustic monitoring with a detection probability specific to their location and their species of interest.

**Keywords**

Detection probability · Propagation modelling · Sonar equation · Underwater acoustics · Bio-acoustics

## Introduction

Assessing the population density and abundance of cetaceans is crucial for understanding and protecting these marine mammals. Cetaceans are an important part of ocean ecosystems, and their populations can be affected by a variety of factors such as human activities, climate change, and pollution. By understanding the distribution, abundance, and trends of cetacean populations, better conservation policies and management plans can be implemented to protect and preserve these species for future generations. However, studying cetaceans can be challenging due to their mobility, habitat use, and the limitations of traditional monitoring methods (Zimmer 2011). Therefore, it is essential to develop and use new, efficient methods for assessing cetacean populations. One such method is passive acoustic monitoring (PAM), which is particularly useful for monitoring vocal species. This method is increasingly being used by the scientific community to assess cetacean population density as it provides a noninvasive, cost-effective, and long-term monitoring approach.

The density of a species (which is equivalent to the total number of animals in a given area, the abundance) can be defined as the ratio of the number of individuals by the area they occupy. The size of the area considered plays an important role as it may considerably influence the density result. Estimating the size of the area covered by a single acoustic detector is therefore one of the obstacles to address before being able to derive accurate animal abundance and density estimations from acoustic data alone. Another important parameter required to estimate cetacean density and abundance is the rate at which animals produce sounds (Jacobson et al. 2017). There is little information in the literature about sound production rates of wild

animals as it is difficult to measure this parameter experimentally and this aspect will not be covered by this chapter. Yet, one should remain aware that sound production rates are a potential source of significant bias in density and abundance estimations as some animals may not be detected because of a low sound production rate making them silent while crossing the area monitored.

## Detection Probability Modelling: Theoretical Considerations

### Global Approach

The purpose of this study is to model the detection probability associated with a given detector and a given study (mooring location, deployment period, species of interest). To accurately model the detection probability, one should consider several parameters such as noise level (NL), detection threshold (DT), as well as animal-specific parameters source level (SL), directional loss (DL), and behavior features (position in the water column and head movements). The objective is to produce a 2D probability field around the mooring position. To do so, one can consider a set of 2D profiles corresponding to a set of bearings uniformly distributed around the detector (0° to 360°). The 2D detection probability field can then be derived by modelling a 1D detection probability function $g(r)$ for each 2D profile and by merging the results for the entire set of profiles.

Modelling the detection probability along a 2D profile requires precise information about the transmission loss (TL). This can be achieved by either in situ measurements or numerical modelling using underwater acoustic propagation models. In situ measurements are expensive and time-consuming and therefore cannot be realized for small-scale or low-budget projects. Underwater acoustic modelling, on the other hand, is a reliable and cost-effective approach to estimate the transmission loss and will therefore be used in this project.

### Signals of Interest

Cetaceans produce a large variety of sounds for a variety of purposes (Zimmer 2011) which leads to a large variety of specific detectors. Most studies tend to focus on echolocation clicks using click detectors. Currently, the most commonly used click detectors are continuous porpoise detectors (C-PODs) which are fully automatic, static, passive monitoring systems that recognize click trains of porpoises, dolphins, and other toothed whales. The detection probability model developed focuses on echolocation clicks similar to the type of click emitted by porpoises. In other words, the model can be considered to be dedicated to species emitting narrow-band and relatively high-frequency clicks. (The range of validity of the model could be widened in the future.)

## Sound Propagation Modelling

A large variety of sound propagation models (computer solutions to the wave equation) have been developed over time. As most vocalizing species (the acoustic sources of interest) produce sound at high frequencies (few kilohertz or above), the infinite-frequency approximation, ray theory, is the most suitable kind of model for the purpose of this study. The ray model considered is the model BELLHOP.

BELLHOP is a state-of-the-art beam-tracing model for predicting acoustic pressure fields in ocean environments. This acoustic ray theory-based model uses a finite number of acoustic rays emanating from a source to derive a transmission loss field. This model is particularly suitable for high frequencies and allows for range dependence in the top and bottom boundaries as well as in the sound speed profile (Porter 2011). The ray theory is based on a high-frequency approximation and therefore can only produce reasonably accurate results in the region of validity of this approximation. A first-order assumption as well as an often-quoted rule is that the acoustic wavelength should be substantially smaller than any physical scale in the problem (Jensen et al. 2011). Therefore, the wavelength should be small compared to the water depth and the bathymetric features of the area of interest.

## Reciprocity Hypothesis

The situation to model can be described as follows:

- A detector is moored at the position ($lon_{mooring}$, $lat_{mooring}$).
- A vocalizing animal is located at a random position ($lon_{animal}$, $lat_{animal}$).
- The objective is to model the probability of a single click emitted by the animal to be detected by the detector.

This situation is represented in Fig. 1 where the propagation of sound goes from the right side to the left side.

In order to estimate the detection function $g(r)$ along the profile (x-axis), one needs to compute the transmission loss at the receiver location (*blue point* on Fig. 1) for a large set of random positions of the source along the profile. Such a strategy, although theoretically feasible, is far from realistic in terms of time and computational effort. Thus, to model this situation for all range of bearings ([0°, 360°]), it is convenient to consider the reciprocal situation where the positions of the vocalizing animal and the detector are exchanged.

The wave equation from which the BELLHOP model is derived offers a very useful property which is reciprocity. This means that, in a given environment, the positions of the source and the receiver can be exchanged without modifying the received field (Jensen et al. 2011). One can show that the ray theory, which is only an approximate solution of the wave equation, preserves the reciprocity property (Jensen et al. 2011).

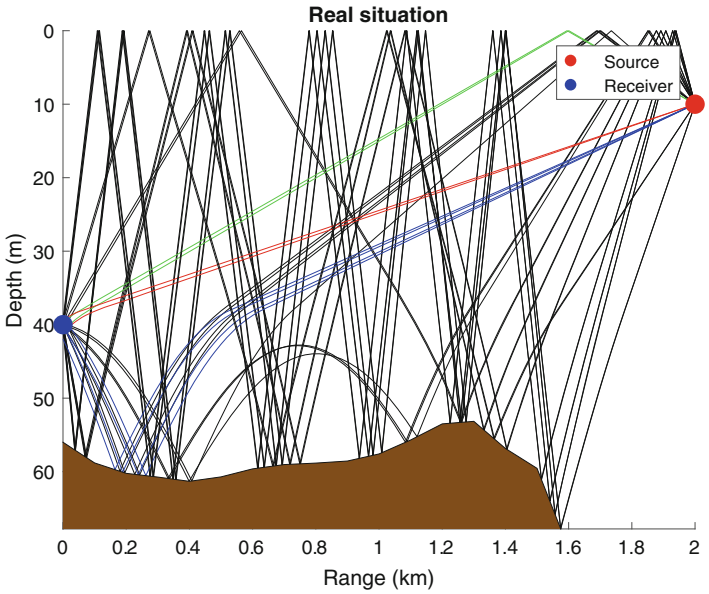



**Fig. 1** Real situation to model. *Red point*, vocalizing animal; *blue point*, detector. *Green lines*, rays hitting surface only; *blue lines*, rays hitting bottom only; *red lines*, direct path from the source to receiver

Considering the reciprocity property, the situation modelled can now be described as follows:

- A vocalizing animal is located at the mooring position ($lon_{mooring}$, $lat_{mooring}$).
- A detector is located at a random position ($lon_{animal}$, $lat_{animal}$).
- The objective is to model the probability of a single click emitted by the animal to be detected by the detector.

In this situation, the transmission loss can be evaluated at different positions of the receiver along the profile with a single simulation (single position of the source).

## Sonar Equation Approach

A usual way to estimate the performances of a PAM device is to use the sonar equation. As devices considered in the model are passive ones, the sonar equation is used in its passive form as stated in Eq. (1). Because it is customary to use decibels in underwater acoustics, all equations presented below are expressed in decibels and thus appear as additive terms. All detectors are assumed to be omni-directional, and no detector processing gain is considered here. Sound sources considered in the model are all assumed to be rotationally invariant (relative to the acoustic axis) so

that quantities relative to the apparent source level (SL) only depend on the off-axis angle which is the angle between the acoustic axis and the direction of the receiver (Zimmer 2011).

$$\mathrm{SE}(\theta, r) = \mathrm{SNR}(\theta, r) - \mathrm{DT} \tag{1}$$

In this equation signal, excess SE corresponds to the difference between the signal-to-noise ratio SNR and the minimal signal level for detection, detection threshold DT. Therefore, a detection occurs if SNR is greater than DT. In other words, a detection occurs if SE $\geq 0$. This equation can be reformulated in Eq. (2)

$$\mathrm{SE}(\theta, r) = \mathrm{ASL}(\theta) - \mathrm{TL}(r) - \mathrm{NL} - \mathrm{DT} \tag{2}$$

where $\theta$ is the off-axis angle and $r$ is the x-axis coordinate along the profile. ASL($\theta$) is the apparent source level (ASL) defined as

$$\mathrm{ASL}(\theta) = \mathrm{SL}_0 - \mathrm{DL}(\theta) \tag{3}$$

with $\mathrm{SL}_0$ the source level on the source main acoustic axis (on-axis) and DL($\theta$) the directional loss for the off-axis angle $\theta$. NL is the ambient noise level.

The 2D TL($r$, $z$) map provided by BELLHOP needs to be reduced to a 1D vector only depending on the $r$ coordinate. To do so, each column of the TL($r$, $z$) map is reduced to the Gaussian weighted mean TL over depth. In other words, considering that the animal is usually swimming at a depth $d$ (mean living depth) with a uncertainty of $\sigma_d$, for a given distance $r$ to the mooring, TL($r$) is defined by

$$\mathrm{TL}(r) = \frac{\sum_i \omega_i \mathrm{TL}(r, i)}{\sum_i \omega_i} \tag{4}$$

where the weights $(\omega_i)_i$ are normally distributed: $(\omega_i)_i \sim \mathcal{N}\left(d, \sigma_d^2\right)$.

It is important to note that in Eq. (2), $\mathrm{SL}_0$ and NL need to be defined with respect to the same reference pressure and frequency bandwidth (usually one should consider a one-third octave band).

Another useful quantity to define is the figure of merit FOM which is the transmission loss that gives a zero signal excess (Jensen et al. 2011).

$$\mathrm{FOM} = \mathrm{SL}_0 - \mathrm{NL} - \mathrm{DT} \tag{5}$$

The on-axis source level is assumed to be normally distributed around the average value $\mathrm{SL}_0$ with a standard deviation $\sigma_{\mathrm{SL}}$. Hence, one can rewrite (5)

$$\mathrm{FOM}(\xi) = \xi - \mathrm{NL} - \mathrm{DT} \tag{6}$$

where $\xi \sim \mathcal{N}\left(\mathrm{SL}_0, \sigma_{\mathrm{SL}}^2\right)$. Equation (2) may then be written as

$$\mathrm{SE}(r, \theta) = \mathrm{FOM}(\xi) - \mathrm{TL}(r) - \mathrm{DL}(\theta) \tag{7}$$

Using the previous definitions, one can finally define the detection function

$$g(r) = \int_{-\infty}^{+\infty} \omega_{\mathrm{SL}}(\xi)\ P[\mathrm{SE}(r,\theta) > 0]\ d\xi \tag{8}$$

where $\omega_{\mathrm{SL}}(\xi)$ is the probability density function of the on-axis source level and can be expressed as

$$\omega_{\mathrm{SL}}(\xi) = \frac{1}{\sigma_{\mathrm{SL}}\sqrt{2\pi}} \exp\left\{ -\frac{1}{2}\left( \frac{\xi - \mathrm{SL}_0}{\sigma_{\mathrm{SL}}} \right)^2 \right\} \tag{9}$$

The quantity $P[\mathrm{SE}(r, \theta) > 0]$ is the probability for a signal emitted by a source with an on-axis source level $\mathrm{SL}_0(\xi)$ in a direction resulting in an off-axis angle $\theta$ to be detected. This equation is equivalent to Eq. (10).

$$g(r) = \int_{-\infty}^{+\infty} \omega_{\mathrm{SL}}(\xi) P[\mathrm{FOM}(\xi) - \mathrm{TL}(r) > \mathrm{DL}(\theta)]\ d\xi \tag{10}$$

For a given pair $(\xi, r)$, the probability $P[\mathrm{FOM}(\xi)–\mathrm{TL}(r) > \mathrm{DL}(\theta)]$ can be derived by integrating the off-axis angle distribution $\omega_{\mathrm{OA}}(\theta)$ over all angles matching the condition $\mathrm{FOM}(\xi) - \mathrm{TL}(r) > \mathrm{DL}(\theta)$.

$$P[\mathrm{FOM}(\xi) - \mathrm{TL}(r) > \mathrm{DL}(\theta)] = \int\limits_{\mathrm{FOM}(\xi)-\mathrm{TL}(r)>\mathrm{DL}(\theta)} \omega_{\mathrm{OA}}(\theta)\ d\theta \tag{11}$$

For a directional loss function, $\mathrm{DL} : \theta \mapsto \mathrm{DL}(\theta)$ bijective on the interval $[0, \mathrm{DL}_{\max}]$ (11) becomes

$$P[\mathrm{FOM}(\xi) - \mathrm{TL}(r) > \mathrm{DL}(\theta)] = \int_0^{\mathrm{DL}^{-1}(\mathrm{FOM}(\xi)-\mathrm{TL}(r))} \omega_{\mathrm{OA}}(\theta)\ d\theta \tag{12}$$

where the term $\mathrm{FOM}(\xi) - \mathrm{TL}(r)$ is limited to values less than $\mathrm{DL}_{\max}$. For values greater than $\mathrm{DL}_{\max}$, the condition $\mathrm{FOM}(\xi) –\mathrm{TL}(r) > \mathrm{DL}(\theta)$ holds for all angles $\theta$, cues will be detected from all angles, and $P[\mathrm{FOM}(\xi) - \mathrm{TL}(r) > \mathrm{DL}(\theta)] = 1$. Considering (12), (10) can be written as

$$g(r) = \int_{-\infty}^{+\infty} \omega_{\mathrm{SL}}(\xi) \left( \int_0^{\mathrm{DL}^{-1}(\mathrm{FOM}(\xi)-\mathrm{TL}(r))} \omega_{\mathrm{OA}}(\theta)\ d\theta \right) d\xi \tag{13}$$

The figure of merit can be written as

$$\mathrm{FOM}(\xi) = \xi - F_0 \tag{14}$$

where $F_0 = \mathrm{NL} + \mathrm{TH}$ describes the environmental and PAM system parameters. The nominal figure of merit is then defined as

$$\mathrm{FOM}_0 = \mathrm{SL}_0 - F_0 \tag{15}$$

In order to derive (13), three cases can be differentiated:

1. $\xi < F_0 + \mathrm{TL}(r)$ no detection is possible.
2. $F_0 + \mathrm{TL}(r) < \xi < F_0 + \mathrm{TL}(r) + \mathrm{DL}_{\max}$ cues can potentially be detected.
3. $\xi > F_0 + \mathrm{TL}(r) + \mathrm{DL}_{\max}$ detection always occurs.

For case n°1,

$$P[\mathrm{FOM}(\xi) - \mathrm{TL}(r) > \mathrm{DL}(\theta)] = 0 \tag{16}$$

and (13) reduces to

$$\begin{aligned} g(r) = &\int_{F_0+\mathrm{TL}(r)}^{F_0+\mathrm{TL}(r)+\mathrm{DL}_{\max}} \omega_{\mathrm{SL}}(\xi) \left( \int_0^{\mathrm{DL}^{-1}(\xi - F_0 - \mathrm{TL}(r))} \omega_{\mathrm{OA}}(\theta)\, d\theta \right) d\xi \\ &+ \int_{F_0+\mathrm{TL}(r)+\mathrm{DL}_{\max}}^{+\infty} \omega_{\mathrm{SL}}(\xi)\, d\xi \end{aligned} \tag{17}$$

With the following notations,

$$\Phi : x \mapsto \int_x^{+\infty} \omega_{\mathrm{SL}}(\xi)\, d\xi \tag{18}$$

$$u = \xi - F_0 - \mathrm{TL}(r) \tag{19}$$

(17) can be written as

$$\begin{aligned} g(r) = &\phi(F_0 + \mathrm{TL}(r) + \mathrm{DL}_{\max}) \\ &+ \int_0^{\mathrm{DL}_{\max}} \omega_{\mathrm{SL}}(u + F_0 + \mathrm{TL}(r)) \left( \int_0^{\mathrm{DL}^{-1}(u)} \omega_{\mathrm{OA}}(\theta)\, d\theta \right) du \end{aligned} \tag{20}$$

The first term of Eq. (20) is independent of off-axis angles and integrates over all source levels for which detections occur (Zimmer 2011). Considering the source-level distribution defined by Eq. (9), one can write

$$\omega_{\mathrm{SL}}(u + F_0 + \mathrm{TL}(r)) = \frac{1}{\sqrt{2\pi}\sigma_{\mathrm{SL}}} \exp\left\{ -\frac{1}{2} \left( \frac{u + \mathrm{TL}(r) - \mathrm{FOM}_0}{\sigma_{\mathrm{SL}}} \right)^2 \right\} \tag{21}$$

With this last equation, the detection function can finally be written as

$$g(r) = \Phi(F_0 + \mathrm{TL}(r) + \mathrm{DL}_{\max}) + \frac{1}{\sigma_{\mathrm{SL}}\sqrt{2\pi}} \int_0^{\mathrm{DL}_{\max}} \exp\left\{-\frac{1}{2}\left(\frac{u + \mathrm{TL}(r) - \mathrm{FOM}_0}{\sigma_{\mathrm{SL}}}\right)^2\right\} \left(\int_0^{\mathrm{DL}^{-1}(u)} \omega_{\mathrm{OA}}(\theta)\, d\theta\right) du \tag{22}$$

where the second term of the equation handles the cases where the off-axis has an impact on the detection probability.

To derive this quantity, the off-axis attenuation DL and the off-axis distribution $\omega_{OA}$ need to be defined explicitly.

## Off-Axis Attenuation

Echolocation signals emitted by cetaceans can be classified into two categories, brief broadband signals (40–70 $\mu$s) such as clicks emitted by dolphins and longer narrow-band signals (greater than 100 $\mu$s) emitted by species such as harbor porpoises (Au et al. 1999). Usually, signals from both categories are directional which means that the source is not radiating the same amount of energy in every direction. Thus, the ASL needs to be taken into account when considering the detection probability (Macaulay et al. 2020). As stated in Eq. (3), the ASL depends on the on-axis source level and the DL pattern of the source. This last quantity is usually referred to as the off-axis attenuation.

Limiting this study to the sound beam of odontocetes, the sound projector can be modelled by a plane circular piston of radius $a$. The directivity pattern of a circular piston source of radius $a$ is given by Eq. (23) (Zimmer 2011).

$$\mathrm{D}(\theta) = \left[\frac{2\mathrm{J}_1(\mathrm{ka}\ \sin(\theta)}{\mathrm{ka}\ \sin(\theta)}\right]^2, \theta \in \left[-\frac{\pi}{2}, \frac{\pi}{2}\right] \tag{23}$$

where $J_1$ is the first-order Bessel function of the first kind. As previously mentioned, due to rotational symmetry, this expression only depends on the off-axis angle $\theta$. The DL function is then defined as

$$\mathrm{DL}(\theta) = -10 \log_{10}(D(\theta)) \tag{24}$$

As most studies concerning odontocetes' acoustic signals provide directional index (DI) values, it is convenient to express DL as a function of this property. For a circular piston model with $ka \gg 1$ (high directionality),

$$\mathrm{DI} \approx 20\ \log_{10}(\mathrm{ka}) \tag{25}$$

Hence,

$$\mathrm{DL}(\theta) = -10 \ \log\left(\left[\frac{2J_1\left(10^{\mathrm{DI}/20}\sin(\theta)\right)}{10^{\mathrm{DI}/20}\sin(\theta)}\right]^2\right) \tag{26}$$

This function is not bijective and needs to be approximated in order to be used to derive the detection function.

### Narrow-Band Approximation

To derive detection probability, the off-axis attenuation (DL) function must be reversible on the interval [0°, 90°] (as stated in Eq. (12)). The circular piston model can be transformed into a bijective function by limiting the model to the main lobe. Yet, as (Macaulay et al. 2020) puts it forward a main lobe ([−30°, 30°]) piston model with no side energy largely underestimates detection probability. Moreover, while the piston model cannot mathematically describe off-axis angles greater than 90°, click energy is radiated at those extreme off-axis angles (Macaulay et al. 2020). In order to get a full 360° ASL pattern, the back beam is set to the lowest value of the narrow-band piston model (approximately −35 dB for a directivity index of 23.5 dB).

Thus, the model considered here is a hybrid model built as described below:

1. The circular piston model presented in Eq. (23) is considered.
2. A zero finding algorithm is then used to find the knots of the $\mathrm{DL}_{\mathrm{nb}}$ function by finding the zeros of the numerator.
3. The $\mathrm{DL}_{\mathrm{nb}}$ model is then modified to fit with the injectivity requirement as follows:

$$\mathrm{DL}_{\mathrm{modified}}(\theta) = \begin{cases} \mathrm{DL}_{\mathrm{nb}}(\theta), & \theta \leq \theta_{\mathrm{mainlobe}} \\ \mathrm{DL}_{\mathrm{nb}}\left(\theta = \frac{pi}{2}\right), & \theta \geq \frac{\pi}{2} \\ \mathrm{DL}_{\mathrm{slope}}(\theta), & \theta_{\mathrm{knot2}} < \theta < \frac{\pi}{2} \\ \mathrm{DL}_{\mathrm{interp}}(\theta), & \theta_{\mathrm{mainlobe}} < \theta < \theta_{\mathrm{knot2}} \end{cases} \tag{27}$$

where $\mathrm{DL}_{\mathrm{slope}}(\theta)$ corresponds to $\mathrm{DL}_{\mathrm{nb}}\left(\theta = \frac{pi}{2}\right)$ to which has been added a light decreasing slope to make it injective on the interval $\left[\theta_{\mathrm{knot2}}, \frac{\pi}{2}\right]$. $\theta_{\mathrm{mainlobe}}$ is defined as the greatest angle verifying condition (28)

$$\begin{cases} \theta \leq \theta_{\mathrm{knot1}} \\ \mathrm{DL}_{\mathrm{nb}}(\theta) > \mathrm{DL}_{\mathrm{nb}}\left(\theta = \frac{\pi}{2}\right) \end{cases} \tag{28}$$

and $\mathrm{DL}_{\mathrm{interp}}(\theta)$ is an interpolated value using *pchip* interpolation method.

The piston model and the narrow-band piston model are illustrated below (Fig. 2). One should note that the little decrease of the detection range (white line) next to the

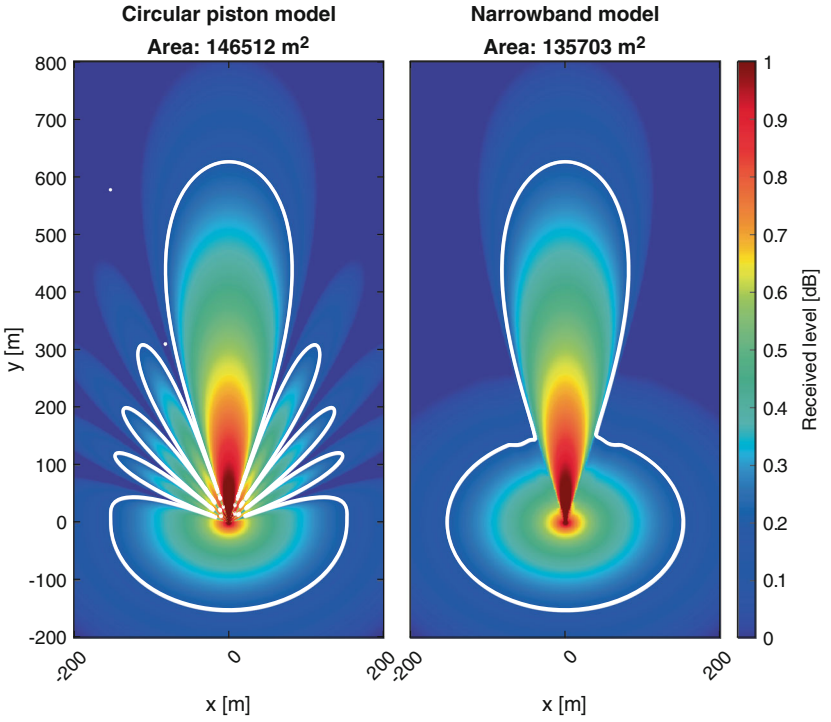



**Fig. 2** Received level computed for a source located at (0,0), facing the y-direction and emitting an echolocation click with $SL_0 = 191$ dB pp re 1 $\mu$Pa @ 1 m and DI = 23.5 dB

main lobe (narrow-band model) is due to an interpolation artifact (for visualization purpose only) and does not represent any knots in the DL pattern.

The white line indicates what the detectable area would be for a PAM study using an automated detection algorithm with a peak-to-peak threshold of 110 dB re 1 $\mu$Pa pp. Received level has been derived assuming a spherical transmission loss law $TL = 20 \log_{10}(R) + \alpha R$ where the absorption coefficient $\alpha$ is set to a casual value of 0.04 dB/m. This configuration is set to be comparable with the porpoise pattern presented in (Macaulay et al. 2020).

## Off-Axis Distribution

In order to model the detection function, one should also consider the off-axis distribution. Two different cases are considered here based on the models presented in (Zimmer 2011). The user of the model can either model the detection probability of arbitrary click considering a uniform spherical distribution or consider the detection probability of click emitted by an animal generally heading toward the detector.

### Uniform Spherical Distribution

Considering that there is no preference in off-axis angles $\theta$ between the source on-axis and the detector direction, the off-axis distribution can be modelled by a uniform spherical distribution (Zimmer 2011).

$$\omega_{\mathrm{OA_A}}(\theta) = \frac{1}{2}\sin\theta \tag{29}$$

### Near On-Axis Distribution

As clicks are more likely to be detected if the animal heads toward the detector, one can also consider a near on-axis distribution. To model this distribution, small variations of the off-axis angles around the main acoustic beam direction due to head motion are considered. To do so, head movements are assumed to be distributed around the nominal on-axis without vertical or horizontal preference. Therefore, off-axis can be modelled by a Rayleigh distribution:

$$\omega_{\mathrm{OA_H}}(\theta) = \frac{\theta}{\sigma_H^2}\exp\left\{-\frac{1}{2}\left(\frac{\theta}{\sigma_H}\right)^2\right\} \tag{30}$$

## Gathering Data

Once the mathematical aspect is implemented, one needs to gather data relative to the situation the user of the model is willing to simulate. Indeed, as shown in section "Detection Probability Modelling: Theoretical Considerations," the model requires several information about the environment to run.

### Environment Data

Sound propagation in the ocean is a very complex phenomenon which is influenced by physical and chemical properties of the medium. Even though the real world can't be accurately described with a finite number of parameters, a sufficient accuracy can be obtained by taking into account main properties influencing the sound propagation. To simulate propagation, the program uses bathymetry data, ocean properties, seabed properties, as well as ambient noise information. In this section, the different sources of information used by the app to run will be presented.

### Bathymetry

Bathymetry can have a significant impact on the propagation of the sound in the ocean. Thus, bathymetric features such as seabed slopes or seamount must be correctly described in order to accurately estimate the transmission loss.

To do so, the model offers two options:

- The users have a bathymetric file describing the area of interest. This file can be then used as an input file (csv or netcdf format required).
- Bathymetry surveys are usually expensive, and therefore, it can be convenient to use a global model. With no user file provided, the bathymetry of the area of interest is automatically retrieved from the GEBCO_2021 global grid.

## Ocean Properties

The transmission loss may be considered to be the sum of two components: the loss due to the geometrical spreading and the loss due to attenuation (Jensen et al. 2011). The geometrical spreading, which traduces the fact that the signal is weakening as it propagates outward from the source, is driven by the geometry of the propagation environment and in a weaker way by the sound speed profile (SSP). The loss due to attenuation can result from either volume attenuation in the water column or loss due to bottom or surface reflections. This second component of the transmission loss is influenced by two factors: the absorption coefficient, usually written $\alpha$ (in dB/km or dB/wavelength), which describes how fast the signal weakens while propagating and drives the volume attenuation, and the SSP which curves ray paths influencing reflections on boundaries (inducing reflection losses) and path length (driving the total volume attenuation along the path).

### Sound Speed Profile

The Mackenzie Eq. (Mackenzie 1981) is used to derive the sound speed profile as a function of temperature, salinity, and depth. Temperature and salinity are automatically queried from online models produced by Copernicus Marine Environment Monitoring Service (CMEMS) for the location of the mooring and the corresponding deployment period. Data are collected from the Global Sea Physical Analysis and Forecasting Product *global_analysis_forecast_phy_001_024*.

### Absorption Coefficient

Similar to the SSP, the absorption coefficient is derived using an empirical formula. The formula used here is the Francois and Garrison formula (Francois and Garrison 1982). As for temperature and salinity, the pH is queried from CMEMS online database with the *globalanalysis-forecast-bio-001-028-daily* model.

## Seabed Properties

As mentioned in section "Bathymetry," reflection of the sound on the seabed needs to be taken into account in order to accurately estimate TL. The type of sediment covering the area of interest can be selected among a list of five: *boulders and bedrock*, *coarse sediment*, *mixed sediment*, *muddy sand and sand*, and *mud and sandy mud*. The classification used is based on the folk 7-class classification from the European Marine Observation and Data Network (EMODnet).

The values associated with each type of sediment are described in Table 1.

**Table 1** Sediment characteristics

| Sediment type | Density ($ton.m^{-3}$) | CWC ($m.s^{-1}$) | CWA ($dB.\lambda^{-1}$) |
|---|---|---|---|
| *Boulders and bedrock* | 2.50 | 3820 | 0.75 |
| *Coarse sediment* | 2.37 | 2122 | 0.88 |
| *Mixed sediment* | 2.03 | 1855 | 0.89 |
| *Muddy sand and sand* | 1.53 | 1708 | 0.91 |
| *Mud and sandy mud* | 1.16 | 1517 | 0.37 |

## Ambient Noise Level

The last parameter to influence the detectability of the signal is the ambient noise level NL. Indeed, as stated in section "Sonar Equation Approach," a signal is assumed to be detected if and only if the SNR of the received level is greater than the DT associated with the detector.

### Root Mean Square Sound Pressure Level

Because of the growing necessity to assess the ambient noise level, a lot of studies use a continuous recorder as an extra recorder in order to monitor the NL in the area monitored by click detectors such as C-PODs. Therefore, the ambient noise level can be estimated from recordings within the app. The metric considered is the average root mean square SPL over the entire recorded period. As the noise level depends on environmental (rain, wind, biologic sources) and anthropogenic factors (traffic intensity, offshore construction), one should consider an audio file extending over a long period.

### Wenz Model

As mentioned earlier, continuous recorders are expensive devices and cannot always be used. For studies with no broadband recordings available, the ambient noise level can be integrated from Wenz curve for an octave band centered on the signal frequency.

## Preliminary Results

In order to evaluate the performance of the model, the estimated detection probability needs to be compared to empirical values for a known configuration. To do so, a first test case is considered. Please note that the configurations modelled are specific to the present study and therefore cannot guarantee the accuracy of the model outside of this scenario. The user is strongly advised to read the results with caution and should keep in mind that some computing artifacts could potentially lead to inaccurate results.

In order to evaluate the accuracy of the model, two metrics are defined:

1. Effective detection range (EDR) is defined here as the mean 50% detection range (maximum range having a detection probability greater than 0.5) over all bearings.
2. Maximum detection range (MDR) is defined here as the mean 5% detection range (maximum range having a detection probability greater than 0.05) over all bearings.

## Test Case 1

The first test case considered is based on a published study (Nuuttila et al. 2018). This study evaluates the detector performance of C-PODs using artificial and recorded harbor porpoise clicks played at a range of distances and source levels. A network of C-PODs located off New Quay headland on the west coast of Wales was used to estimate the click detection probability as a function of distance. The two types of clicks (artificial and recorded) are treated as two different test cases: test case 1a and test case 1b. Test case 1b is the most likely to fit with reality and will therefore be the only one presented in this chapter.

In the paper (Nuuttila et al. 2018), several C-PODs are used to accurately estimate the detection probability. As none of the exact positions are given, an approximate position is considered $(lon, lat) = (-4.370°, 52.225°)$. For both test cases (1a and 1b), the bathymetry is automatically queried from GEBCO_2021 global model for a 2.5 km box centered on the mooring position. As mentioned in (Nuuttila et al. 2018), the C-PODs are deployed 1.5 m above the seabed from 26/04/2012 to 04/05/2012. According to (Nuuttila et al. 2018), the area presents "a generally even, sandy bottom" substrate. Therefore, the seabed property is set to *muddy sand and sand*.

The C-POD manufacturer does not provide any information on the way the detection is affected by ambient noise, though some papers (Dähne et al. 2013; Nuuttila et al. 2018) do evaluate the detection threshold of those detectors as a SPL detection threshold. In other words, the detection threshold estimated in the literature is independent from the noise level. With such a detection threshold, 1 is still valid (SNR is replaced by RL and the casual DT by a detection threshold in terms of received level). The model can then run with $NL = 0$.

The peak-to-peak detection threshold measured by calibration by (Nuuttila et al. 2018) varies from 111 dB to 119 dB re 1 $\mu$Pa peak to peak (pp) (at 130 kHz) across the C-PODs used in the study. The average value of 114 dB re 1 $\mu$Pa pp is used here. (This value is in line with the one given by (Dähne et al. 2013), 114.5 +/− 1.2 dB re 1 µPa pp at 130 kHz.)

### Test Case 1 Settings

The common properties of test cases 1a and 1b are described in Table 2.

**Table 2** Test case 1 – common properties

| Property | Value |
|---|---|
| Frequency | 130 kHz |
| Living depth | 2 m |
| Living depth std | 0.5 m |
| Maximum detection range | 1500 m |
| Detection threshold | 114 dB pp re 1$\mu$Pa |

**Table 3** Porpoise recorded clicks' properties – test case 1b

| Property | Value |
|---|---|
| Source level | 156 dB pp re 1$\mu$Pa @ 1 m |
| Source level std | 9 dB pp re 1$\mu$Pa @ 1 m |
| Directivity index | 23.5 dB |

**Table 4** Porpoise recorded clicks' off-axis properties – test case 1b

| Property | Model | Note |
|---|---|---|
| Off-axis distribution model | Near on-axis | Head movement simulated |
| Off-axis attenuation model | Narrow-band | Directional transducer |

**Table 5** Test case 1b – results

| Metric | Simulation | Reference | Difference | Relative difference (%) |
|---|---|---|---|---|
| EDR | 167 m | 188 m | 21 m | 11.2 |
| MDR | 519 m | 566 m | 47 m | 8.3 |

## Test Case 1b: Recorded Clicks

In test case 1b, recorded clicks (with source levels between 130 and 182 dB re 1 $\mu$Pa) from captive porpoises are played through a calibrated directional transducer, a Reson TC2130. The directional transducer has a narrow beam (about 12.3°) used to replicate the directionality and beam width of a real porpoise. During playbacks, the transducer is continuously rotated from side to side horizontally in an arc of approximately 90° centered on the middle of each C-POD station imitating the sweeping movement of a porpoise head. Properties of test case 1b are presented in Tables 3 and 4.

Simulation results for test case 1b are presented in Table 5.

The model shows a satisfactory accordance with the results from (Nuuttila et al. 2018). MDR error might be due to the definition of the metric. Adjusting the probability threshold (5%) used to derive MDR may lead to a better fit with the empirical results. The probability map describing the probability of a modelled porpoise to be detected according to its position is represented below (Fig. 3).

The detection probability is nearly isotropic due to the nearly flat bottom and the smooth SSP (no acoustic channel).

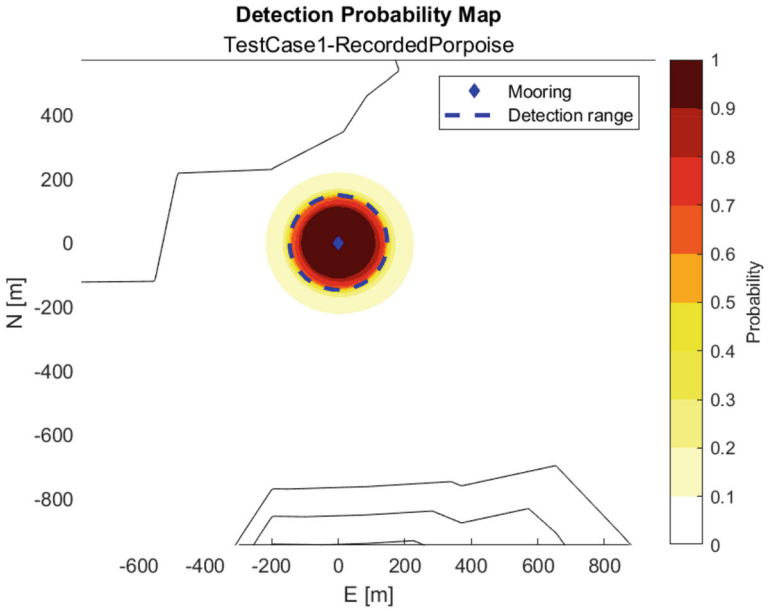



**Fig. 3** Detection probability map – test case 1

## Conclusion

The increasing number of endangered species and unsustainable population declines highlight the need for a better understanding of their distribution and habitat use. To effectively protect these animals, it is crucial to fill the knowledge gaps by acquiring relevant data. PAM, using affordable click detectors like C-PODs, has become an essential technique for studying vocalizing cetaceans. However, there are several challenges that can impede accurate density estimations. One important factor is the estimation of the effective detection area, which directly affects data processing and survey design. A model has been developed to address this issue, and it is handled by a standalone app, which provides tools for modelling the detection probability, offering full control over the different parameters.

It should be noted that the results presented here are preliminary. While the model shows relatively satisfactory results for the first test case considered, it is essential to keep in mind that the signals considered are recorded clicks and may not be representative of real click detection. Furthermore, the accuracy of the model should be tested against a range of frequencies and locations. Finding studies that estimate the detection function while providing enough information to be modelled has been a difficult task and remains the main obstacle to further testing. Ideally, the model should be tested against a dedicated measurement campaign to control all the parameters of the model.

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# Automatic Deep-Learning-Based Classification of Bottlenose Dolphin Signature Whistles

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Frants Havmand Jensen, Piper Wolters, Louisa van Zeeland, Evan Morrison, Gracie Ermi, Scott Smith, Peter L. Tyack, Randall S. Wells, Sam McKennoch, Vincent M. Janik, and Laela S. Sayigh

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**Abstract**

Bottlenose dolphins (*Tursiops truncatus*) produce individually distinctive signature whistles that develop early in life and are used to recognize and maintain contact with conspecifics. Health assessments in Sarasota, Florida (USA), have provided a unique opportunity to record signature whistles of wild individuals of known age, sex, and matrilineal relationships. After 37 years, the Sarasota Dolphin recording library contains 930 recording sessions of 296 individual dolphins. Here, a deep convolutional neural network classifier was trained using a curated subset of 200 different signature whistles from each of

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70 individual bottlenose dolphins. A MobileNetV2 trained on spectrogram data allocated signature whistles to the correct individual with an accuracy of 95.8%. To improve generalization to novel audio datasets, a data augmentation step was implemented that incorporated time stretching, pitch shifting, and mixing database whistles with natural background noise recordings. This augmented model achieved a classification accuracy of 94.5% for high signal-to-noise ratio (SNR) signature whistles, and 92.6% under more realistic conditions with signature whistles mixed with ambient noise at −6 to 12 dB SNR. These initial results are promising and suggest that automatic signature whistle classification techniques could enable acoustic monitoring of movements and habitat use of individual bottlenose dolphins at scale.

**Keywords**

Passive acoustic monitoring · Individual recognition · Machine learning · Population monitoring · Deep learning

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## Introduction

Bottlenose dolphins (*Tursiops truncatus* and *Tursiops aduncus*) produce individually distinctive signature whistles (Caldwell and Caldwell 1965; Caldwell et al. 1990; Gridley et al. 2014) that are recognized by familiar conspecifics (Sayigh et al. 1999) and serve to mediate contact and cohesion among individuals (Janik and Slater 1998; Quick and Janik 2012) as well as to maintain social bonds (Chereskin et al. 2022). Signature whistles are learned early in life and generally remain stable across an animal's lifetime (Sayigh et al. 1990, 2022), facilitating maintenance of differentiated social relationships in complex, cooperative societies (King et al. 2018).

Free-ranging animals produce a range of different whistle types, but on average some 38–70% of all whistles tend to be signature whistles (Buckstaff 2004; Cook et al. 2004; Watwood et al. 2005), and these can typically be differentiated from other whistle types through their temporal bout structure (Janik et al. 2013). Combined with acoustic localization (Quick et al. 2008), this can enable identification of signature whistles from wild animals given enough time spent following individuals (King et al. 2018). However, signature whistles typically account for more than 80% of emitted whistles when animals are isolated from conspecifics (Caldwell et al. 1990; Sayigh et al. 2007), making it much easier to identify the signature whistle of each individual in such contexts.

Sarasota Bay (Florida, USA) constitutes a natural laboratory in which to study wild bottlenose dolphins. The resident community of common bottlenose dolphins (*Tursiops truncatus*) has been studied for more than 50 years as part of a long-term research program (Wells 2020). Periodic brief catch-and-release events that are part of this program have enabled development of a unique dataset of whistles recorded from specific, known individuals with suction cup hydrophones (Fig. 1). Over more than 37 years, the authors have built a recording library that consists of 930 recording sessions of 296 individual dolphins. Of these dolphins, 58% have been recorded more than once (up to 18 times), and half of these (29%) have been recorded over a



**Fig. 1** Bottlenose dolphin being recorded with a suction cup hydrophone on the melon while being temporarily held during health assessments in Sarasota Bay, Florida. (Photo taken by the Chicago Zoological Society's Sarasota Dolphin Research Program under NOAA/National Marine Fisheries Service Scientific Research Permit #15543 to Dr. R. Wells)

decade or more. This recording library is now being converted into a systematic resource, the Sarasota Dolphin Whistle Database (SDWD), containing thousands of validated signature whistles from known animals. The intent of this database is to provide a curated dataset of known dolphin signature whistles from which to train and assess performance of automated algorithms for classification of dolphin signature whistles, in particular deep learning-based algorithms that tend to require large datasets for training and validating models.

Since bottlenose dolphins use signature whistles for identifying conspecifics, they provide an ideal study species to non-invasively track individual animals and even estimate population density acoustically by eavesdropping on their communication system (Longden et al. 2020). However, to do this at scale, automated methods are needed to extract whistles and to identify and classify signature whistles. Traditionally, this has been done by visual inspection of spectrograms and comparison to established catalogues (Sayigh et al. 2022). Several machine learning approaches have been implemented to automate the process, mainly based on dynamic time-warp contour similarity (Buck and Tyack 1993), since the contour has been shown to carry identity information (Janik et al. 2006). Neural network-based algorithms such as ARTWARP (Deecke and Janik 2006) in combination with temporal sequencing information through the SIGID method (Janik et al. 2013) show promise for identifying animal call repertoires (Janik and Oswald 2022). But despite these advances, contour-based methods for identifying signature whistles still have issues – they require an intermediate contour extraction step, and there is still a performance gap between such approaches and visual classification (Kershenbaum et al. 2013; Sayigh et al. 2007).

Recently, deep learning approaches to audio classification have shown performance far beyond many traditional acoustic detection algorithms (Shiu et al. 2020), and deep learning-based detection and classification algorithms have been applied to species from rats (Coffey et al. 2019) to orcas (Bergler et al. 2019) and beyond. The benefits of deep learning-based classification of audio signals are that they can operate directly on time-frequency representations of audio signals, and that they have the capacity to learn to classify signals based on complex time-frequency patterns. Thus, given appropriate training, deep learning-based classifiers might be able to pick up on similar features that humans use for visual classification of signature whistles. Therefore, the objectives of this study are to (1) quantify classification performance of dolphin signature whistles using a deep neural net and (2) evaluate both trade-offs between the number of whistles per individual used in model training and classification performance and the extent to which data augmentation can compensate for sparsely sampled individuals. This information will help guide decisions on suitability of these methods for different conservation applications based on what baseline information is available for different study populations.

## Methods

**Data Sampling** Whistle recordings were obtained during periodic catch-and-release sessions, when small groups of selected dolphins were encircled with a

500-m long, 4-m-deep seine net in shallow water (<2 m depth). Each dolphin was brought aboard a specialized veterinary examination vessel, one at a time, where a standard suite of procedures was carried out, including being weighed, measured, examined by veterinarians, marked if necessary, photographed, and then released (Barratclough et al. 2019; Loughlin et al. 2010; Wells et al. 2004; Wells 2009). Each animal was typically aboard the veterinary boat for less than 1 hour. Acoustic recordings were made with suction-cup hydrophones attached to the melon throughout the operations, whether on the veterinary boat or being held in the water. If for any reason this was not possible, a hydrophone would be held in the water nearby. Up to four individuals were recorded simultaneously on four different audio channels. Total time from encirclement to release (and thus of recording time per catch-and-release session) was typically 2–4 hours, depending on numbers of animals and procedures performed. A variety of hydrophones were used, including both custom-made hydrophones and commercially available hydrophones manufactured by High Tech Inc. (Gulfport, MS), with frequency responses extending at least to 30,000 Hz. Recordings have been made on cassette tapes (1984–89), HiFi VHS tapes (1989–2005), and with digital recorders (since 2006). More detail about recording methods can be found in Sayigh et al. (2007, 2022).

**Data Curation and Labeling** Field recordings were analyzed in the acoustic analysis software program Raven (K. Lisa Yang Center for Conservation Bioacoustics 2022). All analog recordings were digitized (sample rate 96 kHz) to wav files prior to analyzing. Spectrograms (FFT length of 1024, 50% overlap) were viewed in a black and white color scheme using a 10 second page length and up to 32 kHz on the frequency axis, with up to four different audio channels inspected simultaneously on a monitor. Brightness was adjusted individually for each recording until the baseline background noise was just visible, ensuring that any sound above the baseline would be visible. All other settings were kept at Raven's defaults. Whistles were selected and labeled according to the individual that produced them, which was verified by comparing amplitudes of multiple simultaneously recorded hydrophones and by consulting field notes.

The signature whistle of each individual was operationally defined as the predominant whistle type produced during a recording session; typically, the signature whistle makes up >80% of whistles produced during health assessments. Signature whistle variants were defined as whistles that contained key features of the signature whistle but that deviated in some way from the typical contour; examples are embellishments and deletions to the contour (see Sayigh et al. 2022 for more detail). Non-signature whistles were defined as whistles that were visually different from the signature whistle type and were often more variable compared to signature whistles. Finally, signature whistle copies (Janik 1997, 2000) were defined as whistles that matched the signature whistle type of another animal present in the same catch-and-release session. Whistles were classified as possible signatures in rare instances where a signature whistle could not be verified; for example, if an animal produced fewer than ten whistles of the same type or approximately equal numbers of two different whistle types.

**Data Pre-processing** For this study, a subsample of 200 verified signature whistles from each of 70 different dolphins from the Sarasota Dolphin Whistle Database was used. The 200 whistles for each individual were first split into subsets consisting of 150 whistles for training the model; 25 whistles for measuring cross-validation performance for each trained model in order to tune hyperparameters; and 25 whistles that were withheld until the end to measure final model performance. Each audio file was transformed into a spectrogram using a 1024-point FFT with a Hamming window, and a step length of 512 samples for an approximate temporal resolution of 17 ms and spectral resolution of 60 Hz. Spectrograms with a duration less than 3 seconds were zero-padded on with the whistle randomly placed within the frame, and whistles with a duration greater than 3 seconds were cropped to 3 seconds. A median filter was then applied along the frequency dimension for noise reduction. Finally, to increase contrast, pixels that were lower than the 50th percentile of the spectrogram intensities were set to the 50th percentile value before scaling its intensity range to between 0 and 255 and saving as an 8-bit PNG raster image file.

**Model Training** A MobileNetV2 deep learning architecture (Sandler et al. 2018) was used for signature whistle classification (Fig. 2). MobileNetV2 is a deep convolutional neural network with 53 layers and 3.4 million trainable parameters. The model was instantiated via Tensorflow's built-in model MobileNetV2 using randomly initiated weights. The model was then trained to identify individual whistles based on spectrogram images with an input resolution of 484x251 RGB

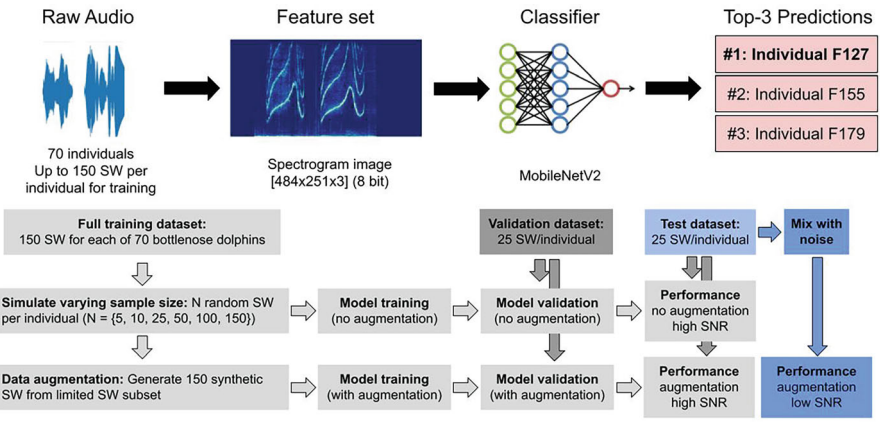



**Fig. 2** Deep learning pipeline and process for training and evaluation classification algorithms. Top: Raw audio files were converted into spectrograms and used to train a MobileNetV2 deep learning classifier that predicts ID of each signature whistle. Bottom: The classifier was trained on 70 individuals with a training dataset of varying numbers of signature whistles drawn from a total sample of 150 different signature whistles per individual. Data augmentation was used to compensate for low sample size. Final model performance was evaluated on new signature whistles (test dataset), optionally mixed with ambient noise from Sarasota

image, using an Adam optimizer and an initial learning rate of 0.0001. The code was written using Python v.3.8, and the Librosa, soundfile, SciPy, and OpenCV Python libraries were used for pre-processing and augmentation. All code is available at github.com/AllenMLI/dolphin_signature_whistles.

**Effects of Sample Size** To evaluate trade-offs between classification accuracy and size of the training dataset, a signature whistle classifier was trained based on a subset of the available training samples for each known individual. For each individual, a random subsample of 5, 10, 25, 50, or all 150 signature whistles were extracted from the full 150-whistle training dataset and used for model training. All other parameters of the training algorithm were kept constant so as to evaluate effects of sample size on final classification performance (Fig. 2).

**Data Augmentation** It was then investigated how well researchers can compensate for reduced variation in training datasets by introducing synthetic variation in individual signature whistles through data augmentation (Fig. 3). For each sample of N randomly selected signature whistles per individual (where N was 5, 10, 25, 50, or 150), each sample was modified 150/N times to achieve a constant training set size of 150 training samples per individual. For each augmented sample, the speed of audio, and hence the duration of the whistle, was changed slightly (with a randomly selected value between −10% and + 10%) using the Librosa *time_stretch* function. Afterwards, the pitch was changed slightly (with frequency multiplied by a randomly selected value between −10% and + 10%) using the Librosa *pitch_shift* function. Finally, mixture augmentation was implemented where the modified signature whistle copy was mixed with a random 3-second sample of underwater noise.

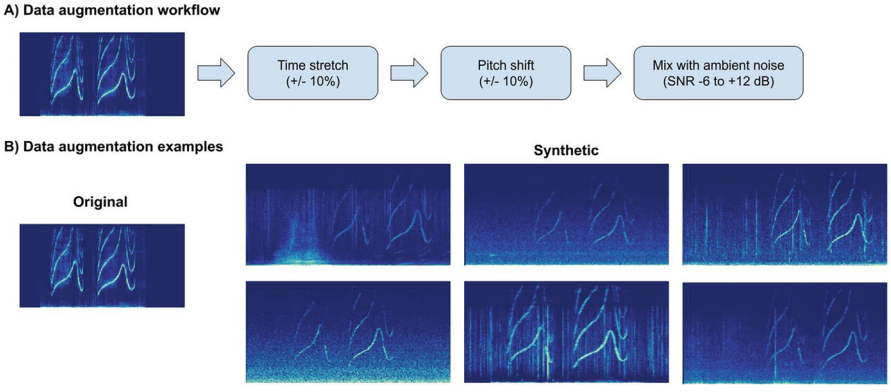



**Fig. 3** Top: Data augmentation combined time stretching (+/−10%), pitch shifting (+/−10%), and mixing known signature whistles with natural background noise (−6 dB to 12 dB SNR) to generate 150 different training examples for each animal. Bottom: Example signature whistle and six examples of synthetic versions including all types of data augmentation

To train the classifier to work across diverse noise conditions, noise samples were procured from seven long-term field sites around the US that were part of the SanctSound project (NOAA 2020), including Stellwagen Bank, MA; Gray's reef, GA; Florida Keys, FL; Santa Carolina Islands, CA; Monterey Bay, CA; Olympic Coast, WA; and Hawaiian Islands, HI. For each site, a random day every month for 12 months was downloaded, and 10 seconds of data every 30 minutes was extracted to form a random selection of noise files. Finally, a 3-second segment was extracted and mixed with the signature whistle exemplar with whistles placed with a random offset (with the entire whistle contained within the 3-second noise segment) and attenuated to result in a randomly selected SNR level between −6 dB and + 12 dB.

**Model Test Datasets** The final model performance was assessed under two separate conditions. The original test dataset of 25 signature whistles from each animal in the model was used as a clean test set to assess performance on unseen exemplars of signature whistles from animals included in the model, at reasonably high SNR given the suction cup recordings.

To assess more realistic field performance in an actual conservation application, the same test dataset files were also mixed with representative underwater noise samples. For this, underwater noise samples provided by the Sarasota Bay Listening Network (Rycyk et al. 2020) and validated to contain no dolphin whistles were used. As during the training phase, whistles were placed with a random offset within the noise file, and attenuated to an SNR of −6 to 12 dB.

## Results

A MobileNetV2 deep learning model was trained to classify different bottlenose dolphin signature whistles directly from spectrograms. Operating on data from suction cup recordings alone, where signal-to-noise ratio is high, the model achieved 98.8% cross-validation accuracy and 95.8% correct classification of whistles in the test dataset. Performance decreased with the number of different signature whistles available per individual, dropping off markedly below 25 signature whistles (Fig. 4).

Using data augmentation to create a synthetic dataset of 150 augmented whistle replicates from a limited subset of source whistles helped improve performance dramatically for low sample size. Using the full dataset of 150 SW per individual yielded a classification performance of 95.9% cross-validation accuracy, on par with the model without augmentation. However, with augmentation, classification accuracy remained above 86% even with only five different source signature whistles per individual (Fig. 4).

Classification accuracy decreased slightly when measured on signature whistles mixed with background noise data at SNR of −6 to 12 dB. Using all 150 whistles per individual, accuracy with more noisy test data was at 92.7%, and decreased for fewer whistles, reaching 82.3% correct classification with only five different signature whistles per individual (Fig. 4).

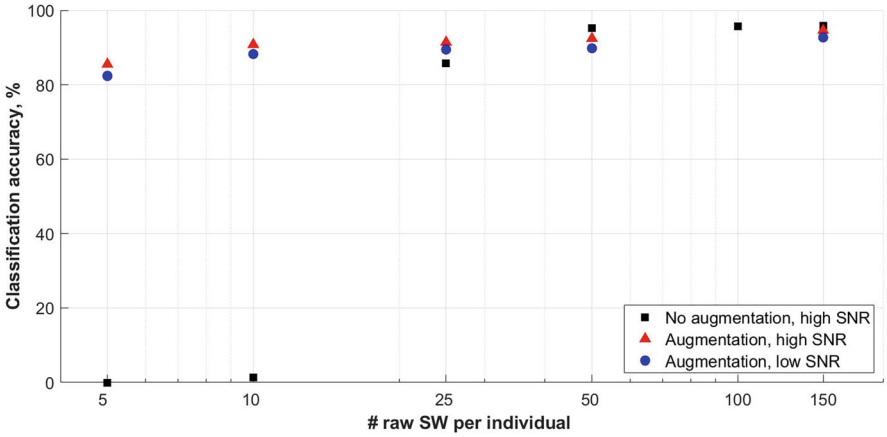



**Fig. 4** Classification accuracy (% correctly classified whistles) as a function of varying sample size for three situations: A classifier trained on limited data per individual without data augmentation (black square); a classifier trained on limited raw data but using data augmentation to generate 150 synthetic SW per individual, evaluated on either a test whistle dataset at high SNR data from the Sarasota Whistle Database (red triangles) or in a more realistic scenario with signature whistles combined with noise at SNR between −6 dB and 12 dB (blue circles)

## Discussion

The Sarasota Dolphin Whistle Database provides a unique dataset for training and assessing machine learning algorithms to automatically classify individual bottlenose dolphins by their signature whistles. Different renditions of the signature whistle of a given individual can contain a wide range of natural variation, including in the number of repeated loops per whistle, different initial and terminal loops, and deletions or additions to the whistle (Sayigh et al. 2022). Such extensive coverage of natural variation in signature whistles from so many individuals is difficult to obtain from other populations of dolphins. Our database of signature whistles from known individuals also includes some signature whistles that are quite similar to one another (especially simpler upsweeps) and while more work is needed to understand how deep learning models differentiate whistles that are similar, this is an issue that could not be resolved through a bottom-up approach of identifying different signature whistles ad-hoc from acoustic recordings.

Highly accurate classification of signature whistles could be achieved with deep learning, with classification accuracy of >92% for a dataset of signature whistles from 70 different dolphins, even under realistic, noisy conditions. For human analysts to maintain the same level of accuracy over long time periods would require extensive training and redundant work-checking, as well as considerable expenditure of time and budget. It is also noteworthy how well augmentation techniques can compensate for small sample sizes, thus giving promise for applying similar approaches in areas where fewer training data are available, or for individuals that

have been more sparsely sampled. Future work is needed both to improve classification accuracy under low-SNR conditions and also to integrate methods for anomaly detection that make it possible to identify signature whistles for which the model has not been trained.

Automated classification of signature whistles has strong potential for conservation applications given that many whistling delphinids studied to date are either known or suspected to produce signature whistles (Janik and Sayigh 2013) and since signature whistles can be used for density estimation using mark-recapture techniques (Longden et al. 2020). Our reported results are promising and suggest that automatic signature whistle classification techniques could enable acoustic monitoring of movements and habitat use of individual bottlenose dolphins acoustically at scale.

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# Measuring Fin Whale Songs in a Noisy Environment

137

Virginia Sciacca, Giuseppe Morello, and Francesco Filiciotto

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**Abstract**

Acoustics is an essential non-invasive approach to study fin whales (*Balaenoptera physalus*). Knowledge of this species' sound repertoire and of its acoustic communication is increasing rapidly with the implementation of a growing number of long-term passive acoustic monitoring (PAM) studies. As

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for other baleen whales, several works performed in the last decade have proven the importance of song studies in the assessment of population status and conservation. Understanding the structure and use of fin whale songs within and between populations and across different oceans represents the next necessary step to improve PAM capabilities in a highly pressured environment. Many differences still exist in the methodologies and tools applied for this purpose. This chapter presents a brief overview of the main metrics and software used to define fin whale songs and to assess the influence of noise on the detection of fin whale song, and the acoustic communication system of the species is presented.

**Keywords**

Fin whale · Vocal repertoire · 20-Hz songs · Acoustic communication · Noise · Conservation · Population ecology

## Introduction

Marine mammals are essential to maintain the ecological equilibrium and to ensure resilience of marine ecosystems. They play a major role in the enhancement of primary productivity in their feeding areas (Roman and Mccarthy 2010). In particular, large deep divers such as sperm whales (*Physeter macrocephalus*) and baleen whales are recognized as "ecosystem engineers": they represent the largest existing marine top predators; they concentrate nitrogen near the surface by fecal plumes; they ensure the vertical transportation and mixing of nutrients from the depth to the surface; and their carcasses sequester carbon to the deep sea, providing habitat and food for many invertebrates (Roman et al. 2014). Among other baleen whale species, the fin whale (*Balaenoptera physalus*) has been studied since the 1960s by acoustically recording and monitoring the typical "20-Hz song pulses" emitted to communicate over long distances (Watkins et al. 1987). The species is protected worldwide under several national and international agreements, and it is globally considered "vulnerable" by the IUCN Red List of Threatened Species (Cooke 2018). However, some populations are more pressured than others, such as the Mediterranean sub-population, which is today listed as "endangered" (Panigada et al. 2021). To enable fin whale conservation worldwide, it will be crucial to monitor the species' distribution, to identify threats, and to provide policymakers with effective mitigation tools. Despite the vast uncertainties that still exist, acoustic monitoring represents one of the most effective methods to understand the species' behavior. It is essential to study its population ecology and conservation status, for example, to assess exchanges between neighboring populations (Castellote et al. 2012a).

In this chapter, a brief review of the studies performed between 1981 and 2022 on fin whale songs, song notes, and temporal patterns from different populations and geographic locations is reported. Methods and tools to detect and classify songs are described and discussed. Finally, the assessment of underwater noise effects on fin whale song studies is examined.

## Fin Whale Vocal Repertoire

The fin whale acoustic repertoire is mainly composed of vocalizations emitted at frequencies close to the lower limit of the human hearing frequency range, established around 16–20 Hz (Fletcher and Munson 1933; Møller and Pedersen 2004); these are referred to as "calls" and have fundamental frequencies in the range between 15 and 142 Hz (Pereira et al. 2020). These calls are produced in irregular series or in stereotyped sequences called "songs" or "bouts" (Watkins et al. 1987). Different types of calls and songs have been recorded across all oceans, showing significant differences in spectral characteristics and repetition rates. As further discussed below, these differences may be related to population-specific vocal repertoires or to the functional role of sound emissions in intraspecies communication. However, the distinction between an irregular call sequence and a song is still very uncertain.

### Calls

The best-known fin whale vocalizations are called "20-Hz pulses," calls centered around 20 Hz, used in long-range communication (Watkins et al. 1987; Edds-Walton 1997). Approximately 20-Hz calls are frequency-modulated downswept pulses, with a fundamental frequency between 17 and 25 Hz, centered around 20 Hz, and an average duration of 1 s (Morano et al. 2012). These sounds are used for intraspecies communication during the breeding period (stereotyped repetitions in songs – see section "Songs") and to keep contact and convey various information at distance (irregular repetitions) (McDonald et al. 1995; Širović et al. 2013). In some regions, a higher-frequency (HF) component also called "overtone," not harmonically related to their fundamental frequency, has been commonly found in association with the 20-Hz pulses (Castellote et al. 2012a; Simon et al. 2010; Širović et al. 2009, 2013; Oleson et al. 2014; Wood and Širović 2022). This could be representative of distinct populations' repertoires and seems to be consistent over time (Oleson et al. 2014). Calls with a frequency higher than 30 Hz have been also reported, and they may be associated with relatively short distance communication (within 20 km) (McDonald et al. 1995; Boisseau et al. 2008) and may have a function during foraging, e.g., the "40 Hz pulse" (Širović et al. 2013; Romagosa et al. 2021). Irregular rumbles were also described by Watkins (1981). These sounds have most of the energy below 30 Hz, and they were described as associated with alarming situations and hydrodynamic sounds produced by the movement of the animal during swimming or feeding.

### Songs

The term "song" is generally used in animal research to describe repeated series of sounds, structured into patterned units of variable complexity, and often nested into a

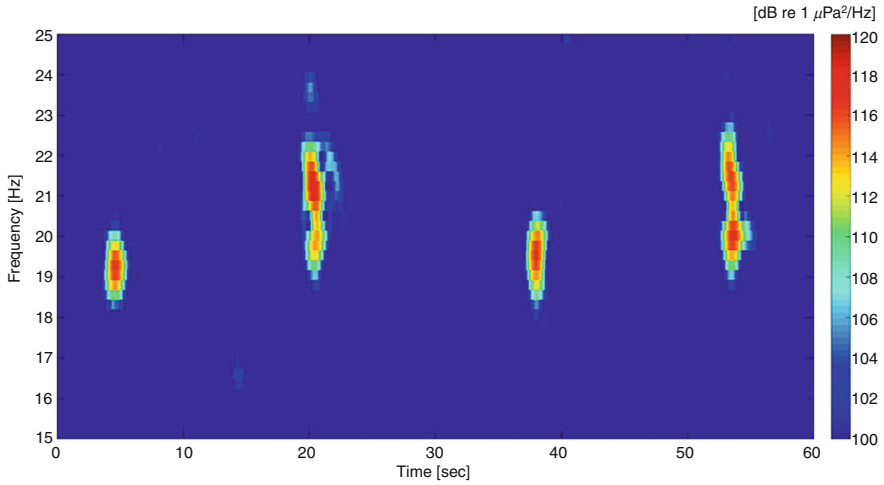



**Fig. 1** Spectrogram of a brief note sequence (BB-C-BB-C) from a song recorded in the Ionian Sea (Central Mediterranean Sea). The spectrogram was created in MATLAB© (sampling frequency, 2 kHz; 8192 FFT points, Hanning window, overlap 98%). Time resolution, 0.04 s; frequency resolution, 0.24 Hz. (Spectrogram from Sciacca et al. (2015))

hierarchical structure. Hence, there is not a common definition for the units that compose a song across different taxa. To perform accurate analysis of any song structure, and of the potential information conveyed, it is essential to characterize its constituent acoustic units (for a review, see Kershenbaum et al. 2016).

Watkins et al. (1987) attributed the definition of fin whale "songs" or "bouts" to the long sequences of stereotyped "20-Hz pulses" spaced apart by at least 2 h. Songs may last for hours, alternated by resting periods of variable length. Several studies described songs as repeated sequences mainly composed by 20-Hz pulses, hereafter called "20-Hz notes" for functional differentiation between a generic call and calls repeated in songs (Castellote et al. 2012a; Weirathmueller et al. 2017).

Most fin whale song studies focus only on the characterization of 20-Hz notes. These notes can be distinguished into a "classic" (C) note (a downswept signal from 23 to 17 Hz of about 1 s duration) and a variation of this note type called "backbeat" (BB), with a constant frequency in the range of 18–20 Hz, lasting about 0.8–1 s (Clark et al. 2002). Following the definition from McDonald and Fox (1999), two consecutive C and BB note types within a song can be defined as "doublet calls." An example of a "BB-C" doublet alternation of 20-Hz song notes produced by Mediterranean fin whales in the Ionian Sea is shown as a spectrogram in Fig. 1.

As for song produced by some other baleen whale species, fin whale songs are thought to be produced by male individuals, with a display function during courtship (Croll et al. 2002).

The acoustic characteristics of the 20-Hz notes (e.g., peak frequency, duration, and intensity), the frequency bandwidth (bw), and the timing patterns seem to vary to a limited extent within a sequence (Clark et al. 2002). Variations in the inter-note

intervals (INIs) (see section "Inter-Note Intervals (INIs)") have been more commonly observed. These changes may occur seasonally within the same population (Morano et al. 2012), but they seem to be higher between different populations, and they seem to be related to geographic location (Širović et al. 2009; Simon et al. 2010; Castellote et al. 2012a).

## Note Detection and Classification Methods

Passive acoustic monitoring (PAM) data can be used to improve knowledge and conservation of baleen whale species and of their endangered populations, for species such as the North Atlantic right whale (*Eubalaena glacialis*) (Mellinger 2004) or the endangered Mediterranean fin whale (Pintore et al. 2021). However, the analysis of large datasets derived by long-term PAM represents a growing challenge. The improvement of standardized tools for the acoustic detection of fin whale sounds represents a necessary step to acquire knowledge on the species' distribution, behavior, population structure, and abundance (Schall and Parcerisas 2022). Traditionally, the detection of fin whale vocalizations has been performed by visual inspection of waveforms, spectrograms, and power spectra analyses. These methods are long and laborious and lead to highly variable errors, depending on inter- and intra-analyst annotation capabilities (Mouy et al. 2014; Schall and Parcerisas 2022). In contrast, automatic acoustic recognition methods have many advantages, especially when enhanced with acoustic tracking techniques (Mellinger and Clark 2000). Since the early 2000s, many software tools have been developed with the aim of automatically detecting and classifying fin whale calls over long data series (e.g., Mellinger and Clark 2000; Urazghildiiev and Clark 2007; Mouy et al. 2014; Best et al. 2022). Most of these methods generally targeted the detection of 20-Hz pulses, which are the best-known signals in the fin whale song repertoire. However, the variability of acoustic habitats and background noise conditions and the rapidly growing amount of PAM data ask for increasingly efficient detection and classification methods.

### Detection Tools for 20-Hz Song

Detection software performances strongly depend on the complexity of the target signal as well as on background noise and sound propagation characteristics in the area (Mouy et al. 2014). Following the Baumgartner and Mussoline (2011) definition, fin whale call detectors can be distinguished in two main types. In the first type, the spectrogram is inspected by using a detection function (DF) represented by a template or kernel (i.e., a spectrographic picture of a certain call). In this case, detections occur when the DF exceeds a given threshold. Detectors of this type include spectrogram cross-correlation (Mellinger and Clark 2000; Mellinger 2001), neural networks (Mellinger 2004), convolutional neural networks (CNN) (Best et al. 2022), and banks of two-dimensional linear finite impulse response filters

(Urazghildiiev and Clark 2007). The second type of detectors works by setting a series of known parameters a priori, such as minimum frequency, maximum frequency, and duration of the signal to be detected. The acoustic signal is then investigated in the search of these parameters. All the signal portions that fit well into the pre-established attributes, based on the similarity to call-type models, are then classified as detections (Baumgartner and Mussoline 2011). Examples of this type of detectors are time-frequency methods based on edge detectors or instantaneous frequency tracking algorithms (Mouy et al. 2014) or detectors based on energy or power spectral density (PSD) signal-to-noise ratio (SNR) (Sciacca et al. 2016; Pintore et al. 2021; Schall and Parcerisas 2022). To account for low-frequency noise (<1 kHz) variability, in many of the existing tools, diffuse narrowband noise (e.g., ship noise) is usually minimized by subtracting a running mean or median from each frequency band of the spectrogram (Mouy et al. 2014). This technique is referred to as pre-whitening or normalizing (Baumgartner and Mussoline 2011). Still, changes in background noise and sound propagation conditions may affect detection capabilities (see section "Song Detection and Noise").

## Classification of Classic and Backbeat 20-Hz Song Notes

Classification of fin whale 20-Hz notes has been usually performed by human operators using spectral measurements. Discrimination between two note types has been mainly based on frequency bandwidth or frequency range. The main distinction performed to study fin whale song occurs between "classic note" (C) and "backbeat" (BB) (Clark et al. 2002). Apparently, the frequency limits of the two note types seem to differ, but there is not a clear cutoff between them. However, when both note types occur, the classic note is recognized as the note with a higher frequency compared to the backbeat (Helble et al. 2020). The lack of a universally accepted definition for the 20-Hz notes is probably due to the variability of the recording settings, geographic locations, and long-term trends in populations' vocal repertoire. Moreover, since the identification of different note types is generally performed by non-standardized methods based on fast Fourier transform (FFT) (e.g., spectrographic representations), time-frequency resolution capabilities are not consistent throughout literature. This causes a high inconsistency in the description of both time and frequency features of 20-Hz notes.

Only a few quantitative methods have been described for 20-Hz note classification. Rankin et al. (2018) proposed the application of a density clustering algorithm developed by Rodriguez and Laio (2014) to the classification of 20-Hz notes, using as spectral variables: centroid frequency, peak frequency, maximum frequency, bandwidth, and centroid peak difference (difference between the centroid frequency and the peak frequency). The same method was employed by Archer et al. (2020) who combined unsupervised density clustering to find note types using supervised classification. A fin whale "doublet" call detector was recently developed in MATLAB (MathWorks, Natick, MA) by Constaratas et al. (2021). Here the detection is based on the cross-correlation between a "doublet" call template and the

spectrogram of the whole signal (Constaratas et al. 2021). This approach can be useful to speed up the analysis of call sequences in highly stereotyped patterned songs. Helble et al. (2020) used a generalized power law detector to identify the start and end time of each note and to create a template of the signals. Then, the authors developed an automated classifier to categorize each detected signal as either a C or BB note, depending on the energy content of the different frequency bins. They found that it is possible to assess a threshold separating the two note types by using the "energy ratio" calculated as the ratio between signal template energy summation in the 10–23 Hz band and in the 10–50 Hz band. More recently, Best et al. (2022) fitted a two-component Gaussian mixture model to measured center frequencies (see section "Frequency Features") of Mediterranean fin whale 20-Hz notes, obtaining a threshold of 19.96 Hz to discriminate between the two types. This work showed that the center frequency may be used as a fundamental parameter to discriminate between note types. The main limitations to the establishment of a common classification framework are the definition of standardized protocols and metrics for note and song measurement and the assessment of background noise influence on established methodologies. Below is a review of the main approaches to song note classification and background noise assessment applied in literature.

## How to Define Fin Whale Songs

The differences in fin whale populations' vocal repertoire and the difficulties in obtaining reliable and common classification methods have resulted in a vast number of different approaches in exploring the features of fin whale songs. Furthermore, it is not yet known what measurable differences are between songs and irregular sequences of calls. Often the latter have been defined as sequences of calls with variable frequency features and inter-pulse interval (IPI). Irregular sequences sometimes include pulses with a peak frequency higher than the classic 20-Hz notes, ranging from 16 to 40 Hz (McDonald et al. 1995; Weirathmueller et al. 2017).

Irregular call sequences have been documented in the presence of transiting groups of whales interacting acoustically, each one emitting a different call type (McDonald et al. 1995). Soule and Wilcock (2013) recorded and tracked (in Northeast Pacific Ocean) several sequences in the 16–30 Hz frequency range, with peaks at 17, 19, and 24 Hz and with irregular IPIs. These irregular sequences contained calls with frequency components exceeding 24 Hz, defined as "24 Hz calls." These calls were recorded in association with the acoustic tracking of two near whales traveling together. The authors propose that these calls may be produced by immature males and that they could indicate social behavior other than courtship display. This work supports the hypothesis that the definition of "songs" relies on their biological function and that only long stereotyped 20-Hz note sequences (sometimes coupled with overtones) may be associated to male courtship display (Croll et al. 2002). However, given the very low number of observations, many uncertainties still exist concerning irregular call sequences. Hence, most work has focused on the analysis of regularly spaced call sequences (i.e., songs) including

"20-Hz notes" with identifiable and patterned repetitions (e.g., Weirathmueller et al. 2017; Širović et al. 2017; Constaratas et al. 2021). The most common steps in this analysis are the classification of note types in terms of frequency features and the measurement of the INIs. Recent studies (Helble et al. 2020; Best et al. 2022) considered a combination of these two approaches for the identification of recurring patterns and the study of their variation over time. An overview of the main metrics used for these purposes is reported in the next paragraphs. Call duration has been less commonly used to describe songs; hence it will not be discussed further in this chapter.

## Frequency Features

Several frequency parameters have been used to describe fin whale song notes, with only a few consistencies over the years and geographical areas. Table 1 shows a review of the frequency features of fin whale 20-Hz and higher-frequency (HF) song notes measured in research articles between 1987 and 2022. As shown in Table 1, one of the most investigated features is the peak frequency ($F_p$) corresponding to the frequency of the signal's maximum power. Taking as reference $F_p$, the frequency band attenuated by 3 decibels (dB) can be calculated, by finding the two frequencies where the spectrum level reaches 3 dB below the peak value (Crane and Lashkari 1996; Helble et al. 2020). This parameter shows how energy concentrates around the peak frequency. A small 3-dB bandwidth indicates a "tone-like" signal, and a wider value shows a more impulsive nature, respectively, meaning that the signal energy is more concentrated in a few frequency bins or more uniformly distributed over the entire band (Crane and Lashkari 1996). The center frequency ($F_c$) is the frequency value corresponding to half the distance between the initial ($F_{start}$) and the final ($F_{end}$) frequency bins. As described in section "Classification of Classic and Backbeat 20-Hz Song Notes," as an example, Best et al. (2022) show that $F_c$ may be better suited than $F_p$ for the discrimination between the two note types (C and BB). Also, the highest ($F_{max}$) and the lowest ($F_{min}$) frequency values and call frequency contours have been sometimes estimated manually by spectrogram annotation. Knowing $F_{max}$ and $F_{min}$, the overall note frequency bandwidth can be measured as the distance between these two parameters in Hz (see Table 1).

Rankin et al. (2018) proposed a method to reduce uncertainties across different analysts when measuring spectral features using spectrograms. Particularly, these authors proposed a spectrogram annotation method which allows the user to establish the percentage of signal energy to include in the selection. The same method was adopted by Archer et al. (2020) to account for differences in background noise levels among different deployment locations. All frequency features are normally measured by means of a FFT, and hence their precision depends on the frequency step selected. However, having considered this, energy-based parameters such as $F_p$, $F_c$, and 3-dB bandwidth produce more easily replicable measurements than contour-based ones.

**Table 1** Review of frequency features

| Authors | Year | Area | $F_C$ | $F_P$ | $F_{start}$ | $F_{end}$ | $F_{max}$ | $F_{min}$ | bw | 3 dB bw | HF |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Watkins et al.** | 1987 | Various | | | 23 | 18 | 25 | 17 | 5 | | |
| **Thompson et al.** | 1992 | Gulf of California | | | 30–58 | 17–26 | 58 | 17 | 19 | | 18–310 (range) |
| **Clark et al.** | 2002 | NW Med | | 21(C) | | | 24 | 19 | | 5 (C) | |
| | | | | 20 (BB) | | | 24 | 19 | | 3 (BB) | |
| **Širović et al.** | 2009 | Antarctica | | | | | 28 | 15 | 13 | | 88 and 99 (Fp) |
| **Castellote et al.** | 2012a | NA | 22 | 21 | | | | | 6 | | |
| | | W MED | 22 | 21 | | | | | 5 | | |
| **Archer et al.** | 2020 | N Pacific and Chucki Sea | 24 (C) | 23 (C) | | | 34 (C) | 17 (C) | | | |
| | | | 19 (BB) | 19 (BB) | | | 26 (BB) | 14 (BB) | | | |
| **Helble et al.** | 2020 | CP (Hawaii) | | 23 (C) | | | | | | 13 (C) | |
| | | | | 16 (BB) | | | | | | 6 (BB) | |
| **Constaratas et al.** | 2021 | New Zeeland | | 21–24 (C) | | | | | | | 74–80 |
| | | | | 19–22 (BB) | | | | | | | 64–91 (Fp) |
| **Wood and Širović** | 2022 | Antarctic Peninsula | | | | | | | | | 85–89 (FP range) |
| **Best et al.** | 2022 | NW Med | 18–22 | | | | | | <6 | | |

A review of the main frequency features measured between 1987 and 2022 on fin whale 20-Hz notes and overtones (HF notes). $F_c$, $F_p$, $F_{start}$, $F_{end}$, $F_{max}$, and $F_{min}$, respectively, indicate the central, peak, starting, ending, maximum, and minimum frequency values expressed in Hz. Bandwidth (bw) indicates the difference in Hz between $F_{max}$ and $F_{min}$. The 3-dB bw is the width of the band at 3 dB from Fp. C and BB are classic pulse and backbeat notes. All entries are mean values, rounded to the closest integer value for comparison. Helble et al. (2020) only report median values

## Inter-Note Intervals (INIs)

The time lag between two consecutive 20-Hz notes within a song can be defined as inter-note interval (INI). Among all 20-Hz note features, the INI is the most variable feature between song repertoires and different fin whale populations (Castellote et al. 2012a; Pereira et al. 2020). Many different protocols are described in the literature to define the variation in the intervals between the two main fin whale 20-Hz note types. Besides the use of a variety of PAM systems (from deep-sea fixed observatories to bio-logging tags), different time features have been considered as a reference for the measurement of INI in fin whale song studies.

Table 2 shows a review of INI measurements described in studies between 1987 and 2022, performed as differences between various time features of consecutive 20-Hz notes: start times ($T_s$) are the most commonly used features, corresponding to the beginning of each note; central note times ($T_c$) have been sometimes used, by measuring the center of the note as it is computed and selected from the spectrogram (Castellote et al. 2012a; Rankin et al. 2018); and peak times ($T_p$) have been also considered by some authors, and these can be defined as the times of the maximum pressure level within a given frequency band (Clark et al. 2002; Best et al. 2022).

The time features of the notes are obtained by analyzing either the waveform (e.g., maxima of cross-correlation lag times between two notes), the energy envelope (e.g., the time of the maximum energy within a time window – $T_p$), or the spectrogram (Pereira et al. 2020) of a given acoustic signal. INI measurements have been performed by user annotation (directly from spectrogram image processing) or by automatic analysis (defined also as "automatic time measurement" or ATM in Table 2). Some authors adopted a combination of automatic signal detection/

**Table 2** Review of time features for INI measurement

| Authors | Year | Area | T res | $T_s$ | $T_c$ | $T_p$ | ATM |
|---|---|---|---|---|---|---|---|
| **Watkins et al.** | 1987 | Various | n.s. | x | | | |
| **Thompson et al.** | 1992 | Gulf of Cal. | n.s. | x | | | |
| **Clark et al.** | 2002 | NW Mediterranean Sea | 0.1 | | | x | |
| **Morano et al.** | 2012 | W- N Atlantic | 0.15 | x | | | |
| **Castellote** | 2012a | N Atlantic and W Med | n.s. | | x | | x* |
| **Delarue et al.** | 2013 | Bering Sea and Chucki Sea | 0.064 | x | | | x* |
| **Oleson et al.** | 2014 | North Pacific | n.s. | x | | | x* |
| **Rankin et al.** | 2018 | Pacific Ocean | 0.025 | | x | | x* |
| **Furumaki et al.** | 2021 | Chucki Sea | 0.37 | x | | | |
| **Wood and Širović** | 2022 | W Antarctic Peninsula | 0.1 | x | | | |
| **Best et al.** | 2022 | NW Med | 0.03 | | | x | x |

A review of the time features used for inter-note interval measurement in different fin whale song studies worldwide. The study areas and time resolution: expressed in seconds (s) ("n.s." if not specified) are reported for each study. The alternative use of note starting time ($T_s$), central note time ($T_c$), or note peak time ($T_p$) is described, together with the use of automatic time measurement (ATM) systems, including partially automatized spectral analysis (marked with *)

selection methods and manual measurements, here defined as partially automatized spectral analysis (Table 2). In this last case, MATLAB (MathWorks©)-based custom algorithms and spectrogram editing software such as Triton (Oleson et al. 2014; Wood and Širović 2022) or XBAT (Castellote et al. 2012a) or other commercial software such as SpectroPlotter (JASCO Applied Sciences©) (Rankin et al. 2018) were used. Another fundamental difference in the measurements of INIs performed across literature is the time resolution of the spectral analysis performed, relying on selected FFT parameters (e.g., window length and overlap) and on the sampling frequency of the data. These features appear highly variable, and sometimes they are completely omitted (more commonly in earlier studies), as reviewed in Table 2.

## Pattern Identification

Following the measurement of note frequency features and INIs, the occurrence and definition of recurring patterns in spectral features and note alternation can be searched within each song. Watkins et al. (1987) described the long 20-Hz patterned series as calls with a single repeated INI value ("singlet") or sometimes as "doublets" of two alternating INIs. Thompson et al. (1992) also defined the patterned 20-Hz pulse series as pairs or doublets, alternating short and long INI duration. They also showed that doublet and singlet patterns may alternate within the same song. The understanding of these patterns for many fin whale populations' repertoires has increased over the years. "Triplet" patterns were also identified. Triplets are note sequences having at least two distinct INIs alternating, with one of them repeated two or more times (Širović et al. 2017). The definition of doublet, singlet, and triplet patterns is useful to group the songs as a function of their INIs and to understand the possible seasonal fluctuations (Širović et al. 2017) and inter-annual variations in different geographical areas and populations.

Recently, it has been shown that there is a strong relationship between note type and INI (Helble et al. 2020; Best et al. 2022). As an example, Helble et al. (2020) showed that in the "BB-C doublet" songs analyzed in their study, the BB-C interval has a shorter INI than the C-BB interval and that both doublet and singlet songs can occur with a unique note type such as BB-BB and C-C. Other authors showed that there may be more diversity in the alternation of note types than simple singlets or doublets, as observed in Mediterranean fin whale songs (Best et al. 2022). Hence, the traditional definition of singlet, doublet, and triplet songs may not be sufficient to estimate differences in song repertoires and that note type should be further considered for future studies. This is relevant when songs are used to isolate different populations, to define individuals' acoustic features (Širović et al. 2017), or to monitor long-term changes in population acoustic behavior. There is also some evidence that song patterns change over time. Seasonal variations (e.g., presence/absence of songs and INI increases) have been observed in western North Atlantic (Morano et al. 2012) and North Pacific oceans (Clark et al. 2002; Oleson et al. 2014; Weirathmueller et al. 2017). A synchronous increase in INIs was instead observed across the North Pacific during late fall and winter, and it was suggested as a feature

of the mating display (Oleson et al. 2014). Similar variations were observed by Morano et al. (2012) in the North Atlantic, and these authors suggest that seasonal INI variations may be associated with different purposes of communication. In addition, INI and frequency features have been shown to exhibit inter-annual variations worldwide, with a linear increase of INIs and a constant decrease of peak and centroid frequencies in some fin whale populations, as reviewed by Best et al. (2022). Simultaneous INI and peak frequency inter-annual shifts were found by Weirathmueller et al. (2017) and by Wood and Širović (2022). Only inter-annual INI variation was found in fin whale songs recorded in Northwestern Mediterranean Sea with no evidence of seasonality in song production and INI changes (Clark et al. 2002; Castellote et al. 2012a; Best et al. 2022), suggesting that differences across populations might arise culturally (Best et al. 2022).

## Noise Assessment in Fin Whale Song Studies

Most of the acoustic noise produced by large ships and seismic sources overlap with fin whale emission frequency band (Castellote et al. 2010; Sciacca et al. 2016). Noise may heavily alter fin whale signal detection and classification, especially when signal detection techniques based on template correlation or similar methodologies are applied (see section "Detection Tools for 20-Hz Song") (Weirathmueller et al. 2017; Best 2022). The acoustic masking induced by noise, including both natural and anthropogenic noise sources (waves, earthquakes, currents, sonars, vessels, etc.) not only alters whales' intraspecies communication, but it may also affect the detection space, thus reducing our capability to study fin whale song notes (Helble et al. 2020; Sciacca et al. 2015; Širović et al. 2015). A brief review of the studies describing background noise measurements, detection range estimation, and their variation in relation to fin whale song studies is reported in the next paragraphs.

### Measuring Noise Levels

To estimate communication masking means to define how much noise reduces the ability of a receiving animal to recognize the sounds of a conspecific (Clark et al. 2009). Similarly, PAM detection and classification capabilities depend on the possibility to receive and discern target signals from interferences, in the constant presence of random diffuse noise (Zimmer 2011).

Therefore, it is initially important to define the most suited metrics to measure background noise levels, considering target signal characteristics and study aims. Merchant et al. (2015) provided a thorough review of the main signal processing techniques, statistical methods, and metrics to measure acoustic habitats for PAM purposes. Assuming that the ability of PAM systems to accurately record sound is limited by the properties of the system itself, the authors suggest statistical analyses of sound levels to compare different acoustic habitats and time-series representations to describe discrete events and trends in sound levels. Concerning statistical sound-

level analysis, the use of root mean square (RMS), median, and percentile distribution appears particularly useful to assess the potential extent of acoustic masking (Clark et al. 2009). Furthermore, the use of spectral probability density (SPD), power spectral density (PSD), and 1/3-octave frequency band analysis is also suggested and described for spectral representation (Merchant et al. 2015). Time-series analysis is instead effective to analyze transient biological and anthropogenic sounds. The most used representation in time-series is the spectrogram, which shows how PSD sound levels vary as a function of the time. Spectrograms can be used to track trends in sound levels over specific frequency bands (Merchant et al. 2015). As an example, long-term spectral averages (LTSAs) are commonly used to compare sound levels over large time scales (Širović et al. 2013, 2015; Wood and Širović 2022). A small number of studies describe background noise measurements with the aim of evaluating noise impact on fin whale song note detection, acoustic behavior, and intraspecies communication. Castellote et al. (2012b) compared to background noise levels at four different recording locations in the Northwestern Mediterranean Sea. They investigated the occurrence of changes in the acoustic behavior of fin whales in response to shipping and airgun noise. To compare noise levels to the different sites and to correlate them with fin whale behavior, average received levels (root mean square dB re 1 μPa/5-min) were estimated, and PSD percentiles of background noise were computed for the frequency range 1–50 Hz, with a frequency resolution of 1 Hz. Other examples are shown in the next section "Song Detection and Noise," focusing mainly on the effects of noise on song detection capabilities.

## Song Detection and Noise

Acoustic detection is influenced by background noise, sound transmission path, and distance between the emitter and the receiver (Clark et al. 2009). To understand the limits of fin whale song detection studies and enable comparison across different areas, it is essential to assess the variability of detection ranges for each site and recording context and to assess transmission loss (TL) by modeling for each relevant context (Širović et al. 2015). Even discrete events, such as the passage of a ship, can cause changes in background noise levels. As a result, the choice of the metrics to compare detection range variations and fin whale songs between different sites should preferably consider both the short-term and the long-term variabilities of noise levels and of the overall soundscape dynamics. Temporal and spatial ocean sound dynamics affect sound propagation and influence the acoustic detection area of fin whale vocalizations (Miksis-Olds et al. 2015). This variability may also affect the measurement of the acoustic features of fin whale songs. Different note types with different frequency characteristics could have different propagation features. Such differences may also occur for the different frequency components of the potential noise sources occurring in the detection area. Hence, the study of song patterns should consider background noise levels and propagation conditions along with their spatial and temporal variability. To do so, several articles quantified the occurrence of whale calls by comparing the acoustic power in a frequency band

including the whole typical 20-Hz pulse signal, with different neighbor noise frequency bands above and below fin whale band (Širović et al. 2009; Simon et al. 2010). Širović et al. (2015) calculated average background noise levels (in 1 Hz bins) averaged over 5 s and as daily means at the recording locations to model detection range at different background noise conditions for each site. Finally, these authors used average background noise levels varied from −5 to +10 dB to test the variability of the detection ranges. They observed that fin whale detection range varied by 7–22%, while the detection area varied by 1–12% under different noise assumptions.

Stafford et al. (2007) modeled the maximum detection range for different baleen whales' calls in the Gulf of Alaska. They modeled transmission loss (TL) at selected frequencies, characteristic of each call type. For fin whales, the authors used a reference frequency of 25 Hz and source levels and monthly average ambient noise levels from literature (Stafford et al. 2007). Samaran and colleagues (2010) also modeled the range over which blue whale low-frequency calls (frequency modulated three-component 28 to 18 Hz calls) can be expected to propagate. These authors considered how modeled detection range changed with the monthly received levels, frequency, and source level of whale call in relation to the seasonal ambient noise levels received at the station and the seasonal propagation conditions along the transmission path from source to receiver. On the other hand, ocean dynamics may consistently change with varying operating depth and season of the year, depending on the receiver location.

Sciacca et al. (2015) also investigated the correlation between background noise levels and acoustic detection of fin whale in the Mediterranean Sea. In this study, the recording location was at high depth in Central Mediterranean Sea, where variations of the sound velocity profile (SVP) are significant only above the seasonal thermocline. The recorder was installed in an almost flat sea bottom, and due to the inter-seasonal homogeneity of the SVP, this allowed reliable estimates of the propagation of low-frequency sounds from a source close to the surface (Sciacca et al. 2015; Viola et al. 2017). Sciacca et al. (2015) measured median values and percentile distribution of noise PSD as useful indicators of the typical background noise trends. The SPL in the 1/3-octave band centered at the frequency of 20 Hz was selected for noise analysis, and the percentile distribution of integral noise amplitude (dB re 1 μPa) in this frequency interval was measured every 10 min. Fin whale calls were detected visually by trained operators through the investigation of the spectrograms. Measured values of median noise were compared to the presence and absence of fin whale calls, showing that 20-Hz calls were more often detected in lower background noise conditions. By applying a transmission loss model, the typical detection range was also estimated, and its variation was studied at the different percentiles of noise levels recorded. The estimated typical detection range was reduced by about 10 km with a 6 dB increase in the median noise levels.

In addition, acoustic behaviors, i.e., source levels and vocalizing depths of baleen whales, should be better understood (Stafford et al. 2007) to perform reliable studies on low-frequency communication sounds. In the past years, several studies have been conducted to estimate the source levels for fin whale vocalizations, which

might vary depending on their biological function (Charif et al. 2002), and different estimates are available for the 20-Hz pulses worldwide: 159–184 dB re 1 μPa at 1 m (Charif et al. 2002), 186 dB re 1 μPa at 1 m (Watkins et al. 1987), 189 ± 4 dB re 1 μPa at 1 m (Širović et al. 2007), 189 ± 5.8 dB re 1 μPa at 1 m (Weirathmueller et al. 2013), and 190.5 ± 7.4 dB re 1 μPa at 1 m (Garcia et al. 2019).

It has been shown that higher-frequency calls may exhibit completely different source levels with respect to the 20-Hz notes. As an example, fin whale 130 Hz upsweep vocalizations may show a source level about 20–25 dB lower than 20-Hz pulses (Simon et al. 2010; Garcia et al. 2019). It follows that these signals have different detection ranges, and their frequency band is also subject to different noise-level conditions. They may be undetected in spectrograms where the 20-Hz notes are visible (Garcia et al. 2019). Similarly, differences of 3–7 dB in the received levels of C and BB 20-Hz pulses have been described by Clark et al. (2002) and Weirathmueller et al. (2013). Therefore, C and BB pulses from the same emitting whale could propagate differently and be diversely subjected to background noise influence.

## Conclusions

The establishment of common metrics and methods to define the patterns of fin whale songs worldwide, accounting for both INI and note type, will favor the assessment of inter-seasonal and inter-annual variations in populations' repertoires. This chapter underlined that information concerning time-frequency resolution in the analysis performed on fin whale song notes should always be described to enable large-scale and long-term comparisons.

As shown, custom-developed algorithms are often used for song analysis and noise measurement. This points out that available commercial and open-source software tools are still not sufficiently adaptable to the wide range of recording contexts and dataset specificities. Furthermore, custom algorithms should be fully described in scientific articles, tested, and optimized (e.g., to reduce false-positive detections) for the specific dataset (Schall and Parcerisas 2022). The improvement of knowledge devoted to the conservation of fin whale populations (as enabled by PAM) would certainly benefit of the sharing of these tools and information, following FAIR principles (findability, accessibility, interoperability, and reusability) (Wilkinson et al. 2016). New automatic methods for note classification and pattern identification may be also developed by using deep-learning methods such as CNN (Best et al. 2022), which are increasingly spread.

A low number of fin whale song studies focused on background noise measurement and available detection space variations. This lack can be solved by selecting a subsample of available note sequences, based on higher SNR, to assess indexes of call abundance or note spectral features (Širović et al. 2015; Helble et al. 2020; Kowarski and Moors-Murphy 2021). Still, when measuring fin whale songs, it would be important to perform statistically relevant measurements of background noise levels. This is crucial to assess the typical detection range on significant time

scales to define whales' distance with respect to the receiver. With a view to fin whale song studies, it is also important to know the source levels of the different note types and their variability. This also enables proper definition of song notes' acoustic features and comparison between different fin whale populations.

Based on the review performed, future studies may also focus on finding a clearer definition of the differences between songs and irregular sequences, coupling PAM data with behavioral observations from visual surveys and bio-logging tags. This will improve our ability to interpret the biological significance of songs. It is commonly accepted that male baleen whales use features of their songs to promote their physical condition, as proved in other species (Clark et al. 2019). However, this hypothesis does not have exhaustive demonstrations in fin whale singing observations. Many questions about the ecological and biological significance of songs and related energetic costs are still wide open.

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# Acoustic Baseline Monitoring of Marine Mammals Offshore West Africa: Using an Autonomous Surface Vessel

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David Hedgeland and Chris Pierpoint

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**Abstract**

A marine mammal baseline monitoring program was initiated offshore São Tomé and Príncipe in 2018–2019. An AutoNaut autonomous or uncrewed surface vessel (USV) was deployed by Seiche Ltd. The aim of this study was to identify occurrence and distribution of marine mammals as part of a wider baseline data

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collection program, which also included the use of fixed location acoustic recorders, moored to the seabed for monitoring at specific locations. The USV provided an opportunity to collect data over an area of 6840 $km^2$ between the two islands and off the east coast of São Tomé. The minimal physical footprint of the USV equipment shipment, deployment, and autonomous or remote control provided substantial cost and safety advantages compared to conventional vessel-based towed passive acoustic monitoring (PAM) methods—which would otherwise have involved logistical resources and challenges. Assembly, launch, and recovery activities were based at a location with access to a small sandy beach. A two-person team with local support mobilized, serviced, and demobilized the USV. The mobility and unobtrusive nature of the AutoNaut platform enabled successful acquisition of PAM data over a wide area. Vocalizations of humpback whales, sperm whales, beaked whales, and delphinid species were recorded throughout the survey.

**Keywords**

Marine mammals · AutoNaut · Passive acoustic monitoring · Baseline · Autonomous · Uncrewed surface vessel · Physical footprint · Safety

## Introduction

The Democratic Republic of São Tomé and Príncipe is a country of two islands in the South Atlantic Ocean, 200 km off the west coast of Africa. Among a growing number of visitors to the islands are humpback whales, other marine mammal, and sea turtle species. Previous studies have identified the seasonal presence of humpback whales (*Megaptera novaeangliae*) in the waters around the islands (Carvalho et al. 2011). Humpback whales are known to visit the region between July and November each year. Less information is available for the local presence of other marine mammal species that occur in the wider West Africa region. The region is inhabited by a diverse range of cetacean species (Bamy et al. 2006; de Boer 2010; Weir 2011). These include at least one other baleen whale Bryde's whale (*Balaenoptera brydei*); sperm whale (*Physeter macrocephalus*); dwarf sperm whales (*Kogia sima*), Cuvier's beaked whale (*Ziphius cavirostris*); several "blackfish" delphinids such as short-finned pilot whale (*Globicephala macroryhnchus*) and false killer whale (*Pseudorca crassidens*); several dolphins including bottlenose (*Tursiops truncatus*), Clymene (*Stenella clymene*), and pantropical spotted dolphin (*Stenella attenuata*).

Most of the available information on marine mammal occurrence in West Africa has been gathered by visual observation, either from onshore locations looking out to sea or by personnel going out to sea onboard vessels or by identification of stranded animals. Visual monitoring continues to provide important information on the

seasonal presence/absence and the distribution of species, but has its limitations and potential sources of bias—visibility and sea state conditions, group size, and the surfacing behavior of animals strongly influence the likelihood of their detection at the surface. Both dedicated and opportunistic observations can also be limited in temporal duration (often focused on the seasonal presence window) and spatial extent. In contrast, autonomous passive acoustic monitoring (PAM) technologies provide opportunities to collect marine mammal data continuously over extended periods and distances, independently of weather and sea conditions. The fundamental bias of these methods is that only animals that vocalize at sufficiently high level above background sounds in the vicinity of the hydrophone recorder, are available for passive acoustic detection. A well-chosen PAM system, however, can offer a cost-effective means of collecting long-term and/or wide areal datasets, the analysis of which can address multiple objectives. For example, these data may be applied to validate existing understanding of known species seasonal presence, to confirm the presence or absence of other species, and to understand baseline soundscape conditions and characteristics.

The wider study of marine mammal occurrence and soundscape characteristics combined the use of fixed passive acoustic recorders and an autonomous or uncrewed surface vessel (USV). Fixed recorders were placed at three nearshore locations off the east coast of São Tomé and collected data over a 12-month period. Concurrently, the USV collected data over shorter time periods but over a wider area between the islands of São Tomé and Príncipe. While the use of fixed PAM recorders has become increasingly common in recent years, USVs are a relatively new maritime platform for marine mammal monitoring. Therefore, in addition to targeted species and soundscape monitoring objectives, this study aimed to demonstrate the application of USV technology for marine mammal PAM.

The fixed PAM equipment was selected based on past experience and performance of their use, ease of manual handling, deployment, and recovery via medium sized vessels that would be available locally through tourism, hotel, and diving organizations and facilities.

Nearshore locations of the fixed PAM units were identified and selected following reviews of available information, discussions with academic researchers who had previously worked in the area and engagement with local groups and fishing communities. Aside from identifying the most likely places humpback whales would be detected, distances offshore were constrained by transit durations, with vessel availability and safe operating conditions limited to daylight hours.

An engagement and information sharing program was implemented with project personnel meeting with local organizations and fishing communities prior to and between field visits to explain the purpose of the studies and specifically what the acoustic monitoring equipment looked like. Throughout the project, continued engagement with partner personnel, local national agencies, hotel and logistics staff all helped to ensure the project was safely and successfully completed.

## AutoNaut USV and PAM System

A USV is effectively a remotely controlled mobile platform, from which hydrophones can be deployed to record underwater sound. The absence of crew onboard reduces safety exposure to personnel. The USV is controlled by a pilot located onshore, either in the same country of operation or elsewhere. The AutoNaut is a 5-m, wave-propelled USV, which towed a matched, linear pair of broadband hydrophones with 0.5 m separation, mounted on a 27-m cable and was piloted via a satellite link from the United Kingdom (Fig. 1). The PAM system was powered by 12 V batteries that were charged using solar panels mounted on the AutoNaut's deck. Acoustic and positioning data were recorded onboard the USV using the Seiche μPAM sound recorder running the PAMGuard software. The sensitivity of the ceramic hydrophone elements was −196.5 dB re: 1 V/μPa, and their preamplifier gain was 40 dB. Acoustic signals received at the hydrophones were digitized using a National Instruments NI-9222 DAQ at a sampling rate of 125 kHz per channel. The PAM system had an end-to-end frequency response of 25–62,500 Hz(−3 dB). The signals from both hydrophones were recorded continuously, as 16-bit wav-format files. Extraction and analysis of marine mammal vocalizations was carried out offline, once the USV had returned to shore and the data were downloaded.

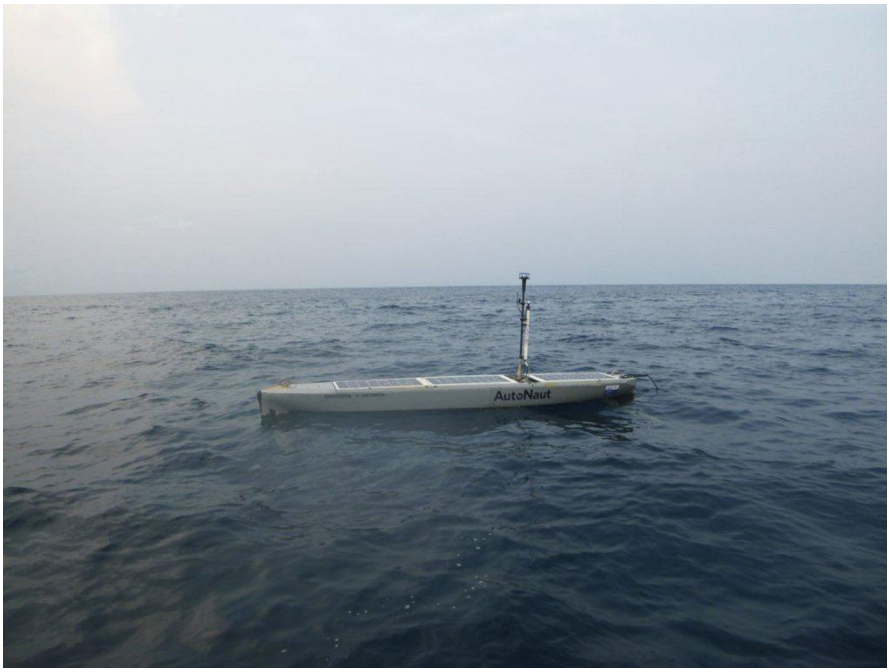



**Fig. 1** AutoNaut offshore São Tomé

## Survey Logistics and Monitoring Effort

The AutoNaut was mobilized from the grounds of a local hotel at Santana on the east coast of São Tomé. The USV and a launch and recovery trolley arrived on site in three packing containers and were assembled onsite. Vehicle movement via the trolley from a workshop area to the beach launch site was managed in close collaboration with hotel management and staff. Deployment was implemented by maneuvering the trolley into the water until the USV was able to float free. Its nearshore movements were controlled via a Wi-Fi controller, prior to hand-over to the remote pilot who thereafter navigated using an Iridium satellite connection to the vessel. In addition to positional command and control information, regular status messages were relayed via the satellite link to monitor PAM system functionality.

The marine mammal survey was planned as a series of equal-angle zigzag line transects to distribute survey effort systematically across the survey area. Transit routes were planned between the offshore survey area and the launch and recovery site at Santana, to provide additional information on marine mammals in the nearshore areas. The predetermined route was 1725 km in length, and the transect lines within the survey area were to be repeated as time allowed. The realized survey effort was 75 days duration and was carried out as a series of three deployments (Data Periods 1–3) from October 2018 to February 2019. Data was acquired for 67% of the planned mission duration. The AutoNaut traveled 1772 km while making approximately 1500 h of sound recordings. Additional sound recordings and trackline effort were excluded from the analysis data-set due to elevated background sound levels when USV propulsion was assisted by its electric motor, primarily during periods with flat seas and strong oceanic currents. Further data collection periods were excluded when the AutoNaut was approached by other vessels, events that were identified from long-term spectral average plots, headphone playback and manual inspection of spectrograms. The realized trackline and distribution of survey effort are shown in Figs. 2, 4, 6, 8, and 9. The AutoNaut attempted to follow the predetermined zigzag survey tracklines, but at times this was challenging for a number of reasons, including:

(a) Strong and unpredictable oceanic currents. AutoNaut attempted to hold position when faced with adverse currents and calm seas with little wave action with which to propel itself along the trackline. The vessel would circle or would drift off course and then use its electric motor to return to the trackline once the current speed changed direction or wave conditions improved.
(b) Interaction with other maritime users. Small boats approaching the USV and/or taking it under tow towards port.
(c) Mechanical failure that compromised the wave-propulsion system affected the ability to make way against the prevailing currents, under wave propulsion or by using the electric motor.

A summary of the sound recording effort across the three Data Periods is shown in Table 1.

**Table 1** Summary of the sound recording effort across the three Data Periods. "Deployment Days" is the number of days that the AutoNaut was deployed at sea; "Recording Days" is the number of days on which sound recordings were made. Total Survey Distance and Recording Time (the total effective survey effort) exclude periods with noise levels that would hamper marine mammal detection

| | Data period 1 | Data period 2 | Data period 3 |
|---|---|---|---|
| Start date | 24-Oct-2018 | 26-Nov-2018 | 21-Jan-2019 |
| End date | 08-Nov-2018 | 01-Jan-2019 | 17-Feb-2019 |
| Deployment days* | 16 | 37 | 28 |
| Recording days* | 16 | 33 | 26 |
| Σ survey distance | 516 km | 739 km | 517 km |
| Σ file size | 624 Gb | 1164 Gb | 903 Gb |

## Analysis of Marine Mammal Vocalizations

### Humpback Whale Song

Humpback whales produce a wide range of tonal and impulsive vocalizations, with most energy between approximately 50–2000 Hz and higher frequency harmonics that can exceed 24 kHz (Au et al. 2006). Vocalizations include sporadic calls (Silber 1986; Dunlop et al. 2007) and song of hours duration in which unit sounds are repeated in a complex structure of phrases and themes (e.g., Payne and McVay 1971; Cholewiak et al. 2012). To search for humpback vocalizations the sound recordings were first decimated from 125 kHz to 8 kHz (IIR Butterworth, 4th order, 3500 Hz low pass), providing an analysis band of 0–4000 Hz. The PAMGuard "whistle and moan detector" (WMD; Gillespie et al. 2008, 2013) was applied to the decimated data to identify candidate humpback whale sounds within the frequency band 100–2000 Hz. A long-term spectral average (LTSA) was plotted (1024-point FFT, 100% hop, frequency resolution 7.8 Hz, time resolution 128 ms, 5-s averaging), and the output from the WMD was overlaid. The analyst inspected clusters of WMD events on short-term spectrograms and reviewed audio over headphones to verify and annotate humpback whale detections. The recordings either side of each WMD cluster were also examined, and in many cases lower amplitude detections were identified that had not triggered the automatic detector.

### Delphinid Whistles and Pulsed Calls

A similar process was used to detect and extract delphinid tonal and pulsed calls from the full bandwidth data. Whistles and pulsed call vocalizations are utilized by oceanic delphinid species as communication signals (e.g., Herman and Tavolga 1980; Herzing 1996; Lammers et al. 2003; Rankin et al. 2008). Whistles are frequency and amplitude modulated, narrowband tonal harmonic calls. Pulsed calls (or burst-pulse calls) are rapid series of broadband clicks with very short

inter-click intervals. The pulse repetition rate of pulsed calls displays as a tonal signal with harmonic or sideband structure, when viewed on spectrograms with narrow analysis bandwidth (i.e., low temporal resolution) (Watkins 1967), rendering them amenable to detection using a whistle contour detector such as the WMD (Gillespie et al. 2013). The sound recordings were first processed using an LTSA with the following parameters: FFT length 1024 points, Hann-windowed, 100% hop, frequency resolution 122 Hz, time resolution 8.2 ms, 5-s averaging. The WMD was then applied to identify candidate tonal events within the frequency band 2–35 kHz, exceeding a trigger threshold of 8 dB above background level. The large volume of delphinid vocalizations precluded manual verification of all candidate acoustic events, although inspection of the LTSA enabled the analyst to delimit periods of effective survey effort, i.e., periods unaffected by background sound from close vessel interactions or the operation of the AutoNaut's electrically powered propulsion pod. The encounter rate used for delphinids was detection positive minutes (DPM) of effective survey effort that included WMD events, with systematic adjustments applied to account for stereotyped false triggers caused by electrical system noise. The false triggers consisted of continuous narrowband tones or pulses. These events were excluded from the WMD database table as a series of SQL queries, where an event's start and end frequencies were restricted to within one of three frequency bands (4.8–6.3 kHz, 10.1–11.1 kHz, 16.5–17.5 kHz).

## Sperm Whale Vocalizations

The data were decimated to a sampling rate of 48 kHz and processed using a click detector (CD; Gillespie and Leaper 1997) to identify the regular click trains and coda click sequences of sperm whales (Whitehead 2003). The CD trigger threshold was set to 10 dB above background level in the pass-band of a "trigger filter" (2nd order IIR Butterworth 2–20 kHz band-pass), creating click data clips. Each clip included the click energy passed by a wider band "pre-filter" (4th order IIR Butterworth 500 Hz high-pass). These data were stored in binary files. The binary files were examined using PAMGuard's Offline Viewer mode. Click bearing, amplitude, and inter-click interval (ICI) were plotted against time to identify click train events. Sperm whale clicks were distinguished from broadband delphinid echolocation and other transient sounds by viewing a corresponding spectrogram (0–65.2 kHz, FFT length 4096 samples, 50% hop, Hann window function, frequency resolution 30.5 Hz, time resolution 16.4 ms). Detection events were delimited manually and annotated by vocalization type ("sperm whale click train," "coda," etc.).

## Delphinid and Beaked Whale Click Trains

A second CD was used to process the full bandwidth recordings and detect the click trains of delphinid and beaked whale species. Oceanic delphinids produce series of short (~20–50 μs), broadband clicks with most energy between 10–150 kHz

(Au 1993; Baumann-Pickering et al. 2010). In contrast, beaked whale species produce narrower band pulses, generally at a lower and less variable repetition rate than delphinids, with species-specific frequency modulation (Baumann-Pickering et al. 2013); for example, Cuvier's beaked whale emit FM upswept pulses of duration 200 μs, centre frequency of 42 kHz and −10 dB bandwidth of approximately 22 kHz (Zimmer et al. 2005). During analysis, the CD trigger threshold was set to 10 dB above background level in the pass-band of a 'trigger filter' (8th order IIR Butterworth 20 kHz high-pass), creating click data clips. Clip energy was extracted from the output of a 'pre-filter' (8th order IIR Butterworth 1 kHz high-pass). These data were stored in binary files and examined in Viewer mode. Three click-type classifiers (Gillespie and Caillat 2008) were applied to highlight clusters of clicks with the characteristics of delphinid echolocation, beaked whale pulses and catch-all 'mid-frequency clicks' respectively. The classifiers used various parameters including click length, number of zero-crossings, zero-crossing frequency sweep, energy band comparison, peak frequency and peak frequency bandwidth. Individual clicks from each cluster were examined using waveform, power spectrum and Wigner plots, time v click bearing and time v ICI displays. A spectrogram display was also helpful, and ultrasonic sounds were played back through headphones at reduced speed, lowering their pitch. Detection events were delimited manually, annotated ('delphinid click train', 'beaked whale', etc.) and written to a database. Click parameters were extracted for all beaked whale events to attempt to assign these vocalisations to particular species, using the ROCCA plug-in for Pamguard (Oswald et al. 2007; Rankin et al. 2016). ROCCA enabled the measurement of 17 parameters for clicks manually selected within the CD interface. Detection of Cuvier's and other beaked whale species are presented in more detail elsewhere (Pierpoint et al. 2021).

## Species Detection Rates

The detection outputs for each of the species of interest was analyzed offline to calculate detection positive minutes (DPM, by species-group). DPM represent the number of effective survey effort minutes in which a species or species vocalization was detected. This was expressed as a percentage of total effort minutes, either overall or per day, and per 5 x 5 km grid cell. DPM thereby describe both the spatial distribution of marine mammals, and track variation in the numbers of marine mammal acoustic detections over time.

## Humpback Whale

Humpback whales were only detected acoustically in Data Period 1, between the 25th October and 5th November. Detections were clustered in the west and south-western parts of the survey area, to the north of São Tomé (Figs. 2 and 3). Humpback whale vocalizations consisted of varied tonal sounds, mainly with the frequency

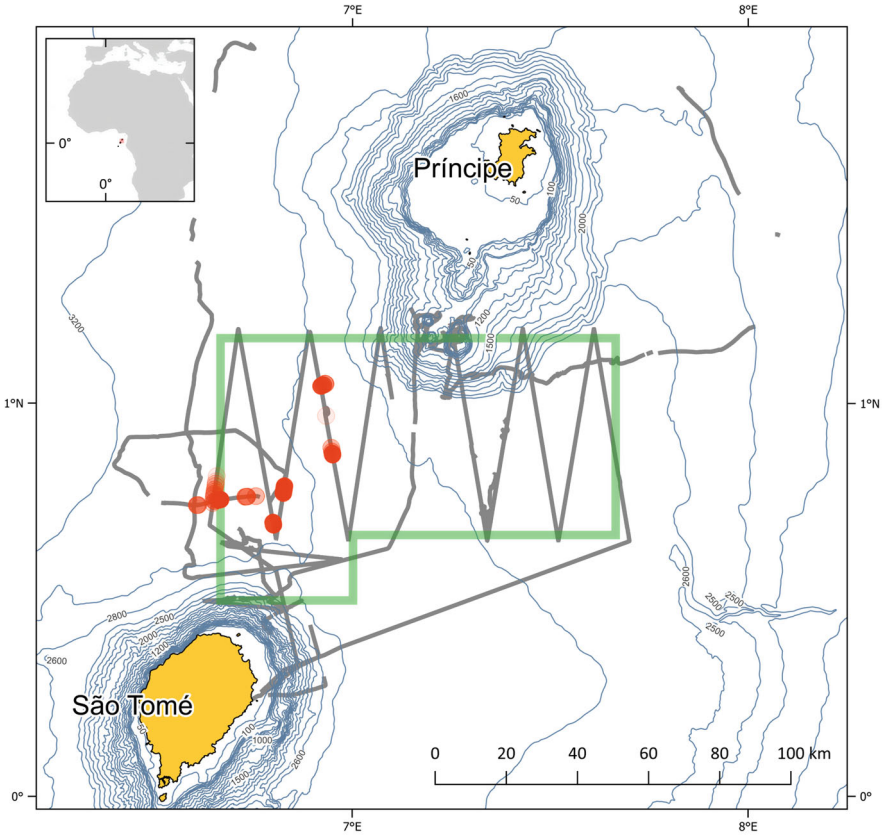



**Fig. 2** The effective survey effort track of the USV and locations from which humpback whales were detected acoustically. The green polygon delimits the core survey area. Bathymetric basemap: GEBCO (2015)

band 200–1500 Hz. Vocalizations were relatively low amplitude, with low signal-to-noise ratio and lacking overtones, suggesting they were produced by distant whales. Series of song units were recorded with repetitive phrasing, i.e., similar song units or combinations of units were repeated in sequence. From the phrasing and consistent spacing between song units, in most cases a single whale was believed to have been detected. On a few occasions, a second whale was also audible.

## Sperm Whale

The relative abundance of sperm whales appeared to increase over the course of the survey. Sperm whale vocalizations were recorded on 37% and 36% of days in Data Periods 1 and 2 respectively, and on 58% of days in Data Period 3. Sperm whale

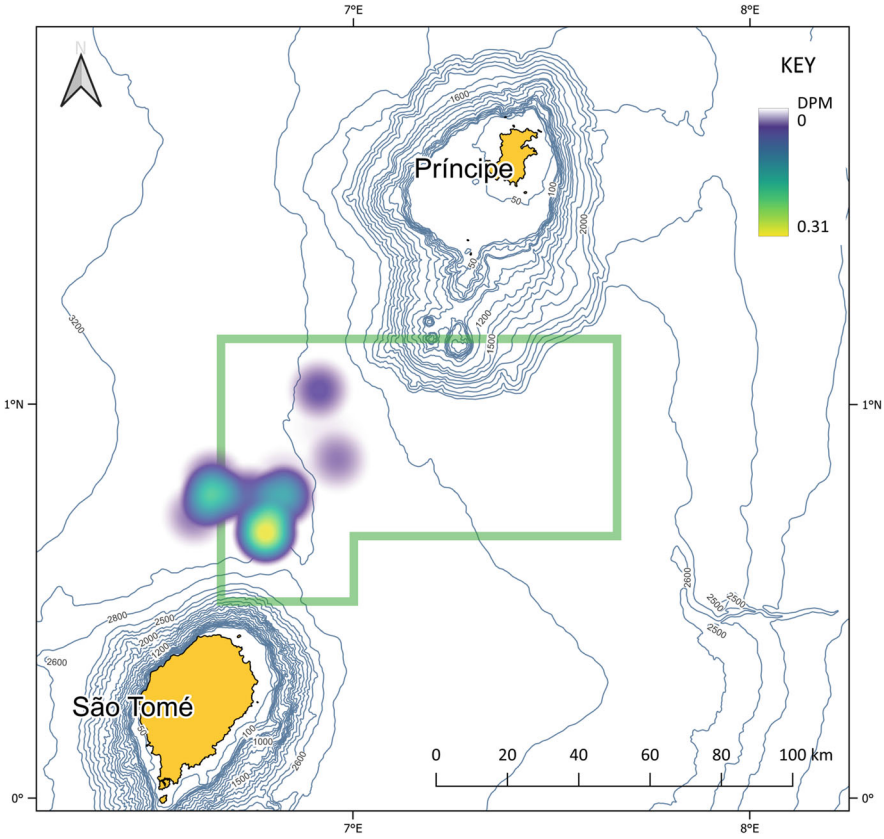



**Fig. 3** Heatmap interpolation of humpback whale detection rates within 5 x 5 km grid cells

detections were widespread and there were several parts of the survey area in which relatively high encounter rates were recorded (Figs. 4 and 5). Throughout the survey, however, extended periods with sperm whale detections were followed by periods of several days without any whales being recorded. In this respect, sperm whale detection occurrence was sporadic, and there was no evidence that aggregations of whales occupied particular areas more than others. Acoustic detections were categorized as "usual click trains," "codas," and "slow clicks" (Whitehead 2003). Usual click trains tended to be recorded for continuous periods of more than 1–2 h at a time. Multiple whales were recorded clicking simultaneously during most encounters. Some usual click trains were interspersed with "creaks," accelerated click rates that are associated with prey capture. Coda vocalizations were recorded between or concurrently with, periods of continuous clicking. "Surface creak" or "chirrup" vocalizations (i.e., "creaks" that were not the termination of a usual click train) were heard among the codas and were included in the same category. There was a

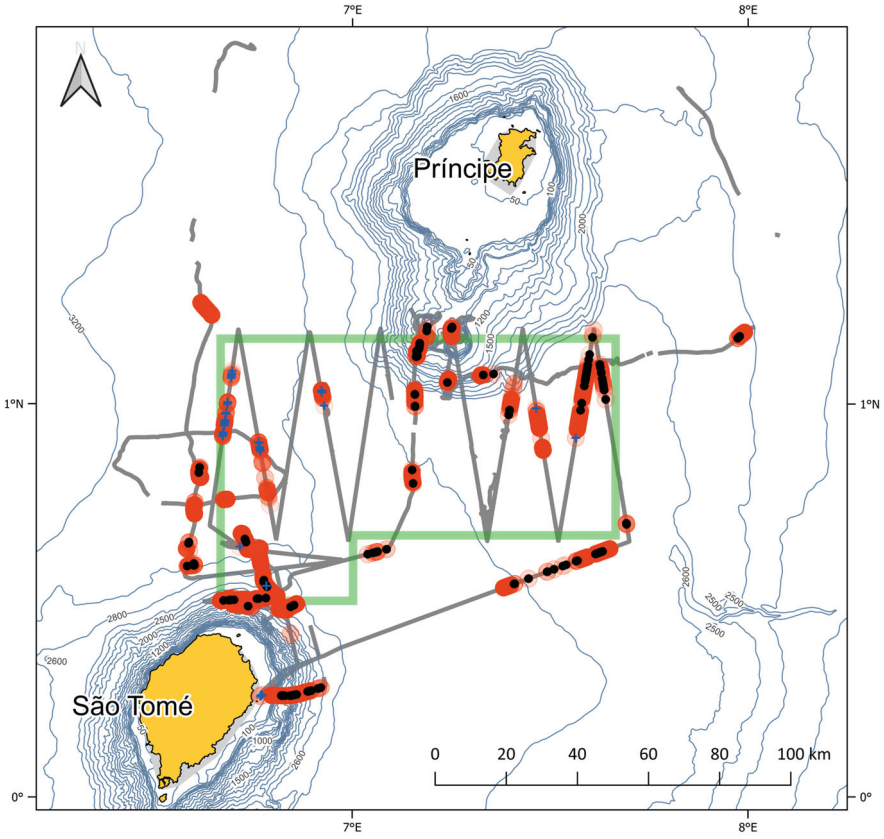



**Fig. 4** Locations from which sperm whales were detected acoustically. Black points and blue cross symbols represent detections of coda and slow click vocalizations respectively

distinct change in the proportions of different categories of vocalization during the survey period. Slow clicks were the common sperm whale sounds detected in Data Period 1 but were very rarely heard during Data Periods 2 and 3. In other studies slow clicks have been heard only in the presence of mature and maturing male sperm whales (Weilgart and Whitehead 1988). In most cases it was evident from the regularity of inter-click intervals that a single whale was slow clicking at any one time, and in a very few cases, two whales. Conversely, the incidence of both usual click trains and codas increased after Data Period 1, with the highest detection rates having been recorded in Data Period 3. Multiple whales were then recorded vocalizing simultaneously, with codas and usual click trains. Codas appear to be produced by all sex and age classes of sperm whale and are commonly recorded at low latitudes during the vocal exchanges of social units that include female and immature whales (Whitehead 2003).

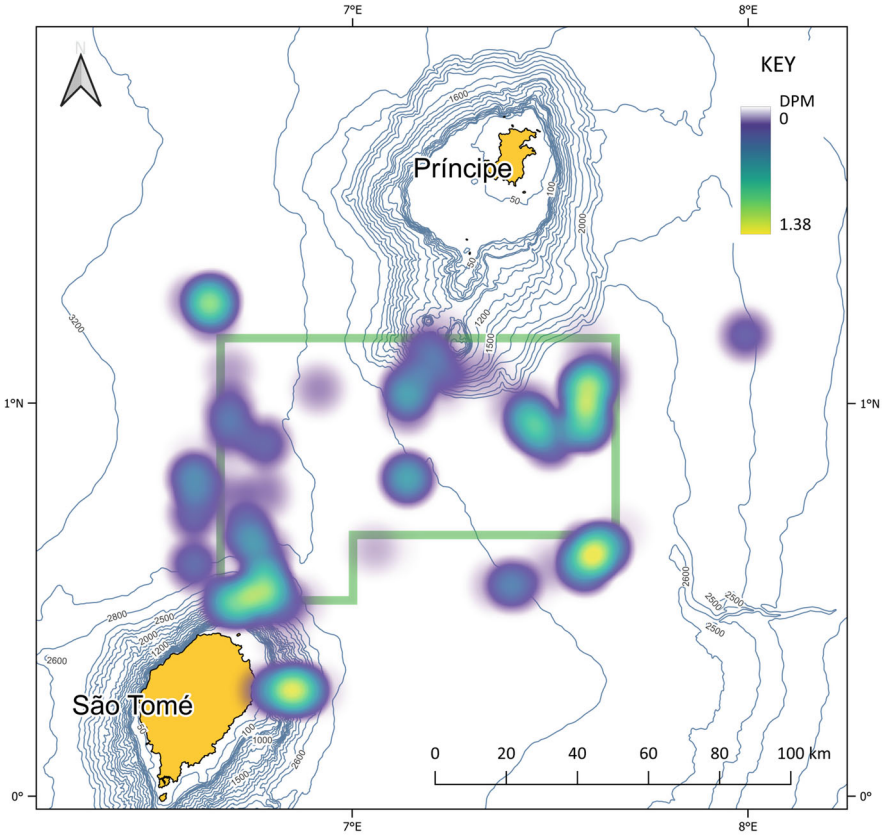



**Fig. 5** Heatmap interpolation of sperm whale detection rates

## Beaked Whale

Beaked whale detections were not uncommon and occurred on 28 of 75 survey days. Acoustic encounters were widespread on the shelf edge and in oceanic habitats above the abyssal plain (Figs. 6 and 7). The average water depth at which acoustic detections were recorded was 2519 (sd 555) m. Thirty-nine of 47 encounters that yielded a sufficient number of click measurements were attributed to Cuvier's beaked whale; the remaining eight encounters were categorized as "unidentified beaked whales." Cuvier's beaked whales were detected on four (25%), fourteen (42%), and eight (31%) survey days in data periods 1–3 respectively.

A key identification parameter was the inter-click interval (ICI). The usual foraging click trains of Cuvier's beaked whale (i.e., excluding buzz-phase clicks

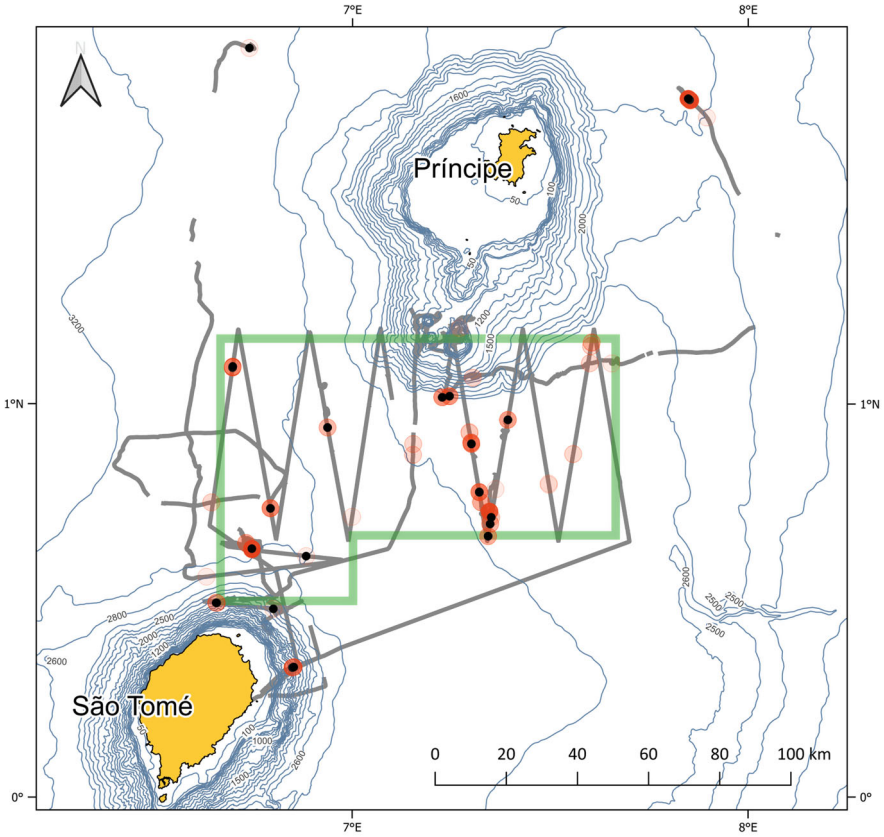



**Fig. 6** Locations from which beaked whales were detected acoustically. Black points represent detections of Cuvier's beaked whale

with increasing repetition rate) have a stable regular ICI of approximately 500 ms (Zimmer et al. 2005; Hildebrand et al. 2015). Median ICI during encounters in this survey was 537 (467–640) ms. The two *Mesoplodon* beaked whale species known to inhabit the region, Blainville's beaked whale (*M. densirostris*), and Gervais' beaked whale (*M. europaeus*) tend to produce echolocation clicks at higher repetition rates (Johnson et al. 2006; Gillespie et al. 2009). Centre and peak frequency during Cuvier's beaked whale encounters averaged 37 kHz and 39 kHz, respectively. Click waveforms during these encounters tended to have 11–15 zero-crossings and an FM sweep rate of 108 kHz $ms^{-1}$.

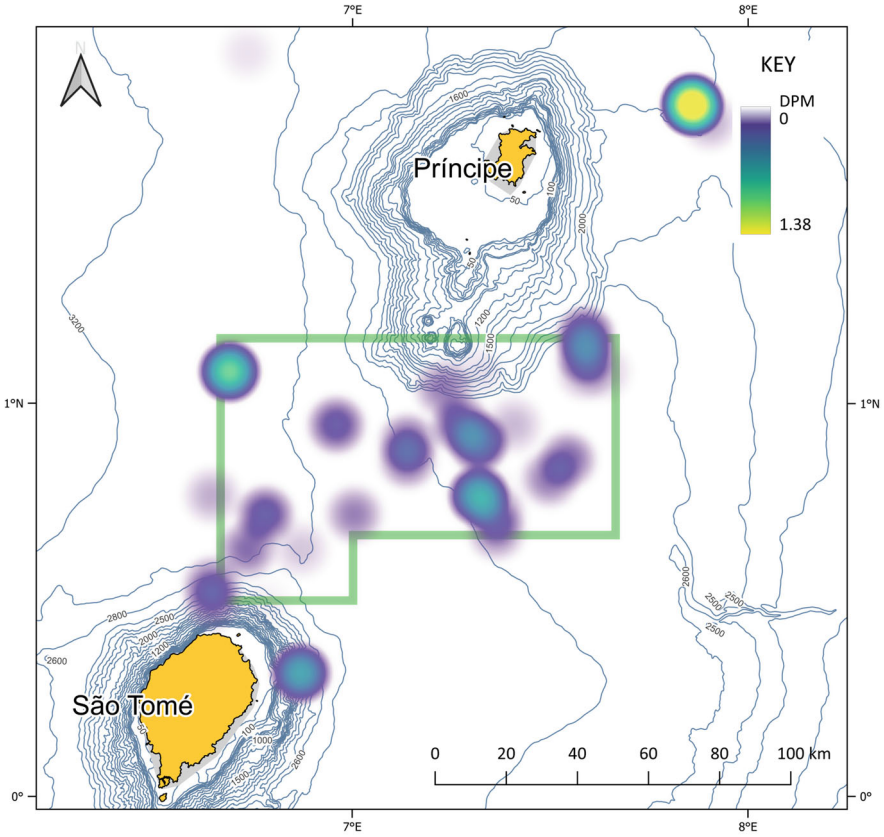



**Fig. 7** Heatmap interpolation of beaked whale detection rates

## Delphinids

Delphinids were the most frequently detected marine mammals in each data period and were recorded every day. Delphinid detection rates were higher, however, during Data Period 3 than in the previous two deployments. Vocalizations included frequency-modulated whistles, pulsed calls, and broadband echolocation click trains. Whistle calls were highly variable and occupied a frequency band from approximately 1.5 kHz to 30 kHz.

The spatial distribution of delphinid acoustic detections was widespread (Fig. 8), but when plotted as a function of survey effort in 5 x 5 km grid cells, a clustering of acoustic encounters was evident north and north-east of the island of São Tomé, including waters above the steep continental shelf break (Figs. 9 and 10).

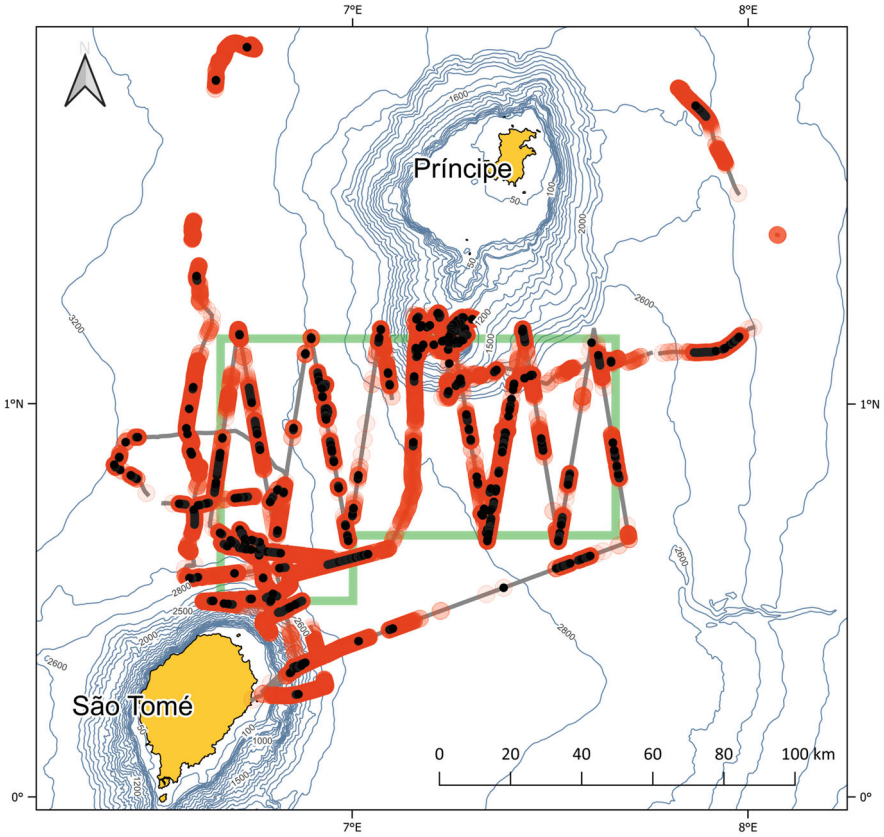



**Fig. 8** The effective survey effort track of the USV and locations at which delphinid whistles (red) and echolocation click trains (black) were detected acoustically

## Study Outcomes

This survey presented challenging conditions for a wave-propelled USV; sea conditions were often calm, oceanic currents strong and unpredictable, and overcast skies hampered solar charging of the vessel's battery power supplies. The vessel also experienced some mechanical issues that affected its propulsion, and survey transects were occasionally halted due to interference from local boat operators. None of these factors had been experienced to such an extent on previous surveys in other areas, and led to the vessel being deviated from its planned systematic survey transect lines.

Despite these challenges, the AutoNaut completed 1772 km of effective trackline effort and collected 1500 h of sound recordings. It remained at sea for 75 days, which was approximately 2/3 of the planned duration of the survey. Survey effort was fairly

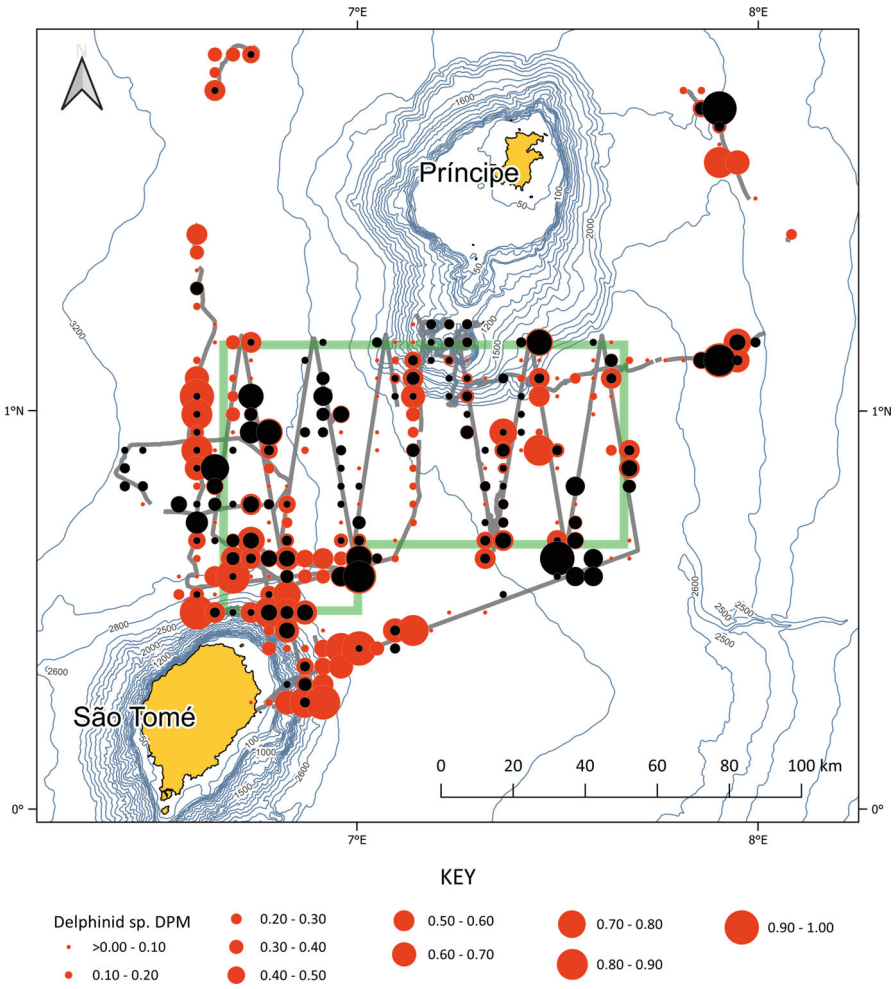



**Fig. 9** Proportional circles representing the detection rate of delphinids within 5 x 5 km grid cells. Red symbols represent whistles and black symbols click trains

evenly distributed across planned survey area, although it was not possible within the operations period to repeat the coverage of the transect lines as planned. The baseline data showed that marine mammals were regularly encountered in and around the survey area, and the following species and species-groups were recorded: humpback whale, sperm whale, beaked whales including Cuvier's beaked whale and delphinid species.

Humpback whales were only detected acoustically from late October to early November, supporting the findings of existing studies (Carvalho et al. 2005; Rosenbaum and Collins 2006). Detections were clustered in the west and south-western parts of the survey area, north of São Tomé. Sperm whales, beaked whales,

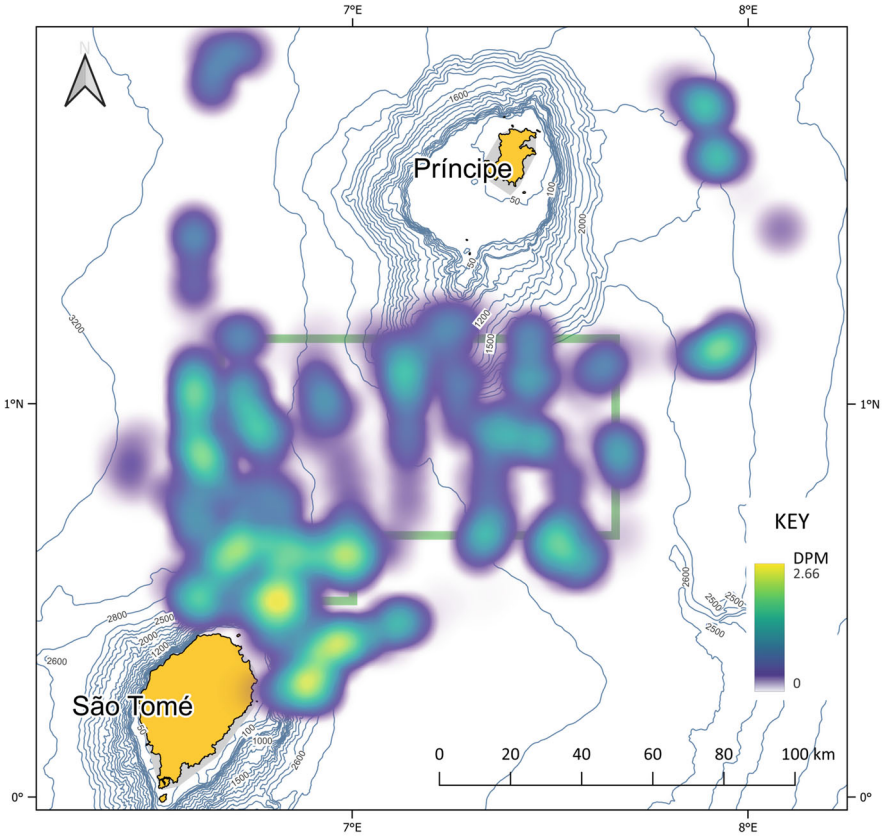



**Fig. 10** Heatmap interpolation of delphinid detection rates

and delphinid species were recorded throughout the survey and were widely distributed within the survey area. The vocalizations of sperm whales suggested that groups of female and immature animals occupied the survey area for extended periods, and that adult male sperm whales also visited the region. Beaked whale detections were not uncommon; the majority were attributed to Cuvier's beaked whale. Prior to this survey there were very few previous data on beaked whale occurrence in the region. Delphinids were recorded on a daily basis. Acoustic detections of delphinids were not differentiated by species, but likely represented encounters with a range of taxa as the Gulf of Guinea is inhabited by at least 17 species.

In conclusion, the AutoNaut baseline survey yielded a wealth of passive acoustic data and new information on marine mammals in the region. It was determined that a number of diverse species-groups inhabited the area from October 2018 to February 2019. In combination with the fixed PAM efforts, the USV contributed to achieving the overall program objectives of validating current understanding of seasonal

presence of humpback whales, confirming the presence/absence of other marine mammals and understanding the baseline soundscape characteristics. In addition, the USV study successfully demonstrated the use of uncrewed technology for marine mammal acoustic monitoring.

**Acknowledgments** We would like to acknowledge the important contribution made to this marine mammal monitoring survey by field staff based on the island of São Tomé and to BP Exploration (STP) Ltd. and partners, Kosmos Energy (STP) Ltd. and ANP-STP, for supporting this work during 2018–19.

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# Measuring Fluctuation in Humpback Whale Presence in Hawaii Based on Song Chorusing Levels

139

Marc O. Lammers, Eden J. Zang, Anke Kügler, and T. Aran Mooney

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**Abstract**

Approximately half of the North Pacific humpback whale population migrates annually to the Hawaiian Islands to breed during the winter and spring months. This Hawaii Distinct Population Segment (DPS) was removed from the US Endangered species list in 2016. Around that same time, this and other North Pacific DPSs experienced substantial climate-driven ecosystem disturbances on their Northeast Pacific feeding grounds, resulting in high mortality and disrupted migration patterns. In Hawaii, humpback whale presence has been monitored

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continuously since 2014 by recording the levels of male song chorusing with acoustic moorings deployed off Maui. Changes in chorusing levels during the past eight breeding seasons indicate that humpback whale presence around Maui declined by approximately 50% between 2015 and 2018 and that the annual duration of whale presence in Hawaii across that period was reduced by up to 1 month. Chorusing levels mostly recovered in subsequent years, but the timing of the migration and the whales' residence time in Hawaii has continued to fluctuate. These results suggest that the Hawaii DPS is still adjusting to the ecosystem disturbances it experienced several years ago or may be at the beginning of a shift in migration patterns in response to climate change.

**Keywords**

Humpback whale · Hawaiian Islands · Chorusing · Population fluctuations · Climate change

## Introduction

The Hawaiian archipelago is an important breeding ground for humpback whales in the North Pacific (Herman and Antinoja 1977; Calambokidis et al. 2008). Estimates suggest that approximately half of the North Pacific population migrates to the Main Hawaiian Islands annually during winter and spring to mate and socialize (Herman and Antinoja 1977; Calambokidis et al. 2008). This part of the population is known as the Hawaii Distinct Population Segment (DPS) and is one of nine humpback whale DPSs that was removed from the US Endangered species list in 2016 after being deemed recovered following the end of commercial whaling (NOAA 2016). Coincidentally, around that same time, humpback whales in the North Pacific experienced substantial ecosystem-wide disturbances on their Northeast Pacific feeding grounds off Alaska, Canada, and the Western USA caused by the convergence of a marine heat wave, a strong El Niño, and a warming period in the Pacific decadal oscillation (PDO) cycle that altered ocean currents and the flow of nutrients (Gentemann et al. 2017). These factors resulted in reduced prey availability on the feeding grounds that led to unusually high whale mortality (Savage 2017) and disrupted migration patterns in the population that affected whale presence on the Hawaiian breeding ground (Cartwright et al. 2019; NOAA 2019; Kügler et al. 2020; Frankel et al. 2022).

Male humpback whales produce an acoustic display of song, which forms a chorus when multiple whales sing simultaneously and becomes the dominant source of acoustic energy in the frequency band between 80 and 1500 Hz in Hawaiian waters during winter and spring, masking other biological, anthropogenic (e.g., vessel noise), and physical (e.g., wind-driven noise) sources of sound (Au et al. 2000; Au and Green 2000; Kaplan et al. 2017). The function of song remains debated but is believed to play an important role in the mating system of humpback whales by signaling to females and/or mediating male-male interactions (e.g., Darling et al. 2006; Cholewiak et al. 2018; see Herman 2017 for a comprehensive review on the function of song). Acoustic monitoring of song has been successfully

used to document trends in humpback whale presence in Hawaii and elsewhere, both temporally and spatially (Clapham and Mattila 1990; Au et al. 2000; Lammers et al. 2011; Seger et al. 2016; Noad et al. 2017; Kügler et al. 2020). Humpback whale presence in Hawaii has been monitored continuously since 2014 by recording the levels of male song chorusing on bottom-moored acoustic moorings deployed off the island of Maui (Kügler et al. 2020). Here we explore how humpback whale presence in Hawaii was affected during an 8-year period spanning the time before (2015), during (2016–2018), and after (2019–2022) the ecological disruptions experienced by the population. We examine song chorusing levels observed during each season to establish how whale presence varied between years.

## Methods

The study used data obtained from an ecological acoustic recorder (EAR) (Lammers et al. 2008) deployed at a depth of 12 m near Olowalu, Maui, to track the relative amplitude of song chorusing (Fig. 1). The EAR recorded for 30 s every 5 min at a sampling rate of either 25 or 50 kHz and was refurbished with new batteries and storage drives approximately every 6 months. The amplitude and timing of chorusing levels were used as indicators of relative whale abundance by relating measured decibel levels in the 0–1500 Hz band with visually derived whale density estimates as described in Kügler et al. (2020, 2021). In brief, a custom Matlab program (MATLAB and Statistics Toolbox Release 2015b, The MathWorks, Inc., Natick, MA, USA) downsampled 30 s recordings to 3 kHz sample rate and converted sample values to pressure, then calculated the root mean square sound pressure levels (RMS SPL) and converted the result into dB re 1 $mPa^2$ (following standards described in Ainslie et al. 2018). The data were averaged daily, and a 5-day running average was calculated for each day to smooth daily variation.

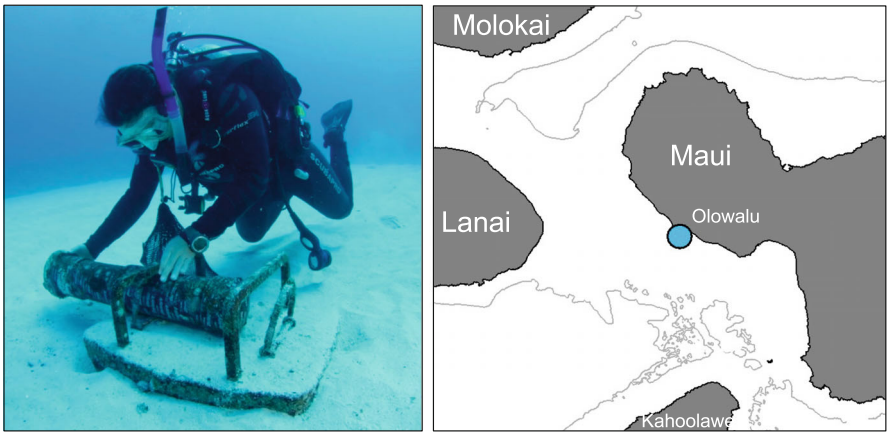



**Fig. 1** EAR deployment location off Olowalu, Maui. (Photo credit: Jason Sturgis)

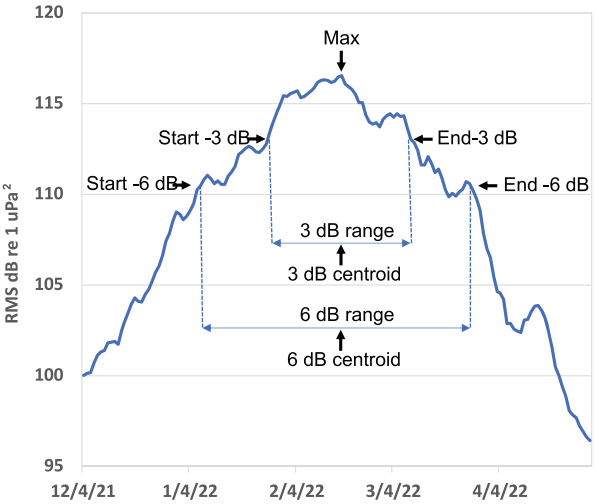



**Fig. 2** Humpback whale song chorusing decibel levels recorded between December 2021 and April 2022 off Maui showing max, −3 dB and −6 dB points

Data from each breeding season (December to April) when chorusing levels remained above other sources of ambient noise (e.g., vessel traffic, wind, etc.) were evaluated to establish: a) when peak chorusing (max) occurred; b) the number of days within 3 dB and 6 dB before and after the max; c) the total number of days within 3 dB and 6 dB of the max (range); d) the centroid of the 3 dB and 6 dB ranges; and e) the dates and number of days that chorusing levels in the 0–1500 Hz band were above 110 dB, a level reached during all 8 years (Fig. 2). These metrics were tabulated and used as a proxy for comparing the relative abundance of whales, the timing of their arrival and departure, and the relative duration of the whale season each year.

## Results

Chorusing levels varied during the eight winter breeding seasons that occurred between November 2014 and May 2022 (Fig. 3). Data from some time periods are not available because the EAR stopped recording either due to a malfunction or low batteries. The levels recorded during this period reveal that peak chorusing declined by more than 6 dB between 2015 and 2018 and that the annual duration of whale presence across that period was reduced by up to 1 month. Kügler et al. (2021) previously determined that this 6 dB decrease corresponded with an approximately 50% decrease in whale abundance, which matched with estimates obtained by other researchers for the same period (Frankel et al. 2022). Chorusing levels mostly recovered in subsequent years, but the timing of the migration continued to fluctuate annually (Fig. 3).

Annual chorusing statistics are presented in Table 1. The date of peak chorusing level varied by up to 1 month (between February 11 in 2015 and March 11 in 2020)

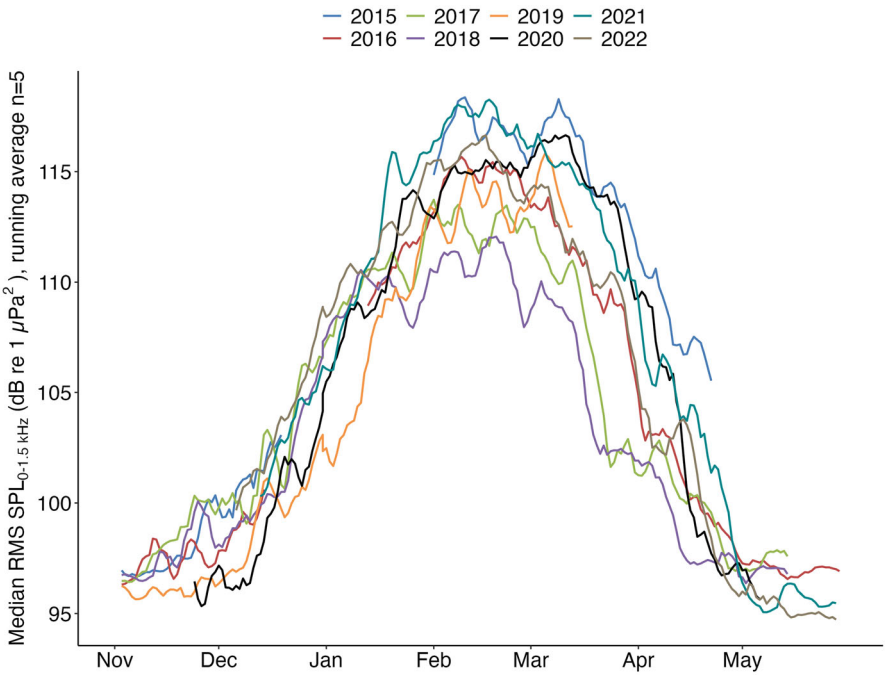



**Fig. 3** Humpback whale song chorusing decibel levels recorded during eight breeding seasons between 2014 and 2022 off Maui

during the 8-year period. The data also reveal that 2015 had the highest maximum chorusing level, which has not been surpassed since the ecosystem disruptions in the Northeast Pacific that began later that year. In addition, the 2015 season lasted longer than all subsequent seasons, with chorusing levels remaining above 110 dB until April 7, as well as 36 and 50 days lapsing between the chorusing max date and the −3 dB and −6 dB dates, respectively, which is longer than in any season since then. In 2020, the season peaked and persisted later than all other seasons, with a −3 dB end date of March 26, and both a 6 dB and 3 dB centroid date of February 23, which is later than all other years. Between 2016 and 2018, the season began progressively earlier, as revealed by the −6 dB start and 110 dB start dates. In 2019, the beginning of the season was considerably later than in 2018 and has begun progressively earlier in the following 3 years.

## Discussion

The results show that the abundance, timing, and residence period of humpback whales' seasonal migration to Hawaii have fluctuated since a marine heat wave combined with an El Niño and warming PDO event to disrupt the ecosystem in the Northeast Pacific feeding grounds between 2014 and 2017 (Gentemann et al. 2017).

**Table 1** Summary of chorusing statistics between 2015 and 2022. N/A indicates when data were not available

| | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 |
|---|---|---|---|---|---|---|---|---|
| Start -6 dB | N/A | 16 Jan | 2 Jan | 29 Dec | 21 Jan | 19 Jan | 16 Jan | 7 Jan |
| Start -3 dB | 1 Feb | 31 Jan | 28 Jan | 8 Jan | 29 Jan | 23 Jan | 28 Jan | 27 Jan |
| Max | 11 Feb | 10 Feb | 23 Feb | 20 Feb | 6 Mar | 11 Mar | 18 Feb | 17 Feb |
| End -3 dB | 18 Mar | 8 Mar | 15 Mar | 10 Mar | N/A | 26 Mar | 14 Mar | 9 Mar |
| End -6 dB | 1 Apr | 19 Mar | 19 Mar | 18 Mar | N/A | 30 Mar | 24 Mar | 19 Mar |
| Centroid 6 dB | N/A | 16 Feb | 9 Feb | 6 Feb | N/A | 23 Feb | 18 Feb | 11 Feb |
| Centroid 3 dB | 23 Feb | 18 Feb | 20 Feb | 7 Feb | N/A | 23 Feb | 19 Feb | 16 Feb |
| 110 dB start | N/A | 17 Jan | 7 Jan | 10 Jan | 28 Jan | 20 Jan | 12 Jan | 6 Jan |
| 110 dB end | 7 Apr | 18 Mar | 15 Mar | 6 Mar | N/A | 30 Mar | 31 Mar | 27 Mar |
| 110 dB range | N/A | 2 Mar | 8 Mar | 24 Feb | N/A | 11 Mar | 19 Mar | 21 Mar |
| Start -6db to max (days) | N/A | 26 | 53 | 54 | 53 | 53 | 34 | 42 |
| Start -3 dB to Max (days) | 10 | 11 | 27 | 44 | 49 | 49 | 22 | 22 |
| Max to end −3 dB (days) | 36 | 28 | 21 | 19 | N/A | 16 | 25 | 21 |
| Max to end −6 dB (days) | 50 | 39 | 25 | 27 | N/A | 20 | 35 | 31 |
| −3 dB range (days) | 45 | 38 | 47 | 62 | N/A | 64 | 46 | 42 |
| −6 dB range (days) | N/A | 64 | 77 | 80 | N/A | 72 | 68 | 72 |
| Max dB | 118.54 | 115.89 | 113.63 | 112.15 | 115.87 | 116.77 | 118.45 | 116.54 |

These fluctuations have continued in recent years, with a general trend toward an earlier end to the breeding season in Hawaii, albeit with exceptions (e.g., 2020). This suggests that the Hawaii DPS may still be adjusting to the ecosystem disturbances it experienced several years ago (Cartwright et al. 2019; Kügler et al. 2020; Frankel et al. 2022). Alternatively, it could point to the beginning of a more long-term shift in migration patterns in response to climate change (von Hammerstein et al. 2022). Some whales may be forced to seek prey in different high latitude feeding areas, thereby affecting where, when, and/or how much time they spend on the Hawaiian breeding grounds. Lastly, it is also possible that annual fluctuations in migration could simply reflect normal variation resulting from a recovered population having reached carrying capacity (Begon et al. 2006; Kügler et al. 2020).

The extended duration of the 2020 breeding season coincided with the economic downturn that resulted from the COVID-19 pandemic. In Hawaii, Covid-related

lockdowns began in mid-March 2020, which coincided approximately with the peak of chorusing that year (March 11). Nearly all commercial vessel activity was halted temporarily by the end of March, which altered the amount of vessel traffic and noise experienced by whales during this time. This correspondence is interesting but may or may not be related. It is possible that less vessel noise prompted more whales to sing or for them to persist in Hawaii longer. Alternatively, the season may have been trending later due to factors unrelated to Covid, such as prey availability during the previous summer on the feeding grounds. Regardless, the timeline of this season's end is intriguing. If the timing of the whale's departure was indeed influenced by reduced human activity, it would be a stark indication of how vessel traffic affects whales. On the other hand, such a late departure, if driven by oceanographic/prey factors, would suggest that these are fluctuating even in the absence of a marine heat wave or an El Niño event, as was the case in 2014–2017.

The results presented indicate that continued monitoring is warranted to understand the long-term trends in this DPS, including in other parts of the Hawaiian archipelago where the population dynamics may be different. Lammers et al. (2023) recently reported the presence of a likely substantial number of humpback whales in the northwestern end of the Hawaiian archipelago. Sustained acoustic monitoring in that part of the archipelago would help determine whether whales in the Main Hawaiian Islands and Northwestern Hawaiian Islands are influenced in the same way by the climatic factors impacting North Pacific humpback whales.

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# Background Noise Impacts Harbor Porpoise Detections in Passive Acoustic Monitoring

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Mel Cosentino, Jakob Tougaard, and Magnus Wahlberg

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**Abstract**

Passive acoustic monitoring (PAM) is instrumental to monitor marine mammals, with many applications, including impact of noise. Noise, however, also affects the ability of PAM to detect sounds of interest. The performance of a PAM algorithm was tested by feeding it recordings of ship noise merged with harbor porpoise (*Phocoena phocoena*) clicks. Three groups of porpoise clicks ("click trains") were inserted every 10 sec in files containing ship noise, one at a time (n = 1180 min). The recall (probability of detecting porpoises in 10-sec bins) was

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modeled using a generalized linear model with a binomial distribution and one explanatory variable: noise in one of four frequency bands (<20 kHz, 20–80 kHz, 80–150 kHz, and the decidecade centered on 16 kHz). The best model had the 20–80 kHz band as the explanatory variable, but noise in all bands masked detections when exceeding 90 dB re. 1μPa, becoming a cofounding factor. Therefore, decreased detections at these noise levels cannot be attributed unequivocally to effects of noise on the behavior of porpoises but may be due to masking of the PAM instrument. However, PAM can be used, as long as the noise levels above 20 kHz are monitored and accounted for.

**Keywords**

Sound · Underwater noise · Detections · Ship noise · Monitoring · PAM

## Introduction

Monitoring wide-ranging cetaceans is a difficult task, and using traditional visual methods alone is limiting, especially for small-sized animals with cryptic behavior. The development of affordable passive acoustic monitoring (PAM) devices that can be deployed for long periods was a game changer, becoming instrumental to monitoring many cetacean species, such as the harbor porpoise (*Phocoena phocoena*). Harbor porpoises produce only one type of sound: polycyclic, narrow-band clicks. These clicks have peak and centroid frequencies between 100 and 150 kHz, with little or no spectral energy below 100 kHz, duration of about 100 μs, and − 3 dB bandwidths between 12 and 20 kHz (Hansen et al. 2008; Kyhn et al. 2013; Møhl and Andersen 1973; Villadsgaard et al. 2007). Clicks are emitted in trains in a narrow, forward-oriented beam (Au et al. 1999, 2006). A click train is a group of clicks with regular or gradually changing inter-click intervals (Koschinski et al. 2008), and the patterns in click production appears to be context dependent (Clausen et al. 2010; Sørensen et al. 2018; Verfuss et al. 2005). These characteristics make harbor porpoises' sounds well suited for PAM.

PAM systems have several advantages over visual surveys, as they can be used during bad weather conditions (e.g., high waves, fog) as well as at night. Data collection can be accomplished with no supervision, and therefore, the performance is not affected by the experience or levels of alertness of the observer. Moreover, because cetaceans spend most of their lives underwater, animals sometimes are only detected acoustically or are detected acoustically before they are detected visually (e.g., Yack et al. 2013).

Existing PAM devices that can be used specifically to study harbor porpoises can be divided into two main types: devices that only log data about acoustic events and those that record continuously. The most used data logger is the C-POD and its predecessor the T-POD (Chelonia Ltd., Cornwall, UK). T- and C-PODs have been successfully used for many applications, including understanding the impact of underwater noise (Dähne et al. 2013; Gallus et al. 2012; Koschinski et al. 2008; Kyhn et al. 2008; Tougaard et al. 2009). Wideband continuous recorders opened up

the possibility of more detailed sound analysis. Not only can the raw data be audited and reanalyzed using different parameters (e.g., detection thresholds), but it is also possible to study the temporal and spectral characteristics of the background noise. In turn, this can be used to better understand how noise impacts the target species.

Harbor porpoises are common in coastal and shelf waters (Bjørge and Øien 1995; Hammond et al. 1995), so they are regularly exposed to vessel traffic (Wisniewska et al. 2018). Ship noise is typically assessed at low frequencies (e.g., the EU descriptors for a healthy marine environment are measured at 63 and 125 Hz, Dekeling et al. 2014). This could suggest that porpoises would not be affected, as their best hearing is at high frequencies (Kastelein et al. 2010). However, recent studies carried out with devices that can record the full broadband signals from ship noise have shown that ship noise extends to much higher frequencies (Hermannsen et al. 2014) and that porpoises respond negatively, responding behaviorally or vocally, when noise exceeds certain levels (Dyndo et al. 2015; Wisniewska et al. 2018). Although high frequencies travel short distances (Urick 1983), porpoises are often close to vessels, and so it is expected that ship noise also contributes to masking porpoises' ability to detect sounds of interest (Erbe 2015).

Masking does not only affect the animals but also the PAM system itself. This phenomenon is well-known yet remains understudied (Clausen et al. 2019; Sarnocinska et al. 2016). If ship noise masks the PAM system, noise becomes a confounding factor, and reduced detections cannot be directly interpreted as reduced vocal behavior or displacement of harbor porpoises. There is, therefore, a need to further disentangle noise impact on detections and impact on the animals.

To address this, the performance of a PAM detector/classifier algorithm was investigated under different noise conditions on files containing ship noise that were merged with real harbor porpoise clicks. By doing this, the same number of porpoise detections was expected throughout the entire recording of a passing vessel. Thus, any decrease in or lack of detections would be a direct result of noise masking.

## Ship Noise and Porpoise Clicks

The data used here were obtained from five deployments carried out in Danish inner waters in 2015: four in the Great Belt area (55.3615 N, 10.9655 E) in August (n = 1), September (n = 1), and November (n = 2) and one in the Little Belt area (55.5295 N, 9.7535 E) in August (Fig. 1). The equipment used was one ST 202HF (Ocean Instruments, New Zealand), recording continuously at a sampling rate of 576 kHz. The ST hydrophone had a sensitivity of −172 dB re: 1 V/μPa.

The deployment location in the Great Belt is close to a shipping lane, so ship passages were selected to ensure only one vessel was passing at a time, assessed by the pattern of increasing and decreasing noise levels (Fig. 1). A total of 25 individual ship passages were selected, amounting to 430 min of recordings, covering different time periods in five deployment days.

Three porpoise click trains were merged with ship noise to prevent pseudo-replication. These trains were also extracted from the data and had different durations, signal-to-noise ratios, amplitude variations, repetition rates, and number of




**Fig. 1** Map of the study area. The rhomboids indicate the location of the acoustic recorders

clicks. Only one click train was merged at a time and inserted every 10 sec in each file. Periods with porpoise detections in the original files were removed to prevent errors in the interpretation of the results, which left in almost 1180 min of data.

## Noise Levels

For each 10-sec bin (n = 7076), the noise levels (dB re: 1μPa$^2$/Hz) were estimated from the power density spectrum (Welch average, 512-point FFT, Hann window, 50% overlap) for three frequency bands: 0–20 kHz, 20–80 kHz, and 80–150 kHz, as well as the decidecade centered on 16 kHz. The 16 kHz decidecade band was used because Wisniewska et al. (2018) identified it as a useful proxy for predicting porpoise reactions to ship noise, when noise in that band exceeds 96 dB re. 1 μPa$^2$/Hz.

## Porpoise Detections

The data consisted of *.wav* files of 10 or 30 min in duration, which were analyzed using D-PorCCA. D-PorCCA is a new analytical tool specifically developed for analyzing acoustic data of species that emit narrowband high-frequency clicks, like

the harbor porpoise. D-PorCCA includes a transient sound detector (adapted from Gillespie et al. 2008 by Parcerisas 2021), a porpoise click classifier (Cosentino et al. 2019), and a click train detector (Cosentino 2020). The click train detector groups clicks into acoustic events, which are then classified as either high- or low-quality porpoise click trains, based on the output of an algorithm that favors gradual changes in centroid frequency, amplitude, and inter-click intervals (Cosentino 2020).

## Noise Impact

The recall (probability of porpoise detection in a given 10-sec bin) was modeled using a generalized linear model with a binomial distribution and one explanatory variable: noise level in one of three frequency bands (<20 kHz, 20–80 kHz, or 80–150 kHz) or the decidecade centered on 16 kHz. The Akaike information criterion (AIC) value was used to compare the performance of the models with each other.

### Detections

The number of porpoise clicks detected decreased as the noise levels increased with passing vessels. The effect was visible when looking at individual vessel passages (Fig. 2), especially for increasing levels of noise in the 20–80 kHz frequency band. The models predicted reduced detections as a function of noise (Fig. 3) in all bands, and the model with the lowest AIC value (AIC = 3886.69) had noise in the 20–80 kHz frequency band as the only explanatory variable. The next best model was the one with noise in higher frequencies 80–150 kHz (AIC = 3972.76), then the decidecade centered on 16 kHz (AIC = 4070.34), and last the band <20 kHz (AIC = 4075.84).

### Disentangling Impacts

Wisniewska et al. (2018) found that the decidecade centered on 16 kHz could be used as a proxy to identify the noise levels at which porpoises respond behaviorally (by moving away from the noise) or becoming silent. The models in this study indicate that noise masks porpoise detections when it reaches about 90 dB (dB re: $1\mu Pa^2/Hz$) in all frequency bands, but especially the 20–80 kHz band (Fig. 3). This means that when noise surpasses these thresholds, masking occurs (Fig. 4).

## Conclusions

Ship noise extends far beyond the low frequencies at which it is typically monitored, reaching frequencies at which porpoises react both by changing their behavior (e.g., moving away) and also by becoming silent (Dyndo et al. 2015; Wisniewska et al. 2018). However, reduced detections (or no detections at all) during noisy periods cannot be safely explained as an effect of noise in the behavior of porpoises. At high

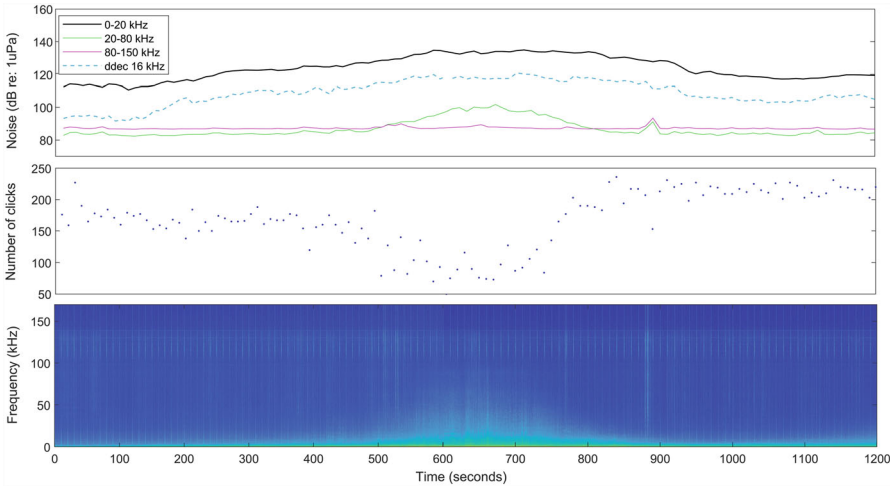



**Fig. 2** Ship noise and porpoise detections. Top panel: background noise levels (dB re: 1μPa$^2$/Hz) in three frequency bands (0–20 kHz, 20–80 kHz, and 80–150 kHz) and the decidecade centered at 16 kHz. Center panel: changes in the number of clicks detected in 10-second bins. Bottom panel: long-term spectral average of the merge files, showing the passing of an individual ship recorded on August 12, 2015, at 3.56 am in the Great Belt area. The vertical lines with energy between 100 and 150 kHz are the inserted porpoise click trains

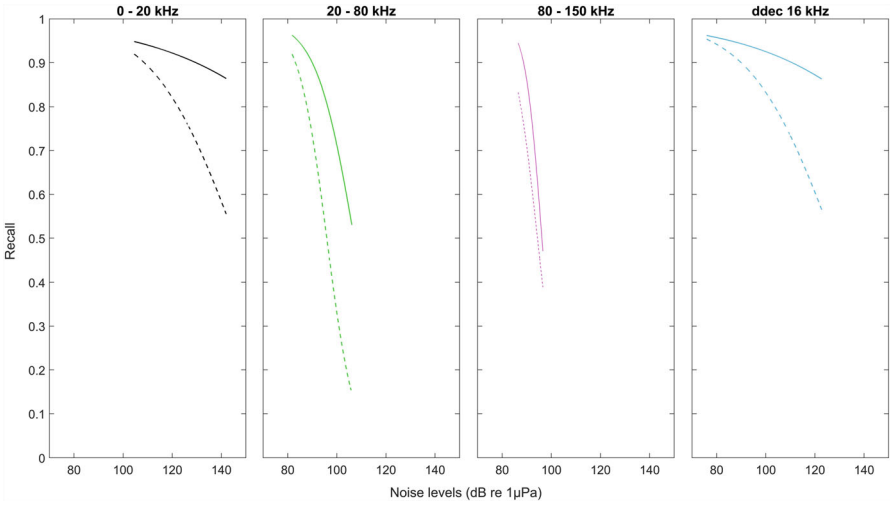



**Fig. 3** Recall (probability of detecting porpoises) as a function of ship noise. Modeled recall shows a decrease in detections as noise (dB re: 1μPa$^2$/Hz) increases in different frequency bands: <20 kHz (AIC = 4075.84), 20–80 kHz (AIC = 3886.69), 80–150 kHz (AIC = 3972.76), and the decidecade centered at 16 kHz (AIC = 4070.34). Solid lines include porpoise click trains of high and low quality. Dashed lines include only porpoise click trains of high quality

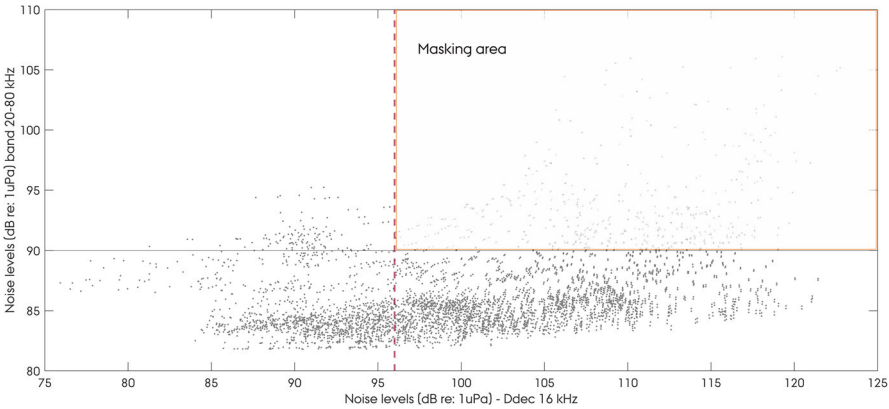



**Fig. 4** Masking. Noise levels (dB re: 1μPa$^2$/Hz) in the ddec centered at 16 kHz vs in the 20–80 kHz frequency band. The shaded area shows where noise masks the acoustic reaction of porpoises

received noise levels, the noise masks the PAM system's detectability, becoming a confounding factor.

As long as frequencies above 20 kHz are monitored and accounted for in the analysis, PAM can be used for monitoring the impact of noise on animals.

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# Methods to Measure Underwater Sound Sources from Oil and Gas Activities

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**Abstract**

In 2021, the International Association of Oil and Gas Producers (IOGP) published a report in which measurement procedures are described that enable the acoustic characterization of underwater sound sources associated with seismic acquisition, high-resolution geophysical surveys, and production of oil and gas. These procedures are designed to provide consistency and comparability of independently carried out measurements. An initial review of existing methods revealed that there was some commonality in reported measurements of airgun arrays and high-resolution geophysical sources, but very few reports or publications included all relevant parameters. Achieving comparability in measurements requires precise nomenclature and clear documentation of the various choices made concerning measurement geometry, instrumentation, and processing. The measurement procedures describe requirements and recommendations for equipment selection, calibration, placement in depth and range, sampling rates, and operational measurement procedures for the three source types. Signal analysis methods are specified, as well as reporting content and formats for the acoustic results and relevant metadata. Where possible, calculation procedures are described to determine source metrics from the measured acoustic field, which can be used as input for environmental impact assessments for future projects.

**Keywords**

Measurement procedure · Underwater sound · Sound sources · Sound measurement · Seismic exploration · Geophysical surveys · Oil and gas platforms

## Introduction

Activities related to offshore oil and gas exploration and production introduce sound into the marine environment. Since the early 1990s, there has been increasing environmental and regulatory focus on the effects on marine life of that and other man-made sound in the sea. Assessment of potential effects relies on the proper quantification of the dose of sound to which animals are exposed. Achieving consistency in measurements for underwater sound sources depends strongly on being precise with nomenclature and carefully documenting the required measurement geometries, instrumentation, and processing. Lack of standardization of terminology and of procedures for measuring or predicting and reporting of underwater sound leads to potential inconsistencies and uncertainties in the reported numbers (Ainslie et al. 2017). This hampers effective regulation and mitigation of underwater sound.

The International Association of Oil & Gas Producers (IOGP) promotes the use of standard procedures for underwater sound measurements for activities related to offshore oil and gas exploration and production (E&P). Under its Sound and Marine Life Joint Industry Programme (JIP) (http://www.soundandmarinelife.org/), a consortium consisting of TNO (Netherlands), CSA (USA), UNH (USA), and JASCO (Canada) and independent consultants produced a series of reports describing standard terminology (Ainslie et al. 2018a) and procedures for data processing (Ainslie et al. 2018b) and reporting (Ainslie and de Jong 2018). All reports are available online at http://www.soundandmarinelife.org/library.aspx.

This chapter introduces recently published measurement procedures for underwater sound sources associated with oil and gas exploration and production activities (de Jong et al. 2021). It follows ISO 18405:2017 of the International Organization for Standards (ISO) and the terminology described in (Ainslie et al. 2018a).

## Overview of Report Contents

The reported procedures provide a scientific guide on measuring underwater sound generated by oil and gas activities. Therefore, these are supported by and further support existing standards, which are referenced throughout. The final product is a report that describes all details of the procedures to be followed, with the goal of consistency and comparability across projects. The report contains the following sections:

1. Introduction (Background, Objective, and Scope).
2. Terms and definitions (Sound source types, Guidance on characterizing sound source signals, SL, SPL, PL).
3. Instrumentation (Selection of hydrophones, Preamplifiers, Filters, ADC, Recording system, Calibration).
4. Sensor deployment (Mooring geometries, Flow noise suppression, etc.)
5. Measurement procedures
   - General considerations and requirements
   - Procedure for measuring type A and B sources
   - Procedure for measuring type C sources
   - Required auxiliary measurements
6. Data processing (Received sound metrics)
7. Calculation of source output metrics
8. Uncertainties
9. Reporting

## Selection of Source Type and Source Metrics

Standard procedures are proposed for measurement of underwater sound associated with the following three types of sound sources:

1. **Type A sources**: Used for geophysical exploration surveys ("seismic acquisition"), such as airgun arrays, water guns, sparkers, boomers, some sub-bottom profilers, and marine vibrators. These typically emit most acoustic energy at frequencies below 1 kHz at a pulse repetition rate below 1 Hz. Their low frequency leads to a large beam width.
2. **Type B sources**: Used for high resolution geophysical (HRG) surveys, such as multi-beam echo-sounders, side scan sonars, and some sub-bottom profilers. These typically emit most acoustic energy at frequencies above 1 kHz at a pulse repetition rate above 10 Hz. The high frequency is typically associated with a narrow beam, which places constraints on the measurement procedure. The processing required for back-propagation to a source property also depends on frequency.
3. **Type C sources**: Associated with production platform activities. These are typically stationary sources that emit underwater sound as an unintended by-product of their activity, such as pumping and drilling. These are treated as a single composite source, similar to the ISO 8297 engineering method for measuring airborne sound power levels of multisource industrial plants.

The association of a source with one of these three types is based mainly on its frequency band and corresponding directivity. This association has implications for the selection of appropriate measurement and processing procedures. In cases where the association in uncertain, the user can make a pragmatic choice.

The measurement and analysis procedures have been developed on the basis of an initial literature review of existing methods, combined with the experience of the developers. The literature review revealed that there was a fair degree of commonality in reported measurements of airgun arrays (type A) and HRG sources (type B), but very limited commonality in reported measurements of oil and gas production activities (type C). Very few reports or publications included all relevant parameters.

The procedures are applicable for measurements of the underwater sound of these sources in or close to operational conditions, for which the measurement equipment is to be deployed in the operational area, with limited impact on operations. The measurement procedures describe requirements for selection of equipment, calibration, placement in depth and range, sampling rates, and operational measurement procedures for the three source types. Signal analysis methods are specified, as well as reporting content and formats for the acoustic results and relevant metadata.

The scope is limited to measurements of sound pressure, for the purpose of source characterization. It does not include procedures for measurement of sound particle motion. This relatively new measurement technique is addressed in the Interim Best Practice Guide for Underwater Particle Motion Measurement for Biological Applications (Nedelec et al. 2020).

Characterization of the sources in terms of a full coherent source waveform, facilitating full reconstruction of the transmitted waveform, including phase information, is considered to be achievable only under carefully controlled conditions, similar to those for the collection of the Svein Vaage data set (Vaage et al. 1983; Prior et al. 2019). Such measurements, whether for arrays or single source elements, are outside the scope of this project.

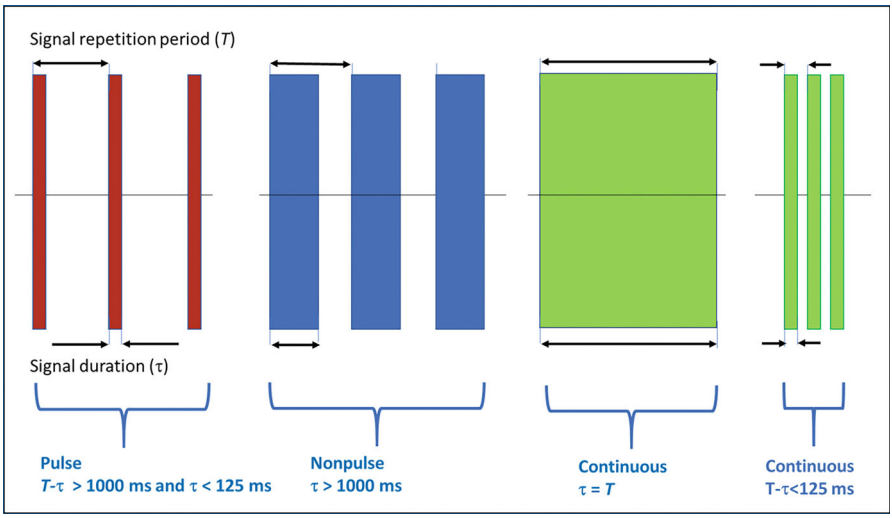



**Fig. 1** Schematic representation of signals, with different signal repetition periods ($T$) resulting in classification as either pulse, nonpulse, or continuous signal. See also Table 2

Hence, measurement procedures for underwater sound sources must collect all data that is needed to perform this calculation. The measurement geometry needs to be optimized, to limit the inherent uncertainties associated with the source calculation, and it needs to be carefully documented and the environmental acoustic parameters properly characterized.

The source characteristics based on sound pressure are expressed in different acoustic metrics, depending on the nature of the sound pressure field. Sources of type A and B often emit *pulse* or *nonpulse* sound. ISO 18405 does not define these terms, while sources of type C often emit *continuous* sound. Within this project no consensus could be achieved on a formal classification of signals as *pulse*, *nonpulse*, and *continuous*, but guidance is provided for making this characterization, according to the signal duration ($\tau$) and the silence duration ($T - \tau$), with $T$ the signal repetition period (defined as the reciprocal of the signal repetition rate), see Fig. 1 and Table 1.

## Preparation for Source Measurements

Because of the variety of sources, operational circumstances and limitations, and specific project requirements, the measurements need to be carefully prepared. Information needs to be gathered on the source to be measured, including its operational settings, and the operational environment and conditions in which the measurements are to be performed. The required information is elaborated in the following subsections. The details of the measurement plan should be determined based on this information.

**Table 1** Categorization of signals as pulses (short duration signals with a long inter-signal interval), nonpulses (long duration signals with a long inter-signal interval) and continuous signals (signals with a short or nonexistent inter-signal interval). The gray areas represent signals that are not clearly in one category or another. In these gray areas, practical considerations may determine the most appropriate choice. See also Fig. 1

| Inter-signal interval | Signal duration $\tau$ < 125 ms | 125 ms < $\tau$ < 1000 ms | Signal duration $\tau$ > 1000 ms |
|---|---|---|---|
| $T - \tau < 125$ ms | continuous | Continuous | continuous |
| 125 ms $< T - \tau <$ 1000 ms | user-defined | user-defined | user-defined |
| $T - \tau > 1000$ ms | pulse | user-defined | nonpulse |

**Table 2** Defined geometric parameters for Fig. 2 and in text

| Symbol | Description |
|---|---|
| $L_x$ | Horizontal source dimension (length) |
| $L_y$ | Horizontal source dimension (width) |
| $L_z$ | Vertical source dimension |
| $z$ | Source depth ($z \leq L_z$) |
| $H$ | Water depth |
| $r$ | Slant range (hydrophone distance) |
| $\phi$ | Vertical observation angle |
| $\theta$ | Horizontal (azimuthal) observation angle |
| $h$ | Hydrophone minimum height above sea floor |
| $R = r \cos \phi$ | Horizontal hydrophone distance (far field) |
| $D = z + r \sin \phi$ | Hydrophone depth ($D \leq H - h$) |
| $A =$ max ($L_x$, $L_y$, $L_z$) | Maximum source dimension |

## Description of the Source and Its Operational Settings

The following questions need to be answered.

- Under which of the three source types (A, B, and C) can the sound source to be measured be categorized?
- For type C sources (production facilities):
  - Which facility components and operations are to be included in the source characterization?

- What are the main geometrical dimensions of the source?
  - What are the dimensions of the smallest closed surface (rectangular box) that spatially bounds the source?
  - What are the three-dimensional coordinates of the assumed acoustic center, relative to the geometry of the actual source?
- What information is available on the underwater sound radiated by the source?
  - What is the frequency range of the source?
  - Can its radiated sound signals be classified as "pulse," "nonpulse," or "continuous"?
- At which locations and over which time durations will the source be operated?
- At which settings will the source be operated and to what extent can its operation be adapted for the measurements?
  - Sources of types A and B are controlled by an operator; thus, the vessel from which the source is deployed may have the possibility to adapt its route and timing to optimize the measurements.
  - Source of type C produce underwater sound as an unintended by-product of their activities, which may or may not be controlled by an operator, generally fixing the location and operation time.

## Description of the Operational Environment and Conditions

- At what location can the measurements be performed?
- What is the local water depth?
  - How does it vary over the operational area?
  - How is it affected by tidal variations?
- What information is available on the local sea water?
  - Acoustic models require local sound speed, as a function of depth, and absorption.
- What are the local sea floor features (are there rock outcrops or local channels)?
  - Generally, it is best to place measurement systems at locations where depth and seabed features are least variable.
- What information is available on the local sea floor composition?
  - Acoustic models require local sediment properties in terms of density, compressional (and shear) wave speed, and absorption, as a function of sediment depth, (i.e., geoacoustic profile).
- What information is available on the local weather conditions?
  - Acoustic propagation is affected by wind-generated surface waves.
  - Weather conditions may lead to practical limitations for measurements (safety and positional stability of measurement equipment).
- What information is available on other activities in and surrounding the operational environment?
  - Is it likely that the measurements are disturbed by operation of other sources in the project area or by nearby vessel traffic?

## Specification of the Requirements for Source Characterization

- Which operational conditions are of interest for the measurements?
  - The operational conditions need to be sufficiently representative in measurements to address environmental impact assessment.
- Which frequencies are of interest for the source measurements?
  - The selection of the measurement frequency range needs to match with the frequency range of interest for environmental impact assessment.
- Which source metrics are of interest?
  - Source metrics depend on the class of signals ("pulse," "nonpulse," or "continuous").
  - Source metrics depend on the metrics of interest for environmental impact assessment.
  - Local regulators may require specific metrics.
- What are the uncertainty requirements for the reported metrics?
- Which are the aspect and elevation angles at which the source characteristics are to be measured?

## Frequency Range

Since the aim is to characterize sources for environmental impact assessments, the frequency range of interest is driven by the sensitivity of marine life.

This report advises a **frequency range** for characterizing sources that at least includes the **10–50 kHz decidecade bands** (ranging from 8.91 to 56.2 kHz).

This requires a minimum sample rate above 113 kHz, which is practically achievable, because many off-the-shelf audio-recorders offer a 128 kHz sample rate.

Where possible, the acoustic frequency range should be extended up to the 160 kHz decidecade band (i.e., up to 178 kHz, which requires a minimum sample rate above 356 kHz), to cover the hearing range of high frequency marine fauna, such as the harbor porpoise.

Note that type B sources that operate at primary frequencies above 160 kHz may also radiate sound at lower frequencies, within the hearing range of marine life (Deng et al. 2014).

## Measurement Geometry

The measurement procedures focus on characterizing the far field source sound emissions rather than the effect of the operating environment on the received sound levels. The measurement distances shall be far enough from the sources that near-field effects are negligible, but also near enough to avoid poor signal-to-noise ratio (SNR) in the measurements and uncertainties associated with quantifying long-range propagation effects. A specific difficulty in dealing with long-duration signals is that source position of moving sources at the time of sound signal emission

becomes difficult to define. Nevertheless, if the source-receiver geometry variation is small over the signal repetition period, longer-duration signals can also be accommodated under the measurement procedures specified.

Figure 2 schematically illustrates the detailed measurement geometry and partners with Table 2 to define the geometric parameters.

### Measurement Distance

Ideally, source characterization measurements are made with sensors in the far field of the source, so that it can be described as a point source. As a first requirement (from ANSI/ASA S1.20:2012 developed by the Acoustical Society of America (ASA) and accredited by the American National Standards Institute (ANSI)), the minimum range $r$ that is required for measuring in the far field, should fulfil:

$$r > \pi A^2/4\lambda \tag{1}$$

with acoustic wavelength $\lambda = c/f$ (for sound speed $c$ and frequency $f$) and $A$ the maximum source dimension $A = \max(L_x, L_y, L_z)$.

In addition to the requirement to measure in the acoustic far field, there may be external requirements to maintain a safe minimum distance between source and hydrophone. For example, for the safety of personnel and equipment, or for the practicality associated with deploying and retrieving moorings near production activities.

This minimum specified distance depends on source type. For type A the recommend distance is 100 m and for type B it is 50 m. Moreover, the minimum distance of 100 m allows for the type A measurement zone to be of sufficient size to accommodate the low signal repetition rate, which is not necessary for type B sources.

The above requirement for the acoustic far field does not necessarily apply to type C sources that may contain multiple incoherent source mechanisms. The maximum dimensions associated with these source mechanisms are generally smaller than the maximum dimensions of the complete platform and unknown at the time when the measurement locations are planned. Hence, the frequency dependent minimum range for measuring in the far field (Eq. 1) is not required. For type C sources, hydrophones shall be deployed at a horizontal distance greater than or equal to 1.5 times the maximum horizontal dimension ($L_x$, $L_y$) of the surface that encloses the source, measured from the position on the contour that is closest to the hydrophone.

Hydrophone locations shall be selected such that no objects block a line-of-sight from the hydrophone to the full surface that encloses the source or affect the sound pressure measurements via acoustic reflections.

Hydrophone locations shall be selected such that the water depth along the trajectory between source and hydrophone is close to uniform or varies with a gradual slope.

### Vertical and Horizontal Observation Angles

Measuring the full directionality of source emissions is impractical and beyond the scope of the proposed measurement procedures. Instead, five main observation directions are defined, one keel and four horizontal aspects (see the plan view in Fig. 2).

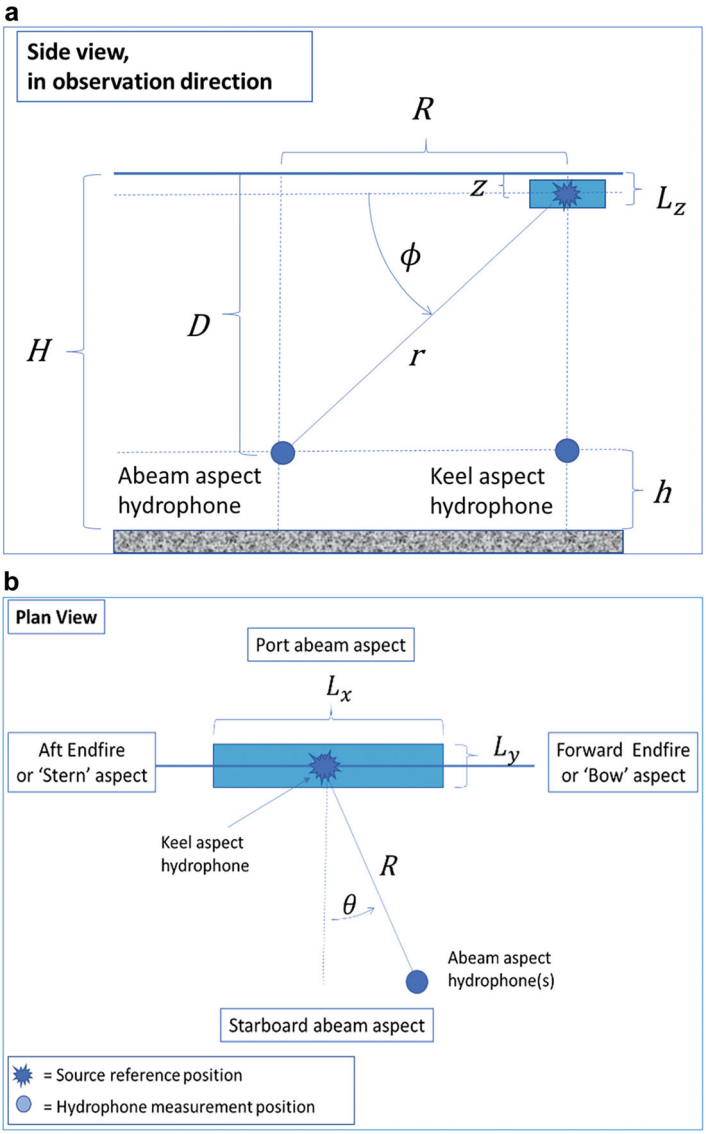



**Fig. 2** Schematic side view (top) indicating source traveling toward the reader, and plan view (bottom) of the measurement geometry indicating source traveling left to right. The sound source is contained within a (blue) bounding box (with length $L_x$, width $L_y$, and height $L_z$). Far field sound emission characteristics are determined for a hypothetical point source inside the box, which is assumed to be at a distance $z$ below the water surface. The figure parameters are defined in Table 2.

Keel aspect addresses the sound emission nominally in the vertical direction ($\phi = 90°$), directly below the source. This direction is of particular interest for type A and B sound sources because animals situated directly beneath a source will often experience the highest sound levels.

The four main horizontal aspects (bow, stern, and port and starboard abeam) are defined to characterize sound emissions mainly in horizontal directions. These measurement geometries also prescribe associated vertical angles, but the goal is to characterize sound emissions that mainly propagate horizontally to assess the exposure of animals at larger distances from the source. The bow is in front of the source and the aft is behind the source. Looking to the bow, port is to the left and starboard is to the right of the source, both "abeam" aspects (port and starboard) are perpendicular to the source track. For ships, abeam aspect is the direction in which the radiated noise level measurements of ships are made, according to the ISO 17208-1 standard. For the type A and B sources, the abeam definition is adapted to indicate the bearing projected at right-angles from the bow and aft line, outward from the center of the source. For type C sources, the horizontal directionality needs to be specified case-by-case.

Sound source measurements are to be taken from different aspects:

- For type A sources from at least three aspects: either the bow or stern aspect, either the port or starboard abeam aspect, and the keel aspect.
- For type B sources the procedure requires measurements from at least two aspects: either the port or starboard abeam aspect, and the keel aspect.
- For type C sources the user should define the aspects.

The vertical observation angle ($\phi$), see Fig. 2, needs to be representative of sound emission angles that lead to long-range propagation. Those propagation paths are relatively shallow, but to obtain reliable measurements it is necessary to maintain a minimum $\phi$ to avoid excessive surface interference effects that occur at very shallow angles.

Therefore, in *deep* water (see section "Water depth") the following three abeam measurement angles are to be selected: $\phi = 15°$, $30°$ , and $45°$, which are based on ISO 17208-1 geometry specified for radiated noise level measurements of ships.

The opportunity to measure the abeam aspect source emissions at these three angles is limited by the minimum measurement distance ($R$) necessary to be in the acoustic far field of the source, the water depth ($H$), the distance between the hydrophone and the sea floor ($h$), and the source depth ($z$).

The hydrophone geometry for abeam aspect measurements in *shallow* water (see section "Water depth") uses a single hydrophone positioned at horizontal distance $R$ off the source track and at depth as close as possible to the seabed to maximize the angle $\phi$ of the direct acoustic path.

There is no general standard for selecting a minimum height ($h$) of the lowest hydrophone above the sea floor. Proximity of the hydrophone to the sea floor may affect the measurements due to contributions of bottom-reflected sound, interface waves, and other sea floor vibrations that can cause hydrophone accelerations leading to unwanted noise. Alternatively, flow noise on hydrophones close to the sea floor can be reduced because water currents there are often lower. Hence, the effects of hydrophone position relative to the sea floor will need to be evaluated for each project to ensure they are accounted for appropriately.

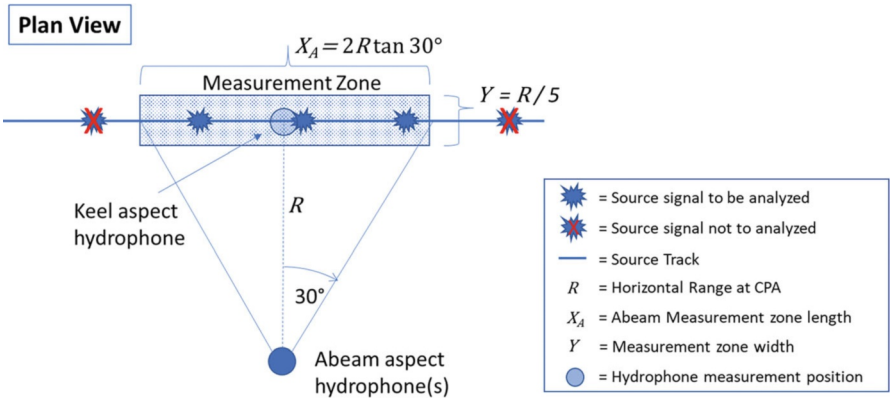



**Fig. 3** Plan view of abeam and keel aspect measurement zone relative to hydrophone locations

For moving sources (types A and B), a measurement zone is defined along the portion of the track subtending an angle of $\pm 30°$ relative to line segment between the abeam hydrophone position and the source's CPA position, as shown in Fig. 3. The measurement zone is centered on the source track and is given a width of $Y = R/5$ to accommodate up to $\pm 10\%$ range deviation that might result from the source being towed slightly off-track. This allowance is expected to result in maximum measurement errors less than 1 dB. However, when measuring type B sources with narrow cross-track beam widths, the off-track deviation may have to be smaller to ensure the keel hydrophone samples the beam.

### Water Depth

This report defines minimum water depths for both *deep water* and *shallow water* measurement geometries.

The minimum water depth allowed for the *deep water* geometry is required to obtain measurements of at least $\phi = 45°$ vertical angle (see Fig. 2) when the source reaches its closest point of approach (CPA), corresponding with horizontal distance $R$, to the abeam hydrophones. This means that the water is *shallow* when the water depth is less than the horizontal measurement distance $R$ plus the source depth ($z$), plus the distance of the hydrophone above the sea floor ($h$).

The minimum water depth allowed for the *shallow water* geometry is defined to obtain measurements of at least $\phi = 15°$ vertical angle (ref Fig. 2) when the source reaches its CPA.

## Requirements for Statistical Analysis of Source Output

Multiple measurements over time are required to evaluate statistical variations caused by:

- Variations of the source signal (a sensor at a fixed distance from a type C source can be useful to provide information on source signal variations over time).
- Geometry variations (for type A and B sources that are generally towed or mounted on a moving platform), including variations due to small-scale seabed features and geoacoustic parameter differences that affect seabed reflectivity.
- Variations in environmental conditions (wind, currents, etc.)

Multiple sensors are required to evaluate statistical variations caused by:

- Evaluating the source metrics in different aspects and frequency bands.
- Statistical uncertainties in propagation loss estimation.

## Data Processing (Received Sound Metrics)

The term "data processing" is used here to mean the conversion from recorded sound signals to processed metrics of received sound at the position of each hydrophone. The focus is on metrics for describing sound pressure signals that are suitable for the quantification of their possible biological impact, and for conversion to source output metrics. Both applications are topics of scientific investigation. Future updates of this report may include additional metrics. Specification of the time period and the frequency range over which these metrics are calculated depends on the signal type (*pulse*, *nonpulse*, or *continuous*, see Table 1), as well as on whether the source is moving (type A and B) or stationary (type C) during the measurements.

The report specifies the data processing steps that are required to quantify these metrics, starting from the digitized representation of the measured sound pressure at the output of the ADC at the end of the measurement chain.

### Metrics for Pulse and Nonpulse Signals

The main received sound metric to be calculated for pulse and nonpulse signals is sound exposure level ($L_{E,\,p}$).

The *temporal analysis window* (TAW) is a time window corresponding with signals captured while the source was within the spatial measurement zone (Fig. 3). The TAW may include multiple pulse or nonpulse signals, as illustrated within the blue box of Fig. 4.

The *temporal observation window* (TOW) is a time window that contains a complete received signal of a *single* pulse, as illustrated within the green boxes of Fig. 4, or the full sequence of signals of interest for a single nonpulse transmission nonpulse. The reported statistics are calculated from the metrics of multiple individual TOWs.

Other metrics such as sound pressure level ($L_{p,\mathrm{rms}}$), peak sound pressure level ($L_{p,\mathrm{pk}}$), or SEL with a specified (auditory) frequency weighting ($L_{E,p,\mathrm{w}}$) may be

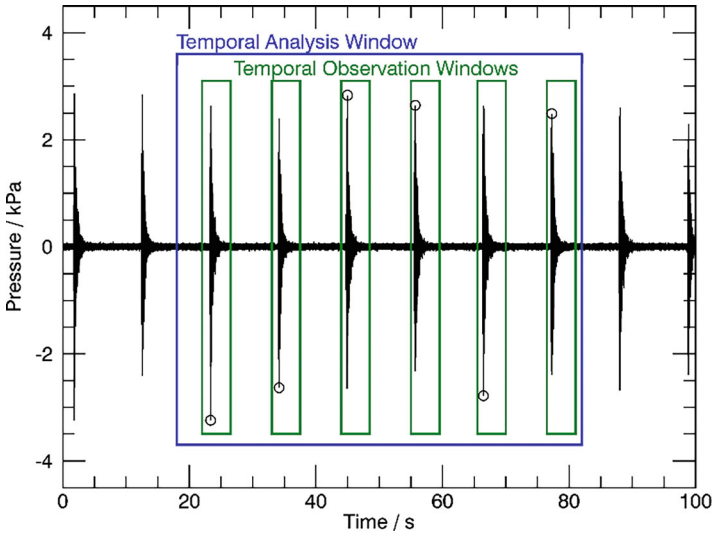



**Fig. 4** Illustration of time windows for pulse and nonpulse signals (example for a sequence of airgun pulses). Black curve: sound pressure time series; the black circles indicate the (compressional or rarefactional) peak sound pressures of single pulses within each individual pulse's respective TOW. Green boxes: TOW for $L_{E,\,p}$ for six complete pulses. The blue box spanning the six pulses represents the selected TAW, during which the source was within the spatial Measurement Zone

required. It may also be of interest to determine additional metrics such as, e.g., kurtosis, see (Prior et al. 2019; Martin et al. 2020; Müller et al. 2020).

## Metrics for Continuous Signals

The main received sound metric to be calculated for continuous signals is sound pressure level ($L_{p,\mathrm{rms}}$).

For continuous signals from stationary sources (type C), $L_{p,\mathrm{rms},60\ \mathrm{s}}(t)$ shall be calculated over a series of time intervals (TOW) $T = 60$ s. The TOW duration is selected in correspondence with the guidelines for observation of ocean sound proposed by the International Quiet Ocean Experiment (IQOE 2019).

## Calculation of Source Output Metrics

Underwater sound sources are typically quantified in terms of their source level. Since 2017 the term source level has been defined in an international standard for underwater acoustic terminology [ISO 18405]. This defines source level ($L_S$) in terms of the product of the square of the distance and the mean-square sound

pressure at that distance in the acoustic far field of the sound source "if placed in a hypothetical infinite uniform lossless medium of the same density and sound speed as the real medium at the location of the source, with identical motion of all acoustically active surfaces as the true source in the true medium." This cannot be measured in practice, but it can be calculated from measurements of sound pressure level ($L_p$) in a real environment. If there is one and only one sound source contributing to the sound field, source level can be calculated by adding the propagation loss ($N_{\mathrm{PL}}$) between the source and receiver locations in that environment to the measured sound pressure level

$$L_S = L_p + N_{\mathrm{PL}}. \tag{2}$$

The main sound source metric to be calculated for pulse and nonpulse signals is "energy source level" ($L_{S,E}$), equal to the sum of sound exposure level ($L_E$) and propagation loss ($N_{\mathrm{PL},E}$) for the measurement environment and geometry (ISO 18405)

$$L_{S,E} = L_E + N_{\mathrm{PL},E} \tag{3}$$

Propagation loss is usually assumed to be a frequency-dependent, time-invariant property of the environment and geometry over the duration of the measurement period. Frequency domain representations of $N_{\mathrm{PL}}$ and $N_{\mathrm{PL},E}$ for the same measurement geometry and environment are equal ($N_{\mathrm{PL}} = N_{\mathrm{PL},E}$) and can be determined from:

- Measurements on a source with a known source level, in the same environment with the same measurement geometry. Note: In an environment with multipath propagation, the three-dimensional directionality of this source needs to reproduce the directionality of the actual source.
- An appropriate numerical propagation model that accounts for source and receiver geometry, sea surface and seabed reflections, and acoustic absorption, where possible validated (or calibrated) for the measurement environment.
- An appropriate semiempirical ("practical spreading law" (Ainslie et al. 2014)) model, calibrated by measurements at multiple distances from the source. This approach is considered suitable for measurements of type B (HRG) source measurements in which propagation loss is dominated by spherical spreading and absorption in sea water.

Note that ISO 18405 provides no definition for source metrics resulting from measurements of zero-to-peak sound pressure level, hence these are not included.

Two different type C source configurations are considered:

(a) Floating platforms in deep water (Fig. 2, top).
(b) Bottom mounted platforms that extend over the water depth.

The source level of a *floating platform* can be determined following the approach for measuring ship source level (ISO 17208-1 and 17208-2) when measured in deep water. The source level can be defined in shallow water as well; however, a standard procedure for measuring ship source level in shallow water is still under development (ISO 17208-3).

For *bottom-mounted platforms* in shallow water, calculation of source metrics faces the same problems as observed for offshore pile driving. The scope of the ISO standard for measuring radiated underwater sound from percussive pile driving (ISO 18406) does not include the calculation of source level metrics. The Annex to ISO 18046 explains that it is not yet clear how to provide an entirely robust definition of source level for percussive pile driving. Sound sources that penetrate both the water surface and the seabed are intimately connected to the environment. This means that if the environment changes, so does the source output. Hence these sources cannot be properly characterized independent of the environment in which they operate. Nevertheless, source characterization may be required to map the sound distribution around their operational location. In that case, it is advised to adopt an approach based on inter- and extrapolation of the received sound metrics at the measurement locations to the sound map of the environment.

## Conclusions

- Reports describing standard procedures for characterizing underwater sound sources associated with oil and gas exploration and production activities are available online from http://www.soundandmarinelife.org/library.aspx.
- Regular maintenance of these standards is recommended. ISO standards are reviewed every 5 years. Therefore, feedback from users of these standards is appreciated, to support future updates, incorporating lessons learned.
- In the event that procedures cannot be precisely followed, it is imperative that all deviations from the standard procedures be thoroughly documented and reported. The importance of following these standard procedures is the provision of comparability between studies and results.

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# A Global Library of Underwater Biological Sounds (GLUBS): An Online Platform with Multiple Passive Acoustic Monitoring Applications

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**Abstract**

Aquatic ecosystems contain some of the world's most diverse environments, and their soundscapes are often teeming with sounds from a wealth of biological sources. Passive acoustic monitoring (PAM) is an increasingly accessible technique that offers an unprecedented, non-extractive means to "observe" these habitats, many of which are too deep, dark, turbid, or remote to sample easily with other methods. Applications to assist analysis of PAM data already exist (e.g., reference libraries, data portals, discussion forums), machine learning code is increasingly more available, and citizen science programs are broadening public interest in underwater sound. However, individually, these resources do not realize their full potential. To help address this limitation, a working group for a Global Library of Underwater Biological Sounds (GLUBS) has proposed an open-access web-based single-point-of-contact platform to integrate and expand these applications to help broaden and standardize scientific and community knowledge of underwater soundscapes and their contributing sources. This paper presents a summary of a meeting of the GLUBS working group that was

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held at "The Effects of Noise on Aquatic Life, 2022," including some of the core values, initial targets, points for design, data management issues, and potential avenues for stakeholder engagement.

**Keywords**

Underwater soundscape · Soniferous fauna · Artificial intelligence · Bioacoustics database · Biodiversity · Ecological informatics · Reference library · Outreach · Collaboration

## Background

Marine and freshwater environments form some of the world's most extensive and diverse realms, often teeming with a variety of biological, anthropogenic, and geophysical sounds that can help us monitor the presence, number, and behavior of individual species, inform on biodiversity trends, and observe the type and quality

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of local habitat at the recording site (e.g., Greenhalgh et al. 2020; Linke et al. 2018; Mooney et al. 2020; Rountree et al. 2020b; Williams et al. 2022). However, the world's aquatic ecosystems are experiencing rapid changes as our planet progresses through the Anthropocene epoch, and in response, their inhabitants are changing their distribution, behavior, and contributions to the soundscape (Duarte et al. 2021). Passive acoustic monitoring (PAM) is a technique for sampling aquatic systems that can be particularly effective in deep, dark, turbid, and rapidly changing or remote locations (ACT 2007; Bolgan and Parmentier 2020; Gordon et al. 2018; Lin et al. 2019; Mellinger et al. 2007; Van Parijs et al. 2009). Technological developments of the last two decades have significantly advanced the logistical, acquisitional, analytical, and economic facets of PAM (Chapuis et al. 2021; Lindseth and Lobel 2018; Waddell et al. 2021; Wall et al. 2021), and as a result, collection of PAM data has increased dramatically in both deployment number and data volume, to a point that traditional acoustic data processing techniques are becoming ineffective (Mooney et al. 2020; Parsons et al. 2022). The recent increased volume of data and focus on aquatic ecoacoustics have also led to the discovery of many undescribed sounds from unknown sources that contribute to underwater acoustic diversity, particularly in coastal environments, the sources of which remain largely unknown (e.g., Bolgan and Parmentier 2020; Di Iorio et al. 2021; Rountree et al. 2020a). There is much space for collaboration among the growing research community asking questions in this area; the co-creation of a Global Library of Underwater Biological Sounds by this community could potentially serve to facilitate many of the goals of PAM scientists.

New methods of processing aquatic PAM data are continually being developed and multiple applications have recently been published: Reference libraries that document groups of taxa are increasingly being created (e.g., Looby et al. 2023a; Mellinger and Clark 2006; Ocean Networks Canada 2021; Sayigh et al. 2016; Vigness-Raposa et al. 2012); focus groups have developed event extraction, detection, characterization, localization, and density estimation machine learning algorithms (e.g., Frazao et al. 2020; Lin et al. 2021b; Shiu et al. 2020; Sun et al. 2022); data portals to upload long-term and short-term recordings are online (e.g., Darras et al. 2022; OPUS 2022); software to extract soundscape characteristics from long-term recordings are becoming standardized (e.g., Merchant et al. 2015; Miksis-Olds et al. 2021); and forums are being created to encourage discussions about PAM analysis (e.g., https://bioacoustics.stackexchange.com/).

At the same time as these PAM advances, some wider-reaching ecological tools now have the potential to include acoustic information. The World Register of Marine Species (WoRMS; Costello et al. 2013), for example, provides a list of all aquatic and semiaquatic species, including a selection of their characteristics or "traits," the Ocean Biodiversity Information System (OBIS; Fornwall 2000) can map the regional or global distribution of these species, and Fishbase (Froese and Pauly 2022) provides biological and ecological information on adult finfishes around the world. A web portal that keeps an up-to-date set of links to tools demonstrating the latest best practice in PAM data processing and sharing could be of great value to the research community.

Individually, all these applications offer a wealth of information to better understand aquatic soundscapes and the soniferous species that contribute to them. However, to maximize their use and streamline analysis processes, these tools need to be integrated on a cohesive platform (Parsons et al. 2022). Previous efforts to create enduring PAM-related applications, such as acoustic libraries, have often been unable to secure sufficient ongoing funding to continue updates and maintenance once the initiator departs the host institute and therefore become outdated or without license (Bianco et al. 2019; Frasier 2021). To overcome these issues requires significant initial and long-term investment, driven by substantial user endorsement and uptake, stakeholder engagement, appropriate governance on national and international levels, and scalable infrastructure to cope with the expansion of PAM (Parsons et al. 2022).

A working group for the Global Library of Underwater Biological Sounds (GLUBS), initiated by the International Quiet Ocean Experiment (IQOE; Tyack et al. 2015), has proposed the development of a web-based platform to bring these applications together in a single point-of-contact interface and to facilitate the integration of a new wave of PAM data mining, extraction, and artificial intelligence (AI) techniques (Parsons et al. 2022). A similar approach is underway to collate museum collections and allow the querying of their data for characteristics and specimens of individual species (vertnet.org). For PAM, GLUBS would be a critical step forward in collating, documenting, understanding, and mapping the world's underwater soundscapes and their sources, as they respond to increasing pressure from climate change, fishing, aquatic resources development, and various pollutants (e.g., plastics and anthropogenic noise). For individual researchers that lack data storage support, such as the servers of large institutions, this platform will provide a much-needed avenue to ensure the longevity of samples of biological sounds. GLUBS will also facilitate growth in knowledge and understanding of aquatic soundscapes among the potential stakeholders through sharing soundscape data and the sounds that contribute to them, sharing the algorithms that can analyze the data, and open discussions about issues, new developments, and the ever-occurring question, "What's that sound?".

The following discussion summarizes the focus points discussed at a GLUBS workshop on July 16, 2022, following "The Effects of Noise on Aquatic Life, 2022" conference in Berlin, Germany. This 1-day GLUBS meeting was attended by 20 partners (7 in person, 13 online). The authors of this report comprise the attendees, plus several contributors to specific subgroup discussions included below.

## GLUBS Vision and Framework

The overarching GLUBS objective is to provide a holistic approach to PAM that is as useful for as many stakeholder sectors as possible. This includes, for example, processing and analysis packages for researchers of varying experience and disciplines, application-based information to assist management decisions, knowledge, and the opportunity to contribute for citizen scientists and soundbites for musicians and artists. To do this, GLUBS will be a web-based application programming

interface (API), serving as an umbrella platform that links external software applications and purpose-built GLUBS software packages that perform multiple complementary functions. It will host dedicated GLUBS data storage and links to external databanks, such as existing sound libraries and national sound repositories, from which it would have the functionality to record metadata, extract data from long-term recordings, and store "snippets" (i.e., relatively short-term recordings of individual sounds, such as a fish sound or humpback whale song). The GLUBS framework will also include software to fill functionality gaps left between the integrated applications.

Approaching such a multidisciplinary objective from an individualistic or discipline-focused approach may not unveil the functionality required by all users and thus could limit stakeholder uptake needed to ensure enduring support. Therefore, a decision was made to achieve community consensus on the structure and scope of such a platform, prior to implementing any design. To achieve this, a bioacoustic community appraisal was designed to collate the experiences, knowledge, and needs of stakeholders and will develop the GLUBS structure with consideration of the community's views. Therefore, GLUBS will be a product, designed by a consortium of bioacousticians and implemented under the auspices of the IQOE.

## Core Values

To ensure outcomes are as equitable and valuable as possible, the platform will be based on open-science principles and follow a series of core values and endorsed/standardized principles, essential to modern research. The GLUBS platform would therefore:

- Follow FAIR (Findability, Accessibility, Interoperability, and Reusability of digital assets) principles (The Turing Way 2021; Wilkinson et al. 2016).
- Be as inclusive and transparent in its design and implementation as feasible.
- Ensure identifiers are used to track usage and provide recognition for contributors.
- Minimize national bias (e.g., incorporate multiple languages, wherever possible).
- Prioritize net environmental benefit, which may include being carbon neutral, and ensure outputs follow standardized and community-endorsed processes, where possible. For example, GLUBS would follow International Standards Organization (ISO) definitions (International Organization for Standardization 2014) and use IQOE-endorsed applications.

## Framework

The final platform design, contributing applications, and how they can be integrated into the platform are part of the engagement process and will be chosen and defined over the coming year. However, GLUBS will be structured and written in a programming language that facilitates integration with external applications and not necessarily exclusive to individual packages (e.g., this would allow potential data

portals to connect to GLUBS and their users could interact with GLUBS). There is no intention to supersede existing applications, but, where possible, GLUBS will help enhance or expand them, typically to include functionality required to produce desired outputs of GLUBS. For example, this could include supplying libraries with new sounds; highlighting target (data-limited) "wanted" species; developing soundscape analysis tools for processing software to include novel, established metrics; creating generalized analysis tools that can visualize acoustic data from different databases; or providing AI groups with training datasets to test and evaluate and benchmark their new algorithms.

The framework will be categorized into essential and preferential functions and applications. Different levels of implementation will match stage-gated funding to ensure that, if initial funds were limited, the core structure is in place and will have the technical capacity to be expanded to include additional, predesigned levels, when sufficient funds are secured. This will ensure core functionality is available to as many users as quickly as possible. Core features have been broadly identified as:

1. User front-end interface with query structure (to identify user objectives) and metadata forms
2. Data portal(s) potentially being different for sounds and soundscapes
3. Links to external databases that can interactively improve the observation of marine biodiversity and ecosystems, such as WoRMS, Fishbase, and OBIS
4. Integrated internal and external reference libraries
5. Training datasets for AI algorithm development
6. Event extraction/machine learning software and links to external AI groups
7. Graphical interface(s) for scientists/public/artists to visualize and download information
8. Software package for soundscape analysis to produce data products (e.g., sound levels, acoustic indices, visualizations)
9. Forum (internal or external) for bioacoustics discussion

## Initial Targets: Collating Information on Known Soniferous Species and Unidentified Sounds

To assist the goal of documenting as many biological aquatic sounds as possible, the GLUBS working group (WG) set two initial targets. The first was to produce and maintain a list of all aquatic species confirmed or anticipated to produce sound underwater. The WoRMS comprises a searchable database of selected characteristics (known as "traits") for all extant and extinct aquatic and semiaquatic species, for which the distribution of species with this trait can be mapped through OBIS. Including underwater sound production as a searchable trait would facilitate distribution mapping of soniferous species for a specific region or globally. One could also choose a location and extract a list of all confirmed or likely soniferous species known to occur there or choose a species and investigate how their sounds vary through space and time. Integrating this information with a reference library of

sounds would present all the sounds that one might expect to be present in a recording at a given location and time. To begin to achieve this objective, a GLUBS sub-working group has produced a database of the extent to which all known aquatic species produce sound (Looby et al. 2023b). To achieve this, each species was assessed to determine in what way they answer the question "Does the species actively or passively produce sound under natural conditions while submerged in water?" following set definitions of active and passive sound production in natural and unnatural conditions (see Looby et al. 2022 for definitions). The trait has six categories according to whether a species has been confirmed or is considered likely to produce sound (Table 1). This database can be updated on WoRMS as reports of new species confirmed to exhibit soniferous behavior are released and later incorporated with sound recordings.

The second initial target is to promote the reporting of sounds from unknown sources. Unidentified sounds can provide valuable information on the richness of the soundscape, the acoustic communities that contribute to it, and behavioral interactions among acoustic groups (Bolgan et al. 2020; Di Iorio et al. 2021; Sun et al. 2022). However, unknown, cryptic, and rare sounds are rarely target signals for research and monitoring projects and are, therefore, largely unreported (Parsons et al. 2022). To address this knowledge gap, GLUBS will facilitate a series of forums to promote research of such signals, make knowledge of their characteristics more accessible, and increase awareness of their occurrence. The first of these forums is a special issue on unidentified underwater biological sounds that is being developed for the journal *Frontiers in Remote Sensing* (https://www.frontiersin.org/journals/remote-sensing).

## Ongoing Design, Development, and Implementation Discussions

### The Snippet, the Snapshot, and the Soundscape

Passive acoustic monitoring encompasses the assessment of individual sounds and the soundscape through time and space. Short acoustic recordings (here called "snippets") may record individual sounds, while a longer recording (e.g., minutes, hours, days) can provide a "snapshot" of a soundscape and include numerous sounds, and recordings that extend into months or years provide the opportunity to capture all the cycles (e.g., naturel diel, tidal, lunar, seasonal patterns in biological noise) and other fluctuations (e.g., passing vessels, earthquakes, cyclones, seismic surveys) in the soundscape, across many time frames. In its simplest context, the soundscape is defined as a sound or combination of sounds that forms or arises from an immersive environment (International Organization for Standardization 2014). It is important to note that characteristics of the soundscape vary with time and so the length of the recording may affect whether the recording captures all the potential variation and sounds that naturally occur at the recording site (Parsons et al. 2016). In contrast, soundscapes are degrading due to human influence over timescales of months to years (Gordon et al. 2018), and the PAM community is currently

addressing the challenge to develop rapid assessment tools that can detect both anthropogenic impacts to soundscapes and the "sound of recovery" to healthy soundscapes (Lamont et al. 2022a).

**Table 1** Trait and category definitions for species sound production characteristics in the World Register of Marine Species, based on our current state of knowledge regarding sound production

| Identifier | Presented information/field | Definition |
|---|---|---|
| Trait | Species exhibits underwater soniferous behavior | Does the species actively[a] or passively[b] produce sound under natural conditions[c] while submerged in water? |
| Category 1 | Unknown or undetermined | There is no known acoustic[d], morphological, or physiological[e] study of sound production for this species, and its likelihood of soniferous behavior based on an ancestral state reconstruction analysis[f], lineage, or evolutionary records has not been assessed, or this species has yet to be categorized by an expert source |
| Category 2 | Does not or is unlikely to produce sound under natural conditions | Species has been the subject of one or more studies to assess its acoustic behavior[d] morphology and/or physiology[e], none of which have provided evidence of sound production either actively[a] or passively[b], and/or species does not have a likelihood of soniferous behavior based on an ancestral state reconstruction analysis that results in a probability below 0.5[f], lineage, or evolutionary records |
| Category 3 | Likely to produce sound under natural conditions but unconfirmed | There is no known acoustic documentation[d] to confidently validate sound production under natural conditions[c] by this species, but it is likely to exhibit natural soniferous behavior based on an ancestral state reconstruction analysis that results in a probability of 0.5 or higher[f], lineage, evolutionary records, morphological characteristics, physiological characteristics[e], documented sound production behavior under artificial conditions[g], and/or documented sound production behavior with some uncertainty[h] |
| Category 4 | Produces passive sound under natural conditions | There is validated[i] acoustic documentation[d] that this species produces passive[b] sound under natural conditions[c], but no validated documentation to confirm that it actively[a] produces sound |
| Category 5 | Produces active sound under natural conditions | There is validated[i] acoustic documentation[d] that this species produces active[a] sound under natural conditions[c], but no validated documentation to confirm that it passively[b] produces sound |

(continued)

**Table 1** (continued)

| Identifier | Presented information/field | Definition |
|---|---|---|
| Category 6 | Produces active and passive sound under natural conditions | There is validated[i] acoustic documentation[d] that this species produces active[a] and passive[b] sound under natural conditions[c] |

[a]Active sound: produced deliberately in association with a particular behavior or situation, frequently with specialized sonic organs or structures, and are generally used for communication; also referred to as intentional, deliberate, or specialized sounds (Looby et al. 2022)
[b]Passive sound: may not be associated with specialized sound-producing structures nor with specific behaviors or situations, though they may still serve some signal function; also referred to as incidental, unspecialized, or mechanical sounds (Looby et al. 2022)
[c]Natural conditions: in the absence of artificial manipulation, such as an electrical current or direct handling
[d]Acoustic study or documentation: documentation in the peer-reviewed or gray scientific literature of the presence or absence of audible sounds through listening live or on a recording in captivity or the field (Looby et al. 2022)
[e]Morphological or physiological study or documentation: documentation in the peer-reviewed or gray scientific literature of the presence or absence of morphological and/or physiological structures with sound production functions (Looby et al. 2022)
[f]Ancestral state reconstruction analysis: the extrapolation back in time from measured characteristics of individuals to their common ancestors using statistical techniques to, in this case, recover information on the likelihood of soniferous behavior (https://doi.org/10.1371/journal.pcbi.1004763 Joy et al. 2016; https://doi.org/10.1643/i2020172 Rice et al. 2022)
[g]Artificial conditions: involving human intervention or manipulation, such as an electrical current or direct handling
[h]Uncertainty: doubt or uncertainty expressed by researchers conducting examinations for sound production about their conclusions (e.g., that the species they examined was correctly identified, that they correctly identified active versus passive sounds; https://doi.org/10.1007/s11160-022-09702-1 Looby et al. 2022)
[i]Validated: The researchers' conclusions have no associated uncertainty in the documentation (see above)

Although all these recording durations provide valuable information, the primary initial function of GLUBS is as a global library of biological sounds, created both through the integration of existing libraries and from new dynamic contributions. The library thus has a focus of separating and presenting individual sounds to assist in source allocation. The individual sounds may then be used to analyze the components of soundscapes in recordings of varying duration. Further, although the focus of GLUBS is on biological sounds (or "biophony"), soundscape analysis is also often conducted with the consideration of geophony and anthropophony (Lin et al. 2021a; b; Pijanowski et al. 2011). For algorithms to be able to recognize biophony in soundscapes among background sounds, geophony and anthropophony will also need to be considered. As the future goal, the platform may extend to the collection and analysis of long-term soundscapes and the associated metadata; however, for the immediate future, GLUBS will aim to focus on snippets of individual sounds while bearing in mind their context in soundscapes.

The assessment of biodiversity, habitat type, and condition are novel and complex issues in the underwater environment. Ecoacoustics is the investigation of the natural

and anthropogenic sounds that shape a location's soundscape and the physical environment and species assemblage at that location, across multiple spatial and temporal scales. While numerous terrestrial PAM "ecoacoustic" techniques have been applied successfully (Farina et al. 2018; Kahl et al. 2021; Potamitis 2014; Sueur and Farina 2015), the results have been mixed when applied to aquatic recordings (► Chap. 37, "Anthropophony Effects on Acoustic Metrics in a Marine Soundscape"). Complexity, spectral and temporal variation, and various metrics of sound levels can contribute to what may be a site-specific reflection of physical biodiversity and condition. Therefore, GLUBS must hold the ability to assess and potentially incorporate new methods, as they become available, and their utility is evaluated.

Calibration of equipment is critical if sound levels are to be evaluated and environmental metadata is also required if source levels are to be calculated. Uncalibrated equipment has valid applications for detecting acoustic events or changes in soundscape characteristics over time within a single recording. How the functionality of GLUBS is applied to the assessment of biodiversity and the quantification of sound levels will be decided during the design process, with the inclusion of soundscapes potentially held as a future ambition of GLUBS. There may also be the need to attach provisos with any outputs to identify their limitations and avoid misinterpretation. At this stage therefore, the intention for the design of GLUBS is that it will have gated input options, so that calibrated and uncalibrated data can be included, and there will be the option to input metadata if it is available. This will maximize the accessibility and utility of the platform.

## Sound Categorization

Recent advances in technology have increased accessibility of underwater PAM to a variety of scientists (with different fields of expertise), managers, citizen scientists, artists, and the general public. As a result, reports of sounds may originate from physicists, biologists, ecologists, managers, or musicians, each of whom is recording for their own objectives and reporting using their own experience. There are currently no standards for classifying or reporting sounds and so these may be categorized onomatopoeically, from spectral or temporal components of the sound, or simply using the common name of the source species (Parsons et al. 2022; Looby et al. 2023c). However, while categories related to species' common names can resonate among local communities, common names and onomatopoeic names can be too subjective, are not widely applicable, and are sometimes meaningless to non-English speakers. While in some countries such names have had success for birds and amphibians (e.g., see online use of BirdNet and FrogID), it has not been successfully realized for mammals and even less so for fishes and invertebrates. To be able to systematically include sounds into GLUBS, a standardized method of sound categorization is needed, preferably based on a statistical method. It is anticipated this will use a multitiered naming system, similar to scientific naming of species (i.e., a decision tree process), which is based on the sound rather than the source. Common or colloquial names could then be applied as secondary levels of

naming to maintain public engagement and understanding, in the same way as with Latin and common species names.

## Data Storage, Management, and Governance

As individual PAM datasets approach volumes of terabytes per site, per year, the potential for collection to outweigh storage capacity at a single node or databank is high, if not already occurring. Data storage capacity is also growing rapidly, and volume may not be a long-term future issue, but it is currently a major consideration (Havlik et al. 2022; Wall et al. 2021). However, PAM data are also transient and cannot be recollected in the true sense, and soundscape characteristics (such as ambient sound levels) are difficult to hindcast without the data. Thus, from an information perspective, to ensure their reusability, all raw data need to be appropriately archived in highly durable storage. The cost of data collection often outweighs that of storage, so there is a trade-off between minimizing replication of data to save storage space (and minimize environmental impact) and maximizing the retention and availability of information. Additionally, minimizing restrictions on contributors is one of the best ways to promote user uptake and endorsement. Some national governments have realized this and already ensure that sufficient space is allocated for important PAM data, such as the US National Centers for Environmental Information (within the United States' National Oceanic and Atmospheric Administration) for ongoing programs that acquire significant data volumes (Wall et al. 2021). It is important for GLUBS to be able to access these data sources and potentially add value to them.

Data managers often recommend one server (on a cloud or hosted server) and one hard copy backup of a dataset, with a web-based copy available for sharing. Although cloud data storage is relatively inexpensive, there is a cost each time the data are retrieved for analysis and this increases with the speed at which data are downloaded. The cost versus benefit of data accessibility should be analyzed and this cost/benefit ratio is currently unknown. Such an analysis may suggest an approach of processing as much data as possible at the outset, before uploading to the cloud, and storing the resulting metadata and outputs on the GLUBS platform. An additional and possibly more advantageous option is to exploit various levels of data storage and archival processes and tailor these to the types of data sources GLUBS will encounter. For example, remote links to national databanks may make it possible to process datasets without downloading and extract required metrics and sound snippets directly, depending on the respective storage method. This type of access could be key to minimizing time and data replication. Software links could be provided to research institutes to process their data in-house and upload the outputs. This would potentially limit GLUBS server requirements to include the reference library sound snippets, and eventually metadata, and soundscape metrics from these two sectors (national data collection programs and research institutes) of large-scale data collectors. Additional data contributors will be individual researchers, hobbyists, and citizen scientists who lack resources for data storage themselves. GLUBS

would need to store what is currently a nebulous, but fast-growing data stream. Developing a strategy to address these points is nontrivial and identifying the full needs is problematic until a full design of the GLUBS structure has been finalized. However, ongoing system maintenance and data management (excluding future expansions and developments) clearly require full-time staff.

The proposed level of data sharing and reporting on the web – across multiple regional, national, and community boundaries – necessitates an understanding of the licensing and sovereignty requirements. When sharing and publishing on this scale, it is important to note that all agreements are not created equal and the authority of some licenses is not applicable outside of its region of copyright origins. To avoid bespoke data-sharing agreements, GLUBS will rely on existing licensing frameworks that provide a level of international compatibility, such as the Creative Commons (CC) suite. No single CC license or waiver will be made mandatory as a data owner's legal jurisdiction may prohibit their involvement in the platform. For example, the "CC0" license waiver is not legal in Australia. It is therefore important to produce a table of different agreements and uses of data across all participating communities or countries to develop a flow process outlining the authority and governance and a "user-friendly" way to identify the possible avenues a stakeholder could contribute to the platform.

From an environmental perspective, the GLUBS core values require its designers, developers, and contributors be aware of the environmental cost of data collection and attempt to minimize the platform's impact. This could be achieved through optimizing face-to-face/online meetings, translating recordings of meetings into multiple languages and storing them to enhance inclusivity, sharing recording equipment, engaging local collaborators to conduct deployments (rather than sending researchers long distance), assessing and minimizing the environmental cost of data storage, and minimizing use of disposable items (e.g., batteries, mooring systems). The environmental aspirations of GLUBS and documenting emissions related to platform development and ongoing data collection will be considered on an iterative basis to maintain the platform is of net environmental benefit. Any environmental data and archived recordings of meetings will be stored and made publicly available via the GLUBS platform.

## Artificial Intelligence Needs

Serving as a platform for developing AI algorithms has been identified as a key feature of GLUBS. This requires the creation of manually annotated datasets from diverse marine environments that can serve as training and testing data of AI-based sound detection, classification, and separation applications. Many automated detectors for particular sound sources have been successfully developed. However, although one detector that works in one place has significant value, for single sound detection and species identification, to date, multiple research studies have shown that there is no single detector solution for all sources and locations. Recently, BirdNet (Kahl et al. 2021) and the LIFEPLAN project (https://www.helsinki.fi/en/

projects/lifeplan) have made significant advances in this area, for bird calls. Lauha et al. (2022) describe the development of a pipeline for a global identification model that can be adjusted to detect local calls when supplemented with annotated local recordings.

To develop this field in the aquatic environment, there is therefore a need to identify the various objectives of acoustic signal processing within GLUBS; to engage multiple AI groups in aquatic, terrestrial, and avian research; and potentially to leverage the state-of-the-art AI tools used in music and speech information retrieval. Such data sharing has previously proven successful as evidenced by open competitions for developing algorithms (e.g., Frazao et al. 2020, www.dcase.community/challenge2022/index) and the software applications already available in this field (e.g., Bird Sounds Global, Wikiaves, OrcaSound, etc.). As with open forums and competitions, such engagement is conducted with the ethos that GLUBS outputs provide mutual benefit to the AI groups.

Although data from different contributors will hold characteristics (e.g., sampling frequency, system frequency response) that may affect algorithm performance and compatibility, it is possible to create benchmark AI datasets. These will require samples from all potential environments (e.g., with different background noise, habitat, propagation characteristics from multiple geographical areas), must be carefully identified, and ideally meet as many criteria from an AI developer's wish list of characteristics as possible. To this end, any GLUBS AI dataset should include the following:

- Clear objectives of audio signal processing and suggested metrics to assess performance
- Many (i.e. potentially tens of thousands) examples of the target signals
- Many examples of context (backgrounds, other sounds such as those produced by nontarget species)
- Representative scenarios in which the model will be used (noise, seasonal changes, etc.)
- Comprehensive, validated (false and missed detections) test sets that also reflect the expected usage scenarios
- Annotation (start/end times, lower- and upper-frequency limits) throughout the training data that can include mixture of strong and weak labeling
- An annotation style that generally matches the expected output of the machine learning model

There are several avenues through which datasets can be annotated: by those that created the dataset, by the GLUBS network, or even by citizen scientists. It will be important to develop a roadmap to identify how each of these options can contribute to the datasets and engage contributors.

Sounds of rare, data-deficient species can provide key ecological information and should not be ignored or overlooked. Data augmentation methods will be trialed for such species, noting that few-shot learning and unsupervised learning methods are available. Automatic clustering techniques like scikit-maad (Ulloa et al. 2021) can

be used to identify and group (or "cluster") matching rare sounds and can be used as a first step to build machine learning detectors of rare sounds. Further, it may be necessary to develop and compare AI algorithms that can account for these data-deficient sounds in the assessment of soundscapes, with those that do not. However, as confidence in AI detections improves with the size and quality of the training dataset, it may be necessary to tier databases to highlight and potentially target such sounds for ongoing development.

## Stakeholder Engagement

Historically, several applications and acoustic programs have fallen by the wayside because of lack of user and broader stakeholder uptake (Parsons et al. 2022). However, some platforms remain successful, providing valuable lessons on community engagement and potential partners for moving forward (e.g., DOSITS.org). As GLUBS is planned to be a multidisciplinary and multi-sector platform and therefore requires continued engagement at multiple levels, it is also important to not only engage but to provide recognition to those who contribute data and information. This practice will promote confidence in the platform and potentially improve user opportunities (e.g., citations of contributors' sounds to improve their chances of winning grants).

### National PAM Strategies

Multi-decadal long-term, non-military PAM datasets are few (Caiger et al. 2020, www.acoustic.aodn.org.au/acoustic/); however, several national government departments are dedicating resources toward ensuring that these types of programs can be set up, maintained, and even expanded into the future (e.g., Wall et al. 2021), highlighting the increasing acceptance of ocean sound as an Essential Ocean Variable (EOV; Tyack et al., in prep.), increasing the provision of baseline or reference soundscape data, and advancing this notion to freshwater and brackish environments. The development of national strategies in collaboration with a standardized global platform like GLUBS could increase user uptake.

### Scientists

As stated above, the decision has been made to engage the broader bioacoustics community to ensure the core element of the platform (acoustics research) meets their needs, and the best way to achieve this is through a co-design and endorsement process. However, this needs to be an ongoing ethos and the best way to ensure this is through providing recognition and credit, where appropriate. For example, persistent identifiers (PIDs), digital object identifiers (DOIs), open researcher and

contributor identifiers (ORCIDs), and Research Organization Registry (ROR) tags can be added to contributed recordings, metadata, or algorithms, with the agreement that anyone using the data associated with these identifiers is required to cite their origins whenever published.

## Citizen Science

There are several different levels at which public engagement can occur (e.g., data collection and analysis, individual citizen science, passive consumption, artistic and science communication projects), and there are many members of the public that can be involved for these processes. In the underwater PAM realm, this has most recently been achieved through the Google Arts "Calling in our Corals" project, presented recently at COP 27 (www.unfccc-events.azureedge.net/COP27_87813/agenda). There are also multiple ways to incentivize citizen scientists/public users. Virtual community competitions with points rankings, or online rewards (e.g., virtual toys that increase in size with contribution), for example, have already been shown to thrive to increase validation datasets. Additionally, citizen scientists, keen to become fully engaged with PAM programs, could contribute sound recordings with access to low-cost recording systems (e.g., Hydromoths, GoPros) that are being assessed to identify their potential level of scientific contribution (e.g., Chapuis et al. 2021; Lamont et al. 2022b).

The capacity to crowdsource the transcription of GLUBS text into multiple languages as has been successful elsewhere is highly appealing and, similar to other aspects of GLUBS, such as data collection, could be achieved through social media (Hirschberg and Manning 2015). At the community level, sound is a highly engaging medium and holds potential for successful outreach projects, such as educational presentations with area-specific themes (Scowcroft 2021) and presentations on sound produced in the local language, potentially for specific indigenous communities (Ocean Networks Canada 2021). Such presentations could be provided to local institutions, schools, or libraries to increase engagement. Coordinated media and online outreach efforts could also lead to higher user engagement and extent (Looby et al. 2023c). Collectively, these aspects highlight a need to discuss the level at which these stakeholder groups wish to be engaged and what would they want in the final product.

## Case Studies to Demonstrate Utility

To provide examples of the potential utilities of the planned GLUBS system and to assist end users in understanding the platform's process management, a series of case studies were developed that also fit within the PESTEL (Political, Economic, Social, Legal, Environmental, Technological) framework (Richardson 2006). One such objective is delineating essential fish habitat and estimating biomass of a spawning

aggregation of a commercially or recreationally important soniferous species, which is presented below. These case studies were designed to provide a broad overview of the flow process that highlights how GLUBS might link existing and new applications. For brevity, they do not detail all the methodological subtleties and additional validation data that users may collect.

For example, in the case study below, relative and absolute abundance estimation of "calling" fishes using PAM is a complex and nuanced problem that is dependent on numerous behavioral, physiological, and environmental factors that are species and site specific. As such, methodological detail for abundance estimation techniques (e.g., call counting, fish chorus sound levels, combining threshold limits with chorus sound levels), and validation studies needed to support the transfer between call detections or chorus levels to abundance estimates (e.g., source level, caller spatial distribution, call rates, signal duration, propagation losses, ratio of calling: non-calling fish based on age and gender), can be found elsewhere (e.g., Erisman and Rowell 2017; Parsons 2010; Parsons et al. 2009, 2010, 2012, 2013; Rowell et al. 2017, 2020; Sprague and Luczkovich 2011). For clarity, here, a "call" is defined as being an acoustic advertisement signal related to spawning behavior (e.g., Parsons et al. 2013), and a "chorus" follows the definition of continuous or discontinuous choruses by Cato (1978) and McCauley (2001), respectively.

## Case Study – Fisheries – Identifying Essential Fish Habitat and Estimating Biomass

The following case study assumes a researcher has collected one or more, long-term, calibrated recordings from one or more sites within an estuary that holds a spawning aggregation of a known soniferous species, with some reported acoustic characteristics. GLUBS would utilize multiple applications within the platform to provide information that assists in producing either a relative or absolute estimate of the numbers of fish(es) at the recording sites based on their calls, and this changes through the spawning season. The flow process for this procedure would be as follows (steps below are visualized in Fig. 1):

1. User inputs metadata information and selects query topic (GLUBS stores metadata).
2. GLUBS automatically links to WoRMS/OBIS to query all soniferous species expected at the location and time of the recording.
3. WoRMS/OBIS sends a list of all known and likely soniferous species reported that could be contributing to sounds in the recording to GLUBS portal.
4. GLUBS portal queries internal and external reference libraries for known acoustic characteristics of known and likely sound-producing fish species (as well as other sounds).
5. Libraries produce details on all reported sounds of known soniferous species likely present and the user selects which of these species they are interested in, i.e., species that produce sounds similar to those in the user's recordings.

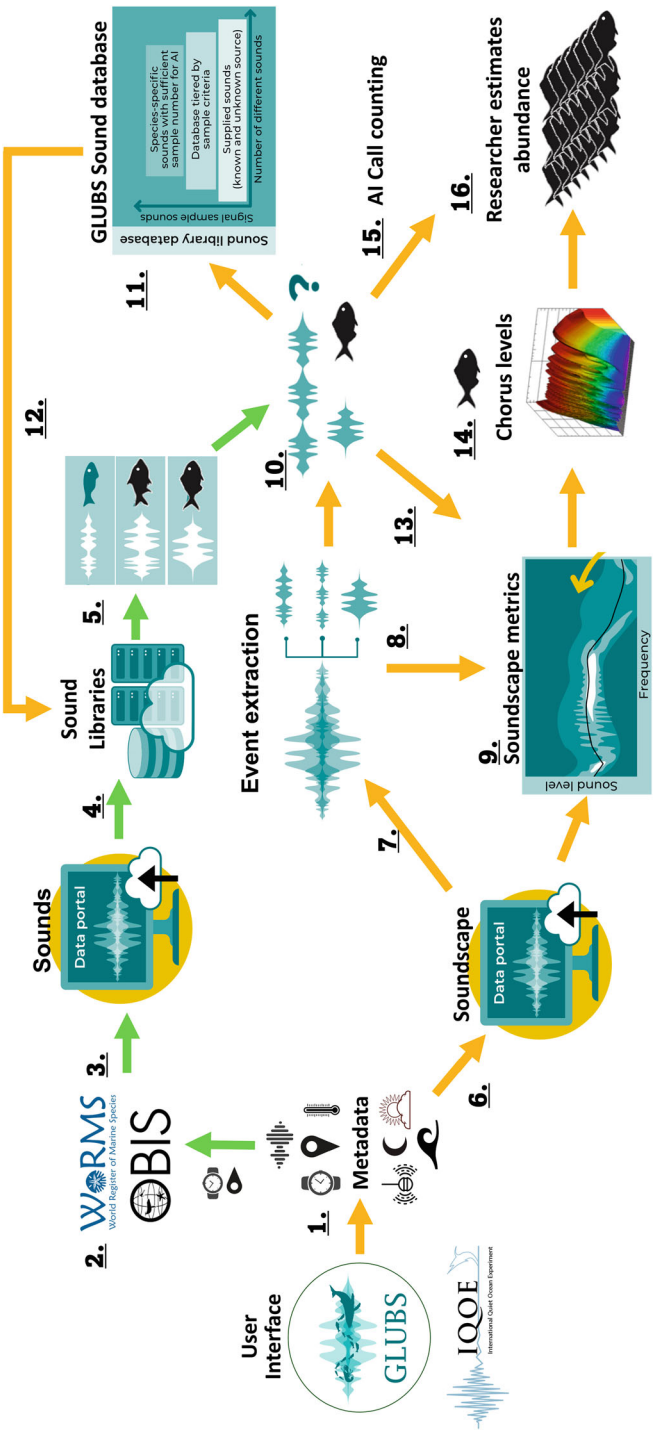



**Fig. 1** Broad GLUBS workflow for a case study (outlined in numerical steps in section 6.1) on estimating fish essential habitat and biomass. Further species-specific research may be required by the researcher between steps 14 and 16

At the same time:

6. The user is forwarded to a soundscape data portal to upload the entire dataset.
7. For all sound types detected in the recordings:
   - Event extraction algorithms identify time periods when mass phenomena (fish choruses) occur and the frequency band of the chorus that contain 90% of the energy.
   - Integrated external AI algorithms pull individual sounds (fish calls) from the dataset.
8. These times and frequency bands are inserted into a soundscape analysis tool to assist processing of the entire dataset.
9. Initial soundscape data products are exported from software, such as percentile plots of sound levels through time and sound levels within the frequency bands and times denoted by the event extraction algorithms.

Combining the reference data and the snippets extracted from the uploaded dataset:

10. Known sources of the individual sounds and choruses are identified by the user either through personal knowledge or from the information provided by the reference libraries.
11. User either (a) validates the fish in the recording or (b) deems sounds to be of unknown origin and examples of both types of calls are forwarded to the GLUBS sound database to begin/expand AI dataset.
12. Selected reference sounds for known and unknown sources are forwarded to GLUBS and external libraries.

Analysis of fish calls:

13. Source identified as the fish species queried by the user is forwarded to GLUBS software.
14. Daily chorus sound pressure levels (over the time and frequency band determined in 7 and 9) are then exported.
15. AI software runs an automated detector through the recording to count the number of fish calls and the estimated call rates (for periods when individual calls can be discriminated).
16. Daily call detection rates and chorus levels are exported for analysis outside GLUBS, and the two methods are compared to investigate the relative change of fish acoustic activity through time, noting the species- and site-specific constraints, nuances, and validation data, described above.

## Financial Support

There are three tiers of activities that require funding: (1) implementation, (2) ongoing maintenance and platform development, and (3) stakeholder engagement and user uptake. These costs (e.g., salary for dedicated and contracted staff, data storage,

and infrastructure) would initially be administered through the IQOE and, in some cases, founding partner institutes, ensuring it is not the responsibility of a sole institute. Funding for associated research and development may be administered through any partner within the GLUBS consortium to work toward a mutual goal. It would comprise three types of team members, supported through multiple institutions and funding mechanisms: (1) dedicated full-time staff to implement and maintain GLUBS (e.g., curator, systems manager/developer/web designer), (2) contracted part-time staff to implement GLUBS (e.g., web designer, maintenance manager), and (3) collaborative researchers/developers co-funded from one or more of a consortium of partner institutions to research and develop GLUBS (some of which operate associated applications).

The costs could be met by one or more of three envisaged dominant sources of funding: industry partners and regulators, philanthropic interests, and government support. In essence, it is expected that the GLUBS WG would provide mission documents and members could seek grants and funding from stakeholders in their region. Once the GLUBS system is operational, commercial avenues of funding could be accessed to cover ongoing costs. Commercial sources of funding could include advertising space within the interfaces, royalties from aspects of the platform used for commercial gain, income from commercial products developed through GLUBS, and, although the intention is for this platform to be open access, subscription fees to use one or more aspects of the platform. There is also the possibility that industry partners, who typically hold large datasets of PAM data, may be interested in funding the assessment of historic datasets via data mining for individual target species. Industry partners may also be able to engage regulators to push for funding to implement GLUBS. Industry and regulators have shared interest in better understanding factors such as impacts of noise, the distribution and migratory pathways of key species, or increasing baseline datasets on sound levels, and funding to help achieve these with GLUBS could be sought through industry programs, such as carbon offsets or Good Standing Agreements that have funded research in Australia (e.g., AIMS 2023).

## Working Subgroups

Finally, the GLUBS WG determined that several working subgroups (WSG) would be required to deliver a complete plan for such an ambitious platform. The WSGs that have been identified to date are species list and reference libraries, sound categorization, PAM applications, cyberinfrastructure design, AI needs, funding, and stakeholder engagement.

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**Conflict of Interest** The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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# Correction to: Anthropophony Effects on Acoustic Metrics in a Marine Soundscape

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The chapter was inadvertently published with incorrect name of the first author and this is now corrected as Lara C. Lopes.

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