Investigating the potential effects of marine renewable energy developments on seabirds

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A thesis presented for the degree of Doctor of Philosophy

at the University of Aberdeen

September 2015

Author's declaration

I, Helen Wade, confirm that I composed this thesis, that it has not been accepted in any previous application for a degree, that the work is my own, and that all quotations have been distinguished by quotation marks and the sources of information specifically acknowledged.

Word count of thesis: 65,907

Signed:

Date: 10.09.15

Summary

Investigating the potential effects of marine renewable energy developments on seabirds

Marine renewable energy developments (MREDs) are becoming an increasing feature of the marine environment. MREDs are just one of several human activities in the marine environment that have the potential to impact marine species. Scotland has considerable potential for generating energy from the marine environment in the form of extensive wind, wave and tidal-stream resources. Scotland also hosts numerous internationally important breeding populations of seabirds and Scottish territorial waters represent a key overwintering area for many species. Several MRED sites in Scotland are located in proximity to Special Protection Areas (SPAs) and draft marine SPAs designated specifically to safeguard important populations of seabirds. EU legislation requires that MREDs do not damage the integrity of protected seabird populations within SPAs. However, the potential effects of MREDs on seabirds are not yet fully understood. In this thesis, I aim to address gaps in knowledge regarding how MREDs may affect seabird populations. Firstly, I generate vulnerability and confidence indices to predict the effects of offshore wind, wave and tidal-stream renewable energy on Scottish seabird populations, and identify areas lacking in data. I found that effects of MREDs are species-specific and several species require additional research and monitoring to confidently predict how MREDs may affect their populations. One Scottish species identified as lacking in data is the great skua *Stercorarius skua.* Therefore, this thesis investigates the overlap of breeding great skua movements with MREDs. I found that overall overlap with MREDs was low, and that breeding birds and birds that had failed in their breeding attempt differed in their movements at sea. This indicates

that subsections of great skua populations could be differently affected by MREDs. To further explore the potential for differential effects of MREDs within seabird populations, I investigate individual variation in great skua foraging behaviour and strategies. I also identify marine environmental conditions influencing great skua foraging behaviour. I found evidence for consistency in individual great skua foraging behaviour and preferred foraging habitat, which indicates that individual behaviour, in addition to breeding status, could lead to subsections of great skua populations being differently affected by MREDs. This section of the thesis illustrates the importance of improving our understanding of both population- and individual-level responses to the marine environment in order to more fully understand seabird behaviour and accurately predict potential effects of human activities at sea. Finally, I focus on another area identified as lacking in information in my vulnerability and confidence indices; the use of high current flow environments by seabird species. I use land-based vantage-point surveys to investigate seabird interactions with fine-scale habitat in a leased tidal-stream renewable energy site in Scotland. I found low numbers of diving seabirds foraging in current speeds overlapping with optimal energy generating locations but that auk species and shags are most likely to be vulnerable to effects of tidal-stream renewable energy developments. Overall, this thesis indicates that seabird responses to MREDs are likely to be species-specific and will vary dependent on the development location and design of the energy generating technology. My findings indicate that effects of MREDs will differ dependent on individual foraging strategies, age and life stage of individuals, which implies that MREDs will differently affect subsections of seabird populations.

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Acknowledgements

First and foremost I would like to thank my supervisors, Dr Elizabeth Masden, Dr Angus Jackson and Prof Robert Furness, for all your help and support throughout my PhD. Your nuggets of wisdom, words of encouragement and consistently prompt feedback on work have always been massively appreciated. I feel privileged to have had access to the wealth of knowledge and understanding you represent, and hope that I have absorbed at least some of it!

I owe a special thanks to Soeren Hoejlund – the best technician I could have wished for. I don't know if it was always evident in the rush of fieldwork preparations but your mammoth efforts in getting the skua tracking system working and organising equipment were so appreciated. And I'm sure you'll join Liz and me in being glad to never have to lug another car battery or solar panel up the Old Man path ever again!

I owe a debt of thanks to all the lovely people who have helped with my fieldwork and to those who collected data on Hoy and Stroma - often in less than pleasant weather conditions! Firstly, a massive thank you to Eliza Leat for taking time out of your own jam-packed PhD schedule to help fit data loggers and harnesses to bonxies on both Foula and Hoy. The amazing results from the tracking work wouldn't have been possible without your expertise. Thanks to Andy Knight and Lee Shields at RSPB Orkney; to Scottish Natural Heritage for permission to conduct fieldwork on Hoy; to Chris Booth for being so forthcoming with help and information about the Stourdale great skua colony; and to all parties for being so accommodating in supporting the tracking work being undertaken there.

Others who I thank for help with fieldwork include Astrid Harendza, Matthew Easton, Neil James, Jason McIlvenny, Mona Larsen, Ted Piper, Andrew Commin, Andrew Guerin, Jo Robertson, Lauren Mackay and Rebecca Crawford. I must also thank Aileen Adam for molecular sexing of great skua feathers and William Simpson for being an excellent skipper in the troublesome waters of the Inner Sound.

I would like to thank the Marine Renewable Energy and the Environment (MaREE) project for funding this PhD research and to those tasked with patiently administering funds and paperwork, often amongst disorganised PhD researchers. Thanks in this case are due to Annette Munro, Jacquie Black and Bill Gray for making things happen.

My collaborators at BTO have provided excellent support and advice over the last 4 years. Particular thanks go to Chris Thaxter for numerous phone calls about tag settings, providing R code and for general discussions about the great skua tracking work. In addition, I would like to thank Viola Ross-Smith and Niall Burton. Also thanks to Willem Bouten and the UvA-BiTS team at the University of Amsterdam.

My thesis has been enhanced by several datasets and code provided by others. For this, massive thanks go to Matthew Easton at the Environmental Research Institute, Andronikos Kafas and Robert Watret at Marine Scotland, Kylie Scales and James Bowcott at Plymouth Marine Lab, and Sam Patrick at the University of Gloucester for giving up their own precious data and coding creations. I also thank Paul Johnson at the University of Glasgow for imparting invaluable statistics advice.

There are also so many people I owe thanks to for brief (and not so brief) discussions about data, writing, stats, and for overwhelming levels of support (and cake) when the going was tough. Special thanks go to Alex Robbins, James Waggitt (never forget the Bonus pig), Ewan Edwards, Lucy Quinn, Kylie Scales and Vicky Warwick-Evans. Also thanks to Sjúrður Hammer, Paul Jeram, Nina O'Hanlon and Adam Cross for welcoming me into their Glasgow office. I would also like to thank Beth Scott, Evelyn Philpott, Rowena Langston and Paul Thompson for giving up time very early on in my PhD to talk through ideas and provide valuable advice to get me started on my PhD journey. And of course, I mustn't forget to thank Astrid for all the boller and porridge, and for joining me in the experience of living in the most beautiful place ever!

To my friends outside the PhD bubble, who don't really know what I'm doing or why it's taking so long, thank you for being so patient and not abandoning me. I can't wait to be able to see you all properly again!

Then of course to my absolutely wonderful family. Words aren't good enough to express my appreciation for your unwavering faith in me and for never failing to support me on whatever crazy (ad)venture I take on. You are amazing.

And finally, to James. At this point I struggle to find words. I could not have done this without you. I can't believe how lucky I am to have your support. I don't know what else to say other than thank you, thank you, thank you.

Chapter 1:

General Introduction

1.1 Marine renewable energy

Marine renewable energy developments (MREDs) are becoming an increasing feature of the marine environment. The rapid development of the marine renewable energy industry is a response to the need to reduce reliance on finite fossil fuels, limit the effects of anthropogenic climate change and to increase energy security (Gill 2005; Grecian *et al.* 2010; Scottish Government 2011b). In this thesis I use MREDs to mean offshore wind farms, tidal-stream arrays and wave energy devices. Although to date developments have mostly been of offshore wind farms because the engineering, operational and financial aspects of wind turbines are well understood, developments of tidal stream arrays are expected within the next few years at a small number of suitable localised sites. Development of wave energy devices have still to solve significant engineering problems, but may eventually become widespread (Gill 2005). MREDs are just one of several human activities in the marine environment that have the potential to impact marine species. Other human activities include commercial fishing and shipping activity.

Scotland has considerable potential for generating energy from the marine environment in the form of extensive wind, wave and tidal-stream resources (ABP Marine Environmental Research 2008; Shields *et al.* 2009). As such, the Scottish Government has developed an advanced marine plan (Scottish Government 2011a) structured around achieving the target of generating 100% of electricity through renewable sources by 2020 (Scottish Government 2011b). Scotland represents a global leader in terms of plans for deployment and testing of

existing and developing technologies for commercial generation of energy from marine resources. For example, Scotland hosts the European Marine Energy Centre in Orkney (EMEC 2014), and is incentivising marine renewable energy development through initiatives such as the Saltire Prize (Scottish Government 2013c).

1.2 Marine renewable energy and seabirds

In addition to substantial marine energy resources, Scotland hosts numerous internationally important breeding populations of seabirds and Scottish territorial waters represent a key overwintering area for many species (Mitchell *et al.* 2004; Forrester & Andrews 2007). Several areas of seabed leased or proposed for MREDs in Scotland are in proximity to Special Protection Areas (SPAs) and draft marine SPAs designated specifically to safeguard important populations of seabirds (Scottish Government 2013a; JNCC 2014a; Scottish Natural Heritage, Joint Nature Conservation Committee & Marine Scotland 2014). This is particularly true in the Pentland Firth and Orkney region, where 13 of the total 48 Scottish SPAs with seabirds identified as qualifying features are located near to the greatest density of leased and proposed MRED sites in the UK (The Crown Estate 2011) (see Figure 3.1). As such, seabird populations could be affected by renewable energy developments. However, EU legislation, in the form of the EC Birds Directive (2009/147/EC), requires that developments do not damage the integrity of protected seabird populations breeding within SPAs (Scottish Natural Heritage 2014). Consequently, this legislation could pose a conflict between energy developers and conservation agencies, and presents a challenge for consenting bodies and marine spatial planners.

The effects of MREDs on seabirds are not yet fully understood. This is particularly true for wave and tidal-stream energy generation, as the technology is still under development and

deployments have not yet reached commercial scale. Owing to the relative infancy of the wave and tidal-stream renewable energy industry, there have been limited environmental monitoring programmes undertaken to assess potential effects on seabirds (Witt *et al.* 2012). However, evidence from more advanced offshore wind farm monitoring suggests the potential for both positive and negative effects on seabird populations (Inger *et al.* 2009). Owing to the rapid development of the marine renewable energy industry in Scotland it is imperative that potential risks to seabirds are estimated, species most likely to be vulnerable identified, and mitigation considered.

1.3 Potential effects of marine renewable energy developments on seabirds

1.3.1 Offshore wind devices and potential effects on seabirds

Offshore wind energy generating devices are commonly horizontal-axis, three-blade turbines mounted on a monopole or jacket structure; similar to those widely deployed at terrestrial wind farms. Offshore wind turbines are considerably larger than onshore structures with blades currently ca. 50-80m in length, mounted on a hub ca. 90-125m above sea level (Federal Maritime and Hydrographic Agency & Federal Ministry for the Environment, Nature Conservation and Nuclear Safety 2014; 4C Offshore 2015). As such, birds in flight may be at risk of colliding with moving turbine blades when flying at altitudes between ca. 20-200m above sea level. Blades rotate at maximum speeds of ca. 12rpm, with blade tips rotating at much greater speeds (ca. 300km h^{-1}) (DanTysk Offshore Wind 2015; European Wind Energy Association 2015). Offshore wind turbines are generally pile-driven into the seabed, although floating platforms tethered to the seabed by moorings are currently in development. Floating platforms will extend the possible range of OWFs from the shore by enabling devices to be deployed in deeper water than is currently possible with pile-driving methods (European Wind Energy Association 2013).

The potential effects of wind farms on seabirds could be positive and negative. Collision with turbine blades whilst in flight is perceived to be the greatest risk presented by OWFs to seabirds (Everaert & Stienen 2007; Furness, Wade & Masden 2013). Owing to the speed at which turbine blades move, particularly the tips, collision with wind turbines is likely to be fatal. As a k-selected species (high adult survival, delayed maturity and low reproductive rate), high adult mortality caused by collisions with OWFs could significantly affect population levels (Sandvik, Erikstad & Saether 2012; Satterthwaite, Kitaysky & Mangel 2012). However, quantifying the number of birds colliding with turbines is problematic as carcasses falling into the sea are likely to be carried away on currents. Seabird carcasses are also known to sink after an average of eight days, or more quickly if scavenged by other seabirds, marine mammals and fish (Wiese 2003). As such, surveying for carcasses to quantify collision rates is impractical. Alternatively, still and video cameras mounted on turbines (including infra-red) and radar have been deployed on and around wind turbines in an attempt to quantify collisions (Hill *et al.* 2014). However, cameras may have a limited view of the turbine area, and images require time-consuming post-collection review by human observers to identify collisions. In addition, radar isn't always capable of reliably resolving flight lines ending in a collision, and distinguishing between species can be problematic (Federal Maritime and Hydrographic Agency & Federal Ministry for the Environment, Nature Conservation and Nuclear Safety 2014). Risk of collision is likely to vary among species, with some species flying more often at altitudes overlapping with the area swept by OWF turbine blades (Garthe & Hüppop 2004; Furness *et al.* 2013). However, determining which species may be most at risk of collision based on flight altitudes is problematic as we have a limited understanding of the flight altitudes at which seabirds fly, and how this may be influenced by weather conditions, life stage (e.g. breeding or migrating), age and foraging behaviour. This fine-scale understanding of flight altitudes is required to accurately predict collision risks for

seabird species. There is also a lack of knowledge regarding macro-avoidance of individual turbines (close range) by seabirds in general, but more importantly, at the species-specific level. Recent projects such as the Offshore Renewables Joint Industry Programme (ORJIP; http://www.carbontrust.com/orjip) are aiming to address some of these knowledge gaps and results are intended to improve estimations of collision risk for seabirds and refine predictions regarding how seabird populations may be affected by collisions with offshore wind turbines.

Conversely, there is a greater understanding of micro-avoidance rates of seabirds in response to OWFs. Evidence indicates that some species show greater avoidance of OWF areas than others (e.g. red-throated diver) (Petersen & Fox 2007; Leopold, Dijkman & Teal 2011), whilst some species respond to avoid OWFs at greater distances than others (e.g. eider/greylag geese) (Larsen & Guillemette 2007). It has been shown that avoidance of OWFs is unlikely to greatly impact migrating seabirds, as the distance required to avoid an OWF is relatively small compared to the overall distances travelled during migrations (Masden *et al.* 2009). As such, avoidance behaviour is unlikely to increase the energetic output of a migrating bird to any detrimental level. Conversely, breeding birds could be at greater risk of incurring fitness costs caused by avoiding OWFs (Desholm & Kahlert 2005; Masden *et al.* 2010). This is because breeding birds undertake repeated foraging trips from the colony (Grant, Trinder & Harding 2014). Therefore, should an OWF be located between the foraging location of a breeding seabird and its colony, that individual may be required to repeatedly extend its foraging trips throughout the breeding season to avoid the OWF (McDonald *et al.* 2012). As breeding seabirds are known to be under energetic constraints during breeding, with some seabirds considered to be working at an energetic maximum (e.g. Hamer, Furness & Caldow 1991), even relatively small extensions to foraging trips could

cumulatively impact individuals throughout the breeding season with potentially negative consequences for reproductive output (McDonald *et al.* 2012). This could subsequently negatively affect population growth. The effects of avoiding OWFs on breeding seabirds will likely increase according to the number of OWFs avoided; with potentially cumulative fitness and reproductive costs incurred.

In addition to energetic costs incurred during avoidance of OWFs in flight, seabirds may also incur costs of avoiding OWFs in terms of habitat loss; most importantly in terms of foraging habitat loss, but also in the loss of moulting and overwintering locations. Effects of habitat loss caused by avoidance behaviour are likely to be species-specific. Direct foraging habitat loss could occur if OWFs are located in productive areas of the marine environment; for example in frontal regions (Scales *et al.* 2014b; a), which could reduce the occurrence or availability of foraging habitat for seabirds. This is of particular concern for breeding birds that have a restricted area over which to forage when constrained to return regularly to a nest, and therefore are restricted in their foraging options (McDonald *et al.* 2012). Exclusion from or destruction of foraging habitat caused by OWFs could negatively impact reproductive output if breeding birds are unable to adequately provision chicks, which has the potential to negatively impact seabird population growth (McDonald *et al.* 2012). Non-breeding birds are more flexible in their foraging behaviour and movements, and are considered less likely to be at risk from foraging habitat loss. However, should numerous OWFs be located in a particular region, the cumulative effects of habitat loss could impact even flexible foragers. Indirect effects of habitat loss to seabirds could occur if OWFs are deployed in habitat used by fish prey species (e.g. sandy habitat favoured by sandeels or spawning locations), which could reduce prey availability for seabirds.

In addition to habitat loss caused by the presence of wind turbine structures, seabirds could also be excluded from or disturbed in areas of vessel and helicopter traffic (Schwemmer *et al.* 2011; Furness *et al.* 2013). Vessel and helicopter traffic occur in and around OWFs during pre-consent surveys, construction and maintenance activities, and eventually during decommissioning of devices. Vessels and helicopters are used to transport personnel and materials to and from OWF sites. The peak of activity occurs during construction and can last several months depending on the size of the wind farm. Some species of seabird are known to be excluded from areas of high vessel traffic (e.g. red-throated diver; Schwemmer *et al.* 2011; Mendel *et al.* 2014), and if these areas of vessel activity overlap with key foraging locations could reduce the fitness of birds affected. Alternatively, some species may flush from a localised area in response to vessel and helicopter traffic. Repeated disturbance through flushing from the water in response to vessel and helicopter traffic could have fitness costs for individuals caused by increased energetic expenditure owing to repeated take-offs and flight, or in the form of a reduction in foraging efficiency and food intake caused by disturbance to foraging activity (Ronconi & Clair 2002; Bellefleur, Lee & Ronconi 2009; Schwemmer *et al.* 2011). Again, fitness costs are likely to impact breeding birds to a greater extent than non-breeding birds, owing to the high energy expenditure and limited activity budget of breeding individuals. However, repeated disturbance could have the potential to affect over-winter survival rates, particularly if birds are unable to move to other areas; for example during annual feather moult when birds are flightless.

As an alternative to habitat loss, OWFs could provide improved foraging opportunities for seabirds should they increase the biodiversity and abundance of marine species (Danish Energy Agency 2006, 2013). This could occur through several mechanisms, including turbine structures and scour protection acting as artificial reefs and fish aggregation devices (FADs)

(Wilhelmsson, Malm & Öhman 2006; Langhamer, Wilhelmsson & Engström 2009; Langhamer & Wilhelmsson 2009; Reubens *et al.* 2013a; b; Russell *et al.* 2014). In addition, OWFs could act as *de facto* marine protected areas (*df*MPAs) if fishing activity is prohibited (Inger *et al.* 2009; Grecian *et al.* 2010). These mechanisms could provide additional or improved foraging habitat for seabirds, which could have positive fitness consequences for those seabirds able to take advantage of them. As OWFs currently tend to be located relatively close to shore (compared to the distances breeding seabirds travel offshore to forage), this has the potential to positively affect breeding seabirds by providing increased foraging opportunities close to breeding colonies. This could benefit seabirds by reducing the length and duration of foraging trips, which could lead to an increase in chick provisioning rates and greater attendance at the nest; all of which are known to increase breeding productivity and chick survival (Elliott *et al.* 2009; Chivers *et al.* 2012; Kuhn *et al.* 2014). OWF structures also represent additional roost opportunities for seabirds, which may be particularly useful for species' constrained in the distances they can travel from shore (Grecian *et al.* 2010; van Deurs *et al.* 2012). For example, shags and cormorants do not have waterproof plumage and so require land or structures at sea on which to dry their wings. These roost structures can act to extend the foraging range of these species and enable access to previously inaccessible foraging opportunities (Grecian *et al.* 2010; van Deurs *et al.* 2012). However, it is worth bearing in mind that attraction of seabirds to OWFs in pursuit of foraging or scavenging opportunities, and attraction to roost structures could act to increase the risk of collision with turbines.

1.3.2 Wave devices and potential effects on seabirds

Wave energy generating devices comprise a broader range of designs (Fig. 1.1) than wind energy technology because of the relative infancy and developmental stage of the industry.

Wave energy structures include devices that float on the surface of the sea but are fixed to the seabed via cable moorings or fixed structures (e.g. Pelamis; Power Buoy: http://www.oceanpowertechnologies.com/); devices that function by allowing water to enter cavities in the structures from both above and below the surface (e.g. Limpet; Wave Dragon: http://www.wavedragon.net/); and devices that are fixed to the seabed but have moving parts that occur in part of or throughout the entire water column (e.g. Oyster: http://www.aquamarinepower.com/) (see also Grecian *et al.* 2010 and McCluskie, Langston & Wilkinson 2013 for more detail). Wave devices do not have a high profile above the sea surface, with current designs reaching heights no greater than ca. 15m above the surface, whilst other device designs are located entirely underwater (Grecian *et al.* 2010). Despite their low profile, many devices are intended to operate in large arrays that could cover relatively large areas of the sea surface and seabed. Because of the variety of wave energy device designs, it is more challenging to accurately predict the potential impacts of wave energy generation on seabirds. However, we can draw on evidence from the offshore wind industry and other marine activities to inform predictions of how seabirds might be affected. In this thesis, I mostly consider wave energy devices that are deployed offshore rather than attached directly to the coastline (e.g. Limpet); although some of the potential risks discussed will be relevant to those technologies.

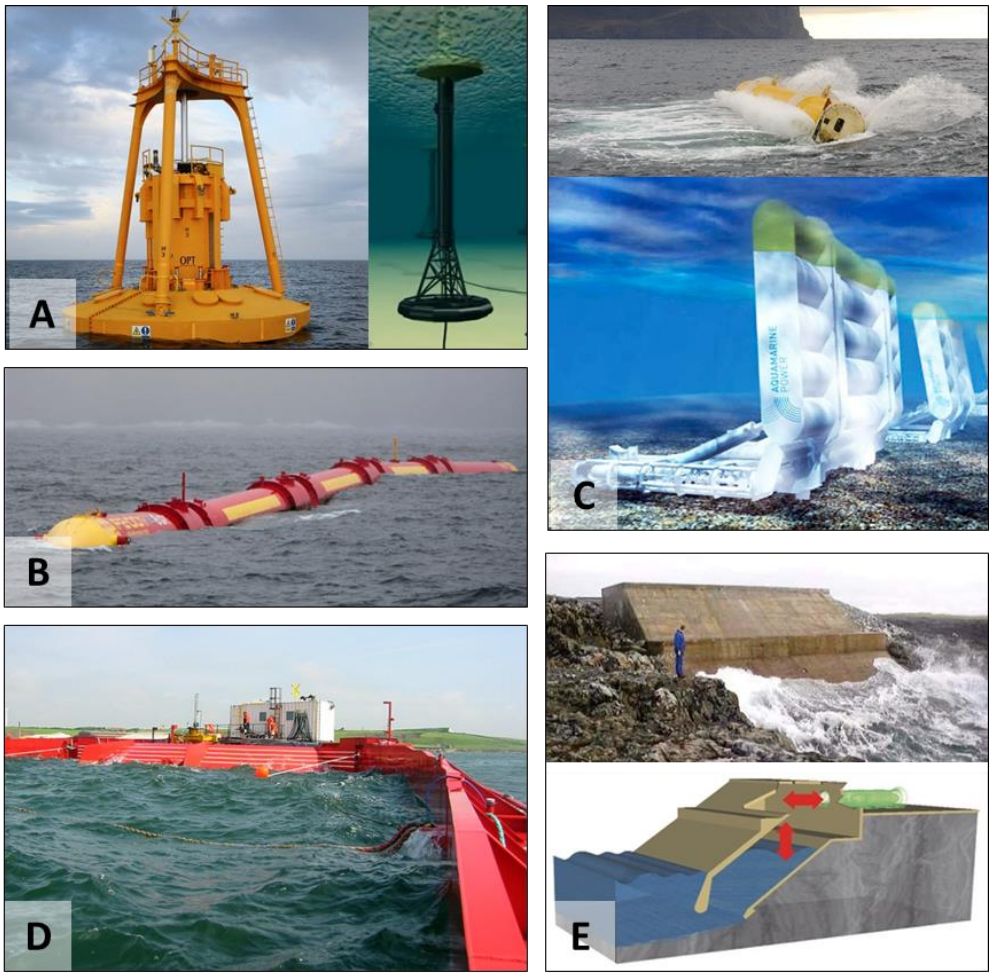

Figure 1.1: Examples of wave energy generating devices: A) Power Buoy; B) Pelamis; C) Oyster; D) Wave Dragon; E) Limpet.

As with offshore wind farms, the potential effects of wave energy developments on seabirds could be positive and negative. Because of their low profile above the water and the absence of moving structures extending above the sea surface, birds in flight are less likely to collide with wave energy devices than with wind turbines (Grecian *et al.* 2010). However, collision in flight could prove to be a risk to birds that regularly fly at low altitudes over the water, particularly during poor visibility (e.g. poor weather, at night). Manoeuvrability of birds in flight may also affect collision rates if devices are encountered unexpectedly (e.g. during poor visibility), with more manoeuvrable species potentially able to avoid close encounters with devices. Conversely, diving birds are likely to be at greatest risk of collision

with wave energy devices (Langton, Davies & Scott 2011). For example, devices designed to move unpredictably up and down on the surface of the water or in the water column in response to wave action could unexpectedly strike diving and surfacing birds; injuring or killing them. The risk of this type of collision will likely be increased if devices are deployed in dense arrays with minimal areas of clear water for birds to surface in. For devices deployed in the surf zone (e.g. Limpet and Oyster) the risk of diving birds colliding with devices is likely to be reduced, as most seabirds avoid diving and sitting in areas with breaking waves. However, some species, such as shags, cormorants and seaducks, may be at risk as they sometimes forage in the surf zone. Mooring cables and cavities are often features of wave energy device designs and as such, entrapment and entanglement are potential risks for seabirds (Grecian *et al.* 2010). For example, seabirds have been observed becoming entangled in cables associated with fishing vessels; with wings becoming wrapped around metal cables (Melvin *et al.* 2010). Although the risk of fatality as a result of entanglement with moving cables on fishing vessels is compounded by birds being dragged underwater and drowned, the risk of injury to birds as a result of encountering difficult to detect mooring cables is a risk that wave energy devices may pose to diving species. Entrapment of seabirds in fishing nets and creels (particularly naïve juvenile birds) have also been reported, which suggests that diving seabirds may be at risk of becoming trapped in cavities in wave energy devices (Galbraith, Russell & Furness 1981; Murray, Wanless & Harris 1994; Weimerskirch & Sagar 1996; Žydelis *et al.* 2009; Sonntag *et al.* 2012).

Costs of avoiding wave energy developments in flight, as a result of habitat loss owing to the presence of structures, will be broadly the same as those outlined for OWFs; with the caveat that wave energy developments will likely displace fewer seabirds owing to their less obtrusive, low profile above the sea surface. However, some species, such as red-throated

divers, will likely avoid wave energy developments as they are known to be extremely sensitive to disturbance caused by human activities and structures at sea (Roycroft, Kelly & Lewis 2004). Red-throated divers (and other diver species) may also avoid areas where they perceive a lack of open water (caused by the presence of wave energy devices on the sea surface) on which to land and take off (Furness *et al.* 2012). As such, these sensitive species could incur fitness costs if they are excluded from foraging habitat. Negative impacts of disturbance caused by vessel and helicopter traffic on seabirds at wave energy developments will also be similar to those described for OWFs (Furness *et al.* 2012; McCluskie *et al.* 2013); although helicopter traffic is less likely to be an issue at wave energy developments. Leased and proposed sites for wave energy developments are largely located in closer proximity to the shore than the majority of OWFs. As a result, different seabird species may be affected by wave energy developments than OWFs; with nearshore species most likely to be at risk. Nearshore species could be particularly affected if they have a limited foraging range (e.g. shags). As such, habitat loss or disturbance caused by wave energy developments could impact these species to a greater extent than those with the capability to forage over larger areas. Overall, it is worth bearing in mind that in comparison to OWFs, seabird avoidance of wave energy developments has not yet been quantified as devices have not been deployed in large arrays.

As with OWFs, wave energy developments could provide improved foraging opportunities for seabirds. As several wave energy devices are designed to float on the surface, devices could act as FADs, whilst bottom mounted structures may act as artificial reefs (Wilhelmsson *et al.* 2006; Langhamer *et al.* 2009; Langhamer & Wilhelmsson 2009; Furness *et al.* 2012; Reubens *et al.* 2013a; b). Development areas are also likely to act as *df*MPAs where fishing is prohibited owing to the risk of entanglement between fishing gear, wave devices and

moorings (Inger *et al.* 2009; Grecian *et al.* 2010). These mechanisms could act to increase the biodiversity and abundance of marine species located within an area of wave energy device, and consequently attract seabirds in search of improved foraging opportunities (Danish Energy Agency 2006, 2013). As with OWFs, improved foraging opportunities could benefit seabirds by reducing foraging trip duration and providing a predictable source of food. Wave energy devices extending above the sea surface could also be used as roost structures, in another similarity with OWFs (Jackson 2014). However, given the inherent movement of wave energy devices above the sea surface, these structures will not provide predictable roost sites as opportunities to rest on devices will depend on sea states being below a threshold at which the device is stable and not washed by waves (Jackson 2014). As such, wave energy developments consisting of moving structures may not extend the range of seabird species (e.g. shags and cormorants) to the same extent as OWFs. However, as with OWFs, it is worth considering that any attraction to wave energy devices could increase risk of collision, entanglement and entrapment for seabird species.

1.3.3 Tidal-stream devices and potential effects on seabirds

Owing to the relative infancy of the tidal-stream energy generating industry, there are a variety of device designs currently in development and testing (Fig. 1.2) but no commercial arrays have yet been deployed. Most of the more advanced tidal-stream device designs incorporate horizontal-axis turbines (e.g. Atlantis: http://atlantisresourcesltd.com/; Andritz Hydro Hammerfest: http://www.hammerfeststrom.com/), which are either mounted on structures fixed to the seabed or attached to floating structures. Other device designs consist of an open-centre turbine with enclosed blades (e.g. Openhydro: http://www.openhydro.com/home.html) and hydrofoils that are suspended in the water column (e.g. PulseTidal: http://pulsetidal.com/). Because horizontal-axis turbines represent

the most advanced tidal-stream technology (with plans to deploy the first commercial array of horizontal-axis turbines in Scotland in 2016; Meygen Ltd. 2014) this thesis focuses on the impacts of this type of device on seabirds. However, other device designs are borne in mind when discussing potential effects.

Horizontal-axis tidal-stream turbines are smaller than OWF turbines, with turbines planned to be deployed by Meygen reaching a maximum height of ca. 26m. Turbine blades will be mounted ca. 13-16m above the seabed, reach lengths of ca. 8-10m, and the tip of the blades will be a minimum of 8m below the sea surface at lowest astronomical tide (Meygen Ltd. 2014). Blades rotate at a speed of ca. 12m s^{-1} (Wilson *et al.* 2006). Tidal-stream energy generating devices are located in areas that experience high velocity current flows (>2 m s^{-1}; Fraenkel 2006), which are termed tidal-races. Tidal-races are often located where water is forced around headlands or into narrow channels, and are characterised on the surface by slick laminar flowing water, upwellings/boils and turbulence. Tidal-races are extremely dynamic environments and are very challenging to work in (Embling *et al.* 2012).

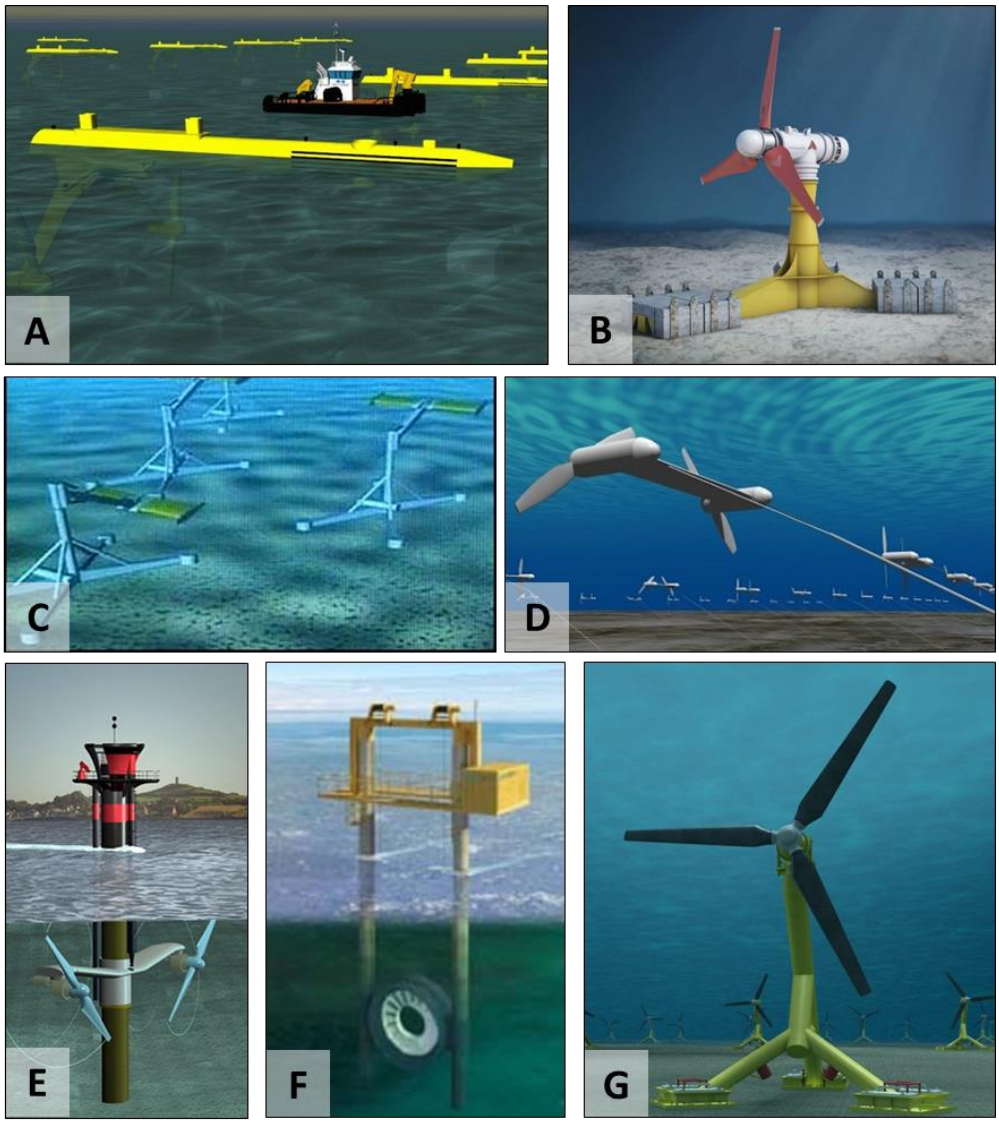

Figure 1.2: Examples of tidal-stream energy generating devices: A) Scotrenewables tidal turbine; B) Atlantis; C) and D) examples of hydrofoil designs; E) Seagen; F) Openhydro; G) Andritz Hydro Hammerfest.

Given that most tidal-stream devices are located underwater, birds in flight are not at risk of colliding with turbines. However, seabirds may be at risk of colliding with fixed structures floating on the sea surface, akin to the risk presented by floating wave energy devices (e.g. Scotrenewables tidal turbine; http://www.scotrenewables.com/). Diving seabirds are predicted to be at greatest risk from tidal turbines, as diving species could collide with turbine blades, resulting in injury or fatality (Langton *et al.* 2011; Waggitt & Scott 2014). The

risk to diving seabirds is compounded by poor visibility in tidal-races caused by turbulence and bubbles, thus making underwater structures difficult to detect. In addition, as turbines will be located in fast flowing water, this will increase the speed at which birds may encounter devices, thus reducing the time available to avoid structures. Diving birds are also potentially at risk from entrapment if devices have enclosed cavities (Galbraith *et al.* 1981; Weimerskirch & Sagar 1996; Sonntag *et al.* 2012). The risk of collision to diving birds will be influenced by both species' diving behaviour and the location of the tidal-stream device in the water column. For example, for most tidal-stream devices, benthic foraging seabirds will be at risk of collision during the ascent and descent phases of their dive, when they overlap with the location of rotating blades or moving hydrofoils (Fig. 1.3). However, once foraging on the seabed and below turbine height, benthic species are less likely to collide with devices. In the case of a device design like the Scotrenewables tidal turbine, benthic foragers are potentially at risk of colliding with or entangling in mooring cables, which may be extremely difficult to detect and avoid (Melvin *et al.* 2010). On the contrary, pelagic foragers could be at risk of collision with turbines and hydrofoils during most of their dive (Fig. 1.3). Collision risk could also be increased if individuals are focused on foraging, which could result in reduced detection rates. Given the dynamic nature of tidal-races only some seabird species may overlap with tidal-stream turbines; for example, species that forage in areas of high current flow (Fraenkel 2006; Furness *et al.* 2012). Identifying these at-risk species will be key to predicting the potential impacts of tidal-stream developments on seabird populations.

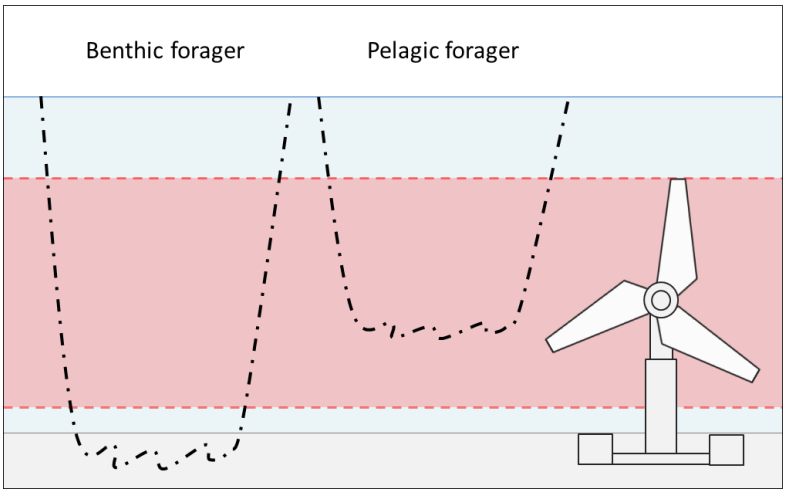

Figure 1.3: Example dive profile (black dashed lines) of benthic and pelagic foraging seabirds indicating the sections of the dive during which individuals may be at risk of collision with tidal-stream turbines. Red section indicates the collision risk zone.

Avoidance of tidal-stream developments owing to disturbance caused by structures will not be a risk associated with devices located fully underwater, although designs with floating structures could prompt avoidance behaviour and subsequent habitat loss similar to that predicted for some wave energy generating devices. As with wave energy developments, tidal-stream developments will be located relatively close to shore, and as such, ship traffic could disturb seabird species in similar ways to that predicted for wave energy developments (Furness *et al.* 2012; McCluskie *et al.* 2013).

In addition to habitat loss caused by disturbance or avoidance of structures and vessel activity, seabirds may be affected by physical changes to tidal-race habitats caused by energy extraction from the environment by tidal-stream devices (Langhamer, Haikonen & Sundberg 2010; Shields *et al.* 2011; McCluskie *et al.* 2013; Scott *et al.* 2014). Energy extraction could reduce the velocity of current flows or alter their trajectories (Shapiro 2010; Copping *et al.* 2014; Yang *et al.* 2014), which could affect the occurrence of turbulent

shearlines, upwellings/boils and eddies (Skov & Prins 2001; Garthe *et al.* 2009). This has the potential to reduce foraging opportunities for seabirds as it is thought that these visible surface cues are the manifestation of physical hydrodynamic features that act to advect and aggregate prey, or bring prey items closer to the surface within easier reach of foraging seabirds (Alldredge & Hamner 1980; Wolanski & Hamner 1988; St. John, Harrison & Parsons 1992). In addition, altering current flows by introducing devices into tidal-races could modify sediment transport processes, which could increase sediment suspended in the water column (Gill 2005; Shields *et al.* 2011). Suspended sediment could affect foraging success for seabirds in tidal-races as increased turbidity reduces visibility and could affect prey detection. In addition, the risk of collision could increase owing to a reduced ability of seabirds to detect and avoid devices. This increased turbidity may affect pelagic foragers, such as auks, to a greater extent than some benthic foragers, such as cormorant species, which are known to be able to switch from visual to tactile foraging with little reduction in prey capture success (Grémillet *et al.* 2005, 2012). Improving our understanding of the use of tidal-races by seabirds, knowledge of seabird diving behaviour and identifying the abilities of seabirds to detect and avoid obstacles underwater are key to predicting how seabirds may be affected by tidal-stream energy developments.

As with OWFs and wave energy developments, tidal-stream developments could provide improved foraging opportunities for seabirds. As with several wave energy designs, tidal-stream devices designed to float on the surface will likely act as FADs (Wilhelmsson *et al.* 2006; Langhamer & Wilhelmsson 2009), whilst bottom mounted structures may act as artificial reefs (Langhamer *et al.* 2009; Reubens *et al.* 2013a; b; Russell *et al.* 2014). Development areas are also likely to act as *df*MPAs where fishing is prohibited owing to the risk of entanglement between fishing gear and devices (Inger *et al.* 2009; Grecian *et al.*

2010). These mechanisms have the potential to increase the biodiversity and abundance of marine species located within tidal-stream energy developments, which could attract seabirds into the area in search of improved foraging opportunities (Danish Energy Agency 2006, 2013). As with OWFs and wave energy developments, improved foraging opportunities have the potential to benefit seabirds by reducing foraging trip durations and providing a predictable source of food. However, as has been discussed for OWFs and wave energy developments, attraction of seabirds into the vicinity of tidal-stream developments has the potential to increase negative associations with devices, including collisions and disturbance.

1.4 Monitoring the potential effects of marine renewable energy developments on seabirds

Monitoring the effects of MREDs on seabirds includes surveying seabird distribution and behaviour. Before-after-control-impact studies (BACI) are often used in environmental monitoring and involve surveying seabird use of leased MRED sites and control areas to establish a baseline prior to construction and operation of energy generating devices. Subsequent monitoring during construction and operation is used in combination with baseline data to detect changes in seabird behaviour and distribution caused by MREDs. A range of methods may be employed to monitor seabirds in MRED sites. Surveys are most often conducted from vessels using standardised European Seabirds At Sea (ESAS) sampling methods (Tasker *et al.* 1984; Leopold *et al.* 2011). Monitoring may also be undertaken from aircraft (Petersen & Fox 2007), vantage points on land (Inger, Harrison & Bearhop 2010) or fixed platforms at sea (Krijgsveld *et al.* 2011). These methods usually employ distance sampling techniques to estimate seabird abundance and distribution. Relatively novel techniques using laser range finders (Skov *et al.* 2012), radar (Christensen *et al.* 2004), and

thermal imaging cameras (Petersen *et al.* 2006) are increasingly being used to detect and quantify seabird encounters with wind farms, whilst still image cameras (Jackson 2014) and seabed sonar platforms (Williamson *et al.* 2014) have been employed to assess seabird interactions with wave and tidal-stream energy devices. These methods incorporate advantages and disadvantages in their application; including the ability to sample large areas and potential difficulties associated with applying detection functions to reliably estimate seabird abundance. The most appropriate methods are site-specific and depend on the information being sought. The ability to detect an effect of an MRED site on seabird populations can be an obstacle to the success of survey efforts; as low numbers of observations of species may prevent analyses to detect effects from being undertaken. Some of the issues and limitations inherent in survey and monitoring designs will be demonstrated and discussed in Chapter 5 and the General Discussion.

Establishing overlap of seabird movements with MREDs can also be addressed by tracking individual birds using GPS data loggers (Thaxter *et al.* 2012b; Soanes *et al.* 2013b) or recording flight paths of individuals from boats by following birds leaving breeding colonies (Perrow, Skeate & Gilroy 2011). An advantage of using data loggers is the ability to track individuals from known colonies, which enables an estimation of the overlap of specific seabird populations with MREDs. This is particularly useful when assessing effects of MREDs on specific SPA populations, as required by EU legislation.

A key aspect of seabird monitoring is linking species' distributions and behaviour to habitat, so that habitat relationships can be used to predict the distribution of seabird species under different scenarios and in different locations. Both tracking and survey data can be combined with remotely sensed marine environmental data (such as sea surface

temperature and chlorophyll *a* abundance) or hydrodynamic model data to identify key foraging locations and establish habitat preferences (e.g. Patrick *et al.* 2014). This information can be incorporated into species distribution models (SDMs), which can be used in marine spatial planning to identify areas appropriate for conservation (Arcos *et al.* 2012) . Population-level SDMs can also be used to assess potential effects of MREDs on seabirds, although it is worth bearing in mind that seabird presence in a MRED site does not necessarily equate to risk. For example, based on GPS data and habitat modelling, common guillemots may be deemed to be present in a particular habitat overlapping with a tidal-stream development site. However, some or all of those birds may not be foraging (for example, owing to individual variation in foraging behaviour or life stage of the individual), which would put them at low risk of collision with tidal-stream turbines. As such, SDMs can be a useful tool to identify habitat and areas at sea that could be important to a particular species; although without more detailed information on the age-structure of populations and the potential for individual variation in foraging behaviour, estimates of vulnerability to MREDs based on SDMs should be treated with some caution.

1.5 Outline and aims of this thesis

This thesis aims to address gaps in knowledge regarding how MREDs may affect seabird populations. The thesis considers potential effects at a range of spatial scales, in different habitats and across life stages to improve predictions of which seabird species may be most vulnerable to MREDs in a Scottish context. The thesis compiles existing information on seabirds to answer specific questions related to the potential effects of MREDs but also contains new data specifically collected to address knowledge gaps with respect to how seabird species may be affected by the increasing development of the marine renewable energy industry.

The thesis contains four data chapters. As these chapters have been written to stand alone, some information may be duplicated across chapters. Sections of Chapter 2 have been published as two peer-reviewed papers, on which I am a co-author (Furness *et al.* 2012, 2013), and Chapter 3 has been published as a peer-reviewed paper, on which I am lead author (Wade *et al.* 2014) (see Appendix 3 for copies of published papers).

I have also contributed to several reports during my PhD (Furness & Wade 2012a; b; Thaxter *et al.* 2012b) and am a named co-author on a paper based on fieldwork undertaken for Chapters 3 and 4 (Thaxter *et al.* In review). In addition, I am also a co-author on a manuscript in preparation that incorporates data collected during fieldwork for Chapter 5 (Waggitt *et al.* In prep.).

The thesis begins in **Chapter 2** with an estimation of the effects of the three principal MRED technologies to be employed in Scotland; wind, wave and tidal-stream. This first chapter develops vulnerability and confidence indices to consider the broad-scale effects of MREDs on all Scottish marine bird species but with no distinction according to life stage or age. The indices in this chapter are intended to assist marine renewable energy developers and regulators by identifying species most likely to be at risk from MREDs. This information can be considered in environmental impact assessments and when designing seabird monitoring programmes. It also aims to identify areas where more research and monitoring could be particularly beneficial.

In **Chapter 3**, the thesis moves on to focus on an internationally important seabird species breeding in Scotland; the great skua *Stercorarius skua*. In Chapter 2 the great skua is identified as one of the top ten most vulnerable species to collision with offshore wind

developments and could be affected through disturbance during the deployment of wave and tidal-stream devices. Given that we know very little about great skua movements at sea it is difficult to make accurate predictions regarding the potential effects of MREDs on this species. As over 60% of the global great skua population breeds in Scotland (Mitchell *et al.* 2004), this chapter aims to improve our understanding of how great skuas may be affected by MREDs by investigating the overlap of great skua movements during the breeding season with leased and proposed MREDs in Scottish waters. The chapter also investigates the potential for differing effects of MREDs based on the breeding status of individual great skuas.

Chapter 4 continues to focus on great skua overlap with MREDs by investigating differences in behaviour and foraging strategies of birds throughout the breeding season. This chapter is developed based on evidence found in Chapter 3 suggesting that the impacts of MREDs could differentially affect great skuas depending on their breeding status. The chapter goes on to consider the consequences of individual variation in behaviour and dietary strategies for population-level effects of MREDs and another human activity, commercial fishing. Chapter 4 identifies marine habitat targeted by great skuas during foraging by investigating the relationship between environmental variables and foraging behaviour. A more detailed understanding of great skua movements at sea and establishing links between foraging and habitat can be used to more accurately predict the effects of human activities on great skuas and can be used to inform marine spatial planning decisions.

Chapter 5 focuses specifically on the use of a high current flow environment by seabird species, which is an area identified as lacking in information in Chapter 2. Chapter 5 uses land-based vantage-point surveys to investigate seabird interactions with fine-scale habitat

in a leased tidal-stream renewable energy site in Scotland. The chapter aims to more accurately predict which species may be most vulnerable to tidal-stream energy developments.

Finally, in **Chapter 6,** I draw the key findings from each of the four preceding data chapters together and set my overall findings in a wider context. I also consider some of the challenges encountered during my PhD research and suggest directions for future research.

Chapter 2:

Confidence in assessing the vulnerability of marine bird populations to offshore wind, tidal-stream and wave energy developments

Parts of this chapter have been published as:

Furness, R. W., H. M. Wade, A. M. C. Robbins, and E. A. Masden. 2012. Assessing the sensitivity of seabird populations to adverse effects from tidal stream turbines and wave energy devices. ICES Journal of Marine Science 69: 1466–1479;

and:

Furness, R. W., H. M. Wade, and E. A. Masden. 2013. Assessing vulnerability of marine bird populations to offshore wind farms. Journal of Environmental Management 119: 56–66.

(See Appendix 3 for copies of the published papers)

My contribution to the published papers included conducting literature searches to generate the vulnerability factors and assisting with the writing and editing of the manuscripts. In this chapter I have combined and revised the two published papers. For example, in light of numerous studies being made available since publication, I have greater than doubled the number of studies included in generating the vulnerability indices from 45 to 93. Based on recent information regarding how seabirds react to marine renewables and requests from regulators I have adapted the vulnerability index calculations to more accurately reflect how species may be affected by MREDs and have recalculated all vulnerability indices. In addition to what is covered in the Furness *et al.* (2012, 2013) papers I added an additional aspect that assessed the level of confidence we can place in the data incorporated in four

updated vulnerability factors that are used to generate the vulnerability indices. This additional work will enable the identification of species lacking in data and highlight where more research may be required. For each species, this required assessing all 93 sources of information to establish whether the data collected was for target or related species, how many studies were undertaken, how many sites data was collected at, over how many years was data collected for, and to establish what type of studies were undertaken. This represents a significant extension and addition to the published Furness *et al.* (2012, 2013) papers and reflects requirements of MRED regulatory bodies that were identified by H.Wade.

Abstract

The Scottish Government is aiming to generate 100% of Scotland's electricity through renewable sources by 2020 and has leased or proposed areas of the seabed for marine renewable energy developments (MREDs). However, increases in human activity and development in the marine environment has the potential to affect marine bird populations. Currently, the effects of wave and tidal-stream renewable energy developments are not fully known, although evidence from the offshore wind industry suggests that marine bird populations could be affected by: collisions with devices; disturbance and displacement; altered foraging opportunities; and increased provision of roost sites. In this chapter I use a well-established approach to identify species at-risk by updating indices of sensitivity or population vulnerability to particular hazards published by Furness *et al.* (2012, 2013). An issue with the use of vulnerability indices is that the data included for each species tend not to be equal in quality and/or quantity. To address this, I have also assessed the confidence level associated with the data for each species in four vulnerability factors used to calculate overall vulnerability indices. I found that newly available literature altered the rankings of only a few species and that overall species rankings remained broadly the same. A greater overall confidence in data on how marine birds may react to MRED structures and for what percentage of flight overlaps with turbine blade altitudes was found; although there were a range of confidence levels among species, with some species lacking in data. A lower overall confidence in data regarding how marine birds may react to vessel and/or helicopter traffic associated with MREDs, and how marine bird species use tidal races was found. Allocating confidence levels to data enables knowledge gaps to be established; determine where more research and monitoring is required; and identifies areas where caution in interpreting results is recommended.

2.1 Introduction

The Scottish Government is aiming to generate 100% of Scotland's electricity through renewable sources by 2020. As a consequence, Scotland has developed an advanced marine plan with many areas of the seabed leased or proposed for marine renewable energy developments (MREDs) (Scottish Government 2013b). These include areas for offshore wind farms and for wave and tidal-stream energy generation in coastal waters. Scotland has some of the best wave and tidal-stream resources in the world (Shields *et al.* 2009) and there are currently plans to deploy large arrays of wave and tidal-stream energy generating devices. Wave and tidal-stream energy installations will occupy much smaller areas at sea than offshore wind farms (ICES 2010).

An increase in human activity and development in the marine environment has the potential to affect marine bird populations; particularly given the numerous locations of internationally important populations of breeding marine birds designated as Special Protection Areas (SPAs) in Scotland. SPAs provide legal protection for breeding marine birds (Birds Directive 2009/14/EC); with developments not permitted to negatively impact designated SPA populations. However, several MRED sites are located close to SPA and draft marine SPA sites (JNCC 2014a; Scottish Natural Heritage *et al.* 2014) (Fig. 2.1). Therefore the need to identify and assess potential effects of these developments on marine bird species is imperative. This will ensure that species identified as potentially vulnerable to MREDs are the focus for research and monitoring where populations overlap with MRED sites (ICES 2010).

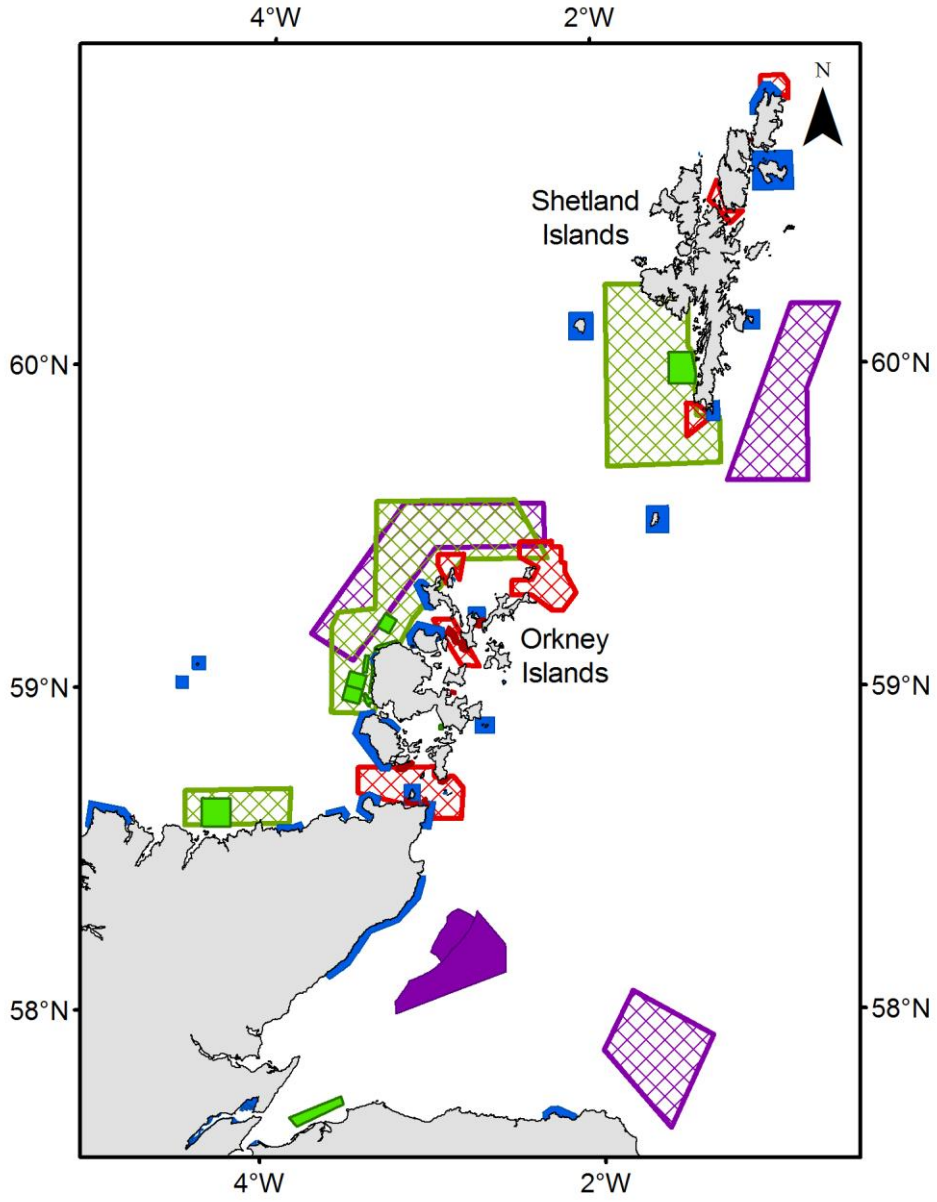

Figure 2.1: Locations of special protection areas (blue polygons) designated for breeding seabird populations in north-east Scotland, and the proximity of leased and proposed marine renewable energy development sites (solid polygons = leased sites; cross-hatched = proposed sites; purple = offshore wind sites; green = wave sites; red = tidal-stream sites).

The effects of wave and tidal-stream renewable energy developments are largely unknown at present as very few deployments and associated monitoring projects have been undertaken (Witt *et al.* 2012). However, evidence from the offshore wind industry suggests that marine bird populations could be negatively affected by collisions with devices (Everaert

& Stienen 2007; Langton *et al.* 2011), disturbance and displacement from foraging and loafing sites as a result of structures and vessel and/or helicopter traffic (Ronconi & Clair 2002; Larsen & Guillemette 2007; Schwemmer *et al.* 2011; McDonald *et al.* 2012), and experience altered foraging opportunities if devices modify flow regimes and sediment transport processes (Gill 2005; Langhamer *et al.* 2010; McCluskie *et al.* 2013). Alternatively, marine birds could be positively affected through improved foraging opportunities if MREDs increase local prey abundances by acting as fish aggregation devices (FAD), through artificial reef effects (Langhamer & Wilhelmsson 2009; Reubens *et al.* 2013a), and if devices increase the provision of roosting sites (Grecian *et al.* 2010; van Deurs *et al.* 2012). Even in the absence of a comprehensive knowledge of potential effects, an assessment of vulnerability to MREDs can be made on the basis of existing information about seabird ecology, and through inference of the effects of devices on the physical environment and lower trophic level organisms (e.g. seabird prey) (WWT Consulting 2010; ABP Marine Environmental Research 2011).

A well-established approach to identify species at-risk is to use indices of sensitivity or population vulnerability to particular hazards (Furness & Tasker 2000; Garthe & Hüppop 2004; Sonntag *et al.* 2012). For example, Garthe & Hüppop (2004) developed a single index of marine bird population vulnerability to offshore wind farms, based on scores of the conservation importance of different species' populations and perceived behaviour-related risk of collision and displacement. They applied this index to marine birds in the southern North Sea. Furness *et al.* (2012, 2013) developed and refined the approach advocated by Garthe & Hüppop (2004) by extending the species list and incorporating new data from recent research on the behaviour of marine birds in response to MREDs. The authors generated four separate indices assessing vulnerability of Scottish marine bird populations

to: 1) collision with offshore wind farms, 2) disturbance and displacement as a result of offshore wind farms, 3) tidal-stream energy developments, and 4) wave energy developments. An important innovation in their approach was the generation of two indices for offshore wind; one estimating collision risk and the other estimating risk of avoidance or displacement. This separation differed to the single index presented by Garthe & Hüppop (2004) and is particularly useful where species most at risk of collision differ from those most at risk of displacement. These indices were developed to assist in the legal process of environmental impact assessment (EIA) required for MREDs; particularly where developments may be at risk of affecting marine bird populations in SPAs.

An issue with the use of vulnerability indices is that the data included for each species tend not to be equal in quality and/or quantity. This can be caused by a variety of reasons; for example, some species may be more difficult to detect, or be lower in abundance than other species. Consequently, there may be fewer observations for some species during surveys and subsequently less robust conclusions drawn regarding those species' behaviour in response to MREDs. Additionally, data may not be available for a particular species and so information may be inferred from closely related species. These issues have implications for data quality and reliability that are not apparent when considering vulnerability indices alone. In order to address the issue of data quality, and to highlight where caution in interpreting results of vulnerability indices is required, I follow the approach taken by Thaxter *et al.* (2012a) and Robbins *et al.* (2014b). These studies incorporate measures of data quality, the accuracy and precision of data collection, consider the number of studies undertaken, and establish how many sites were sampled in order to estimate confidence levels associated with each species' data. Species confidence levels can then be used to

assist in designing EIA monitoring programmes; for example, direct monitoring to focus on species with low data confidence or in areas where information is lacking.

In this chapter I aim to update the four vulnerability indices generated by Furness *et al.* (2012, 2013) by adding newly available information on species' behaviour and responses to MREDs. I also aim to adjust two of the vulnerability index calculations to take into consideration that marine bird avoidance of wind farms is likely to result in a reduced risk of collision, and to recognize that species may differ in their responses to structures and to vessel/helicopter traffic. The purpose in revising and updating these indices is to provide relevant, up-to-date information on marine bird vulnerability to MREDs that can be used by developers and regulators in designing monitoring programmes to fulfil EIA requirements. In addition, I aim to calculate the confidence levels of data for each species included in the vulnerability index calculations. The purpose of calculating confidence levels associated with species' data is to identify marine bird species for which data are lacking and where increased research or monitoring is required to ensure confidence in estimations of responses to MREDs. Results of this study are intended to be useful in directing research and monitoring towards species lacking in data, and to highlight where caution in interpreting vulnerability index results is required.

2.2 Methods

2.2.1 Vulnerability indices

In this chapter, scores from the vulnerability indices presented in Furness *et al.* (2012, 2013) have been reviewed and updated to rank the vulnerability of 38 species to: 1) collision with offshore wind farms, 2) disturbance and displacement as a result of offshore wind farms, 3) tidal-stream energy developments, and 4) wave energy developments. The methods used

broadly follow the approach developed in Furness & Tasker (2000), and implemented by Garthe & Hüppop (2004) in their assessment of the sensitivity of southern North Sea marine bird populations to offshore wind farms. Calculating the indices involved scoring a number of factors considered likely to affect the vulnerability of Scottish marine bird populations to these developments. Wherever possible, scores allocated to each species in each factor were evidence-based, with data extracted from peer-reviewed literature. In cases where the evidence-base was limited, scores were allocated based on the information available. All scores in Furness *et al.* (2012, 2013) were circulated to a group of appropriate experts for review. This review process ensured a consensus of support for the final criteria and scorings from a range of stakeholders, including seabird ecologists and conservationists (see acknowledgements in Furness *et al.* 2012, 2013: Appendix 3). In order to retain expert opinion, scores in this chapter have only been adjusted if strong evidence has become available since the publication of the Furness *et al.* (2012, 2013) peer-reviewed papers.

Breeding and non-breeding (wintering or passage) true seabirds, wintering sea ducks and grebes, and white-tailed eagle have been included in this review (see Appendix 1, Table 1 for scientific names of species included). White-tailed eagle has been included as it feeds extensively over the sea and there is evidence for collision at coastal wind farms (Dahl *et al.* 2012, 2013). The few marine bird species of conservation importance excluded were considered to be unlikely to interact with MREDs in Scottish waters; for example, because their distributions are entirely coastal (e.g. red-breasted merganser *Mergus serrator*), because very low numbers occur in Scottish waters, or because they occur in Scottish waters for very short time periods.

2.2.1.1 Conservation status factors

Four factors were scored by Furness *et al.* (2012, 2013) (and retained in this chapter) to represent the conservation importance of each species in a Scottish context: 1) status in relation to the Birds Directive; 2) UK threat status; 3) percentage of the biogeographic population that occurs in Scotland; and 4) adult survival rate. Despite some correlation between the four factors, each factor represents the conservation importance of seabird species' in a slightly different way. As such, it was deemed that combining the multiple rankings from each of the four conservation status factors would provide a more reliable average score of conservation importance than achieved by relying on a single factor alone. Conservation status factor scores were allocated from 5 (high conservation importance) to 1 (low conservation importance).

Birds Directive Status

The EU Birds Directive acts to protect European wild birds and their habitats; in particular through the designation of SPAs. The scores used in this factor reflect aspects of the conservation status of birds. Species listed in Annex 1 of the Birds Directive are prioritised for conservation protection through European legislation and so were allocated a score of 5. Species qualifying as 'migratory species' but not listed in Annex 1 were allocated a score of 3. Remaining species were allocated a score of 1.

UK threat status

This factor reflects both threat and conservation status of species' in the UK, as given in 'Birds of Conservation Concern 3' (BOCC3) (Eaton *et al.* 2009). The classification of species in BOCC3 sometimes differs from the classification in the previous assessment, BOCC2. These changes were taken into account owing to the implications of changes in status. Scores

were allocated as follows: 1 (green in BOCC2 and BOCC3), 2 (amber in BOCC2 and green in BOCC3), 3 (green in BOCC2 and amber in BOCC3), 4 (amber in BOCC3 and BOCC2), and 5 (red in BOCC3) (see supplementary material; Furness *et al.* 2012).

Percentage of the biogeographic population in Scotland

The percentage of the biogeographic population of each species found in Scotland was assessed using Forrester & Andrews (2007), or by comparing the population estimate in Forrester & Andrews (2007) with the biogeographic population estimates given in del Hoyo, Elliott & Sargatal (1992, 1996). As this metric may vary seasonally we applied the precautionary principle and used the highest seasonal score for each species. Scores were allocated as: 1 (<1%), 2 (1-4%), 3 (5-9%), 4 (10-19%), and 5 (≥20%).

Adult survival rate

Published data on annual adult survival rates (which tend to be robust data for most marine bird species) were included to reflect the vulnerability of species to increases in mortality caused by MREDs above natural mortality levels. Species with low rates of annual adult survival tend to be less vulnerable to additional mortality than species with high annual survival. Data were extracted from scientific literature located in individual species studies or estimated from data for closely related species (see Furness *et al.* 2012). Preference was given to more recent studies and/or studies in the UK if several estimates were available, as survival rates of the same species may differ between populations in different geographical regions. Adult survival was classified according to Garthe & Hüppop (2004) with scores allocated as: 1 (adult survival <0.749), 2 (adult survival 0.75-0.799), 3 (adult survival 0.80-0.849), 4 (adult survival 0.85-0.899), and 5 (adult survival >0.90).

2.2.1.2 Vulnerability factors

Four vulnerability indices were calculated by Furness *et al.* (2012, 2013) and are updated in this chapter: 1) collision risk with offshore wind farms; 2) displacement and disturbance caused by offshore wind farms; 3) vulnerability to tidal-stream energy developments; and 4) vulnerability to wave energy developments. Each index was scored on a different combination of factors used to assess species' vulnerability to MREDs. These are described below.

1) <u>Collision risk with offshore wind farms</u>

Percentage of flight at altitudes overlapping with turbine blades

This factor is widely considered to be of overwhelming importance in determining the risk of collision of marine birds with offshore wind turbines (Band 2012; Cook *et al.* 2012b). Birds that only fly very low over the water will be below the area swept by turbine blades, whereas marine birds that habitually fly at greater heights may experience a greater risk of collision with blades if flight heights coincide with rotor swept areas of a wind farm. Flight altitude scores in Furness *et al.* (2013) were taken from Garthe & Hüppop (2004) and Cook *et al.* (2012b). As in Furness *et al.* (2013), flight altitudes in this chapter are presented as the percentage of flight at altitudes overlapping with turbine blades, which is considered to occur between ca. 20-150 m above sea level (Table 2.2), as opposed to collapsing the data into a 5 point scale (Garthe & Hüppop 2004). Flight altitude percentages represent the mean flight altitude derived from all studies for a given species. Measurements include birds engaged in all activities (for example, foraging, commuting and migrating) and may vary seasonally; although there are too few data available at present to test this possibility. Original values from Furness *et al.* (2013) were retained where appropriate but adjusted where more recent data have become available, for example in peer-reviewed literature or

MRED reports (Appendix 1, Table 2).

Displacement and disturbance as a result of structures

Seabird species differ in their reaction to structures. This behaviour relates in part to the general responsiveness of species to disturbance and in part to their perception of the hazard that structures represent. Disturbance as a result of structures is considered to be equal for offshore wind turbines and wave energy devices. This is likely to prove inaccurate because wave energy devices have a smaller profile than offshore wind turbines, but insufficient data exist regarding species' reactions to wave energy devices. Scores in Furness *et al.* (2013) were taken from Garthe & Hüppop (2004). In Furness *et al.* (2013) disturbance and displacement as a result of structures and vessel and/or helicopter traffic were considered as a combined factor. In this chapter, the factor has been split into two; one considering disturbance and displacement as a result of structures and the other considering disturbance and displacement as a result of vessel and/or helicopter traffic. The split is appropriate given evidence that some species (e.g. gannets *Morus bassanus*) are not consistent in their response to structures and vessels. Original values from Furness *et al.* (2013) were retained where appropriate but adjusted where more recent data were available, for example in peer-reviewed literature or MRED reports (Tables 2.3, 2.4 and 2.5, and Appendix 1, Table 3). Scores were allocated from 1 (limited escape response and minimal risk of disturbance) to 5 (strong escape behaviour and moderate to high risk of disturbance).

Flight agility

This factor takes into account the aerial agility of species and hence their potential to avoid collision with wind turbines at sea. Following Garthe & Hüppop (2004), it is assumed that all

other factors being equal, birds with low flight manoeuvrability are more likely to collide with wind turbines at offshore wind farms than birds with high flight manoeuvrability. Flight manoeuvrability is likely to play a greater role for birds engaged in micro- rather than macro-avoidance of wind turbines; particularly during poor weather conditions and reduced visibility. However, the benefits of greater manoeuvrability will likely be nullified once a bird moves to within a certain distance of a wind turbine as the speed of the blades (particularly tips) will outpace the ability of a bird to avoid collision. A lack of data investigating micro-avoidance of wind turbines prevents a more detailed assessment of how flight agility may influence collision with wind turbines being presented here. Scores follow Furness *et al.* (2013) and were originally taken from Garthe & Hüppop (2004). Species were scored from 1 (very high flight manoeuvrability) to 5 (very low manoeuvrability) (Table 2.2 and see supplementary material; Furness *et al.* 2013). The values presented are considered to be a consequence of morphology rather than behaviour and may vary seasonally (for example, in relation to moult), although such detail is beyond the scope of this assessment.

Percentage of time flying

This factor is assumed to indicate risk of collision because marine birds that spend more time flying while at sea are more likely to be at risk of collision; whether while breeding, migrating, wintering, or as pre-breeders. Scores follow Furness *et al.* (2013) and were originally taken from Garthe & Hüppop (2004). For species not covered by Garthe & Hüppop (2004), Furness *et al.* (2013) calculated scores from data on activity budgets following the procedure outlined in Garthe & Hüppop (2004). Scores were allocated as: 1 (0-20% of time at sea spent in flight), 2 (21-40% spent in flight), 3 (41-60% spent in flight), 4 (61-80% spent in flight), and 5 (81-100% spent in flight) (Table 2.2 and see supplementary material; Furness *et al.* 2013). This factor is likely to vary seasonally, with the literature indicating more flight

activity while rearing chicks than during the incubation period, and more flight while breeding than during winter. Peaks of flight activity occur in migrant species during the migration, while flight activity may be reduced during post-breeding moult. However, these refinements are not yet well enough documented to assess scores separately for different seasons, although that could be a useful development of the method.

Nocturnal flight activity

Although observations exist, detailed data on nocturnal flight activity are not available for many species. Geolocation and GPS logger data are beginning to address this situation, predominantly for large Southern Ocean seabirds (Phalan *et al.* 2007; Mackley *et al.* 2010, 2011), although data are now available for northern gannet (Garthe *et al.* 2012) and black-legged kittiwake *Rissa tridactyla* (Kotzerka, Garthe & Hatch 2010). Similar data are likely to become available for other North Atlantic marine birds, including great skuas *Stercorarius skua*, for which logger deployments have already provided information on migrations and wintering areas (Magnusdottir *et al.* 2012, 2014). Scores follow Furness *et al.* (2013) and were originally published in Garthe & Hüppop (2004). For species not covered by Garthe & Hüppop (2004), Furness *et al.* (2013) used published data where possible and information (often qualitative rather than quantitative) from individual species studies or from handbooks (Cramp & Simmons 1977, 1980; del Hoyo *et al.* 1992, 1996) to calculate scores. Classifications were also moderated by experts. Scores were allocated from 1 (limited flight activity at night) to 5 (much flight activity at night) (Table 2.2 and see supplementary material; Furness *et al.* 2013).

2) Displacement and disturbance caused by offshore wind farms

Displacement and disturbance as a result of structures

See in: 1) Collision risk with offshore wind farms.

Displacement and disturbance as a result of vessel and/or helicopter traffic

Vessel and helicopter traffic are a feature of construction and maintenance activities at offshore wind farms; transporting both materials and personnel to sites. Depending on the size of the offshore wind farm, vessel and helicopter activity may persist for relatively long periods of time during the construction phase of the development, but is likely to be more sporadic during the operational phase of the wind farm. Marine bird species vary in their response to vessel and/or helicopter traffic and this behaviour relates in part to the general responsiveness of the species to disturbance, and in part to their specific response to vessels and helicopters. Species that react strongly to vessel and helicopter activity, either by being repeatedly flushed from the water, or avoiding areas where activity is occurring, may suffer from increased energy expenditure and/or reduced foraging opportunities (Ronconi & Clair 2002; Bellefleur *et al.* 2009; Schwemmer *et al.* 2011). Scores in Furness *et al.* (2013) were originally taken from Garthe & Hüppop (2004). In Furness *et al.* (2013) disturbance and displacement as a result of offshore wind structures and vessel and/or helicopter traffic were considered as a combined factor. In this chapter, the factor has been split into two; one considering disturbance and displacement as a result of structures, and the other considering disturbance and displacement as a result of vessel and/or vessel traffic. This split is appropriate given evidence that some species (e.g. gannets) are not consistent in their responses to structures and vessels. Original values from Furness *et al.* (2013) were retained where appropriate but adjusted where more recent data were available; for example from peer-reviewed literature or MRED reports (Tables 2.3, 2.4 and 2.5, and

Appendix 1, Table 4). Scores were allocated from 1 (limited escape response and flushed only when approached at very short distances) to 5 (strong escape behaviour, regularly flushed when approached at long distances – e.g. ≥1000m, and moderate to high short-term loss of foraging opportunities owing to disturbance).

Habitat specialisation

Marine birds vary in the range of habitats they use, for example associating with particular water masses or frontal systems, and whether they use these as specialists or generalists. Scores follow Furness *et al.* (2013) and were originally presented in Garthe & Hüppop (2004). For species not covered by Garthe & Hüppop (2004) scores in Furness *et al.* (2013) were based on foraging ecology described in single species studies in the literature, or from standard handbook descriptions. Scores were allocated from 1 (tend to forage over large marine areas with little known association with particular marine features) to 5 (tend to feed on very specific habitat features, such as shallow banks with bivalve communities, or kelp beds) (Tables 2.3, 2.4 and 2.5, and see supplementary material; Furness *et al.* 2012, 2013). Scores reflect the species' potential vulnerability to MREDs should a development occur within their foraging range or habitat, with scores allocated as low vulnerability (1) to high vulnerability (5).

3) <u>Vulnerability to tidal-stream energy developments</u>

Displacement and disturbance as a result of vessel and/or helicopter traffic

See in: 2) Displacement and disturbance caused by offshore wind farms.

Drowning risk

Marine birds vary in their risk of drowning. For example, some species appear particularly prone to becoming trapped in nets and pots, whilst others avoid such hazards more successfully. Differences among species are likely to be caused by a range of characteristics, including species' morphology, feeding ecology, and behaviour. It is widely known that juvenile birds are often more prone to such mortality than adults. For example, ring recoveries of juvenile shags *Phalacrocorax aristotelis* occur not infrequently in lobster pots, whereas adult shags are rarely trapped in this way (Galbraith *et al.* 1981). Scoring species in this factor is difficult, as it is impossible to obtain quantitative data on risk levels across species. However, there are published studies reviewing the causes of mortality of marine birds that identify drowning through entrapment in underwater structures and its prevalence in certain species (for example, sea ducks and divers: Žydelis *et al.* 2009). For surface-feeding marine birds the risk of encountering tidal turbines is very low. For diving birds Furness *et al.* (2012) reviewed the literature to identify species most at risk of entrapment and used expert judgement to moderate scores. Scores in this chapter follow Furness *et al.* (2012) and are allocated from 1 (extremely low risk) to 5 (moderate risk) (Table 2.4, and see supplementary material; Furness *et al.* 2012).

Mean and maximum diving depth

Seabirds capable of diving to depths where tidal turbines will be deployed will likely be at greater risk of colliding with these structures, particularly if birds regularly dive at those depths. The depth of deployment of tidal turbines varies with design but is typically 30–50 m below sea surface (Aquatera 2010). Mean and maximum diving depths of seabirds have been recorded for many species, predominantly by deployment of data loggers on breeding seabirds. Where data differed between studies, Furness *et al.* (2012) gave greater

weighting to more recent studies, as they tend to use instrumentation rather than inference (i.e. inferred depth from time spent underwater). Depth categories ensure a spread of species among the five categories and clearly separate species likely to attain depths where tidal turbines are deployed from those unable to access such depths. Scores follow Furness *et al.* (2012) and are allocated as: 1 (surface feeders with maximum diving depth no more than 1 m), 2 (regularly dive to 2 or 3 m but have a maximum diving depth of no more than 5 m), 3 (regularly dive to 5 m but rarely below 20 m), 4 (regularly dive to 20 m but rarely below 30 m), and 5 (regularly dive to 30 m and deeper) (Table 2.4, and see supplementary material; Furness *et al.* 2012).

Benthic foraging

Benthic foraging seabirds may be more likely to interact with tidal turbines than seabirds that do not forage on benthic prey (although see Section 1.3 for discussion concerning risk to pelagic foragers). Categories and scores follow Furness *et al.* (2012), which were chosen to ensure a spread of species among the five categories. Score are allocated as: 1 (<5% benthic foraging), 2 (5% ≤ benthic foraging <20%), 3 (20% ≤ benthic foraging <40%), 4 (40% ≤ benthic foraging <70%), and 5 (70% ≤ benthic foraging <100%) (Table 2.4, and see supplementary material; Furness *et al.* 2012).

Use of tidal races for foraging

There have been relatively few studies of the use of high current tidal flow areas (current speeds >2m s^{-1}; Fraenkel 2006) by foraging marine birds and most of the few published studies relate to birds in the North Pacific; although studies are currently being undertaken in the UK. A key obstacle in establishing the use of tidal races by marine birds is the inherent challenge of monitoring bird behaviour and linking it with physical current flows in such

dynamic environments (Embling *et al.* 2012). In this chapter, newly available data were extracted from the few existing peer-reviewed studies, from MRED reports and unpublished research. Scores previously moderated by experts in Furness *et al.* (2012) were only adjusted if strong evidence had recently become available (Table 2.4 and Appendix 1, Table 5). Scores were allocated from 1 (habitat strongly avoided) to 5 (preferred habitat).

Feeding range

As central place foragers during the breeding season, marine birds are constrained to regularly return to a nest site. In winter and during migration periods, marine birds are less constrained and may travel over enormous distances. However, even in winter some species are limited in the distances they can travel by the need to return to a safe nocturnal roost site or to spend the night out of water (e.g. cormorants). Although the distribution of predictable feeding hotspots may influence habitat quality for marine birds, species with short feeding ranges will more likely be affected by the placement of local MREDs than species with greater foraging ranges. This is because species that forage over large areas can circumvent MREDs with minimal loss of foraging opportunity, whilst species with short foraging ranges may lose higher proportions of their available foraging area if displaced by the presence of devices. Scores follow Furness *et al.* (2012) and are allocated as: 1 (mean range generally over 90 km, and maximum generally over 150 km), 2 (mean range 20–90 km and maximum around 100–150 km), 3 (mean range 10–20 km and maximum around 50– 80 km), 4 (mean range 5–10 km and maximum around 20–50 km), and 5 (mean range less than 5 km and maximum generally less than 20 km). In Furness *et al.* (2012) if cases reported maximum and mean foraging ranges that fell across different scores, greater weight was given to the mean values. Where several estimates were available in the literature but fell across different score bands, Furness *et al.* (2012) gave lower weighting to estimates that

were in review publications rather than original data; lower weighting to estimates reported in unpublished reports, or reports that appeared not to have been peer reviewed; lower weighting to older literature; and lower weighting to estimates derived indirectly from data (such as time spent away from the nest combined with estimated flight speed). Greater weight was given to data derived from direct measures, such as deployment of GPS loggers on breeding birds. Where no data could be found for a particular species, scores in Furness *et al.* (2012) were estimated from data for closely related species with similar ecology (Table 2.4, and see supplementary material; Furness *et al.* 2012).

Habitat specialisation

See in: 2) Displacement and disturbance caused by offshore wind farms.

4) Vulnerability to wave energy developments

Displacement and disturbance as a result of structures

See in: 2) Displacement and disturbance caused by offshore wind farms.

Displacement and disturbance as a result of vessel and/or helicopter traffic

See in: 2) Displacement and disturbance caused by offshore wind farms.

Risk of collision mortality due to structures

Some marine birds may be at risk of injury or death from colliding with wave energy devices, either during low altitude flight or whilst swimming or diving; particularly when visibility is reduced, for example during poor weather or at night. This chapter follows Furness *et al.* (2012) and scores species from 1 (minimal risk of mortality) to 5 (moderate risk of mortality) (Table 2.5, and see supplementary material; Furness *et al.* 2012). Given the nature of wave

energy devices, even a score of 5 on this factor would probably represent a relatively low risk compared to risks such as entanglement in netting.

Exclusion from foraging habitat due to behavioural constraints

Wave energy devices might prevent some marine birds from foraging in important habitat. This may be because they are unable to land or take off readily where devices are present in the water; because other birds have been attracted into the area and affect their foraging; or because they need to spend time avoiding the devices rather than searching for food. For example, diver species require open water for landing and taking off, and may be unable and/or unwilling to land in areas where they perceive that devices block a descent flight onto the water. Alternatively, this impact may be trivial for species that have large foraging ranges and a broad diversity of habitats in which they can feed. Scores in this chapter follow Furness *et al.* (2012) and are allocated from 1 (minimal exclusion) to 5 (moderate exclusion from foraging habitat) (Table 2.5, and see supplementary material; Furness *et al.* 2012).

Benefit from roost platform

Under relatively calm sea conditions, wave energy devices may provide marine birds with a resting platform. Such an opportunity could extend the potential foraging area of some species; for example, cormorants and shags need to return to shore to dry their plumage after a foraging bout and so are limited in the distance they can travel from land. Resting sites at sea could enable these species to exploit areas further from the coast that would otherwise be uneconomical for birds to commute to from the shore. This behaviour has already been reported at the Nysted offshore wind farm in Danish waters. Scores in this chapter follow Furness *et al.* (2012) and classify species into categories from 0 (no significant benefit likely) to –2 (moderate likelihood of species gaining benefit from opportunity to

roost on structures) (Table 2.5, and see supplementary material; Furness *et al.* 2012). Designating –2 as the extreme value, and the use of only three categories, recognises that this effect is likely to have relatively minor influence at the population level.

Benefit from Fish Attraction Device (FAD) effect or biofouling

Wave energy devices are likely to provide shelter for small fish and so have the potential to act as FADs (Rountree 1990; Castro, Santiago & Santana-Ortega 2002; Lokkeborg 2002; Wilhelmsson *et al.* 2006; Langhamer & Wilhelmsson 2009). They are also likely to represent surfaces onto which biofouling organisms will settle. Both situations could attract foraging birds by providing locally high densities of prey, and could explain the reported attraction of some marine birds to structures such as oil platforms (Baird 1990; Wiese *et al.* 2001) and the return of common scoter *Melanitta nigra* to wind farm areas following a period of avoidance (Petersen & Fox 2007). Scores in this chapter follow Furness *et al.* (2012) and classify species into categories from 0 (no significant benefit likely) to –2 (moderate likelihood of species gaining benefit from locally increased prey densities) (Table 2.5, and see supplementary material; Furness *et al.* 2012).

Habitat specialisation

See in: 2) Displacement and disturbance caused by offshore wind farms.

2.2.1.3 Calculating species vulnerability scores

Garthe & Hüppop (2004) calculated species vulnerability scores to offshore wind farms according to Equation 2.1, where *Z* represents the percentage of flight at turbine blade altitudes, flight agility, percentage of time flying, and nocturnal flight activity. *Y* represents disturbance by wind farm structures, ship and helicopter traffic, and habitat specialisation.

$$Overall\ species\ score\ = \frac{1}{4}\sum_{i=Z}^{4} score_i \times \frac{1}{2}\sum_{i=Y}^{2} score_i\ \times conservation\ score$$

Equation 2.1

This recognised that the first four factors all relate to flight ability and flight behaviour, while the last two factors relate to habitat use and susceptibility to disturbance. Thus their index combined both collision risk and disturbance/displacement considerations in a single vulnerability score.

In Furness *et al.* (2013) an alternative approach was used that scored separately for wind turbine collision concern and for disturbance/displacement concern. For collision risk, Furness *et al.* (2013) gave a high weighting to flight at altitudes overlapping with turbine blades (*a*). A lower weighting was given to flight agility, percentage of time flying, and nocturnal flight activity, which are represented by *X* (Equation 2.2).

$$Collision\ risk\ score\ = a\ \times\ \frac{1}{3}\sum_{i=X}^{3} score_i\ \times conservation\ score$$

Equation 2.2

In the case of vulnerability to disturbance/displacement Furness *et al.* (2013) calculated a vulnerability index according to Equation 2.3. In this equation *b* and *c* represent disturbance by wind farm structures, ship and helicopter traffic, and habitat specialisation respectively. The score was then divided by 10 (an arbitrary value) to recognise that the impact of disturbance/displacement at a population level is likely to be considerably less than the

impact of direct mortality from collisions. Therefore, the two scales cannot be compared in a quantitative way, only in terms of species rankings within each scale.

$$Displacement/disturbance\ score\ = \frac{(b\ \times\ c) \times conservation\ score}{10}$$

Equation 2.3

1) <u>Collision risk with offshore wind farms</u>

In this chapter I have split the offshore wind vulnerability factor 'disturbance and displacement as a result of structures, vessel and helicopter traffic' into two separate factors; one concerned with disturbance and displacement as a result of structures (*d*) and the second, disturbance and displacement as a result of vessel and/or helicopter traffic (*e*). This is appropriate given evidence that that some species (e.g. gannets) are not consistent in their responses to both structures and vessels. I have also adjusted the calculation to assess vulnerability to collision risk (Table 2.2) by dividing the percentage of flight at altitudes overlapping with turbine blades by the score for displacement and disturbance as a result of turbine structures. This takes into consideration that species avoiding wind farm areas are unlikely to collide with wind turbines. The higher weighting of flight at altitudes overlapping with turbine blades (*a*) used by Furness *et al.* (2013) has been retained; as have the lower weightings for flight agility, percentage of time flying, and nocturnal flight activity, which are represented by *X* (Equation 2.4).

$$Collision\ risk\ score\ = \frac{a}{d} \times\ \frac{1}{3}\sum_{i=X}^{3} score_i\ \times conservation\ score$$

Equation 2.4

2) Displacement and disturbance caused by offshore wind farms

In this chapter I have also adjusted the calculation for offshore wind farms that assesses vulnerability to disturbance and displacement caused by wind farms (Table 2.3). This is to take into account the now split factor 'disturbance and displacement as a result of structures, vessel and helicopter traffic'. The adjusted calculation gives higher weighting to disturbance and displacement by structures (*d*) and a lower weighting to disturbance and displacement as a result of vessel and/or helicopter traffic (*e*). This is to take into account the permanent nature of wind turbine structures and the more transient nature of vessel and helicopter activity. Habitat specialisation is represented by *c* (Equation 2.5).

$$Displacement/disturbance\ score = \frac{(((c\ \times\ d) + e) \times conservation\ score)}{10}$$

Equation 2.5

3) Vulnerability to tidal-stream energy developments

When calculating an updated vulnerability index of sensitivity to tidal-stream energy developments (Table 2.4), I continued to use the equation in Furness *et al.* (2012) but included updated data in the vulnerability factors of 'use of tidal races' and 'disturbance and displacement as a result of vessel traffic'. Of the 7 vulnerability factors scored, the use of tidal races as foraging habitat (*i*) is likely to be particularly important in determining a species overall vulnerability, as does depth attained during diving (*j*). To take this into account, five factors (represented by *W*) comprising habitat specialisation, disturbance and displacement as a result of vessel traffic, drowning risk, benthic foraging, and foraging range were included in the calculation at similar, but lower importance. The score was then divided by 10 (an arbitrary value) (Equation 2.6).

$$Tidal\ stream\ score = \frac{((i\ \times j)\ \times \frac{1}{5}\sum_{i=W}^{5} score_i) \times conservation\ score}{10}$$

Equation 2.6

4) Vulnerability to wave energy developments

In calculating the vulnerability index of sensitivity to wave energy developments (Table 2.5), I again continued to use the equation in Furness *et al.* (2012) but with updated data in the vulnerability factors of 'disturbance and displacement as a result of structures' and 'disturbance and displacement as a result of vessel traffic'. Of the 7 vulnerability factors scored (and represented by *V*), five represent negative effects (habitat specialisation, disturbance and displacement as a result of structures, disturbance and displacement as a result of vessel traffic, risk of collision mortality due to structures, and exclusion from foraging habitat). The remaining two factors (benefit from roost platform and benefit from FAD effects or biofouling) represent positive effects. No factor seemed more important than any other so the scores were simply summed (Equation 2.7).

$$Wave\ score = \sum_{i=V}^{7} score_i \times conservation\ score$$

Equation 2.7

2.2.1.4 Vulnerability indices not including the conservation importance score

All vulnerability indices incorporate a conservation importance score for each species in generating the overall vulnerability score that is used to rank species vulnerability to MREDs, which includes measures of each species' conservation status in Europe and the UK.

Because the conservation importance score could limit the application of the vulnerability indices to other regions and in the future, indices were also generated that did not incorporate the conservation importance scores in calculating overall vulnerability scores.

2.2.2 Confidence indices

An extensive literature review was undertaken to compile all information available to update four of the vulnerability factors presented in Furness *et al.* (2012, 2013): 1) the percentage of marine bird flight at altitudes overlapping with turbine rotor blades, 2) disturbance and displacement of marine bird species caused by MRED structures, 3) disturbance and displacement of species caused by vessel and/or helicopter traffic, and 4) the use of tidal races by marine birds (see Appendix 1, Tables 2-5 for data gathered and associated references). These factors were updated as they were deemed to be of high importance in determining marine bird vulnerability to MREDs.

The four vulnerability factors were scored at the species level to quantify a level of confidence inherent in the data used to calculate vulnerability indices. By quantifying a level of confidence in the data for each species, this chapter aims to identify where more research may be useful and highlight where caution in interpreting species' vulnerability rankings is required. Within each vulnerability factor for each species, five categories related to the data were scored:

1. It was established whether the majority of data collected for each species originated from the study of the target species, related species, or from the family level. For example, in the case of black guillemot *Cepphus grylle* being the target species, a related species could be pigeon guillemot *Cepphus columba* and data at the family level would refer to auk species. Species were scored as 3 if the majority of data

referred to the target species, 2 if the data referred to related species or data from the family level, and 1 if no published data were available for a particular species.

2. The number of studies for each species was identified. This category was scored according to classifications in Table 2.1.
3. The number of sites at which data were collected was recorded. This category was scored according to classifications in Table 2.1.
4. The mean period of years over which data were collected for each species was included (Tables 2.10 - 2.13 and Appendix 1, Tables 2-5).
5. An overall study type score was calculated based on the level of confidence associated with the methods used in each study, for each species (see Table 2.1 for classification of methods).

Equation 2.8 was used to calculate the overall study type score for each species in the vulnerability factors 'percentage of marine bird flight at altitudes overlapping with turbine rotor blades' and the 'use of tidal races' (Tables 2.10 and 2.13):

$$Species\ study\ type\ score = (E \times 5) + (D \times 4) + (C \times 3) + (B \times 2) + (A)$$

Equation 2.8

Owing to the different types of methods employed in the studies, five categories were used to classify the study type score for these two vulnerability factors. In Equation 2.8, (E) refers to the number of studies with methods associated with high confidence, (D) to the number of studies with methods associated with moderate levels of confidence, (C) to study methods associated with low confidence, (B) to study methods associated with uncertain levels of confidence, and (A) to study methods associated with poor confidence (Table 2.1). Weighting to promote studies with more reliable methods was achieved by multiplying the

number of studies in each category by the values associated with each level of confidence: from 1 (poor confidence) to 5 (high confidence).

Equation 2.9 was used to calculate a study type score for each species in the factors 'disturbance and displacement of marine bird species' and 'disturbance and displacement of species caused by vessel and/or helicopter traffic' (Tables 2.11 and 2.12):

$$Species\ study\ type\ score = (C \times 3) + (B\ \times 2) + (A)$$

Equation 2.9

Owing to the lower number of different method types employed in studies related to these vulnerability factors, only 3 categories were used to classify the study type scores, whilst still promoting studies using more reliable methods. Thus methods in category (A) scored 1, category (B) scored 2, and (C) scored 3 (Table 2.1).

Table 2.1: Categories and scores used to quantify the confidence inherent in data collected for each species, in each of four vulnerability factors. Confidence categories comprised 5 groupings with associated scores: poor (score 1), uncertain (score 2), low (score 3), moderate (score 4) and high confidence (score 5). Capital letters in brackets indicate the study type incorporated in Equations 2.8 and 2.9.

Factor	**Factor attributes**	**Confidence category**				
		Poor (1)	**Uncertain (2)**	**Low (3)**	**Moderate (4)**	**High (5)**
% time flying at turbine height	**Method (study type)**	Anecdotal observation (or unknown method) **(A)**	Observations not recorded in the presence of turbines (indirect study 2) **(B)**	Observations recorded in the presence of turbines (indirect study 1) **(C)**	Study combining results from 5 or more studies/sites to produce modelled flight information **(D)**	GPS or radar (direct study) **(E)**
	No. of studies and sites	<5	5-10	11-20	21-30	>30
	Confidence score	0-15	16-30	31-45	46-60	>60
Disturbance by structures	**Method (study type)**	Anecdotal observation (or unknown method) **(A)**	Observation **(B)**	Before-After-Control-Impact study (BACI) **(C)**		
	No. of studies and sites	0-1	2-4	5-7	8-10	>10
	Confidence score	0-10	11-20	21-30	31-40	>40
Disturbance by vessel and/or helicopter activity	**Method (study type)**	Anecdotal observation (or unknown method) **(A)**	Observation **(B)**	BACI or experimental method **(C)**		
	No. of studies and sites	0-1	2-4	5-7	8-10	>10
	Confidence score	0-10	11-20	21-30	31-40	>40
Use of tidal races	**Method (study type)**	Anecdotal observation (or unknown method) **(A)**	Observation without current data **(B)**	Observation with modelled or inferred current data **(C)**	Study combining results from 5 or more studies/sites with modelled or inferred current data **(D)**	Observation with concurrent current data **(E)**

No. of studies and sites	0-1	2-4	5-7	8-10	>10
Confidence score	0-5	6-10	11-15	15-20	>20

Confidence scores and a related confidence category were established for each species, for each vulnerability factor. These were calculated using scores from the 5 categories assessed for confidence (represented by *U*): study species score, number of studies score, number of sites score, mean period of data collection, and study type score (Equation 2.10).

$$Confidence\ score = \sum_{i=U}^{5} score_i$$

Equation 2.10

An overall confidence category was calculated for each species by summing their confidence scores in each of the four vulnerability factors. This provided an indication of which species were lacking in data and which species were likely to have robust estimates of vulnerability (Table 2.14).

To summarise, the terms used in this chapter with regards to the confidence indices are:

1. Confidence index - an index where species are assigned to a confidence category and score related to the level of confidence inherent in the data contributing to four vulnerability factors.
2. Confidence category - one of five categories that indicate the level of confidence inherent in species data. The categories are: poor, uncertain, low, moderate and high.

3. Confidence scores - are scores associated with each of the five confidence categories: poor (score 1), uncertain (score 2), low (score 3), moderate (score 4) and high (score 5).
4. Combined confidence score – is the resulting score once the confidence scores relating to each vulnerability factor, for each species, are combined. This score gives an overall impression of the level of understanding we have for each species.

2.3 Results

2.3.1 Vulnerability indices

2.3.1.1 Vulnerability to collision with offshore wind turbines

In the updated vulnerability indices white-tailed eagle, lesser black-backed gull, herring gull, great black-backed gull and common gull scored highest on the offshore wind collision risk index. This indicates that these species are most likely to be at risk of collision with offshore wind turbines. The top five highest scoring species in this index were the same as those reported in Furness *et al.* (2013) with the exception of gannet, which dropped down the rankings from 5th to 11th. Common gull moved up one ranking in the updated index to replace gannet as one of the top five highest scoring species (Table 2.2).

Table 2.2: Offshore wind turbine collision risk vulnerability index with species ranked by vulnerability score. 'asl': above sea level. (see Appendix 1, Table 1 for scientific names of species)

Rank	Furness et al. 2013 rank	Species	% of flight at blade height (c.a. 20-150 m asl)	Displacement: structures	Flight agility	% of time in flight	Flight at night	Conservation importance score	Vulnerability score
1	4	White-tailed eagle	24	1	3	5	1	12	864
2	3	Lesser black-backed gull	38	2	1	2	3	16	608
3	1	Herring gull	31	2	2	2	3	16	579
4	2	Great black-backed gull	33	2	2	2	3	15	578

5	6	Common gull	30	2	1	2	3	13	390
6	11	Sandwich tern	15	2	1	5	1	15	263
7	23	Great cormorant	9	1	4	2	1	11	231
8	7	Black-legged kittiwake	14	2	1	3	3	14	229
9	9	Great skua	7	1	1	4	1	16	224
10	8	Arctic skua	5	1	1	5	1	14	163
11	5	Northern gannet	14	4	3	3	2	17	159
12	10	Black-headed gull	18	2	1	1	2	12	144
13	18	Roseate tern	6	2	1	5	1	15	105
14	17	Arctic tern	4	2	1	5	1	17	79
15	13	Great northern diver	7	5	5	2	1	18	67
15	12	Black-throated diver	7	5	5	3	1	16	67
17	14	Common tern	4	2	1	5	1	14	65
18	16	Little tern	4	2	1	5	1	13	61
19	19	European shag	2	1	3	2	1	15	60
19	15	Red-throated diver	7	5	5	2	1	16	60
21	20	Slavonian grebe	4	3	4	2	2	13	46
22	26	European storm-petrel	1	1	1	3	4	17	45
23	28	Leach's storm-petrel	1	1	1	3	4	16	43
24	22	Common eider	3	3	4	2	3	13	39
25	25	Common scoter	6	5	3	2	3	12	38
26	27	Velvet scoter	6	5	3	2	3	11	35
27	24	Common goldeneye	3	4	3	2	3	12	24
28	30	Long-tailed duck	3	4	3	2	3	8	16
29	32	Common guillemot	0	4	4	1	2	16	0
29	33	Razorbill	0	4	4	1	1	16	0
29	21	Greater scaup	0	4	4	2	5	11	0
29	34	Black guillemot	0	3	4	1	1	13	0
29	35	Atlantic puffin	0	3	3	1	1	16	0
29	29	Great-crested grebe	0	3	4	3	2	7	0
29	36	Little auk	0	3	3	1	1	9	0
29	37	Manx shearwater	0	1	3	3	3	17	0
29	31	Northern fulmar	0	1	3	2	4	16	0
29	38	Sooty shearwater	0	1	3	3	3	12	0

2.3.1.2 Vulnerability to displacement and disturbance caused by offshore wind developments

Black-throated diver, red-throated diver, great northern diver, common scoter, common goldeneye, common guillemot and razorbill were the top five highest scoring species in the updated index detailing vulnerability to displacement and disturbance caused by offshore wind developments. These species are most likely to be affected by displacement and

disturbance during offshore wind farm installation and operation. Most of these species match those identified previously in Furness *et al.* (2013) as the most vulnerable species, although common guillemot and razorbill have increased in their vulnerability ranking based on the updated index (Table 2.3).

Table 2.3: Offshore wind farm disturbance and displacement vulnerability index with species ranked by vulnerability score.

Rank	Furness et al. 2013 rank	Species	Displacement: structures	Displacement: vessels and helicopter	Habitat use flexibility	Conservation importance score	Vulnerability score
1	1	Black-throated diver	5	5	4	16	40
1	2	Red-throated diver	5	5	4	16	40
3	3	Great northern diver	5	4	3	18	34
4	4	Common scoter	5	5	4	12	30
5	5	Common goldeneye	4	4	4	12	24
5	11	Common guillemot	4	3	3	16	24
5	12	Razorbill	4	3	3	16	24
8	6	Greater scaup	4	4	4	11	22
9	7	Velvet scoter	5	4	3	11	21
10	8	Common eider	3	3	4	13	20
10	9	Black guillemot	3	3	4	13	20
10	10	Slavonian grebe	3	3	4	13	20
13	17	Atlantic puffin	3	3	3	16	19
14	18	Long-tailed duck	4	3	4	8	15
15	16	Arctic tern	2	2	3	17	14
16	15	Little tern	2	2	4	13	13
17	19	Roseate tern	2	2	3	15	12
17	20	Sandwich tern	2	2	3	15	12
19	21	Common tern	2	2	3	14	11
19	13	European shag	1	4	3	15	11
19	22	Great-crested grebe	3	3	4	7	11
22	28	Northern gannet	4	1	1	17	9
23	24	Black-legged kittiwake	2	2	2	14	8
23	27	Little auk	3	3	2	9	8
23	25	Common gull	2	2	2	13	8
23	14	Great cormorant	1	4	3	11	8
23	23	Great black-backed gull	2	1	2	15	8
28	26	Black-headed gull	2	2	2	12	7
29	33	White-tailed eagle	1	2	2	12	5
29	30	Great skua	1	1	2	16	5

29	29	Herring gull	2	1	1	16	5
29	31	Lesser black-backed gull	2	1	1	16	5
33	32	Arctic skua	1	1	2	14	4
34	34	Manx shearwater	1	1	1	17	3
34	35	European storm-petrel	1	1	1	17	3
34	36	Leach's storm-petrel	1	1	1	16	3
34	37	Northern fulmar	1	1	1	16	3
38	38	Sooty shearwater	1	1	1	12	2

2.3.1.3 Vulnerability to tidal-stream energy developments

Razorbill, European shag, black guillemot, common guillemot and great cormorant were the top five highest scoring species in the updated index assessing species vulnerability to tidal-stream energy developments. These top five species match those identified previously in Furness *et al.* (2012) (Table 2.4).

Table 2.4: Tidal-stream energy development vulnerability index with species ranked by vulnerability score.

Rank	Furness et al. 2013 rank	Species	Displacement: vessels	Drowning risk	Mean and max diving depth	Benthic foraging	Use of tidal races	Foraging range	Habitat use flexibility	Conservation importance score	Vulnerability score
1	2	Razorbill	3	4	5	4	5	3	3	16	13.6
2	3	Shag	4	4	4	5	5	4	3	15	12.0
3	1	Black guillemot	3	4	4	4	5	4	4	13	9.9
4	4	Common guillemot	3	4	5	2	4	2	3	16	9.0
5	5	Great cormorant	4	4	4	5	4	4	3	11	7.0
6	6	Great northern diver	4	4	3	4	3	3	3	18	5.8
7	8	Atlantic puffin	3	3	4	1	4	1	3	16	5.6
8	7	Red-throated diver	5	4	3	4	2	3	4	16	3.8
9	9	Black-throated diver	5	4	3	3	2	3	4	16	3.6
10	10	Little auk	3	3	3	1	4	2	2	9	2.4
11	11	Slavonian grebe	3	4	2	4	2	4	4	13	2.0
12	12	Arctic tern	2	1	1	1	5	4	3	17	1.9
13	13	Common eider	3	4	3	5	1	3	4	13	1.5
14	15	Manx shearwater	1	1	3	1	3	1	1	17	1.5
15	14	Common scoter	5	4	3	5	1	3	4	12	1.5
16	16	Velvet scoter	4	4	3	5	1	4	3	11	1.3
17	17	Northern gannet	1	2	3	1	2	1	1	17	1.2

18	21	Sandwich tern	2	1	2	1	2	3	3	15	1.2
19	19	Great-crested grebe	3	4	2	4	2	4	4	7	1.1
20	18	Common goldeneye	4	4	2	5	1	5	4	12	1.1
21	20	Sooty shearwater	1	1	3	1	3	1	1	12	1.1
22	23	Long-tailed duck	3	4	3	5	1	4	4	8	1.0
23	25	Roseate tern	2	1	1	1	3	4	3	15	1.0
24	22	Greater scaup	4	4	2	5	1	5	4	11	1.0
25	34	Common tern	2	1	1	1	3	4	3	14	0.9
26	29	Common gull	2	1	1	1	4	3	2	13	0.9
27	26	Black-legged kittiwake	2	1	1	1	4	2	2	14	0.9
28	24	Great black-backed gull	1	1	1	1	4	2	2	15	0.8
29	28	Great skua	1	1	1	1	3	2	2	16	0.7
30	27	Herring gull	1	1	1	2	3	2	1	16	0.7
31	31	Little tern	2	1	1	1	2	5	4	13	0.7
32	32	White-tailed eagle	2	1	1	1	3	3	2	12	0.6
33	33	Arctic skua	1	1	1	1	3	2	2	14	0.6
34	35	Black-headed gull	2	1	1	1	3	3	2	12	0.6
35	30	Lesser black-backed gull	1	1	1	1	3	2	1	16	0.6
36	37	European storm-petrel	1	1	1	1	3	1	1	17	0.5
37	38	Leach's storm-petrel	1	1	1	1	3	1	1	16	0.5
38	36	Northern fulmar	1	1	1	1	3	1	1	16	0.5

2.3.1.4 Vulnerability to wave energy developments

Red-throated diver, black-throated diver, great northern diver, razorbill and common guillemot were the top five highest scoring species in the updated index assessing species vulnerability to wave energy developments. These top five species match those previously identified in Furness *et al.* (2012) (Table 2.5).

Table 2.5: Wave energy development vulnerability index with species ranked by vulnerability score.

Rank	Furness et al. 2013 rank	Species	Displacement: structures	Displacement: vessels/helicopter	Risk of collision	Exclusion from foraging habitat	Benefit from roost platform	Benefit from FAD or biofouling	Habitat use flexibility	Conservation importance score	Vulnerability score
1	1	Red-throated diver	5	5	3	3	0	0	4	16	320
1	2	Black-throated diver	5	5	3	3	0	0	4	16	320
3	3	Great northern diver	5	4	3	3	0	-1	3	18	306
4	4	Razorbill	4	3	3	1	0	0	3	16	224
5	6	Common guillemot	4	3	3	1	0	-1	3	16	208
6	5	Common scoter	5	5	2	2	0	-1	4	12	204

7	10	Atlantic puffin	3	3	2	1	0	0	3	16	192
8	7	Black guillemot	3	3	3	3	-1	-1	4	13	182
9	8	Slavonian grebe	3	3	2	2	0	0	4	13	182
10	15	Common goldeneye	4	4	2	2	0	-1	4	12	180
11	12	Greater scaup	4	4	2	2	0	0	4	11	176
12	13	Velvet scoter	5	4	2	2	0	-1	3	11	165
13	18	Common eider	3	3	2	2	-1	-1	4	13	156
14	14	Arctic tern	2	2	2	2	-1	-1	3	17	153
15	16	Northern gannet	4	1	2	1	0	0	1	17	153
16	9	European shag	1	4	3	3	-2	-2	3	15	150
17	11	Little tern	2	2	2	2	-1	0	4	13	143
18	17	Roseate tern	2	2	2	2	-1	-1	3	15	135
19	20	Sandwich tern	2	2	2	2	-1	-1	3	15	135
20	19	Common tern	2	2	2	2	-1	-1	3	14	126
21	24	Long-tailed duck	4	3	2	2	0	-1	4	8	112
22	22	Manx shearwater	1	1	2	1	0	0	1	17	102
23	28	Little auk	3	3	2	1	0	0	2	9	99
24	21	Great cormorant	1	4	3	2	-2	-2	3	11	99
25	26	Great-crested grebe	3	3	2	2	0	0	4	7	98
26	25	Great skua	1	1	1	1	0	0	2	16	96
27	23	Black-legged kittiwake	2	2	1	1	-1	-1	2	14	84
28	32	White-tailed eagle	1	2	1	2	-1	0	2	12	84
29	27	Arctic skua	1	1	1	1	0	0	2	14	84
30	34	Common gull	2	2	1	1	-1	-1	2	13	78
31	30	Great black-backed gull	2	1	1	1	-1	-1	2	15	75
32	37	Black-headed gull	2	2	1	1	-1	-1	2	12	72
33	31	Sooty shearwater	1	1	2	1	0	0	1	12	72
34	33	European storm-petrel	1	1	1	1	0	-1	1	17	68
35	35	Lesser black-backed gull	2	1	1	1	-1	-1	1	16	64
36	36	Leach's storm-petrel	1	1	1	1	0	-1	1	16	64
37	29	Northern fulmar	1	1	1	1	-1	0	1	16	64
38	38	Herring gull	2	1	1	1	-1	-2	1	16	48

2.3.1.5 Vulnerability indices not including the conservation importance score

Overall vulnerability rankings of species', or their region of vulnerability within the indices, did not alter to a great extent when conservation importance scores were not included in calculating vulnerability scores (Tables 2.6-2.9). In all cases the top ten species remained ranked as the most vulnerable species. Long-tailed duck and great-crested grebe increased in vulnerability in relation to wave energy developments and disturbance and/or displacement caused by offshore wind developments.

Table 2.6: Offshore wind turbine collision risk vulnerability index calculated without incorporating species' conservation importance scores. Species are ranked by vulnerability score (see Appendix 1, Table 1 for scientific names of species).

Grouping according to vulnerability score	Rank when conservation score not included	Rank when conservation score included-ten top scoring species in red	Species	% of flight at blade height (ca. 20-150 m asl)	Displacement: structures	Flight agility	% of time in flight	Flight at night	Vulnerability score
1	1	1	White-tailed eagle	24	1	3	5	1	72
2	2	4	Great black-backed gull	33	2	2	2	3	39
3	3	2	Lesser black-backed gull	38	2	1	2	3	38
4	4	3	Herring gull	31	2	2	2	3	36
5	5	5	Common gull	30	2	1	2	3	30
6	6	7	Great cormorant	9	1	4	2	1	21
7	7	6	Sandwich tern	15	2	1	5	1	18
8	8	8	Black-legged kittiwake	14	2	1	3	3	16
9	9	9	Great skua	7	1	1	4	1	14
10	10	12	Black-headed gull	18	2	1	1	2	12
10	10	10	Arctic skua	5	1	1	5	1	12
11	12	11	Northern gannet	14	4	3	3	2	9
12	13	13	Roseate tern	6	2	1	5	1	7
13	14	14	Arctic tern	4	2	1	5	1	5
	14	17	Common tern	4	2	1	5	1	5
	14	18	Little tern	4	2	1	5	1	5
14	17	15	Black-throated diver	7	5	5	3	1	4
	17	19	Shag	2	1	3	2	1	4
	17	15	Great northern diver	7	5	5	2	1	4
	17	19	Red-throated diver	7	5	5	2	1	4
	17	21	Slavonian grebe	4	3	4	2	2	4
15	22	25	Common scoter	6	5	3	2	3	3
	22	26	Velvet scoter	6	5	3	2	3	3
	22	24	Common eider	3	3	4	2	3	3
	22	22	European storm-petrel	1	1	1	3	4	3
	22	23	Leach's storm-petrel	1	1	1	3	4	3
16	27	27	Common goldeneye	3	4	3	2	3	2
	27	28	Long-tailed duck	3	4	3	2	3	2
17	29	29	Common guillemot	0	4	4	1	2	0
	29	29	Razorbill	0	4	4	1	1	0
	29	29	Greater scaup	0	4	4	2	5	0
	29	29	Black guillemot	0	3	4	1	1	0
	29	29	Atlantic puffin	0	3	3	1	1	0
	29	29	Great-crested grebe	0	3	4	3	2	0
	29	29	Little auk	0	3	3	1	1	0
	29	29	Manx shearwater	0	1	3	3	3	0

29	29	Northern fulmar	0	1	3	2	4	0
29	29	Sooty shearwater	0	1	3	3	3	0

Table 2.7: Offshore wind farm disturbance and displacement vulnerability index calculated without incorporating species' conservation importance scores. Vulnerability scores were not divided by ten during calculations to avoid unnecessarily small values. Species are ranked by vulnerability score.

Grouping according to vulnerability score	Rank when conservation score not included	Rank when conservation score included-ten top scoring species in red	Species	Displacement: structures	Displacement: vessels and helicopter	Habitat use flexibility	Vulnerability score
1	1	1	Black-throated diver	5	5	4	25
	1	1	Red-throated diver	5	5	4	25
	1	4	Common scoter	5	5	4	25
2	4	5	Common goldeneye	4	4	4	20
	4	8	Greater scaup	4	4	4	20
3	6	3	Great northern diver	5	4	3	19
	6	9	Velvet scoter	5	4	3	19
	6	14	Long-tailed duck	4	3	4	19
4	9	5	Common guillemot	4	3	3	15
	9	5	Razorbill	4	3	3	15
	9	10	Common eider	3	3	4	15
	9	10	Black guillemot	3	3	4	15
	9	10	Slavonian grebe	3	3	4	15
	9	19	Great-crested grebe	3	3	4	15
5	15	13	Atlantic puffin	3	3	3	12
6	16	16	Little tern	2	2	4	10
7	17	23	Little auk	3	3	2	9
8	18	15	Arctic tern	2	2	3	8
	18	17	Roseate tern	2	2	3	8
	18	17	Sandwich tern	2	2	3	8
	18	19	Common tern	2	2	3	8
9	22	19	Shag	1	4	3	7
	22	23	Great cormorant	1	4	3	7
10	24	23	Black-legged kittiwake	2	2	2	6
	24	23	Common gull	2	2	2	6
	24	28	Black-headed gull	2	2	2	6
11	27	22	Northern gannet	4	1	1	5
	27	23	Great black-backed gull	2	1	2	5
12	29	29	White-tailed eagle	1	2	2	4
13	30	29	Great skua	1	1	2	3
	30	29	Herring gull	2	1	1	3
	30	29	Lesser black-backed gull	2	1	1	3

	30	33	Arctic skua	1	1	2	3
	34	34	Manx shearwater	1	1	1	2
	34	34	European storm-petrel	1	1	1	2
14	34	34	Leach's storm-petrel	1	1	1	2
	34	34	Northern fulmar	1	1	1	2
	34	38	Sooty shearwater	1	1	1	2

Table 2.8: Tidal-stream energy development vulnerability index calculated without incorporating species' conservation importance scores. Vulnerability scores were divided by ten rather than 100 during calculations to avoid unnecessarily small values. Species are ranked by vulnerability score.

Grouping according to vulnerability score	Rank when conservation score not included	Rank when conservation score included-top ten scoring species in red	Species	Displacement: vessels	Drowning risk	Mean and max diving depth	Benthic foraging	Use of tidal races	Foraging range	Habitat use flexibility	Vulnerability score
1	1	1	Razorbill	3	4	5	4	5	3	3	85
2	2	2	Shag	4	4	4	5	5	4	3	80
3	3	3	Black guillemot	3	4	4	4	5	4	4	76
4	4	5	Great cormorant	4	4	4	5	4	4	3	64
5	5	4	Common guillemot	3	4	5	2	4	2	3	56
6	6	7	Atlantic puffin	3	3	4	1	4	1	3	35
7	7	6	Great northern diver	4	4	3	4	3	3	3	32
8	8	10	Little auk	3	3	3	1	4	2	2	26
9	9	8	Red-throated diver	5	4	3	4	2	3	4	24
10	10	9	Black-throated diver	5	4	3	3	2	3	4	23
11	11	11	Slavonian grebe	3	4	2	4	2	4	4	15
	11	19	Great-crested grebe	3	4	2	4	2	4	4	15
12	13	13	Common scoter	5	4	3	5	1	3	4	13
13	14	16	Velvet scoter	4	4	3	5	1	4	3	12
	14	22	Long-tailed duck	3	4	3	5	1	4	4	12
14	16	13	Common eider	3	4	3	5	1	3	4	11
	16	12	Arctic tern	2	1	1	1	5	4	3	11
15	18	13	Manx shearwater	1	1	3	1	3	1	1	9
	18	19	Sooty shearwater	1	1	3	1	3	1	1	9
	18	19	Common goldeneye	4	4	2	5	1	5	4	9
	18	22	Greater scaup	4	4	2	5	1	5	4	9
16	22	17	Sandwich tern	2	1	2	1	2	3	3	8
17	23	25	Common gull	2	1	1	1	4	3	2	7
	23	17	Northern gannet	1	2	3	1	2	1	1	7
	23	22	Roseate tern	2	1	1	1	3	4	3	7
	23	25	Common tern	2	1	1	1	3	4	3	7
18	27	25	Black-legged kittiwake	2	1	1	1	4	2	2	6
	27	28	Great black-backed gull	1	1	1	1	4	2	2	6

19	29	32	White-tailed eagle	2	1	1	1	3	3	2	5
	29	32	Black-headed gull	2	1	1	1	3	3	2	5
	29	29	Little tern	2	1	1	1	2	5	4	5
20	32	29	Great skua	1	1	1	1	3	2	2	4
	32	29	Herring gull	1	1	1	2	3	2	1	4
	32	32	Arctic skua	1	1	1	1	3	2	2	4
	32	32	Lesser black-backed gull	1	1	1	1	3	2	1	4
21	36	36	European storm-petrel	1	1	1	1	3	1	1	3
	36	36	Leach's storm-petrel	1	1	1	1	3	1	1	3
	36	36	Northern fulmar	1	1	1	1	3	1	1	3

Table 2.9: Wave energy development vulnerability index calculated without incorporating species' conservation importance scores. Species are ranked by vulnerability score.

Grouping according to vulnerability score	Rank when conservation score not included	Rank when conservation score included-top ten scoring species in red	Species	Displacement: structures	Displacement: vessels/helicopter	Risk of collision	Exclusion from foraging habitat	Benefit from roost platform	Benefit from FAD or biofouling	Habitat use flexibility	Vulnerability score
1	1	1	Black-throated diver	5	5	3	3	0	0	4	20
	1	1	Red-throated diver	5	5	3	3	0	0	4	20
2	3	3	Great northern diver	5	4	3	3	0	-1	3	17
	3	6	Common scoter	5	5	2	2	0	-1	4	17
3	5	11	Greater scaup	4	4	2	2	0	0	4	16
4	6	10	Common goldeneye	4	4	2	2	0	-1	4	15
	6	12	Velvet scoter	5	4	2	2	0	-1	3	15
5	8	4	Razorbill	4	3	3	1	0	0	3	14
	8	8	Black guillemot	3	3	3	3	-1	-1	4	14
	8	8	Slavonian grebe	3	3	2	2	0	0	4	14
	8	21	Long-tailed duck	4	3	2	2	0	-1	4	14
	8	25	Great-crested grebe	3	3	2	2	0	0	4	14
6	13	5	Common guillemot	4	3	3	1	0	-1	3	13
7	14	7	Atlantic puffin	3	3	2	1	0	0	3	12
	14	13	Common eider	3	3	2	2	-1	-1	4	12
8	16	17	Little tern	2	2	2	2	-1	0	4	11
	16	23	Little auk	3	3	2	1	0	0	2	11
9	18	16	Shag	1	4	3	3	-2	-2	3	10
10	19	14	Arctic tern	2	2	2	2	-1	-1	3	9
	19	14	Northern gannet	4	1	2	1	0	0	1	9
	19	18	Roseate tern	2	2	2	2	-1	-1	3	9
	19	18	Sandwich tern	2	2	2	2	-1	-1	3	9
	19	20	Common tern	2	2	2	2	-1	-1	3	9
	19	23	Great cormorant	1	4	3	2	-2	-2	3	9
11	25	27	White-tailed eagle	1	2	1	2	-1	0	2	7

12	26	22	Manx shearwater	1	1	2	1	0	0	1	6
	26	26	Great skua	1	1	1	1	0	0	2	6
	26	27	Black-legged kittiwake	2	2	1	1	-1	-1	2	6
	26	27	Arctic skua	1	1	1	1	0	0	2	6
	26	30	Common gull	2	2	1	1	-1	-1	2	6
	26	32	Black-headed gull	2	2	1	1	-1	-1	2	6
	26	32	Sooty shearwater	1	1	2	1	0	0	1	6
13	33	31	Great black-backed gull	2	1	1	1	-1	-1	2	5
14	34	34	European storm-petrel	1	1	1	1	0	-1	1	4
	34	35	Lesser black-backed gull	2	1	1	1	-1	-1	1	4
	34	35	Leach's storm-petrel	1	1	1	1	0	-1	1	4
	34	35	Northern fulmar	1	1	1	1	-1	0	1	4
15	38	38	Herring gull	2	1	1	1	-1	-2	1	3

2.3.2 Confidence indices

2.3.2.1 Percentage of flight at altitudes overlapping with turbine blades

The confidence index for data included in the vulnerability factor ‘percentage of flight at altitudes overlapping with turbine blades’ indicates that there is high confidence in data for 6 of the 38 species assessed (Table 2.10). This includes 4 of the top ten species most likely to be at risk of collision with offshore wind turbines (great black-backed gull, sandwich tern, herring gull and black-kittiwake: Table 2.2). There is a moderate level of confidence in data for 11 of the species assessed (Table 2.10), which includes 4 of the top ten species likely to be at risk of collision with turbines (great cormorant, lesser black-backed gull, arctic skua and common gull: Table 2.2). Of the remaining species there is low confidence in 4 of the species assessed, uncertain confidence in 8 species, and poor confidence in 9 species (Table 2.10). The confidence in the data for the remaining two species in the top ten identified as likely to be at risk of collision are classed as low (great skua) and uncertain (white-tailed eagle) (Table 2.10).

Table 2.10: Confidence index for the vulnerability factor 'percentage of flight at altitudes overlapping with turbine blades', which was used to calculate the vulnerability index of species vulnerability to collision with offshore wind turbines.

Species	Species score	Study type score	Number of sites score	Number of studies score	Mean number of years data collection	Confidence score	Confidence category
Arctic skua	3	35	3	3	2	46	Moderate
Arctic tern	3	28	3	3	1.5	38.5	Low
Atlantic puffin	3	20	3	4	2	32	Low
Black guillemot	3	2	1	1	1	8	Poor
Black-headed gull	3	33	3	4	3.5	46.5	Moderate
Black-legged kittiwake	3	56	5	5	2.5	71.5	High
Black-throated diver	3	18	2	2	2	27	Uncertain
Common eider	3	39	3	3	2.5	50.5	Moderate
Common goldeneye	3	2	1	1	1	8	Poor
Common guillemot	3	43	4	5	2.5	57.5	Moderate
Common gull	3	36	4	4	2.5	49.5	Moderate
Common scoter	3	46	4	4	2.5	59.5	Moderate
Common tern	3	49	4	5	2	63	High
European shag	3	20	2	2	2	29	Uncertain
European storm-petrel	3	4	1	1	2	11	Poor
Great black-backed gull	3	49	4	5	2.5	63.5	High
Great cormorant	3	38	3	3	3	50	Moderate
Great northern diver	3	13	2	2	1.5	21.5	Uncertain
Great skua	3	27	3	3	2.5	38.5	Low
Great-crested grebe	3	15	2	2	2.5	24.5	Uncertain
Greater scaup	3	5	1	1	4.5	14.5	Poor
Herring gull	3	54	4	5	2.5	68.5	High
Leach's storm-petrel	3	4	1	1	2	11	Poor
Lesser black-backed gull	3	45	4	5	2	59	Moderate
Little auk	3	10	2	2	1.5	18.5	Uncertain
Little tern	3	3	1	1	2	10	Poor
Long-tailed duck	3	6	1	1	1	12	Poor
Manx shearwater	3	24	3	3	3	36	Low
Northern fulmar	3	32	4	5	2	46	Moderate
Northern gannet	3	57	5	5	2.5	72.5	High
Razorbill	3	42	4	4	0.5	53.5	Moderate
Red-throated diver	3	43	4	4	2.5	56.5	Moderate
Roseate tern	3	4	1	1	1	10	Poor
Sandwich tern	3	47	4	4	2.5	60.5	High
Slavonian grebe	2	5	1	1	1	10	Poor
Sooty shearwater	3	10	2	2	2	19	Uncertain
Velvet scoter	3	14	2	2	1.5	22.5	Uncertain
White-tailed eagle	3	8	1	1	3	16	Uncertain

2.3.2.2 Displacement and disturbance as a result of renewable energy structures

Most of the data contributing to this factor originates from research investigating marine bird responses to offshore wind turbines, with a minority of data derived from research related to wave and tidal-stream energy devices (Appendix 1, Table 3). The confidence index for data included in the vulnerability factor that ranks species sensitivity to displacement and disturbance as a result of renewable energy structures indicates that there is high confidence in data for 9 of the 38 species assessed (Table 2.11). This includes 3 of the top ten species likely to be at risk of displacement from offshore wind farms and wave energy developments (razorbill, common guillemot and common scoter: Tables 2.3 and 2.5). There is a moderate level of confidence in data for 7 of the species assessed (Table 2.11), of which 2 are in the top ten species likely to be at risk of displacement from offshore wind farms and wave energy developments (red-throated diver and black-throated diver: Tables 2.3 and 2.5). A low level of confidence was indicated for 6 species (Table 2.11), of which 2 are listed in the top ten species likely to be at risk of displacement from offshore wind farms and wave energy developments (common eider and Atlantic puffin: Tables 2.3 and 2.5). A level of uncertainty is associated with 13 species assessed, which represents the greatest number of species in each confidence level category (Table 2.11). Of these 13 species, 3 are in the top ten species likely to be at risk of displacement from offshore wind farms and wave energy developments (Slavonian grebe, great northern diver and black guillemot: Tables 2.3 and 2.5). There is a poor level of confidence in the data for the remaining 3 species assessed (Table 2.11), of which all are ranked in the top ten species likely to be at risk of displacement from offshore wind farms and wave energy developments (greater scaup, common goldeneye and velvet scoter: Table 2.3 and 2.5).

Table 2.11: Confidence index for the vulnerability factor 'disturbance and displacement by structures'. This factor was included in the calculations for the two vulnerability indices assessing disturbance and displacement caused by offshore wind farms and species vulnerability to wave energy developments.

Species	Species score	Study type score	Number of sites score	Number of studies score	Mean number of years data collection	Confidence score	Confidence category
Arctic skua	3	4	1	2	2.5	12.5	Uncertain
Arctic tern	2	18	3	3	3.5	29.5	Low
Atlantic puffin	2	15	3	3	4	27	Low
Black guillemot	3	9	2	2	3	19	Uncertain
Black-headed gull	3	16	2	3	3.5	27.5	Low
Black-legged kittiwake	3	27	4	4	4.5	42.5	High
Black-throated diver	3	17	3	3	5.5	31.5	Moderate
Common eider	3	11	3	3	2.5	22.5	Low
Common goldeneye	3	1	1	1	1	7	Poor
Common guillemot	3	30	5	5	4.5	47.5	High
Common gull	3	20	3	3	4.5	33.5	Moderate
Common scoter	3	26	3	4	5.5	41.5	High
Common tern	3	23	4	4	4.5	38.5	Moderate
European shag	3	20	3	3	3.5	32.5	Moderate
European storm-petrel	2	4	2	2	1.5	11.5	Uncertain
Great black-backed gull	3	32	4	4	4.5	47.5	High
Great cormorant	3	27	3	4	5	42	High
Great northern diver	3	9	2	2	3	19	Uncertain
Great skua	3	5	2	2	4.5	16.5	Uncertain
Great-crested grebe	3	6	2	2	4	17	Uncertain
Greater scaup	3	2	2	2	1	10	Poor
Herring gull	3	32	4	4	4.5	47.5	High
Leach's storm-petrel	2	4	2	2	1.5	11.5	Uncertain
Lesser black-backed gull	3	28	4	4	5	44	High
Little auk	2	18	3	3	4.5	30.5	Moderate
Little tern	2	16	3	3	3.5	27.5	Low
Long-tailed duck	3	4	2	2	9	20	Uncertain
Manx shearwater	2	4	2	2	1.5	11.5	Uncertain
Northern fulmar	3	7	2	2	3.5	17.5	Uncertain
Northern gannet	3	31	4	4	5.5	47.5	High
Razorbill	3	28	4	4	4.5	43.5	High
Red-throated diver	3	22	3	4	6	38	Moderate
Roseate tern	2	11	2	2	3.5	20.5	Low
Sandwich tern	3	25	3	4	4.5	39.5	Moderate
Slavonian grebe	2	6	2	2	4	16	Uncertain
Sooty shearwater	2	4	2	2	1.5	11.5	Uncertain

Velvet scoter	3	1	1	1	1	7	Poor
White-tailed eagle	3	8	1	2	6	20	Uncertain

2.3.2.3 Displacement and disturbance as a result of vessel and/or helicopter traffic

The confidence index for data included in the vulnerability factor ranking species sensitivity to displacement and disturbance as a result of vessel and/or helicopter traffic indicates only low, uncertain and poor confidence in data for all of the 38 species assessed (Table 2.12). There is low confidence in data for 6 of the species assessed (Table 2.12); all of which were ranked in the top ten species likely to be at risk of displacement or disturbance as a result of vessel and/or helicopter traffic associated with offshore wind, wave or tidal-stream energy developments (black guillemot, little auk, Atlantic puffin, common scoter, common guillemot and razorbill: Tables 2.3, 2.4 and 2.5). The majority of species assessed (27 of 38) have an uncertain level of confidence in estimations of how they may react to vessels and/or helicopter traffic (Table 2.12). These include 8 of the top ten ranked species (common goldeneye, Slavonian grebe, velvet scoter, black-throated diver, common eider, European shag, great cormorant and red-throated diver: Tables 2.3, 2.4 and 2.5). There is a poor level of confidence in data for the remaining 5 species assessed (Table 2.12), of which 2 species are ranked in the top ten species likely to be at risk of displacement or disturbance as a result of vessel and/or helicopter traffic in offshore wind, wave or tidal-stream energy developments (greater scaup and great northern diver: Tables 2.3, 2.4 and 2.5).

Table 2.12: Confidence index for the vulnerability factor 'disturbance and displacement by vessel and/or helicopter traffic'. This factor was included in the calculations for three of the four vulnerability indices, including assessing disturbance and displacement caused by offshore wind farms , species vulnerability to wave energy developments and species vulnerability to tidal-stream energy developments.

Species	Species score	Study type score	Number of sites score	Number of studies score	Mean number of years data collection	Confidence score	Confidence category
Arctic skua	3	2	1	1	1	8	Poor
Arctic tern	3	4	2	2	1	12	Uncertain
Atlantic puffin	2	16	4	4	3	29	Low
Black guillemot	2	16	4	4	3	29	Low
Black-headed gull	2	6	2	2	2	14	Uncertain
Black-legged kittiwake	2	6	2	2	2	14	Uncertain
Black-throated diver	3	5	2	2	2	14	Uncertain
Common eider	3	9	2	2	1	17	Uncertain
Common goldeneye	3	3	2	2	1	11	Uncertain
Common guillemot	2	16	4	4	3	29	Low
Common gull	2	6	2	2	2	14	Uncertain
Common scoter	3	17	3	3	3	29	Low
Common tern	3	4	2	2	3	14	Uncertain
European storm-petrel	3	3	2	2	1	11	Uncertain
European shag	3	4	2	2	1	12	Uncertain
Great black-backed gull	3	8	2	2	2	17	Uncertain
Great cormorant	3	4	2	2	3	14	Uncertain
Great northern diver	3	1	1	1	1	7	Poor
Great skua	3	4	2	2	2	13	Uncertain
Great-crested grebe	3	4	2	2	1	12	Uncertain
Greater scaup	3	2	2	2	1	10	Poor
Herring gull	3	8	2	2	1.5	16.5	Uncertain
Leach's storm-petrel	3	3	2	2	1	11	Uncertain
Lesser black-backed gull	3	8	2	2	2	17	Uncertain
Little auk	2	12	3	3	3	23	Low
Little tern	3	1	1	1	1	7	Poor
Long-tailed duck	3	6	1	2	1	13	Uncertain
Manx shearwater	3	3	2	2	1	11	Uncertain
Northern fulmar	3	6	2	2	1.5	14.5	Uncertain
Northern gannet	3	6	2	2	1.5	14.5	Uncertain
Razorbill	2	16	4	4	3	29	Low
Red-throated diver	3	5	2	2	2	14	Uncertain
Roseate tern	2	2	2	1	3	10	Poor
Sandwich tern	3	4	2	2	3	14	Uncertain

Slavonian grebe	2	5	2	2	1	12	Uncertain
Sooty shearwater	3	3	2	2	1	11	Uncertain
Velvet scoter	3	8	2	2	1	16	Uncertain
White-tailed eagle	3	2	2	2	3	12	Uncertain

2.3.2.4 Use of tidal races

The confidence index for data included in the vulnerability factor ranking species' use of tidal races indicates a lack of high level confidence in data for all of the 38 species assessed (Table 2.13). There is a moderate level of confidence in data for two of the species (Table 2.13); both of which are in the top ten species likely to be at vulnerable to tidal-stream energy developments (black guillemot and European shag: Table 2.4). A low level of confidence is indicated for six of the species assessed (Table 2.13), of which all are ranked in the top ten species vulnerable to tidal-stream energy developments (great northern diver, black-throated diver, Atlantic puffin, red-throated diver, common guillemot and razorbill: Table 2.4). There is an uncertain level of confidence in estimations of tidal race use for 11 species (Table 2.13). This includes one of the top ten species ranked as potentially vulnerable to tidal-stream energy developments (great cormorant: Table 2.4). Finally, there is a poor level of confidence in the majority of species assessed (19 of 38: Table 2.13), which includes one of the top ten species likely to be vulnerable to tidal-stream energy developments (little auk: Table 2.4).

Table 2.13: Confidence index for the vulnerability factor 'use of tidal races'. This factor was included in the calculations for the vulnerability index assessing species vulnerability to tidal-stream energy developments.

Species	Species score	Study type score	Number of sites score	Number of studies score	Mean number of years data collection	Confidence score	Confidence category
Arctic skua	1	0	1	1	0	3	Poor
Arctic tern	3	3	1	1	4	12	Uncertain
Atlantic puffin	3	13	5	3	2	26	Low
Black guillemot	3	22	4	4	2	35	Moderate
Black-headed gull	1	0	1	1	0	3	Poor
Black-legged kittiwake	3	5	2	2	1	13	Uncertain
Black-throated diver	2	14	3	3	2.5	24.5	Low
Common eider	3	9	3	2	3	20	Uncertain
Common goldeneye	2	4	2	2	1	11	Uncertain
Common guillemot	3	13	4	3	2	25	Low
Common gull	3	1	1	1	1	7	Poor
Common scoter	3	4	2	2	2	13	Uncertain
Common tern	3	6	2	2	3	16	Uncertain
European shag	3	21	4	4	2	34	Moderate
European storm-petrel	1	0	1	1	0	3	Poor
Great black-backed gull	1	0	1	1	0	3	Poor
Great cormorant	3	6	2	2	4	17	Uncertain
Great northern diver	3	16	3	3	2.5	27.5	Low
Great skua	1	0	1	1	0	3	Poor
Great-crested grebe	2	2	2	1	1	8	Poor
Greater scaup	3	4	2	2	1	12	Uncertain
Herring gull	1	0	1	1	0	3	Poor
Leach's storm-petrel	1	0	1	1	0	3	Poor
Lesser black-backed gull	1	0	1	1	0	3	Poor
Little auk	2	2	2	1	1	8	Poor
Little tern	3	6	4	2	2.5	17.5	Uncertain
Long-tailed duck	3	2	2	1	1	9	Poor
Manx shearwater	2	1	1	1	2	7	Poor
Northern fulmar	1	0	1	1	0	3	Poor
Northern gannet	3	4	2	2	5	16	Uncertain
Razorbill	3	10	3	3	2.5	21.5	Low
Red-throated diver	2	14	3	3	2.5	24.5	Low
Roseate tern	3	4	2	2	4	15	Uncertain
Sandwich tern	2	2	1	1	4	10	Poor
Slavonian grebe	2	2	2	1	1	8	Poor
Sooty shearwater	3	1	1	1	2	8	Poor
Velvet scoter	3	2	2	1	1	9	Poor
White-tailed eagle	1	0	1	1	0	3	Poor

2.3.2.5 Overall species confidence

An overall assessment of confidence in each of the species included in the four vulnerability factors indicates that over half of the species assessed (21 of 38) score ≥10 out of a maximum combined confidence score of 20 (Table 2.14). Of these 21 species, 17 are in the top ten of at least one vulnerability index (Tables 2.2 – 2.5). Of the 17 species that score <10 (Table 2.14), eight species feature in the top ten of at least one vulnerability index (Tables 2.2 – 2.5), and include greater scaup, common goldeneye, Slavonian grebe, velvet scoter, white-tailed eagle, Arctic skua, great northern diver and great skua.

Table 2.14: Overall level of confidence inherent in data for each species.

	Vulnerability factor								
	Flight altitude		Structures		Vessels and/or helicopters		Tidal races		
Species	Confidence category	Confidence score	Confidence category	Confidence score	Confidence category	Confidence score	Confidence category	Confidence score	Combined confidence score (max 20)
Common guillemot	Moderate	4	High	5	Low	3	Low	3	15
Razorbill	Moderate	4	High	5	Low	3	Low	3	15
Black-legged kittiwake	High	5	High	5	Uncertain	2	Uncertain	2	14
Common scoter	Moderate	4	High	5	Low	3	Uncertain	2	14
Northern gannet	High	5	High	5	Uncertain	2	Uncertain	2	14
Common tern	High	5	Moderate	4	Uncertain	2	Uncertain	2	13
Great black-backed gull	High	5	High	5	Uncertain	2	Poor	1	13
Great cormorant	Moderate	4	High	5	Uncertain	2	Uncertain	2	13
Herring gull	High	5	High	5	Uncertain	2	Poor	1	13
Red-throated diver	Moderate	4	Moderate	4	Uncertain	2	Low	3	13
Atlantic puffin	Low	3	Low	3	Low	3	Low	3	12
Lesser black-backed gull	Moderate	4	High	5	Uncertain	2	Poor	1	12
Sandwich tern	High	5	Moderate	4	Uncertain	2	Poor	1	12
European shag	Uncertain	2	Moderate	4	Uncertain	2	Moderate	4	12
Black-throated diver	Uncertain	2	Moderate	4	Uncertain	2	Low	3	11
Common eider	Moderate	4	Low	3	Uncertain	2	Uncertain	2	11
Common gull	Moderate	4	Moderate	4	Uncertain	2	Poor	1	11
Arctic tern	Low	3	Low	3	Uncertain	2	Uncertain	2	10
Black guillemot	Poor	1	Uncertain	2	Low	3	Moderate	4	10
Black-headed gull	Moderate	4	Low	3	Uncertain	2	Poor	1	10

Little auk	Uncertain	2	Moderate	4	Low	3	Poor	1	10
Northern fulmar	Moderate	4	Uncertain	2	Uncertain	2	Poor	1	9
Arctic skua	Moderate	4	Uncertain	2	Poor	1	Poor	1	8
Great northern diver	Uncertain	2	Uncertain	2	Poor	1	Low	3	8
Great skua	Low	3	Uncertain	2	Uncertain	2	Poor	1	8
Manx shearwater	Low	3	Uncertain	2	Uncertain	2	Poor	1	8
Great-crested grebe	Uncertain	2	Uncertain	2	Uncertain	2	Poor	1	7
Little tern	Poor	1	Low	3	Poor	1	Uncertain	2	7
Roseate tern	Poor	1	Low	3	Poor	1	Uncertain	2	7
Sooty shearwater	Uncertain	2	Uncertain	2	Uncertain	2	Poor	1	7
White-tailed eagle	Uncertain	2	Uncertain	2	Uncertain	2	Poor	1	7
Common goldeneye	Poor	1	Poor	1	Uncertain	2	Uncertain	2	6
European storm-petrel	Poor	1	Uncertain	2	Uncertain	2	Poor	1	6
Leach's storm-petrel	Poor	1	Uncertain	2	Uncertain	2	Poor	1	6
Long-tailed duck	Poor	1	Uncertain	2	Uncertain	2	Poor	1	6
Slavonian grebe	Poor	1	Uncertain	2	Uncertain	2	Poor	1	6
Velvet scoter	Uncertain	2	Poor	1	Uncertain	2	Poor	1	6
Greater scaup	Poor	1	Poor	1	Poor	1	Uncertain	2	5

2.4 Discussion

In this chapter I have updated the four vulnerability indices published by Furness *et al.* (2012, 2013) by taking into account newly available literature detailing how marine birds may be affected by MREDs. I have greater than doubled the number of sources of information included in the Furness *et al.* (2012, 2013) papers; from 45 sources to 93. Several of these sources were newly available environmental statements and environmental impact assessment reports, in addition to peer-reviewed studies directly investigating potential impacts of MREDs on seabirds. I have also assessed confidence levels associated with data for each species in four vulnerability factors that contributed to the four overall vulnerability indices. I found that despite including information from twice the number of sources referred to in Furness *et al.* (2012, 2013), the newly available literature altered the rankings of only a few species, and overall species rankings remained broadly the same. There appears to be a greater overall confidence in data on how marine birds may react to MRED structures and for what percentage of flight overlaps with turbine blade altitudes.

However, within these vulnerability factors, there were a range of confidence levels among species, with some species lacking in data. There was a lower overall confidence in data referring to how marine birds may react to vessel and/or helicopter traffic associated with MREDs, and how marine bird species use tidal races. In these two vulnerability factors, no species were ranked at high confidence levels but the majority were ranked in uncertain or poor data confidence categories. By allocating these confidence levels, knowledge gaps can be established and areas where caution in interpreting results can be identified.

2.4.1 Vulnerability indices

The vulnerability indices in this chapter have identified white-tailed eagle, lesser black-backed gull, herring gull, great black-backed gull and common gull as the species most likely to be vulnerable to collision with offshore wind turbines. These are broadly the same as the species identified by Furness *et al.* (2013) (Table 2.2) and behaviour common to these species drives their high rankings. For example, all of the highest ranking species spend a high percentage of their time in flight at altitudes overlapping with turbine blades; with at least a quarter of time spent at altitudes where collision could occur. In addition, all of the top five ranked species appear unlikely to be displaced or disturbed by wind farm structures, and four of the five are known to sometimes fly at night (Table 2.2). Great cormorant showed a marked increase in vulnerability to collision with wind turbines in this chapter. This was driven by the species' apparent attraction to wind farm areas and their habit of resting on turbine structures (Petersen *et al.* 2006; Leopold *et al.* 2011; Walls *et al.* 2013). Conversely, northern gannet reduced in vulnerability to collision. This change was driven by the inclusion of the 'displacement caused by structures' vulnerability factor when calculating the collision risk index, as gannets exhibit relatively strong avoidance of wind farm areas.

Adjusting the collision risk calculation to include this factor was based on the assumption that species prone to avoiding wind farm areas are less likely to collide with devices.

Rankings in the vulnerability index assessing displacement and disturbance caused by offshore wind farms identifies black-throated diver, red-throated diver, great northern diver, common scoter, common goldeneye, common guillemot and razorbill as species' most likely to be vulnerable to this threat. These top scoring species match those identified in Furness *et al.* (2013) (Table 2.3), which confirms their strong sensitivity to the risk of displacement and disturbance even in light of newly available data. Common guillemot, razorbill and Atlantic puffin showed an increase in vulnerability in the revised vulnerability index, as a result of further information confirming that auk species show relatively high levels of displacement from wind farm areas (Walls *et al.* 2013). However, Vanermen *et al.* (2013) suggest that despite generally avoiding wind farm areas, auk species may be influenced by a lagged-prey effect, with some birds returning to wind farm sites to forage on increased abundances of prey caused by FAD and reef effects. Increase in food abundance has been reported as driving common scoter return to the Horns Rev wind farm following avoidance of the area for 5 years following construction (Petersen & Fox 2007) and has also been observed in cormorant species (NWP Offshore Ltd 2008; Leopold *et al.* 2011). The revised indices indicate a reduced vulnerability level for shag and great cormorant as a result of the species being observed foraging in wind farm areas and their lack of avoidance response to wind farm structures (NWP Offshore Ltd 2008; Leopold *et al.* 2011; Walls *et al.* 2013).

The updated index ranking species' vulnerability to tidal-stream renewable energy developments indicated no change in the top ten species when compared to Furness *et al.* (2012). All species remained at similar levels of vulnerability; with the exception of common

tern that increased in vulnerability. However, common tern remains at a low overall level of vulnerability to tidal-stream developments with the change likely to have been driven by a slight increase in the score regarding the species' use of tidal races.

The vulnerability index assessing marine bird vulnerability to wave energy developments also ranks the top five species in similar positions to the index in Furness *et al.* (2012). However, Atlantic puffin and common goldeneye slightly increased in vulnerability in the revised index. This was likely caused by increasing evidence that auk species exhibit avoidance responses to structures and tend to be disturbed by vessel traffic. Although overall, seaducks and divers tend to be more disturbed by vessel traffic than auks, auks are more disturbed by vessel movements than gull species or species known to associate with fishing vessels; such as gannets and fulmars. Little tern vulnerability to wave energy developments reduced in the revised index. This was caused by evidence suggesting tern species may be less displaced by wave devices than previously expected, with research on the Pelamis wave energy device indicating that Arctic terns often rested on the structure (Jackson 2014). This was also the case for fulmar, which although was never at high risk from wave energy developments, declined in vulnerability following observations indicating occasional use of Pelamis for resting (Jackson 2014). Shag also showed reduced levels of vulnerability to wave energy developments, which was a result of a reduced displacement score based on evidence that shags forage near to structures in offshore wind farm sites (Petersen *et al.* 2006).

The changes in vulnerability to offshore wind, wave and tidal-stream energy developments are a result of species-specific variability in behaviour. It is recommended that species identified as highly vulnerable should be the focus for further research and monitoring in

order to further understand and clarify the effects of MREDs on marine birds in Scottish waters. Although little overall change was observed in species' vulnerability levels in the revised and recalculated vulnerability indices, I recommend periodic updates to the information the calculations are based on. This will consider changes in species' behaviour that may be observed over longer time periods; for example, as has been observed in common scoter response to wind farms.

There was little change to the vulnerability rankings of species when conservation importance scores were not included in calculating overall vulnerability scores. Long-tailed duck and great-crested grebes increased in vulnerability to wave energy developments and disturbance and/or displacement caused by offshore wind developments. This change in vulnerability was caused by removing the influence of a low conservation importance score in calculating overall vulnerability.

2.4.2 Confidence indices

The four vulnerability factors updated to quantify confidence levels in the data included were chosen as they are deemed to be of high importance in determining marine bird vulnerability to MREDs. Diving depth has also been identified as highly important in determining species' vulnerability to collision with tidal-stream energy devices but is not updated in this chapter as a comprehensive review investigating confidence in dive depth data is currently being undertaken by Robbins et al. (in prep. Examples of species being covered in the review are presented in Robbins *et al.* 2014b).

Of the vulnerability factors assessed, I found a greater overall confidence in information on how marine birds may be displaced or disturbed by renewable energy structures and

information on the percentage of flight spent at altitudes overlapping with turbine blades. There is lower confidence in data detailing how species may be displaced or disturbed by vessel and/or helicopter traffic, and data regarding how marine birds use tidal races. The differing levels of confidence among these factors is likely driven by priorities identified in MRED monitoring programmes and the more advanced stage of monitoring at offshore wind farms, compared to wave and tidal-stream energy developments (of which few have been constructed: Witt *et al.* 2012). For example, measures of flight altitudes essential for collision risk modelling and estimates of displacement based on bird density changes are described in most offshore wind environmental reports, which are driven by EIA requirements. However, more studies collecting reliable and accurate estimates of bird flight altitudes are required (for example by using radar measurements), as observer error can bias altitude data, particularly before structures of a fixed reference height are constructed. Owing to the long-term, fixed nature of renewable energy devices when compared to vessel and/or helicopter traffic, displacement by structures is deemed a greater risk to marine bird species, and as such is prioritised for monitoring. Low levels of data detailing disturbance and displacement caused by vessel and/or helicopter traffic may be a result of the greater ease with which displacement can be measured as a result of fixed structures compared to establishing displacement caused by transient vessel and helicopter traffic. The majority of information describing disturbance and displacement of marine birds by vessels originates from a low number of peer-reviewed studies, designed specifically to assess vessel disturbance and associated flush responses, and which are not related to the renewable energy industry (Ronconi & Clair 2002; Bellefleur *et al.* 2009; Schwemmer *et al.* 2011). Instead, data regarding displacement and disturbance by vessel and/or helicopter traffic collected at MRED sites tends to be anecdotal observations; collected and recorded in an unstructured way. This type of data may not be considered a priority at offshore wind

farms sited relatively far from shore but disturbance and displacement caused by vessel traffic may prove to be of greater impact in shallow, coastal waters, where wave and tidal-stream developments are to be located. This is because coastal marine bird species can be more restricted in their foraging range and specific in the foraging habitat they require (e.g. diver and grebe species, common eider), which could make them more vulnerable to displacement and disturbance if developments are located close to shore. As such, disturbance and displacement caused by vessel traffic and/or helicopter traffic is an area that would benefit from increased research and focused monitoring, particularly in near shore development sites. For example, it could be particularly useful to assess whether designating vessel pathways for MRED traffic is effective in mitigating against marine bird disturbance.

The lower level of confidence in data contributing to our understanding of how marine bird species use tidal races is also an area recommended for further research as this factor is integral in understanding which species may be most vulnerable to collision with underwater energy devices. Tidal-stream environmental reports tend to describe the presence and abundance of species observed in development areas. However, they do not identify associations between species and behaviour, and current velocities or visible surface features generated by current flows. This type of information is fundamental to identifying spatial overlap between marine birds and tidal-stream energy generating devices. Only species that regularly dive in current speeds required for tidal-stream energy generation are likely to be at risk of collision with devices; as such, species' presence in proximity to tidal races does not necessarily indicate overlap with device location. Also of importance is establishing which species target visible surface features generated by the dynamic current flows associated with tidal races (such as upwellings and shearlines) for foraging, as these

species could be vulnerable to changes in current regimes caused by devices (McCluskie *et al.* 2013). A core challenge to understanding marine bird use of tidal races and fast current flows is the dynamic nature of this habitat, which makes it difficult to achieve the ideal situation of monitoring marine birds whilst collecting contemporaneous current velocity data. Where this cannot be achieved, bird observation locations can be combined with hydrodynamic current flow models calibrated with acoustic Doppler current profile (ADCP) data to provide an indication of current speeds experienced by foraging birds. However, fine-scale (10s to 100s m) differences in velocity that marine birds may be responding to; for example, current velocity gradients across shearlines and upwellings, are not resolved in these models. Furthermore, the accuracy of resolution around complex coastlines is not robust, which is problematic if applied to near shore observations. Alternatively, new technology that maps the sea surface and identifies current flow direction and magnitude using radar could be a viable alternative to assess seabird association with current speeds and visible surface features. Although, this method also reports limitations in terms of resolving current velocities and direction around complex coastlines (Bell *et al.* 2014). Monitoring once devices are in place (for example, using cameras mounted on devices) will also be challenging given the poor underwater visibility associated with tidal races. Potentially novel methods (Bell *et al.* 2014; Williamson *et al.* 2014) and an emphasis on improving our knowledge of which species dive in current speeds appropriate for tidal-stream energy generation should be a focus for researchers and developers. Likewise, an improved understanding of marine bird association with fine-scale surface features generated by tidal races is required.

Despite a greater level of confidence in data related to displacement and disturbance as a result of MRED structures and in the percentage of overlap of flight altitudes with turbine

blades, low to poor confidence levels were still indicated for several species; some of which were ranked in the top ten species of at least one vulnerability index. In the case of levels of confidence in flight altitude data, data for great skua and white-tailed eagle are considered to be at low and uncertain levels respectively; despite these species being identified as in the top ten of species ranked as vulnerable to collision with offshore wind turbines. Additionally, several of the top ten marine bird species identified as vulnerable to wave energy developments and to disturbance and displacement as a result of offshore wind farms were classified as having low, uncertain or poor data quality associated with measures of disturbance and displacement caused by structures. These species include greater scaup, common goldeneye, velvet scoter, Slavonian grebe, great northern diver, black guillemot, common eider and Atlantic puffin. Furthermore, all of the top ten ranking species identified as vulnerable to wave energy developments, tidal-stream energy developments, and disturbance and displacement as a result of offshore wind farms were ranked as having low, uncertain or poor data quality associated with measures of disturbance and displacement caused by vessel and/or helicopter traffic (Tables 2.3, 2.4, 2.5 and 2.12). Finally, eight of the top ten species ranked as vulnerable to tidal-stream energy developments were classed as having low, uncertain or poor data quality associated with information on use of tidal races. These species include little auk, black-throated diver, Atlantic puffin, great-northern diver, great cormorant, red-throated diver, common guillemot and razorbill. The species highlighted here should the focus for additional monitoring to increase levels of confidence in vulnerability estimates.

Considering the combined confidence scores for each species (Table 2.14), species with overall high levels of data confidence (those that score over half of the maximum possible) are common species and relatively large in size. Intrinsically, these species are easier to

detect and are regularly observed in MRED surveys. Analysis of collision risk and estimates of vulnerability to displacement for these species are often reported in MRED environmental reports owing to the large number of observations usually recorded for these species. On the contrary, 8 of the 25 species indicated as vulnerable in at least one vulnerability index, score less than half of the maximum possible (≤ 10). These species include greater scaup, common goldeneye, Slavonian grebe, velvet scoter, white-tailed eagle, Arctic skua, great northern diver and great skua. There may be several reasons that explain why the combined confidence scores for these species are low. For example, some species may be difficult to detect during surveys owing to their small size (e.g. storm petrels) or may be more active during sea states not compatible with surveying (e.g. shearwaters above sea state 4). As such, observations of these species and their responses to MREDs are likely to be low in number. Furthermore, some species may be relatively uncommon and only rarely recorded during surveys in MRED sites. This could be because a particular species is rare and so is not observed often, or it may be that the species does not occur in areas designated for MREDs; for example, in the case of coastal marine bird species not occurring in a wind farm located far offshore. In the case of species that do not occur in MRED sites, their level of vulnerability to MREDs is likely to be low. However, the level of confidence in data associated with those species will also be low because few observations have been recorded. Alternatively, rare species from small populations may also score low in confidence rankings, owing to few observations being recorded. However, rare species from small populations may be highly vulnerable to potential impacts of MREDs but may not be recognised as such because of a lack of data informing their responses to MREDs. In summary, species that are data poor could either: 1) overlap with MRED sites but not be abundant or 2) be abundant but not overlap with MRED sites. It is important to establish the distinction between these two categories of data poor species in order to ensure that rare

marine birds overlapping with MRED sites are not incorrectly assumed not to be vulnerable to potential impacts and be scoped out of EIA monitoring. I recommend that species low in data confidence are a focus for further monitoring and research in order to identify reasons explaining their low confidence level.

Conclusions

This study has found that newly available literature altered the vulnerability of only a few species, compared to the indices published by Furness *et al.* (2012, 2013), and that overall species' vulnerability levels remained broadly the same. This indicates that the method of incorporating existing knowledge combined with expert moderation to generate vulnerability indices used by Furness *et al.* (2012, 2013) is robust in estimating species vulnerability to MREDs. However, it is still recommended that vulnerability indices be updated regularly as some species may undergo changes in their response to MREDs in the longer term (e.g. as has been observed in common scoter). Overall, there is a greater level of confidence in data regarding disturbance and displacement caused by energy generating structures and in the percentage of flight altitudes that overlap with wind turbine blades than there is for data regarding disturbance and displacement caused by vessel and/or helicopter traffic and marine bird use of tidal races. Factors with relatively low overall confidence levels should be the focus for increased monitoring and research. Additionally, factors with a greater level of overall confidence but with species classified as low to poor data quality should also be a focus for additional monitoring, to address knowledge gaps and to ensure robust estimates of marine bird vulnerability to MREDs. It should also be borne in mind that species' responses to MREDs may alter over time and that results for species with low to poor confidence levels should be treated with caution. The vulnerability indices and categories of confidence associated with each species presented in this chapter are intended

to be used as a guide for developers and regulators in designing EIA monitoring programmes. They should be viewed as a work in progress to be updated as new evidence is made available or validation of predictions are made; for example the number of collisions with turbines is quantified and the impact of disturbance on foraging success is measured. The vulnerability indices should not be a substitute for site-specific monitoring.

Chapter 3:

Great skua *Stercorarius skua* movements at sea in relation to marine renewable energy developments

This chapter has been published as:

Wade, H. M., E. A. Masden, A. C. Jackson, C. B. Thaxter, N. H. K. Burton, W. Bouten, and R. W. Furness. 2014. Great skua (Stercorarius skua) movements at sea in relation to marine renewable energy developments. Marine Environmental Research 101:69–80.

(See Appendix 3 for copies of the published papers)

Abstract

Marine renewable energy developments (MREDs) are an increasing feature of the marine environment. Owing to the relatively small number of existing developments and the early stage of their associated environmental monitoring programmes, the effects of MREDs on seabirds are not fully known. Our ability to fully predict potential effects is limited by a lack of knowledge regarding movements of seabirds at sea. I used GPS tracking to improve our understanding of the movements at sea of a protected seabird species breeding in Scotland, the great skua *Stercorarius skua*, to better predict how this species may be affected by MREDs. I found that the overlap of great skuas with leased and proposed MREDs was low; particularly with offshore wind sites, which are predicted to present a greater risk to great skuas than wave or tidal-stream developments. Failed breeders overlapped with larger areas of MREDs than breeding birds but the overall overlap with core areas used remained low. Overlap with wave energy development sites was greater than for offshore wind and

tidal-stream sites. Comparison of 2011 data with historical data indicates that distances travelled by great skuas have likely increased over recent decades. This suggests that basing marine spatial planning decisions on short-term tracking data could be less informative than longer-term data.

3.1 Introduction

Marine renewable energy developments (MREDs) are an increasing feature of the marine environment. The rapid development of these industries is being driven by the need to reduce carbon emissions and to increase energy security. In Scotland, the Crown Estate and the Scottish Government are working in partnership to realise the Scottish Government's target of generating 100% of electricity from renewable sources by 2020 (Scottish Government 2011b). An advanced marine plan has been developed for Scotland with sites leased to offshore wind, wave and tidal-stream energy developers, with additional sites proposed for future development (Scottish Government 2013b).

Owing to the small number and limited extent of existing MREDs, the effects of offshore wind, wave and tidal-stream installations on seabirds are not fully known (Fox *et al.* 2006; Witt *et al.* 2012; McCluskie *et al.* 2013). This is despite several leased and proposed MRED sites in north-east Scotland being located in close proximity to special protection areas (SPAs) designated specifically to safeguard breeding populations of internationally important seabird species (JNCC 2014a) (Fig. 3.1). SPA legislation (EC Birds Directive 2009/147/EC) requires that any development does not damage the integrity of protected seabird populations (Scottish Natural Heritage 2014). Some of the potential negative effects

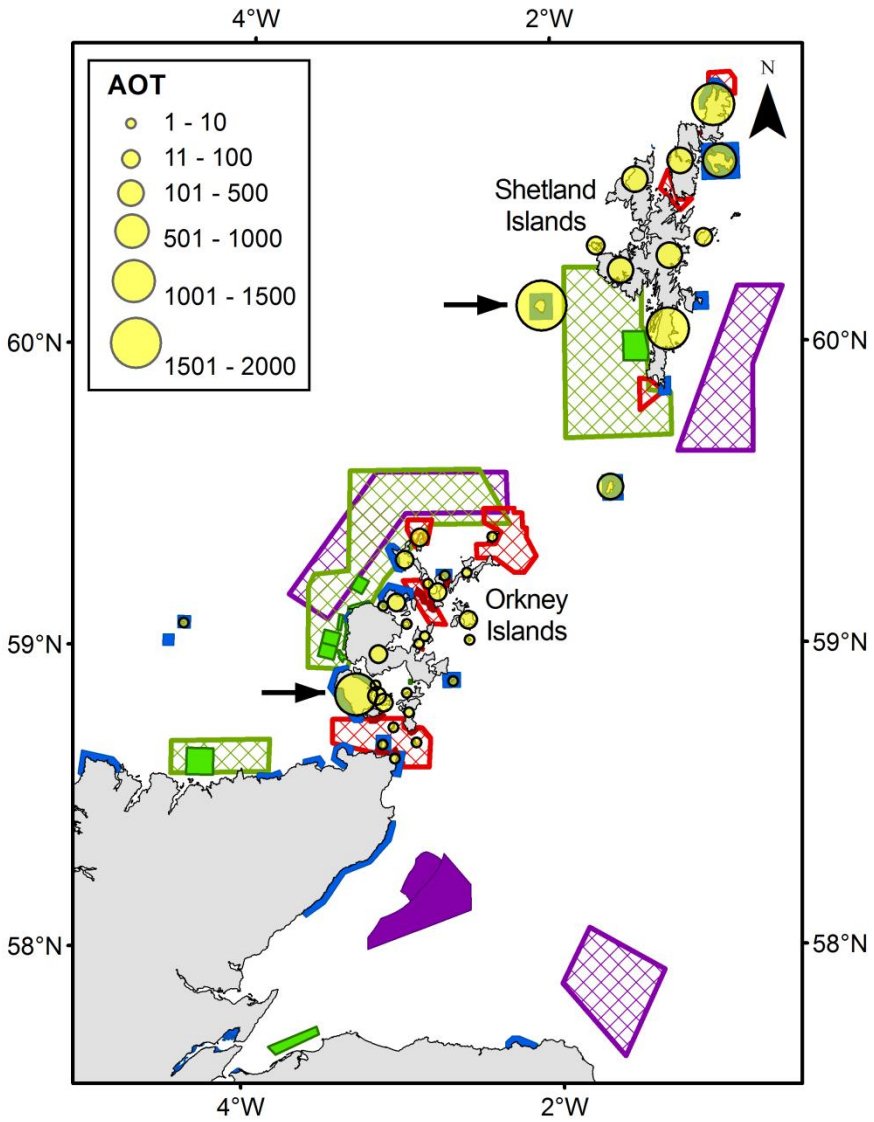

Figure 3.1: Locations of breeding colonies (yellow circles; defined by number of Apparently Occupied Territories (AOT)) and special protection areas (blue polygons) designated for breeding seabird populations in north-east Scotland, and the proximity of leased and proposed marine renewable energy development sites (solid polygons = leased sites; cross-hatched = proposed sites; purple = offshore wind sites; green = wave sites; red = tidal-stream sites). Study colonies indicated with black arrows. See Supplementary Table 1, Appendix 1 for colony sizes and year of AOT count.

of MREDs on seabirds include collision with devices (Everaert & Stienen 2007; Langton *et al.* 2011); displacement from areas where devices are located (Larsen & Guillemette 2007;

McDonald *et al.* 2012); increased flight costs if devices act as barriers to movement (Desholm & Kahlert 2005; Masden *et al.* 2009, 2010); alteration of foraging habitats if devices alter currents and sediment movements (Gill 2005; Langhamer *et al.* 2010; McCluskie *et al.* 2013); and disturbance from increased vessel traffic (Ronconi & Clair 2002; Garthe & Hüppop 2004; Bellefleur *et al.* 2009; Schwemmer *et al.* 2011). Potential positive effects include the increased provision of loafing and roost structures at sea (Grecian *et al.* 2010; van Deurs *et al.* 2012); and increased foraging opportunities if devices act as fish aggregation devices (FADs) (Wilhelmsson *et al.* 2006; Langhamer & Wilhelmsson 2009), artificial reefs (Langhamer *et al.* 2009; Reubens *et al.* 2013a; b) or *de facto* marine protected areas (*df*MPAs) (Inger *et al.* 2009; Grecian *et al.* 2010). Although knowledge of seabird ecology and behaviour allows moderately robust predictions of which species of seabirds are most likely to be at risk from impacts (Garthe & Hüppop 2004; Furness *et al.* 2012, 2013), a lack of knowledge regarding seabird movements at sea limits our ability to identify and quantify potential effects of MREDs on populations (McCluskie *et al.* 2013).

The great skua *Stercorarius skua* is a protected species breeding in Scotland from April to late August/early September, with numerous breeding colonies located in close proximity to leased and proposed MREDs. Although not ranked as one of the most vulnerable species, results in Chapter 2 suggest that great skuas could be vulnerable to MREDs. Based on the uncertainty surrounding data contributing to the vulnerability rankings (Table 2.14), it is important to improve our understanding of great skua behaviour and movements at sea to ensure robust predictions are made regarding how this species may be affected by MREDs. Great skuas have a restricted global distribution with over 60% of the global population (*ca*.16,000 pairs) breeding in Scotland at the most southern part of their range and migrate

to winter off Iberia and northwest Africa (Mitchell *et al.* 2004; Magnusdottir *et al.* 2012). This internationally important seabird population is experiencing low levels of breeding success (Orkney Bird Report Committee 2013; Shetland Bird Club 2013). In Orkney, the breeding population of great skuas has undergone a 23% reduction over the last decade (Meek *et al.* 2011) and numbers at the largest colonies in Shetland have also decreased (JNCC 2014b). These declines are likely to have been driven by a reduced sandeel (*Ammodytes marinus*) abundance (Caldow & Furness 2000), decreases in fisheries discarding and possibly the impacts of climate change (Oswald *et al.* 2008, 2011), which have resulted in lower prey availability for great skuas (Hamer *et al.* 1991; Meek *et al.* 2011). This evidence suggests that great skuas are already under pressure from changing climate and human activities such as fishing, potentially in both their breeding and wintering foraging areas. As such, this species is at risk from additional impacts of MREDs that could compound existing declines in productivity and population sizes. Collision with offshore wind turbines is thought to the greatest risk that MREDs present to great skuas, owing to their tendency to spend the majority of their time at sea in flight (except at night when they sit on the water or return to the colony), and because they fly at altitudes overlapping with turbine rotors (Table 2.2; Garthe & Hüppop 2004; Furness *et al.* 2013). The species is less likely to be negatively affected by wave and tidal-stream MREDs as they are surface feeders and so unlikely to collide with devices on or below the water surface (Tables 2.4 and 2.5; Furness *et al.* 2012). Birds struggling to find sufficient prey or birds from larger breeding colonies travel further to forage than those from smaller colonies or when food is abundant (Lewis *et al.* 2001; Bertrand *et al.* 2012; Wakefield *et al.* 2013). Therefore, great skuas that travel over greater distances and range more widely increase their probability of encountering and potentially being affected by a MRED. Conversely, if MREDs are located in close proximity to breeding

colonies, breeding birds that are constrained to return to their nest and limited in the area over which they can forage (Shaffer, Costa & Weimerskirch 2003), may repeatedly encounter and be affected by MREDs (Masden *et al.* 2010). Given this evidence, different subsections of great skua populations may be affected in different ways by MREDs.

To investigate how great skuas may be affected by MREDs I used tracking technology to improve our understanding of how this species uses the marine environment. I investigated the movements of breeding birds and failed breeders from two of the largest and best studied colonies in the UK (Fig. 3.1; Appendix 2, Table 1) to assess differences according to colony location and breeding status of the bird (actively breeding or failed breeder). I also aimed to investigate the movements of birds according to their nest status (incubating, chick-rearing, following successful fledging of a chick and following a failed breeding attempt). The overlap of great skuas with leased and proposed MREDs was quantified for birds during different breeding stages and from two different colonies. In addition, I compared tracking data with historical data to assess whether great skua movements have changed over time, with the intention of exploring the implications of predicting the effects of MREDs on species based on short-term tracking data.

3.4 Materials and methods

3.2.1 Data collection

Solar-powered GPS data loggers were fitted to 20 breeding great skuas in the period from 03 June 2011 to 14 June 2011. The loggers were attached for the duration of the breeding season. Ten loggers were fitted to birds from Hoy, Orkney, UK (58°52’N, 3°24’W) and ten loggers fitted to birds from Foula, Shetland, UK (60°8’N, 2°5’W). The study sites represent

the largest breeding colonies for great skuas in Orkney and Shetland and are located in close proximity to leased and proposed MRED sites (Fig. 3.1; Appendix 2, Table 1). Birds were caught on the nest using a remote-controlled noose trap and logger attachment took a mean of 25 min (range = 19 – 32 min). UvA-BiTs GPS data loggers (Bouten *et al.* 2013) weighing *ca*. 25g (<3% of adult mass) were attached dorsally using a Teflon® ribbon backpack harness, consisting of a neck loop and wing loops. Only birds with breeding partners in attendance on territory were targeted for trapping. Dummy eggs were temporarily placed in the nest during trapping and logger deployment to prevent damage through trampling or predation. GPS loggers provided location data at intervals ranging from 1 – 30 min. Loggers transmitted data remotely to a base-station via an antenna in each colony until birds left on migration. A control group of 20 breeding individuals (ten from each colony) were monitored to enable adverse effects of the logger and harness attachment to be identified. All 40 birds were marked with uniquely numbered Darvic colour rings for identification purposes and the location of their nest site was recorded by handheld GPS.

3.2.2 Trip characteristics

Movements >500 m beyond a pre-established colony perimeter and lasting >20 min were defined as 'trips'. Trips covering distances >2000 km (n=2), which were an order of magnitude longer than the next longest trips, were excluded from analyses as they biased utilisation distribution (UD) calculations. When comparing movements of birds from Hoy and Foula, trips were categorised as occurring either during breeding or following a failed breeding attempt. When comparing movements of birds from Hoy only, trips were defined in more detail as occurring during incubation, chick-rearing, following successful fledging of a chick or following a failed breeding attempt. For each trip, I calculated the trip duration

(time elapsed between departure and return to the colony), the foraging range (the maximum distance reached from the colony), and the total distance travelled (summation of distances between GPS points along the route). The numbers of trips per day were defined as the number of times a bird left the colony on a foraging trip. When birds did not leave the colony or were away from the colony on a foraging trip lasting longer than a day, a value of zero trips undertaken that day was recorded. Trip metrics were calculated using R package 'trip' ver.1.1-17 (Sumner 2013) and Geospatial Modelling Environment ver.0.7.2.0 (Beyer 2012).

3.2.3 Utilisation distributions

To estimate the core area used by birds in each breeding category (outlined in section 3.2.2) I used kernel density estimates (KDE) to calculate the 50% UD (Harris *et al.* 1990). Owing to variable intervals between GPS fixes and missing GPS locations, data were linearly interpolated to establish regular 10 min time intervals between GPS points. Interpolation of GPS locations eliminated overestimation and bias in habitat/area use as a result of clustering of short-interval GPS fixes in one area. KDEs were calculated using all location fixes within each breeding category (outlined in section 3.2.2) using a fixed smoothing parameter of 10,000m and a grid size of 2,500m. The smoothing parameter and grid size were identified as most appropriately representing the original data through visual assessment of a series of UDs calculated with a range of bandwidths. For birds in each breeding category, I calculated the percentage overlap of the 50% UD with leased and proposed MRED sites (Scottish Government 2013a). Analyses were performed using R packages 'adehabitatLT' ver.3.14 (Calenge 2006), 'adehabitatHR' ver.4.10 (Calenge 2006), and ArcGIS (ArcMap ver.10. ESRI, USA).

3.2.4 Variation in movements

Multiple trips were recorded for each bird. To account for non-independence of data as a result of repeated sampling, I used generalized linear mixed models (GLMM) with bird identity included as a random effect. In analyses comparing the movements of breeding and failed breeders from Hoy and Foula, models were constructed using a Gaussian distribution in R package 'lme4' ver.1.1-5 (Bates, Machler & Bolker 2012). In analyses comparing the movements of birds from Hoy during the breeding season (during incubation, chick-rearing, following successful fledging of chicks or following a failed breeding attempt), models were constructed using a Gaussian distribution in R package 'nlme' ver.3.1-109 (Pinheiro *et al.* 2013). This different model structure was required to compare the movements of birds from the Hoy colony alone as there was a greater variance in trips incorporating freshwater bathing or visits to probable club sites (sites where non-breeders congregate; Furness 1987) than in trips that went directly out to sea, which was not evident when jointly analysing the Hoy and Foula movements. To account for this heterogeneity in variance that was attributed to trip type, a fixed variance structure was applied. In all models trip length (m), duration (min) and maximal distance from the colony (km) were response variables and were log-transformed prior to statistical analysis to conform to assumptions of normality. Significance was assessed using likelihood ratio tests (LRTs) following backward step-wise model selection (Crawley 2007).

All analyses were implemented in R ver.3.0.1 (R Core Team 2013).

3.2.5 Historical data

Productivity on Foula, and data on nest attendance and trip characteristics during chick rearing, were extracted from published and unpublished literature from 1974 – 2011 (Furness 1977; Catry 1997; Hamer 2001; SOTEAG 2012; JNCC 2014b). Trip duration (time away from nest), the number of trips undertaken per day and nest attendance were recorded during nest watches. Where available, productivity data for Hoy were also extracted (JNCC 2014b).

3.3 Results

Of the 20 loggers deployed, data from 17 were analysed. The remaining three loggers (all from Hoy) did not charge their batteries sufficiently for GPS fixes to be taken regularly or the tagged bird left the colony very soon after logger attachment. The duration of data collection ranged from 23 – 106 days. A total of 2039 trips were recorded. Monitoring of the control group of 20 breeding individuals (ten from each colony) found no adverse effects of the logger and harness attachment on territory attendance or breeding productivity during the 2011 breeding season (Thaxter et al., in review). This was despite subsequent monitoring indicating that tagged birds did not return to the colony to breed in the 2 years following logger attachment.

3.3.1 Comparison of movements of breeding birds and failed breeders from Hoy and Foula

A total of 651 foraging trips were recorded for 17 birds of known breeding status from Hoy and Foula between 03 June 2011 and 15 July 2011.

3.3.1.1 *Trip characteristics*

Failed breeders from each colony spent more time at sea, travelled over greater distances and undertook fewer trips per day than birds engaged in a breeding attempt. Birds from Foula tended to spend more time at sea and travelled further than birds from Hoy (Fig. 3.2; Fig. 3.3 A; Appendix 2, Tables 2 and 3).

3.3.1.2 *Variation in movements between colonies and according to breeding status*

The duration of trips differed according to whether a bird was breeding or had failed in its breeding attempt (GLMM: X^2=33.35, d.f. = 1, p<0.001); with breeding birds spending less time away from the colony than failed breeders (Fig. 3.3 B).

The length of trips and the maximum distance reached from the colony during trips also differed according to breeding status (GLMM: X^2 = 12.04, d.f. = 1, p<0.001; GLMM: X^2 = 9.12, d.f. = 1, p<0.01 respectively) (Fig. 3.3 C and D).

The duration of trips also differed according to colony (GLMM: X^2 = 12.35, d.f. = 1, p<0.001); with birds from Foula undertaking trips that lasted longer than trips by birds from Hoy (Fig. 3.3 B). The distance covered during trips and the maximum distance reached from the colony during trips differed according to colony (GLMM: X^2 = 18.15, d.f. = 1, p<0.001; GLMM: X^2 = 18.37, d.f. = 1, p<0.001 respectively) (Fig. 3.3 C and D).

3.3.1.3 *Overlap with marine renewables*

There was no overlap of core areas with leased or proposed offshore wind sites for birds from Foula. A very low overlap with leased sites for offshore wind development was

observed for failed breeders from Hoy (<1%) and a 9% overlap with proposed sites for future offshore wind development (Fig. 3.4 and Table 3.1).

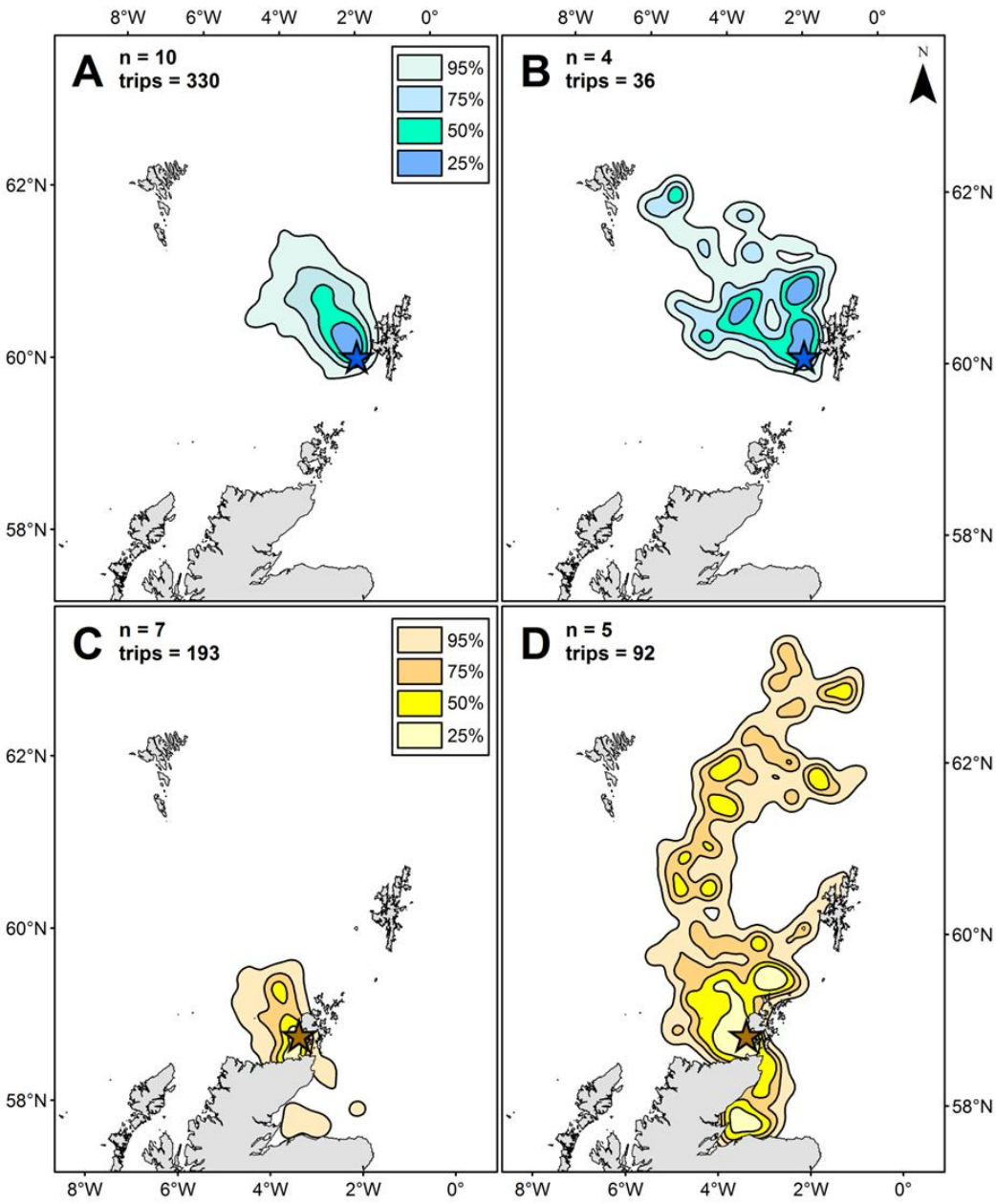

Figure 3.2: Kernel density estimates of great skua movements from Foula (A -B) and Hoy (C-D) during breeding (A,C) and following breeding failure (B,D) until 16 July 2011.

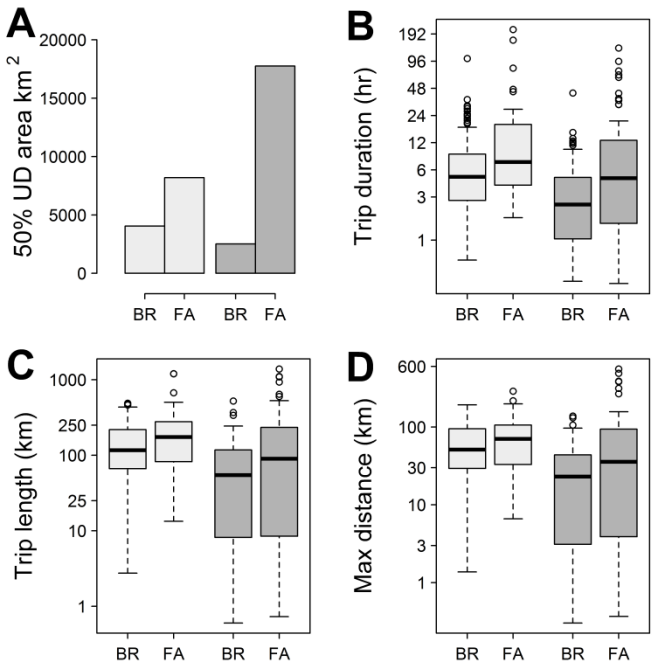

Figure 3.3: Comparison of birds from Foula (light grey) and Hoy (dark grey) during breeding (BR) and following breeding failure (FA) in: A) the core area used, B) the duration of trips, C) the length of trips and D) the maximum distance travelled from the colony.

Only birds from Hoy overlapped with leased sites for wave development (breeders: 3%, failed breeders: 1%). However, breeding birds and failed breeders from both colonies overlapped with proposed sites for wave developments; with a greater overlap observed for birds from Hoy (breeders: 11%, failed breeders: 12%) than for birds from Foula (breeders: 2%, failed breeders: 1%) (Fig. 3.4).

There was no overlap of Foula birds with leased or proposed tidal-stream development sites. Only failed breeders from Hoy overlapped with leased tidal-stream development sites (<1%), and there was a low overlap of breeding birds (5%) and failed breeders (3%) with proposed tidal-stream sites. The limited levels of overlap observed were expected given the small area covered by tidal-stream sites (Fig. 3.4).

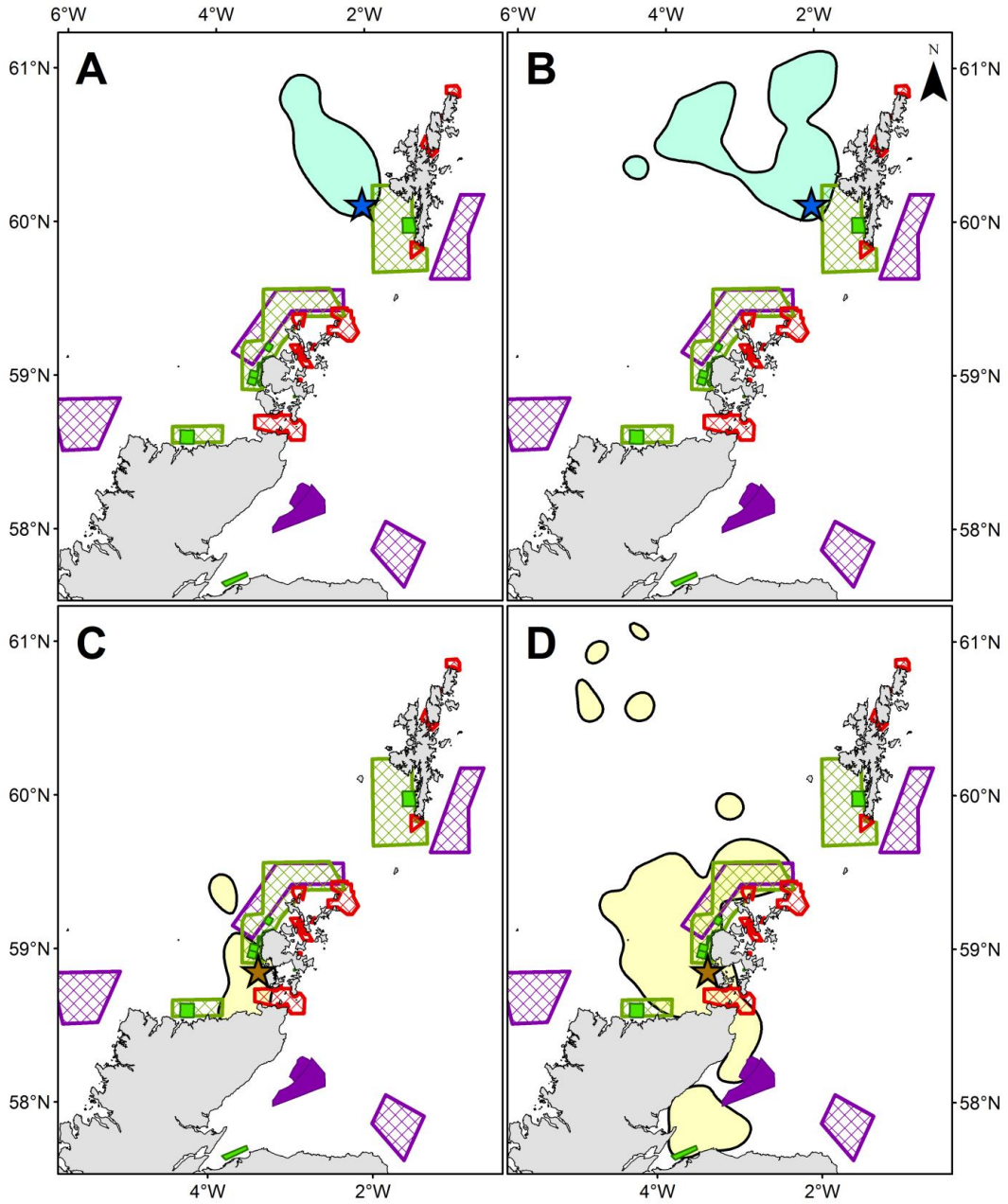

Figure 3.4: Overlap of great skua core areas of use (50% utilisation distribution) with leased and proposed marine renewable energy development sites (solid polygons = leased sites; cross-hatched = proposed sites; purple = offshore wind sites; green = wave sites; red = tidal-stream sites) for birds from Foula (A-B) and Hoy (C-D), during breeding (A,C) and following breeding failure (B,D).

Table 3.1: Percentage of overlap of core areas used (50% utilisation distribution) with leased and proposed offshore wind, wave and tidal-stream renewable energy developments for birds from Foula and Hoy during breeding and once birds had failed in their breeding attempt.

Colony and breeding status	Offshore wind % overlap		Wave % overlap		Tidal % overlap	
	Leased	Proposed	Leased	Proposed	Leased	Proposed
Foula:						
Breeding	-	-	-	1.57	-	-
Failed	-	-	-	1.42	-	-
Hoy:						
Breeding	-	-	2.89	10.55	-	4.81
Failed	0.35	8.56	1.26	11.92	0.09	2.57

3.3.2 Movements of birds from Hoy during the breeding season in relation to nest status

3.3.2.3 *Trip characteristics*

Chick-rearing birds and birds that had successfully fledged chicks undertook more trips per day than incubating birds and birds that had failed in their breeding attempt. Failed breeders used a greater area at sea than incubating birds, chick-rearing birds or birds that had successfully fledged chicks (Fig. 3.5; Fig. 3.6 A; Appendix 2, Tables 4 and 5).

3.3.2.2 *Variation in movements according to nest status*

The duration of trips differed according to the nest status of the bird (GLMM: $X^2 = 11.59$, d.f. = 3, $p<0.01$). Chick-rearing and incubating individuals spent less time away from the nest than failed breeders and birds that had successfully fledged chicks. However, there was no significant difference according to nest status in the distance covered during trips or the maximum distance reached from the colony (Fig. 3.6 B and D).

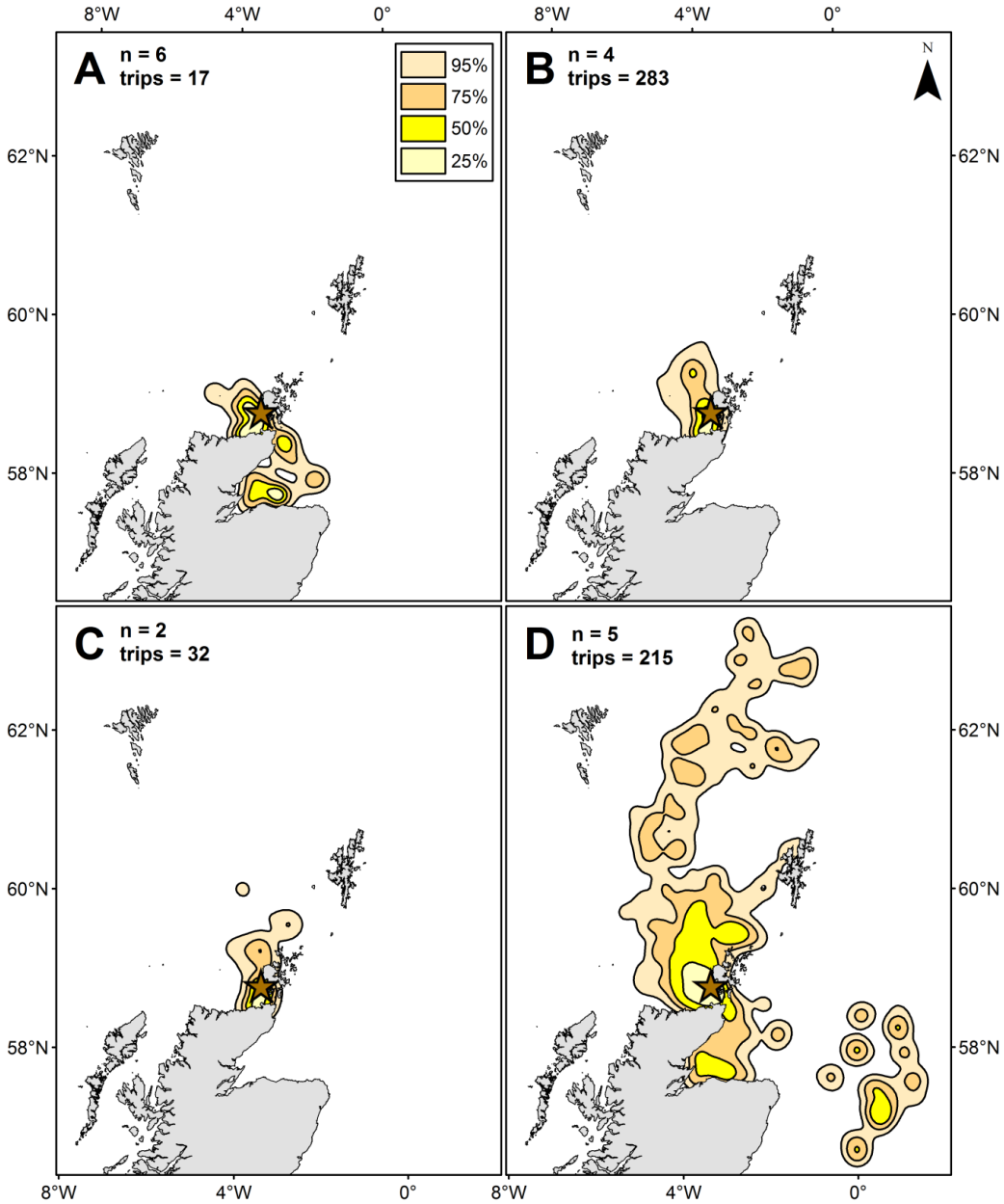

Figure 3.5: Kernel density estimates of great skua movements from Hoy during incubation (A), chick-rearing (B), following successful fledging of chicks (C), and following breeding failure (D) until 16 August 2011.

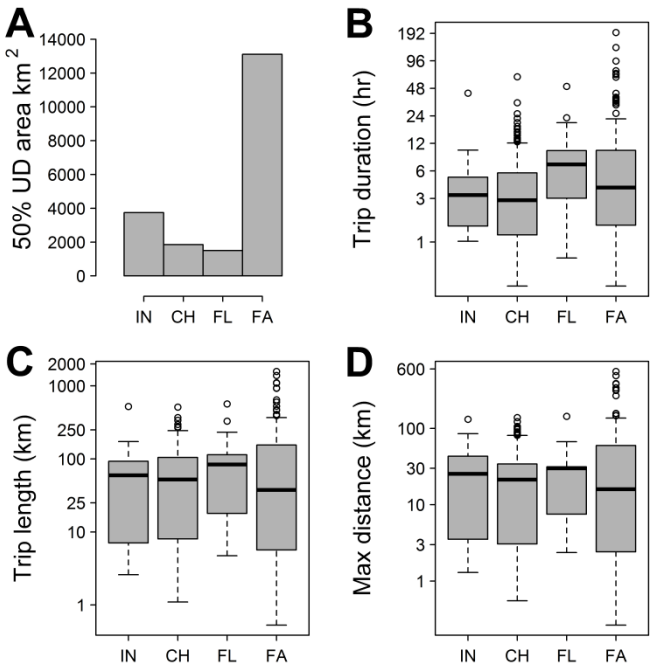

Figure 3.6: Comparison of birds from Hoy during incubation (IN), chick-rearing (CH), following successful fledging of chicks (FL), and following breeding failure (FA) in: A) the core area used, B) the duration of trips, C) the length of trips and D) the maximum distance travelled from the colony.

Table 3.2: Percentage of overlap of core areas used (50% utilisation distribution) with leased and proposed offshore wind, wave and tidal-stream renewable energy developments for birds from Hoy during incubating, chick rearing, once chicks had fledged and once birds had failed in their breeding attempt.

Breeding status (Hoy)	Offshore wind % overlap		Wave % overlap		Tidal % overlap	
	Leased	Proposed	Leased	Proposed	Leased	Proposed
Incubating	-	-	0.43	5.02	-	3.54
Chick-rearing	-	-	3.10	9.64	0.08	7.17
Fledged	-	0.47	1.14	4.99	0.73	12.43
Failed	-	8.77	1.30	12.99	0.14	3.23

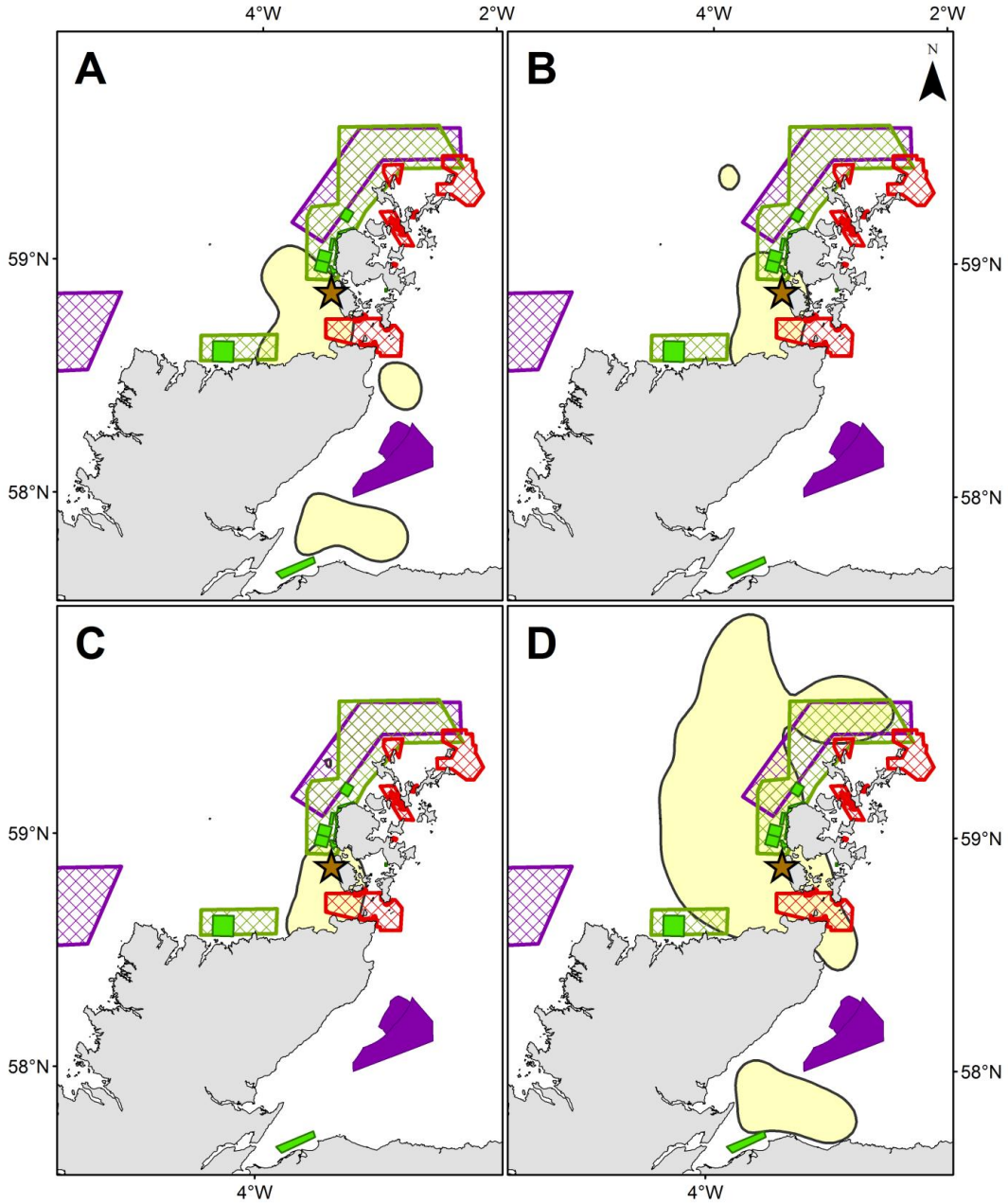

Figure 3.7: Overlap of great skua core areas of use (50% utilisation distribution) with leased and proposed marine renewable energy development sites (solid polygons = leased sites; cross-hatched = proposed sites; purple = offshore wind sites; green = wave sites; red = tidal-stream sites) for birds from Hoy during incubation (A), chick-rearing (B), following successful fledging of chicks (C), and following breeding failure (D).

Birds from Hoy overlapped to a greater extent with proposed sites for all energy types. Overlap with proposed offshore wind sites was only observed for birds that had successfully fledged chicks or had failed in their breeding attempts (<1 and 9% respectively). Birds of all breeding stages overlapped with proposed wave and tidal-stream sites. Incubating birds and successful breeders overlapped with proposed wave sites by 5%, whilst chick rearing birds and failed breeders overlapped by 10% and 13% respectively. For tidal-sites, incubating birds and failed breeders overlapped by less than 4%, chick-rearing birds by 7% and birds that had successfully fledged chicks by 13% (Fig. 3.7).

3.3.3 Comparison of tracking data with historical data

Historical records show an overall negative trend in great skua productivity over time on Foula (JNCC 2014b) and Hoy (Meek *et al.* 2011; JNCC 2014b). This overall decline at both colonies is evident in spite of the different regimes of breeding success in the Shetland Islands and Orkney Islands, which are driven by sandeel abundance (Ratcliffe, Furness & Hamer 1998; Cury *et al.* 2011).

In addition to the negative trend in productivity seen on Foula, there was a concurrent decline in nest attendance. For example, a mean of 1.5 birds were recorded in attendance on territory during spot checks on Foula in 1974, which had declined to 0.7 birds by 2011. There was also a large increase in the median trip duration since the 1970s; from trips lasting less than 40 min to trips lasting over 200 min in the 1990s and 2011. The mean number of trips per day has decreased over time, with 1 trip per day in 2011 compared to between 5 and 8 per day in the 1970s (Table 3.3).

Table 3.3: Comparison between 2011 tracking data and historical data where trip durations were available (all data from Foula unless specified). Annual productivity, attendance and the number of trips undertaken per day shown. All data are for chick-rearing individuals only. 25th and 75th refer to the 25th and 75th percentile of the interquartile range.

Year	Productivity (mean number of chicks per pair)			Attendance Foula (mean number of adults on territory)	Trip duration (h) Foula				Trips per day Foula			n
	Hoy: Stourdale	Hoy: Grutfea	Foula		25th	Median	Mean	75th	Median	Mean	Max	
1974			1.20	1.50	27.00	37.00	39.86	48.50	9.00	8.30	16.00	10
1976			1.30	1.47	18.50	28.00	64.55	76.00	5.50	5.50	8.00	2
1988			0.10	1.03		240.00						30
1994		0.61	0.80	1.20		210.00						6
1995		0.93	1.10	1.30		210.00						6
2011	0.50		0.21	0.70	148.00	245.00	357.70	466.00	1.00	1.32	5.00	9

3.4 Discussion

This study is the first to track individual great skuas throughout the breeding season using GPS technology. I tracked great skuas from two colonies to improve our understanding of how this species uses the marine environment and to predict potential effects associated with MREDs. I also compared movement and productivity data during the tracking period with historical data to assess how great skua movements may have changed over time. I found that the movements of great skuas vary throughout the breeding season, with actively breeding birds remaining closer to the nest and spending less time away from the colony than birds that had failed in their breeding attempt. I also found evidence that movements differ according to breeding colony. Overall overlap with MREDs was low but historical data suggest that great skua movements have changed over time, which could have implications for predicting the long-term effects of MREDs on this species. These data also suggest that

basing marine spatial planning decisions on short-term data could be less informative than longer-term data.

I found that great skuas move differently through the marine environment when actively breeding compared to following a failed breeding attempt. UDs clearly show that failed breeders travelled further and ranged more widely from the colony than active breeders (Fig. 3.2; Fig. 3.5). This is to be expected given that when breeding, birds are central place foragers constrained to regularly return to the nest to incubate eggs or provision chicks, which limits how far the bird is able to travel during foraging trips (Carlson & Moreno 1985; Kacelnik & Cuthill 1990). This behaviour has implications for how great skuas may be affected by MREDs. Failed breeders that range more widely may be at risk of encountering more MREDs than breeding birds and this could result in failed breeders being more affected by MREDs as a result of the cumulative effects of several development sites. Alternatively, if a MRED is located in close proximity to a colony, breeding birds restricted in their foraging range may be forced to encounter a development site more often than failed breeders, who are more flexible in their foraging movements (Fox *et al.* 2006; Everaert & Stienen 2007). This could result in repeated negative effects on breeding birds, for example through displacement or collision with devices, that could affect adult survival and/or chick survival and colony productivity (Masden *et al.* 2010; McDonald *et al.* 2012). The effects of repeated encounters with specific MREDs may be compounded by the greater number of trips undertaken per day by breeding birds when compared to failed breeders; more specifically by chick-rearing individuals and birds that have successfully fledged a chick, than by incubating birds or failed breeders. By increasing the number of trips undertaken per day, adults are likely attempting to respond to additional nutritional requirements of their chicks.

I found that birds that were categorised as having successfully fledged chicks appeared to be undertaking more trips than failed breeders. This may be because they were still provisioning an almost fully grown chick, fledged but still partly dependent on its parents. In addition, I suggest caution should be taken when interpreting the results of the more detailed analyses of nest status, as the categories of incubating birds and birds that successfully fledged chicks comprised small sample sizes.

My analyses suggest that birds potentially differ in their movements according to the colony in which they are breeding. Statistical analyses suggest that when actively breeding, birds from Foula travel over greater distances, spend more time away from the nest and range more widely than actively breeding birds from Hoy (Fig. 3.3 B and D). However, UDs indicate that the converse is true for failed breeders (Fig. 3.2; Fig. 3.3 A). Previous evidence has shown that birds from larger breeding colonies range further from the nest during foraging trips as a result of density-dependent prey depletion around the colony (Ashmole 1963; Lewis *et al.* 2001; Wakefield *et al.* 2013). Therefore, it could be expected that movements by birds from the larger colony of Foula (Mitchell *et al.* 2004) would result in larger core areas of use than for birds from Hoy. Model outputs support this hypothesis (Fig. 3.3 B and D). However, the UD calculations for failed breeders from Hoy do not follow this pattern, with failed breeders from Hoy making use of a larger core area than those from Foula. This lack of consistency between model outputs and UD calculations may be caused by the two colonies not differing in size enough for a clear effect to be seen or perhaps the proximity of prey to the colony differs according to colony location. It is also worth bearing in mind that there is greater variance in the maximal distance travelled from the nest by birds from Hoy; particularly failed breeders from Hoy (Fig. 3.3 D). This greater variance may account for the

greater UD areas calculated for failed breeders from Hoy, and is likely driven by five comparatively long trips to the north of the colony undertaken by three individuals.

I have found evidence for a low overall overlap with MREDs (<13% maximum overlap with core areas used) (Table 3.1 and 3.2); largely owing to the relatively small size of MRED sites compared to the area over which great skuas are capable of ranging. This suggests that great skuas are unlikely to be adversely affected by current development plans for MREDs if they use the marine environment in similar ways across years. UDs indicate a low overlap of core areas used with leased and proposed offshore wind development sites (<10%), which are predicted to represent a relatively high collision risk to this species, as birds tend to fly at altitudes overlapping with turbine swept areas (Fig. 3.4; Fig. 3.7; Table 2.2) (Furness *et al.* 2013). There was no overlap for birds from Foula owing to their consistent north-west direction of movement from the colony and only failed breeders from Hoy overlapped with leased and proposed sites. This could be expected given that failed breeders from Hoy ranged more widely than actively breeding birds thus increasing their chances of overlap. Despite this, the percentage of overlap with offshore wind sites was still low in comparison to the core area over which birds ranged. I found a low overlap of birds from Hoy with leased wave development sites (<4%) but a greater overlap with proposed sites; particularly for failed breeders. Birds from Foula only overlapped to a small degree with proposed wave development sites (<2%), again owing to their consistent movements to the north-west of the colony. A similar low overlap was observed with tidal-stream developments; with no overlap by birds from Foula and overall low levels of overlap with both leased and proposed sites by birds from Hoy (<13% maximum overlap). This low overlap with tidal-stream sites is not unexpected given the small area that these sites cover. In addition to the low levels of

overlap with wave and tidal-stream development sites, these renewable energy types are predicted to present a low level of risk to great skuas (Tables 2.4 and 2.5) (Furness *et al.* 2012). This is largely owing to their limited profile above sea level, which as great skuas are surface feeders, lessens the risk of collision with devices (Grecian *et al.* 2010). In addition, great skuas exhibit a low response to disturbance by structures and vessel traffic (Appendix 1, Table 4) (Furness *et al.* 2012). Wave and tidal-stream developments are therefore unlikely to negatively affect great skuas. Conversely, it is possible that MREDs, particularly wave developments, could benefit breeding great skuas if they act as FADs, artificial reefs or *df*MPAs. For example, if fish abundance increases within the area of MREDs, as has been observed at some offshore wind farms (Reubens *et al.* 2013a), great skuas could benefit from accumulated prey in close proximity to the colony. This has the potential to reduce foraging effort, decrease the time spent away from the nest and subsequently increase productivity. In this study I investigate the movements of adult great skuas during the breeding season; however, there is extremely limited knowledge of the movements of juvenile birds, which therefore restricts predictions regarding how this part of the population may be affected by MREDs. In addition, interactions with MREDs as birds migrate should also be incorporated into predictions regarding the cumulative impacts of MREDs on great skuas throughout their annual cycle.

When comparing the tracking data with historical data there is evidence that great skua movements have changed over time. Chick rearing birds from Foula are spending more time away from the nest than in the 1970s, and are undertaking fewer trips per day. A concurrent decline in the levels of adult attendance at the nest and a reduction in annual productivity was also found (Table 3.3). Cairns (1988, 1992) and Montevecchi (1993) have

concluded that changes in activity budgets are good indicators of changes to marine food supplies. In particular, previous research has shown that increased time spent away from the colony on foraging trips is linked to low prey availability and that a decline in the attendance of adults on their territories reflects an increased difficulty in finding food (Furness & Hislop 1981; Hamer *et al.* 1991; Boyd 1999; Caldow & Furness 2000). Other research suggests that trips per day can be used as a proxy for prey delivery rate (Wakefield *et al.* 2013). Based on this previous research I can conclude that chick rearing great skuas on Foula are currently experiencing lower prey availability than in the 1970s. As Chivers *et al.* (2012) found in black-legged kittiwakes *Rissa tridactyla*, this decline in prey availability is forcing breeding birds to travel greater distances to locate sufficient food to sustain themselves and their chicks, affecting how many trips per day a bird can undertake, and thus reducing the rate of prey delivery to the chick. Increased foraging effort and the subsequent decrease in parental investment has been found to be directly linked to offspring mass and survival (Elliott *et al.* 2009; Kuhn *et al.* 2014), which could be reflected on Foula in the overall decline in annual productivity over time (Table 3.3). In addition to prey availability affecting chick survival, a reduction in nest attendance by adults has previously been proven to increase predation of eggs and chicks by conspecifics and other seabird species (Hamer *et al.* 1991, 2007; Caldow & Furness 2000). This further compounds declines in annual productivity by further reducing chick survival and is likely occurring on Foula. Similar observations of increased foraging effort and reduced attendance in breeding great skuas were recorded during the late 1980's in a period of markedly reduced sandeel availability (Table 3.3) (Hamer *et al.* 1991; Caldow & Furness 2000; Hamer 2001). During this period, it was found that breeding great skua daily energy expenditure averaged fives greater than the basal metabolic rate (BMR) (Caldow 1988), which is suggested to be slightly greater than an

assumed energetic ceiling (Drent & Daan 2002). In combination with the fact that despite increases in foraging effort chick survival remained low, this evidence suggests that during a period of reduced food availability great skuas were physically unable to increase their foraging effort further to maintain reproductive output (Hamer *et al.* 1991). As such, current populations of great skuas may have similarly reached an energetic maximum in terms of effort available to invest in foraging, as it is clear that despite increases in foraging effort over time, productivity has failed to improve and populations are continuing to decline (Mitchell *et al.* 2004; Meek *et al.* 2011). Based on this evidence, great skuas appear to be a species under pressure and therefore vulnerable to additional threats, which makes them potentially sensitive to negative effects of MREDs.

Given the change that I have found in the movements of great skuas during the breeding season since the 1970s, it is likely that areas identified as important in short-term tracking work may alter in the future. This is important to consider when using tracking data to inform marine spatial planning. In addition to long-term changes in movement behaviour, previous studies provide evidence for seasonal and annual variation in the movements of seabird populations and the core areas used (Hamer *et al.* 2007; Pettex *et al.* 2012; Bogdanova *et al.* 2014), which will influence how seabirds are affected by MREDs. Taylor, Terauds & Nicholls (2004) and Soanes *et al.* (2013) also provide evidence that sample sizes of tracked individuals and the number of trips incorporated in analyses can affect the extent of the area identified when calculating UDs, which should be borne in mind when interpreting this type of data. These issues highlight a challenge to, and a shortcoming of using short-term tracking studies in isolation to predict the effects of MREDs on seabird species and incorporating them as a basis for marine spatial planning. A potentially useful method to

address some of these issues is species distribution modelling (SDM). Arcos *et al.* (2012) provide evidence that SDMs can produce useful predictions of seabird hotspots for marine spatial planning even when incorporating only one year of data. However, the authors also acknowledge that SDM performance can differ according to species and that current rapid climatic change could limit the predictive power of such models. I advocate that although tracking data can be useful in marine spatial planning and environmental impact assessments, where possible tracking studies should be conducted over several years, in conjunction with monitoring of populations and distributions at site specific and colony levels to fully assess effects.

Chapter 4:

Habitat association and individual variation in the foraging behaviour of great skuas

Abstract

Human activities at sea are increasing around the world and are altering the marine environment. These activities include the rapid development of the marine renewable energy industry and changes to discarding legislation in the EU Common Fisheries Policy. Both of these activities are likely to affect declining seabird species in the UK, including the great skua *Stercorarius skua*. In order to predict how seabirds may be affected by increasing human activities at sea, an improved understanding of how seabird populations use the marine environment is required. We usually consider interactions between seabird species and marine conditions at the population level by estimating population average responses. However, this approach reveals little about individual strategies and intra-population variation, which could lead to subsections of seabird populations being differently affected by human activities in the marine environment. Great skuas are considered generalist predators with a minority of dietary specialists within the population, which make them an ideal species to assess for the presence of intra-population variation in foraging behaviour. In this chapter I use GPS tracking and remotely sensed environmental data to investigate the foraging behaviour and strategy of great skuas in Scotland to establish whether subsections of the population may be differently affected by human activities at sea. I have found evidence for consistency in individual great skua foraging behaviour and preferred foraging habitat. My results support the theory that although populations may appear to be composed of generalist predators, these populations may actually comprise a population of

individual specialists. Based on consistent differences between individuals, and between breeding birds and birds that had failed in their breeding attempt, my results suggest that subsections of great skua populations could be differently affected by human activities in the marine environment.

4.1 Introduction

Human activities at sea are increasing around the world and are altering the marine environment, with implications for marine species. Extensive discarding of fisheries by-catch, in the form of whole fish and offal, is likely to have inflated some seabird populations (Garthe, Camphuysen & Furness 1996; Furness 2003) and influences seabird behaviour (Votier *et al.* 2004c, 2010, 2013; Bodey *et al.* 2014). The need to reduce carbon emissions and increase energy security has led to the rapid development of the marine renewable energy industry; with increasing areas of the marine environment allocated for marine renewable energy developments (MREDs). MREDs could have positive and negative impacts on the marine environment, including seabird species. For example, seabirds could collide with renewable energy devices (Everaert & Stienen 2007; Langton *et al.* 2011) or be displaced from foraging areas (Larsen & Guillemette 2007; Petersen & Fox 2007). Conversely, MREDs could improve foraging opportunities for a range of marine fauna, including seabirds, should they act as artificial reefs (Reubens *et al.* 2013a; Russell *et al.* 2014), fish aggregation devices (Langhamer & Wilhelmsson 2009) or *de facto* marine protected areas (Inger *et al.* 2009; Grecian *et al.* 2010).

In general, seabirds are in decline in the UK. One seabird species that could be affected by changes in human activities at sea is the great skua *Stercorarius skua* (see Chapter 2 and 3).

The great skua is a declining species breeding in Scotland from April to late August/early September (Mitchell *et al.* 2004; Meek *et al.* 2011; IUCN 2013). The majority of great skua colonies are located in Orkney and Shetland and are of international importance as Scotland holds over 60% of the global breeding great skua population (Mitchell *et al.* 2004; Meek *et al.* 2011). A large proportion of the Scottish population breeds in special protection areas (SPAs), designated specifically to safeguard breeding populations of seabirds. SPA legislation (EC Birds Directive 2009/147/EC) states that developments in the marine environment must not negatively impact these safeguarded populations and that species' populations must remain in favourable conservation status (Scottish Natural Heritage 2014). This is crucial to bear in mind given that several protected great skua colonies are in close proximity to areas leased for MREDs, and that evidence presented in Chapter 3 indicates that movements of great skuas overlap with leased and proposed MRED sites (Fig. 3.1, 3.4 and 3.7).

Great skuas are considered generalist predators, with a minority of dietary specialists within the population (Hamer *et al.* 1991; Phillips *et al.* 1997; Votier *et al.* 2004a; b, 2007). They feed by 'naturally' catching prey themselves, kleptoparasiting prey from other seabird species and by scavenging on fisheries bycatch discarded at sea (Furness 1987; Votier *et al.* 2004b). Extensive dietary studies indicate that great skua diet primarily consists of sandeels when available, discarded demersal whitefish (including whiting and Norway pout), discarded pelagic fish (including herring and mackerel), goose barnacles and other seabird species (including eggs, chicks and adult birds) (Furness 1987; Hamer *et al.* 1991; Phillips *et al.* 1997; Votier *et al.* 2001, 2003). A minority of individuals forage by predating other seabird species (Votier *et al.* 2004a). This foraging strategy has been found to have potentially detrimental effects on the populations of targeted seabird species (Phillips,

Thompson & Hamer 1999; Votier *et al.* 2004b; Meek *et al.* 2011) and increases markedly in times of reduced food availability (Hamer *et al.* 1991; Votier *et al.* 2007). Based on this knowledge of dietary preferences, great skuas are likely to be affected by recent changes to the EU Common Fisheries Policy (CFP), which aims to ban discards by 2019 (Votier *et al.* 2004c; Bicknell *et al.* 2013). Reduced discard availability could cause great skuas to switch foraging strategies to increasingly prey on other seabird species already in decline (Furness 1997; Bearhop *et al.* 2001; Votier *et al.* 2004c; b). In light of evidence presented in Chapter 3 that suggests great skuas could be affected by MREDs, and knowledge of the species' reliance on fisheries discards, it is imperative to improve our understanding of great skua foraging strategies at sea to better understand the implications of changing human activities in the marine environment. In addition, our knowledge of great skua intrapopulation variation in dietary preferences makes the great skua an ideal species to assess for intra-population variation in foraging behaviour and habitat use.

To predict how seabirds may be affected by increasing human activities at sea, an improved understanding of how seabird populations use the marine environment is required. A decrease in the cost of biologging technology and the miniaturisation of tracking devices has led to a rapid increase in the use of this technology to track seabird movements to investigate their foraging behaviour. Based on the principle of area-restricted search (ARS), seabird behaviour during foraging trips can be inferred from the speed of travel between consecutive GPS location fixes, with slower speeds suggesting prey searching behaviour and greater speeds implying transiting flight (Kareiva & Odell 1987). By combining information on foraging behaviour with publically available, fine-scale resolution (spatially and temporally) remotely sensed environmental data it is possible to identify marine habitats

targeted during foraging (Patrick *et al.* 2014). This information is crucial in constructing species distribution models (SDM) for inclusion in marine spatial planning (Arcos *et al.* 2012). SDMs can be used to estimate potential overlap of seabird movements with human activities, predict potential effects of human activities on seabirds and identify areas at sea appropriate for conservation.

We usually consider interactions between seabird species and marine conditions at the population level by estimating population average responses. However, this approach reveals little about individual strategies and intra-population variation (Votier *et al.* 2010). As there is increasing evidence that individuals within populations can markedly differ in their use of the marine environment (Votier *et al.* 2010; Patrick *et al.* 2014), it is important to consider intra-population variation to more accurately estimate the effects of human activities on seabirds. Peer-reviewed studies and evidence presented in Chapter 3 suggests that sub-sections of populations could respond differently to human activities at sea based on individual variation in foraging strategies and behaviour, and differences influenced by breeding status (Votier *et al.* 2010; Patrick *et al.* 2014). As such, it is imperative to better understand individual variation to predict how subsections of seabird populations may be differently affected by human activities, and how populations may respond to conservation measures.

To improve understanding of great skua foraging behaviour and association with marine environmental conditions throughout the breeding season, I have combined fine-resolution GPS tracking technology with remotely sensed marine environment data. I aim to assess:

1) How repeatable individuals are in their foraging behaviour dependent on whether birds are breeding or have failed in their breeding attempt.
2) Investigate population-level relationships with marine environmental conditions to identify foraging habitat.
3) Identify whether relationships with environmental conditions alter throughout the breeding season, which could indicate a change in foraging strategy according to breeding status.
4) Establish the presence of individual variation in foraging behaviour, which could indicate whether an assumed generalist predator population is in fact comprised of individual specialists.

The overarching aim of this research is to establish whether subsections of great skua populations could be differently affected by human activities at sea, according to variation in individual foraging behaviour and variation in foraging behaviour dependent on breeding status.

4.5 Methods

4.2.1 Data collection

4.2.1.1 GPS data

Solar-powered GPS data loggers were fitted to 20 breeding great skuas in the period from 03 June 2011 to 14 June 2011. The loggers were attached for the duration of the breeding season. Ten loggers were fitted to birds from Hoy, Orkney, UK (58°52'N, 3°24'W) and ten loggers fitted to birds from Foula, Shetland, UK (60°8'N, 2°5'W). The study sites represent the largest breeding colonies for great skuas in Orkney and Shetland. Birds were caught on

the nest using a remote-controlled noose trap. Handling for measurements, ringing and logger attachment took a mean of 25 min (range = 19 – 32 min). UvA-BiTs GPS data loggers (Bouten *et al.* 2013) weighing *ca*. 25g (<3% of adult mass) were attached dorsally using a Teflon® ribbon backpack harness, consisting of a neck loop and wing loops. Only birds with breeding partners in attendance on territory were targeted for trapping. Dummy eggs were temporarily placed in the nest during trapping and logger deployment to prevent damage to eggs through trampling or predation. GPS loggers provided location data at intervals ranging from 1 – 30 min. Loggers transmitted data remotely to a base-station via an antenna in each colony until birds left on migration. A control group of 20 breeding individuals (ten from each colony) were trapped, marked and monitored to enable adverse effects of the logger and harness attachment to be identified. All 40 birds were marked with uniquely numbered Darvic colour rings for identification purposes and the location of their nest site was recorded by handheld GPS.

4.2.1.2 Environmental variables

I obtained data for monthly averaged sea surface temperature (SST, °C), monthly averaged chlorophyll-*a* production (chl-*a*, mg m^{-3}), front density, distance from front (km), seabed slope gradient, depth (m), distance from the coast (m) and annually averaged demersal fishing intensity in Scottish waters to explore the relationships between these environmental variables and the flight speed of great skuas. These variables were chosen because they commonly drive productivity at sea or specifically relate to our current understanding of great skua dietary preferences. SST and chl-*a* monthly composite data for June and July 2011 were included as proxy measures of productivity as data on the location of forage fish are not available. SST and chl-*a* monthly composite data values were assigned to GPS

locational data according to the month in which GPS observations were recorded. SST and chl-*a* monthly composites were extracted from the Moderate Resolution Imaging Spectroradiometer (MODIS) website at a resolution of 4 x 4 km (http://modis.gsfc.nasa.gov/). Distance from fronts and front density were calculated over *ca.* 1.1 x 1.1 km area (see methods in Scales et al. 2014). Front density was calculated based on temperature gradients at *ca.* 1.1 x 1.1 km resolution (K. Scales pers. comm.). Bathymetry data for the study area were downloaded from the General Bathymetric Chart of the Oceans (GEBCO) (http://www.gebco.net/) at 1 arc-minute resolution. From these data, depth, seabed slope gradient and distance from coast were calculated at a resolution of 1 x 1 km using the Spatial Analyst toolbox in ArcGIS (ArcMap ver.10. ESRI, USA). Annual average demersal fishing intensity for 2011 was calculated using combined vessel monitoring system (VMS) data and landings information for all UK registered commercial fishing vessels (≥15m length), in ICES areas VIa, VIb, IVa, IVb, IVc, IIa, VIId, and VIIa (fisheries data were provided and processed by Marine Scotland Science; see Kafas *et al.* 2012). VMS data were filtered by speed (0 < speed < 5 knots) to ensure fishing activity was distinguished from steaming (Kafas *et al.* 2012). While demersal fishing activity data were annually averaged, fishing hotspots are persistent over time (Witt & Godley 2007), and so these data provide a proxy of discard availability.

4.2.2 Data analysis

4.2.2.1 Trip characteristics

Derived ground speeds (hereafter referred to as flight speeds) were calculated between consecutive GPS locations as they can be used to infer foraging behaviour (Votier *et al.* 2010; Patrick *et al.* 2014; Bodey *et al.* 2014). Slower flight speeds tend to indicate prey searching

and foraging behaviour, whilst faster speeds indicate transiting flight (Kareiva & Odell 1987; Pinaud & Weimerskirch 2005). Stationary GPS fixes (speeds equal to zero) were excluded from analyses but all speeds >0 were included. This is because great skuas are known to wait behind fishing vessels for catches to be hauled to scavenge on discarded fish (Hudson & Furness 1988, 1989). As such, even slow speeds often removed in GPS tracking analyses (e.g. 1-4 km h^{-1}: Shamoun-Baranes *et al.* 2011; Thaxter *et al.* 2011), could represent birds sitting on the water engaged in scavenging foraging behaviour. This species does not regularly forage at night, so GPS fixes occurring during hours of darkness were removed (Votier *et al.* 2006; Magnusdottir *et al.* 2014). This is because time spent sitting on the water at night could be interpreted as foraging behaviour. Movements >500 m beyond a pre-established colony perimeter and lasting >20 min were defined as trips. Very long trips >2000 km (n=2), which were an order of magnitude longer than the next longest trips, were excluded from analyses as they were identified as outliers during initial model validation and made it difficult to normalise the data for modelling. Nest monitoring throughout the breeding season enabled trips to be categorised as occurring either during breeding (incubation and/or chick provisioning) or following a failed breeding attempt. For each trip, I calculated the trip duration (time elapsed between departure and return to the colony); the trip length (sum of distances between GPS points along the route); the furthest location from the colony (the point the greatest distance from the colony during the trip); and the direction of departure from the colony (average of the first three bearings >1km from the colony). Using the furthest point reached from the colony as a mid-point, trips were also split into outward or return sections. This was based on the assumption that when birds have acquired enough prey they will return to the colony rather than continue searching for additional food. Trip metrics were calculated using R package 'trip' ver.1.1-17 (Sumner

2013) in R ver.3.0.1 (R Core Team 2013), and Geospatial Modelling Environment ver.0.7.2.0 (Beyer 2012).

4.2.2.2 Consistency in foraging behaviour according to breeding status

To estimate the consistency of individual foraging behaviour according to breeding status I assessed four measurements of trip characteristics: 1) trip duration (min); 2) trip length (m); 3) the furthest location from the colony (northings and eastings); and 4) the direction of departure from the colony (degrees). The repeatability (r) of measurements between individual birds in each breeding category was tested against the null hypothesis that between-group variance was equal to within-group variance. Repeatability in trip duration, trip length and furthest location from the colony were estimated using the package rptR ver.0.6.405 (Nakagawa & Schielzeth 2010) in R ver.3.0.1 (R Core Team 2013). Values for repeatability, their standard errors and associated *p*-values are presented. When calculating repeatability in the direction of departure from the colony, location data were linearly interpolated to establish regular 10 min time intervals between GPS points. This ensured that mean bearings were not biased by variable intervals between fixes caused by irregular interval settings and missing GPS locations. As bearings are a circular measure bounded by 0 and 360, a circular ANOVA (circular ver.0.4-7, in R ver.3.0.1: (R Core Team 2013) in conjunction with repeatability calculations (Lessells & Boag 1987) were used to calculate repeatability estimates. *P*-values associated with repeatability measures are not available using this method but standard errors were calculated following Becker (1984).

4.2.2.3 Population- and individual-level relationships between flight speeds and environmental variables

To test the relationships between flight speeds and environmental conditions (see Fig. 4.1 for examples), I fitted generalised linear mixed models (GLMM) with a Gaussian error structure using lme4 ver.1.1-5 (Bates *et al.* 2012) in R ver.3.0.1 (R Core Team 2013). Flight speed (km h^{-1}) was fitted as the response variable and SST, chl-*a*, front density, distance to front, seabed slope gradient, distance to coast and intensity of demersal fishing activity were fitted as continuous fixed effects. Depth was removed as a fixed effect owing to its strong collinearity with distance from coast (Zuur, Ieno & Elphick 2010). Based on the knowledge that most great skuas forage offshore but a minority of the population prey on seabird colonies located on the coast (Votier *et al.* 2004a, 2006), distance from coast was retained as I thought it would explain more variation in individual foraging strategies than depth. An interaction between breeding status (a 2-level categorical variable) and foraging trip direction (a 2-level categorical variable) was included. As multiple trips were recorded for each bird and from each colony, individual and colony were fitted as random effects to account for non-independence of data as a result of repeated sampling. To investigate whether relationships between flight speed and environmental conditions varied between individuals, reaction norms were also fitted within the model (Dingemanse *et al.* 2010). In this study, reaction norms (also known as random slopes) allow both the intercept and the slope of relationships between foraging behaviour (indicated by flight speed) and environmental variables to vary by individual. This allows individual-level relationships with environmental variables to be identified. Random slopes were incorporated to investigate the potential presence of differing foraging strategies within great skua populations.

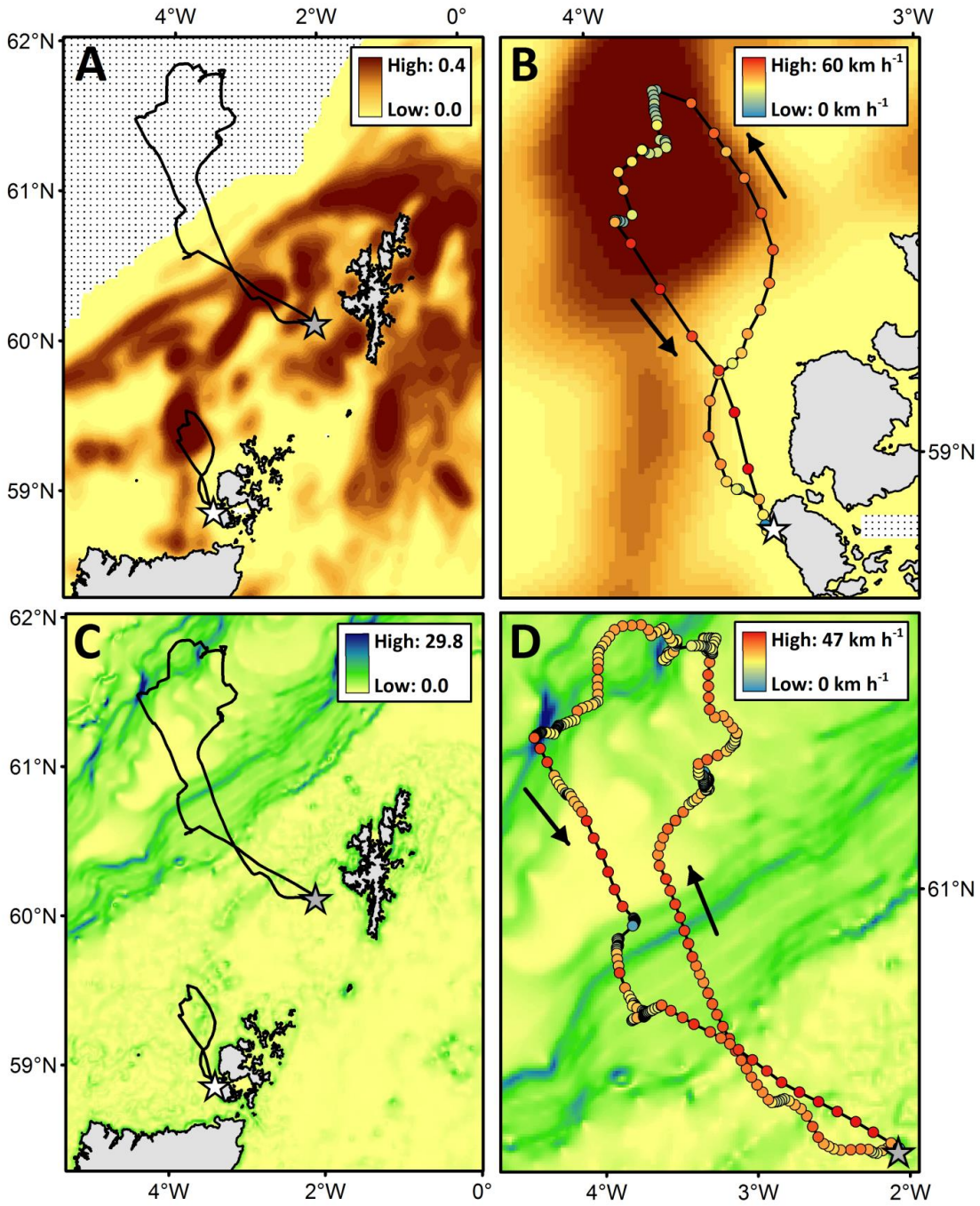

Figure 4.1: Illustrations of two underlying environmental variables included in models; A and B depict demersal fishing intensity, whilst C and D depict seabed slope gradients. Example trips from Foula (grey star) and Hoy (white star) are shown. B and D portray derived ground speeds at each GPS fix location during the trips to illustrate the varying flight speeds recorded and the potential associations with environmental variables. Slow flight speeds (colder colours) are likely to reflect foraging behaviour, whilst faster flight speeds (warmer colours) likely indicate transiting flight. Black and white dotted areas in A and B indicate areas of no data.

Based on knowledge that some great skuas target offshore demersal fisheries discards, whilst others prey on other seabird species located in coastal colonies (Votier *et al.* 2004b, 2008a), individual relationships with demersal fishing intensity and distance from the coast were allowed to vary. In addition, individual relationships with seabed slope were allowed to vary, as during data exploration and initial model testing, seabed slope was extremely significant and explained a relatively large amount of variance in the model. Based on this evidence I made the assumption that seabed slope could also be important in influencing foraging behaviour in great skuas. Individual relationships with these three environmental variables were assessed to compare individual great skua foraging behaviour. All environmental variables were standardised: the sample mean was subtracted from each value and then divided by the sample standard deviation (Zuur, Ieno & Smith 2007). By equally scaling values, or 'centering around subjects', between-subject variation is eliminated to show only within-subject variation (van de Pol & Wright 2009). This enables comparison between individual random slopes, which represent individual-level relationships with environmental variables. Flight speed and chl-*a* were $\log_{10}$ transformed to conform to assumptions of normality and to minimise the effect of sparse extreme values. Demersal fishing intensity and seabed slope gradient were $\log_{10}+1$ transformed to reduce the effect of sparse extreme values and to eliminate negative values (Zuur *et al.* 2007). Significance of fixed effects was assessed using likelihood ratio tests following backward step-wise model selection (Crawley 2007).

4.2.2.4 Population- and individual-level relationships between flight speeds and environmental variables according to breeding status

Evidence presented in Chapter 3 indicates that great skua foraging behaviour is influenced by whether birds are actively breeding or have failed in their breeding attempt. To investigate whether population- and individual-level relationships with environmental variables differ according to breeding status I fitted a further two models. One model was fitted to data collected during breeding (i.e. when birds were incubating eggs and/or provisioning chicks), whilst the other was fitted to data collected once birds had failed in their breeding attempt. The model structure and testing followed that outlined in section 4.2.2.2 but with the exclusion of the categorical factor of breeding status. These additional models were required because although breeding status was considered as an explanatory variable in the first model that contained all tracking data, only the intercepts (not the slopes) of great skua relationships with environmental variables were able to vary. This is because the first model was unable to converge when 3-way interactions between each environmental variable and trip direction and breeding status were included. As such, I was unable to estimate population-level relationships with environmental variables according to breeding status (represented by differences in the slopes according to breeding status). This was also the case for individual-level relationships with environmental variables related to breeding status. Potential differences in relationships between great skua flight speeds and environmental variables according to breeding status are important to consider as foraging strategies may alter throughout the breeding season. These potential changes throughout the breeding season have implications for interpreting how great skuas may be affected by human activities in the marine environment.

In order to identify the proportion of variance explained by the fixed effects included in each model, and the proportion of variance explained by both fixed and random effects included in each model, marginal and conditional R^2 values were calculated following Johnson (2014) and using the MuMIn package ver.1.10.5 (Bartoń 2014) in R ver.3.0.1 (R Core Team 2013).

4.6 Results

Of the 20 loggers deployed, data from 17 were analysed. The remaining 3 loggers (all from Hoy) did not charge their batteries sufficiently for GPS fixes to be taken regularly or the tagged bird left the colony very soon after logger attachment. Between 03 June and 15 July 2011, 651 foraging trips were recorded for 17 birds of known breeding status from Hoy and Foula. The number of trips recorded per individual ranged from 2 to 85 for breeding birds and from 5 to 30 for birds that had failed in their breeding attempt.

4.3.1 Consistency in foraging behaviour according to breeding status

Individual great skuas were consistent in the locations they visited during foraging trips. Breeding birds were highly repeatable in the furthest locations reached during foraging trips (Northings: r = 0.62 ± 0.09, p<0.001; Eastings: r = 0.91 ± 0.04, p<0.001), whilst failed breeders were less repeatable in the furthest locations visited (Northings: r = 0.41 ± 0.15, p<0.001; Eastings: r = 0.57 ± 0.15, p<0.001). Breeding birds showed some consistency in the bearings of their departure from the colony (r = 0.49 ± 0.09) but failed breeders were not consistent in their departure bearing (r = 0.38 ± 0.14). Breeding birds and failed breeders showed less repeatability in trip duration (Breeders: r = 0.29 ± 0.09, p<0.001; Failed breeders: r = 0.26 ± 0.12, p<0.001) and the distances covered during a trip (Breeders: r = 0.36 ± 0.09, p<0.001; Failed breeders: r = 0.23 ± 0.11, p<0.001).

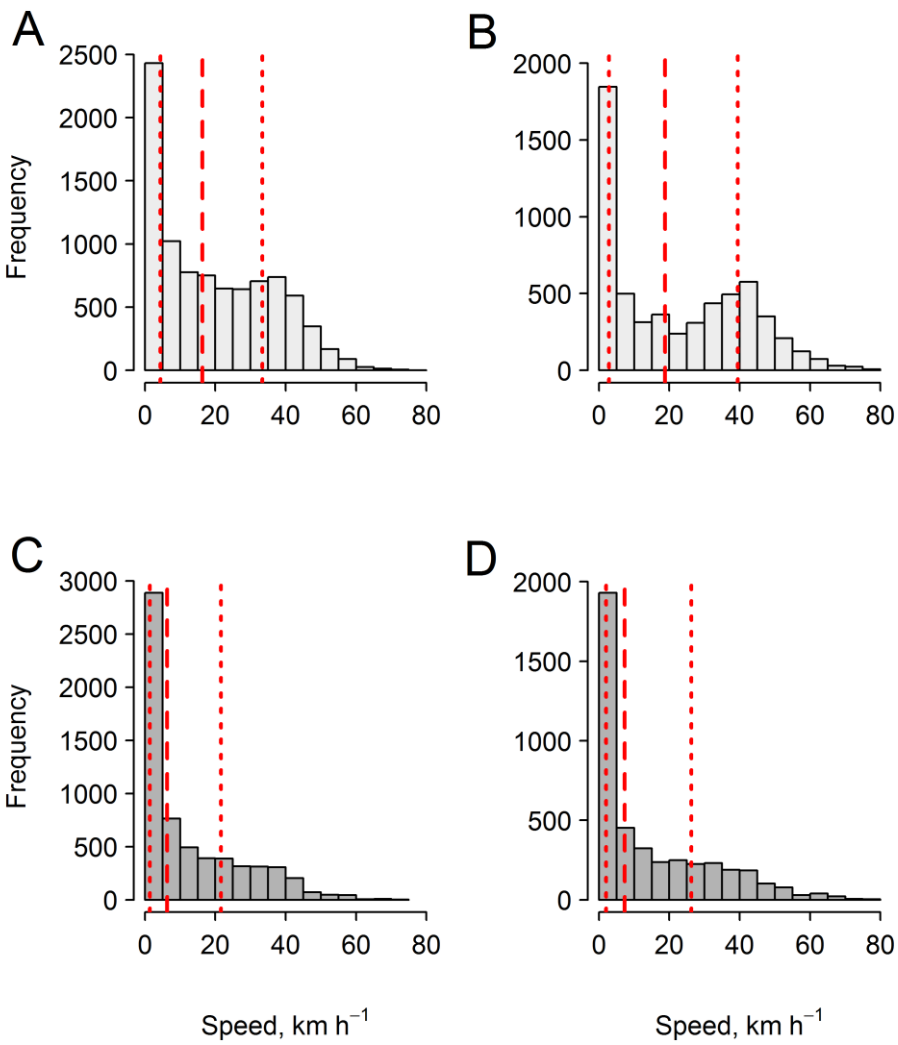

Figure 4.2: Histogram of the flight speeds of breeding birds (A and B) and failed breeders (C and D) on the outward (A and C) and return (B and D) sections of foraging trips. Dashed red line indicates median flight speed. Dotted red lines indicate the upper and lower interquartile range.

Breeding birds travelled faster than failed breeders, with birds flying faster on the return section of foraging trips than on the outward section (GLMM: X^2=4.32, d.f.=1, p<0.05) (Fig. 4.2). Breeding birds leaving the colony flew at a median speed of 16.3 km h^{-1} (Interquartile range (IQR): 4.4 – 33.3 km h^{-1}) whilst birds returning to the colony flew at a median speed of 18.8 km h^{-1} (IQR: 2.8 – 39.4 km h^{-1}). Failed breeders travelled at a median speed of 6.3 km h^{-1} (IQR: 1.4 – 21.6 km h^{-1}) when leaving the colony and 7.3 km h^{-1} (IQR: 2.0 – 26.2 km h^{-1})

when returning. The distribution of speeds differed between breeding birds and failed breeders, with a peak in speeds of travel by breeding birds at *ca.* 40 km h^{-1} that was absent in the distribution of speeds for failed breeders (Fig. 4.2).

4.3.2 Population- and individual-level relationships between flight speeds and environmental variables

Great skua flight speed had a positive relationship with distance from front; with flight speeds decreasing as birds moved closer to fronts (GLMM: X^2=354.6, d.f.=1, p<0.001) (Fig. 4.3 A and B). Despite this relationship, great skuas increased their speed of travel over areas of greater front density (GLMM: X^2=5824.1, d.f.=1, p<0.001) (Fig. 4.3 C and D). Flight speeds reduced as SST and chl-*a* increased (GLMM: X^2=3.9, d.f.=1, p<0.05 and GLMM: X^2=830.7, d.f.=1, p<0.001 respectively) (Fig. 4.3 E, F, G and H). In addition, foraging birds also reduced their speed of travel over increasing seabed slope gradients (Fig. 4.4 A and B), the further they were from the coast (Fig. 4.4 D and E), and in areas of increased demersal fishing intensity (Fig. 4.4 G and H) (GLMM: X^2=8.1, d.f.=1, p<0.01; GLMM: X^2=29.6, d.f.=1, p<0.001 and GLMM: X^2=3.9, d.f.=1, p<0.05 respectively).

At an individual level, most birds shared the population level relationship with environmental variables (Fig. 4.4 C, F and I). However, individual variation was evident, with some birds showing opposing relationships with seabed slope (Fig. 4.4 C) and demersal fishing intensity (Fig. 4.4 I), and some individuals displaying no relationship with distance from coast (Fig. 4.4 F). It is also evident that the range of environmental conditions encountered by foraging birds varies between individuals (Fig. 4.4 C, F and I).

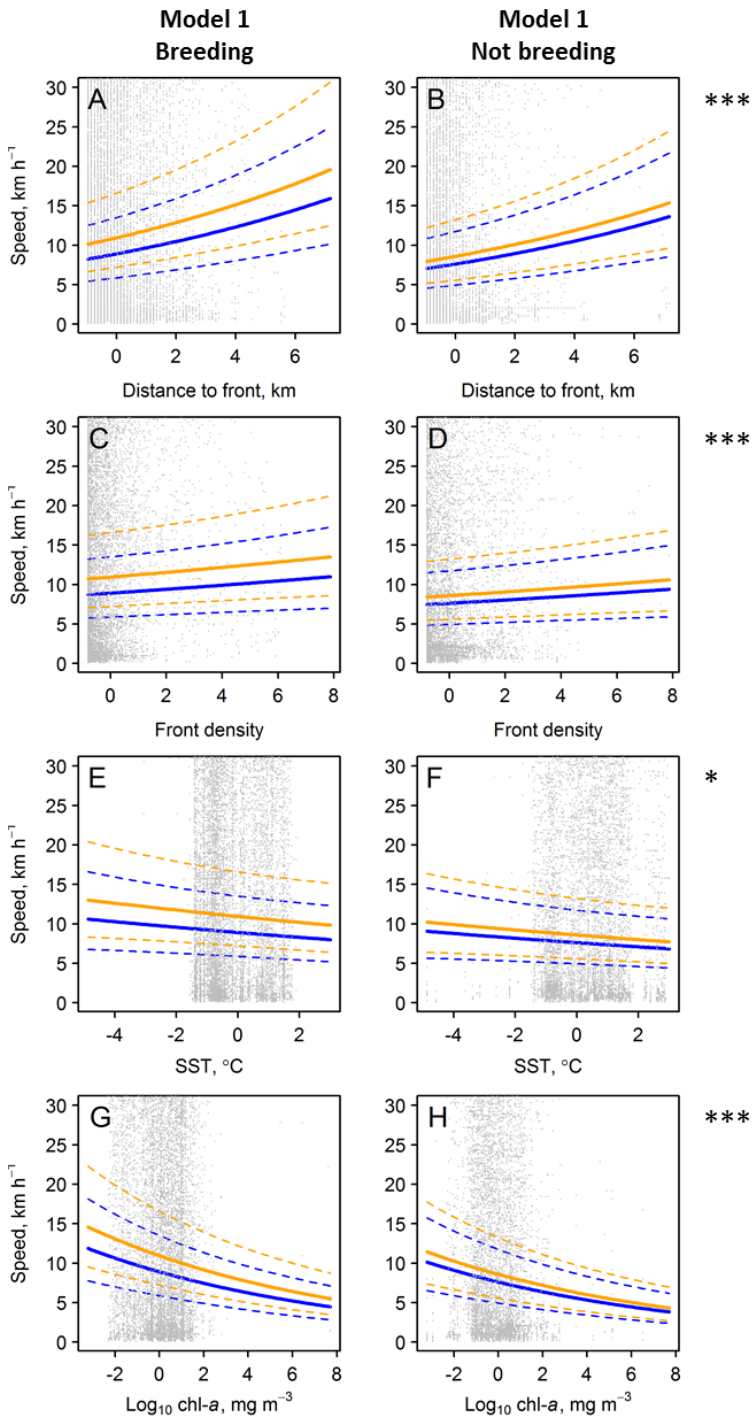

Figure 4.3: All data (both breeding and not breeding) (n = 17). Slopes represent population level relationships with environmental variables. Stars to the right of plots indicate the significance of the relationships between fight speed and environmental variables: *** <0.001; **<0.01; *<0.05; NS>0.05. All variables are standardised (see Methods). Blue lines indicate outward (from the colony) sections of trips and orange lines the return (to the colony) sections of trips. Dashed lines indicate upper and lower confidence intervals. Grey dots represent underlying data.

The marginal R^2 value (proportion of variance explained by the fixed effects only) indicates that the association between flight speeds and environmental variables accounts for 16% of the variance in the data. The conditional R^2 value (proportion of variance explained by both fixed and random effects) indicates that by allowing the intercept and slope to vary by Individual, and accounting for repeated measures between individuals and colonies, an additional 24% of the variance in the data is explained. The final model explains 41% of variance in the data.

4.3.3 Population- and individual-level relationships between flight speeds and environmental variables according to breeding status

Both breeding birds and birds that had failed in their breeding attempt decreased their flight speeds as they moved closer to fronts (Breeding: GLMM: X^2=125.7, d.f.=1, p<0.001; Failed: GLMM: X^2=224.8, d.f.=1, p<0.001) (Fig. 4.5 A and B) and increased their speed of travel over areas of greater front density (Breeding: GLMM: X^2=3692.0, d.f.=1, p<0.001; Failed: GLMM: X^2=2072.0, d.f.=1, p<0.001) (Fig. 4.5 C and D). The negative population level relationship with seabed slope (Fig. 4.6 A and B) was only significant in affecting flight speeds for breeding birds, with breeding birds slowing down over greater seabed slope gradients (Breeding: GLMM: X^2=10.8, d.f.=1, p<0.01; Failed: GLMM: X^2=1.2, d.f.=1, p>0.05). Conversely, the negative relationship with distance from coast (Fig. 4.6 E and F) was consistent across breeding status, with all birds slowing down the further they were from the coast (Breeding: GLMM: X^2=19.3, d.f.=1, p<0.001; Failed: GLMM: X^2=5.2, d.f.=1, p<0.05).

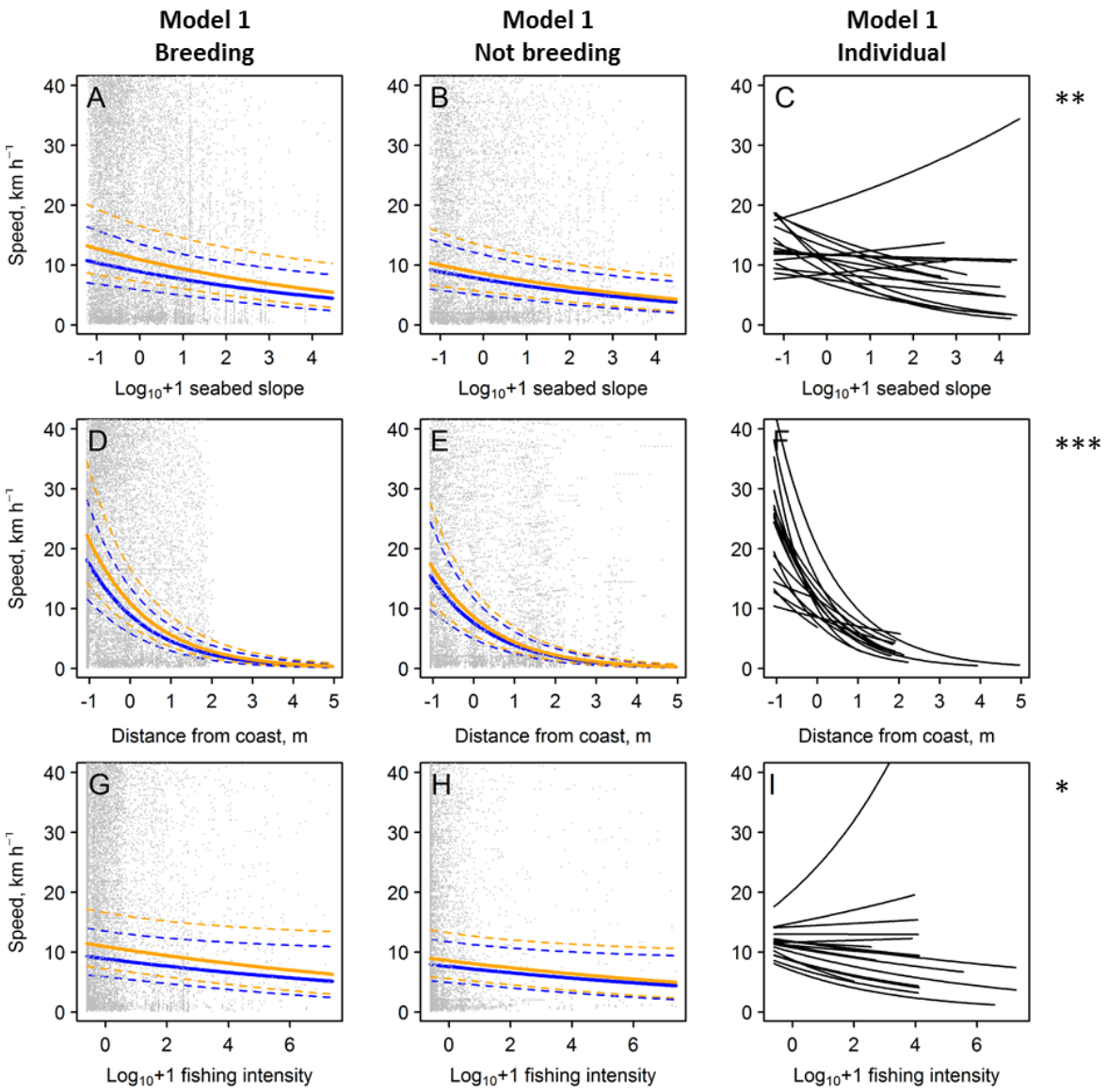

Figure 4.4: All data (both breeding and not breeding) (n = 17). Slopes represent population and individual level relationships with environmental variables. Stars to the right of plots indicate the significance of the relationships between fight speed and environmental variables: *** <0.001; **<0.01; *<0.05; NS>0.05. All variables are standardised. Population-level relationships (A, B, D, E, G and H) are indicated by the blue (outward-from-the-colony sections of trips) and orange lines (return-to-the -colony sections of trips). Dashed lines indicate upper and lower confidence intervals. Grey dots represent underlying data. Black lines (C, F and I) indicate individual-level relationships with environmental variables. Each line represents an individual bird.

However, the relationship between flight speeds and distance from coast was stronger in breeding birds than in birds that had failed in their breeding attempt (Fig. 4.6 E and F). There was no significant relationship between demersal fishing intensity and flight speeds of breeding birds and birds that had failed in their breeding attempt (Fig. 4.6 I and J) (Breeding: GLMM: X^2=0.42, d.f.=1, p>0.05; Failed: GLMM: X^2=3.2, d.f.=1, p>0.05). A different relationship was found between flight speeds and chl-*a* (Fig. 4.5 G and H) according to breeding status, with a positive relationship observed when birds were actively breeding (chl-*a*: GLMM: X^2=395.3, d.f.=1, p<0.001), compared to a negative relationship following breeding failure (chl-*a*: GLMM: X^2=473.0, d.f.=1, p<0.001). SST did not significantly influence flight speeds of great skuas during breeding or when birds had failed in their breeding attempt (Fig. 4.5 E and F).

At an individual level, most breeding birds shared the population level negative trend in relationships with seabed slope, distance from coast and demersal fishing intensity (Fig. 4.6 A and C, E and G, I and K). However, individual variation was evident, with some individuals showing positive relationships or no relationship with the three environmental conditions (Fig. 4.6 C, G and K). Alternatively, there appeared to be greater evidence of individual variation amongst birds that had failed in their breeding attempt (Fig. 4.6 D, H and L), particularly in the relationship with seabed slope (Fig. 4.6 D). Variation in behaviour was also apparent for some failed breeders in the relationship with distance from coast (Fig. 4.6 H) and demersal fishing activity (Fig. 4.6 L).

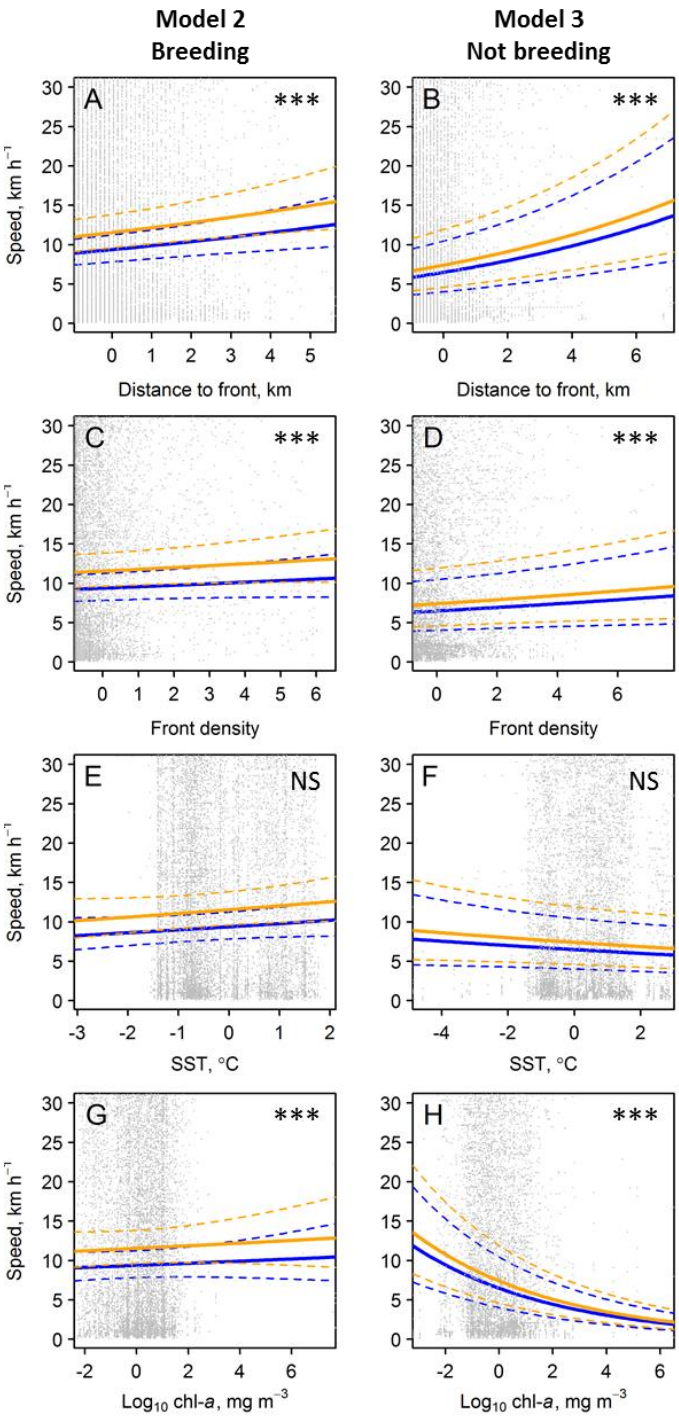

Figure 4.5: Results of the breeding (n = 17) and not breeding models (n = 9). Slopes represent population level relationships between fight speed and environmental variables. Stars within plot boundaries indicate the significance of the relationship: *** <0.001; **<0.01; *<0.05; NS>0.05. All variables are standardised. Blue lines indicate outward (from the colony) sections of trips and orange lines the return (to the colony) sections of trips. Dashed lines indicate upper and lower confidence intervals. Grey dots represent underlying data.

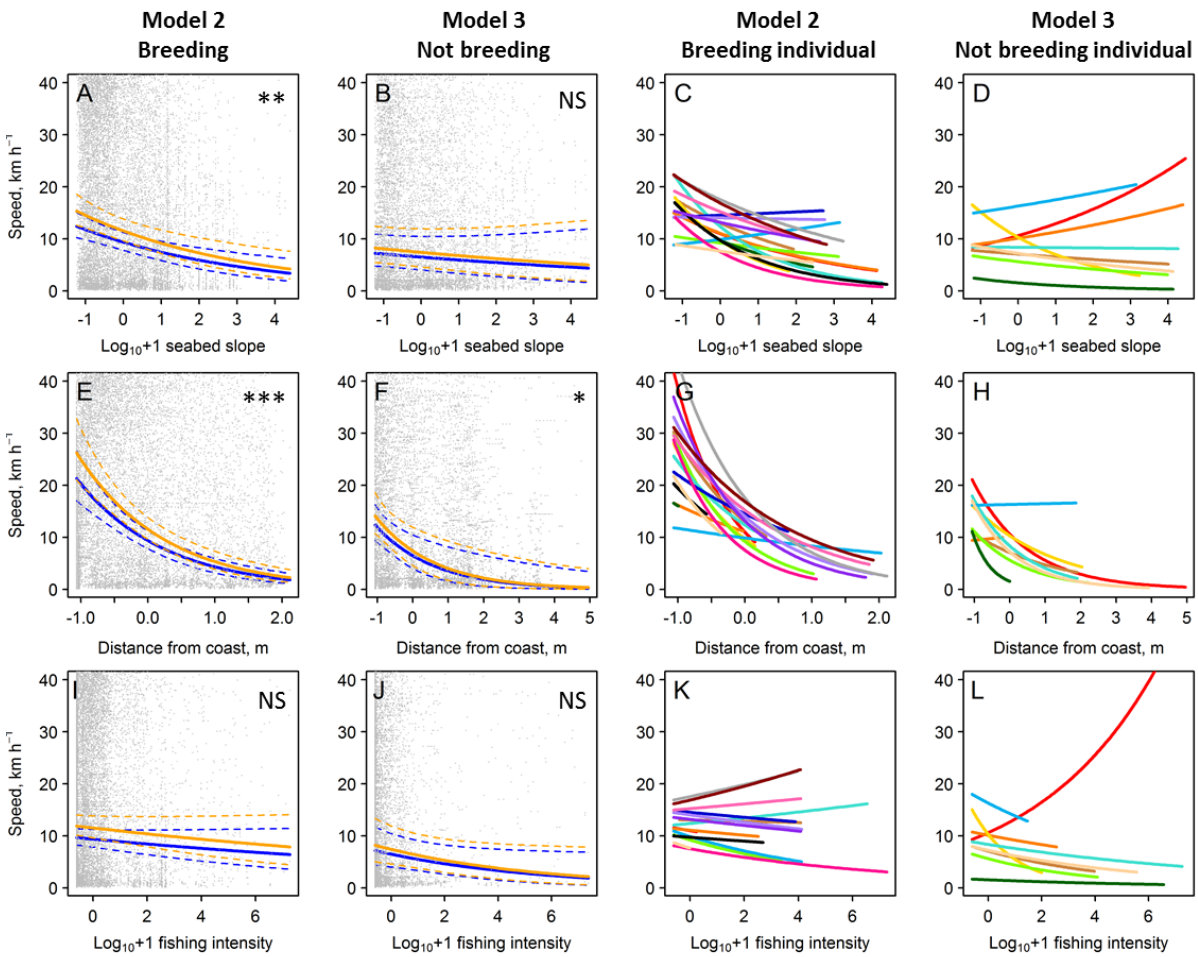

Figure 4.6: Results of the breeding (n = 17) and not breeding models (n = 9). Slopes represent population and individual level relationships with environmental variables. Stars within plot boundaries indicate the significance of the relationships between fight speed and environmental variables: *** <0.001; **<0.01; *<0.05; NS>0.05. All variables are standardised. Population-level relationships (A, B, E, F, I and J) are indicated by the blue (outward-from-the-colony sections of trips) and orange lines (return-to-the -colony sections of trips). Dashed lines indicate upper and lower confidence intervals. Grey dots represent underlying data. Lines in C, D, G, H, K and L indicate individual-level relationships with environmental variables. Each colour corresponds to an individual bird.

4.7 Discussion

Data presented here provide evidence for consistency in individual great skua foraging behaviour and preferred foraging habitat. Based on consistent differences between individuals, and between breeding birds and birds that had failed in their breeding attempt, these data indicate that different subsections of great skua populations could be differently affected by human activities at sea. My results support the theory that although populations may appear to be composed of generalist predators, these populations may actually comprise a population of individual specialists (Bolnick *et al.* 2002). This study represents a step forward from existing studies of this kind on seabird species (e.g. Patrick *et al.* 2014) as it follows individual birds throughout the breeding season from incubation to fledging or failure. As such, it has been possible to assess the consistency of foraging behaviour dependent on the breeding status of birds throughout the breeding period. The large numbers of repeated trips per individual have allowed robust estimates of consistency in behaviour to be established.

Results of this study indicate that breeding great skuas are more consistent in their foraging destinations and in their direction of travel from the colony than birds that have failed in their breeding attempt. There was no evidence of consistency in the duration or the distances covered during foraging trips. These results are similar to discoveries of consistency in breeding northern gannets *Morus bassanus* (Hamer *et al.* 2009; Patrick *et al.* 2014) and reinforces evidence presented in Chapter 3 that breeding birds behave differently to failed breeders. Evidence of spatial foraging consistency in great skuas suggests foraging site fidelity and potentially represents individual differences in preferred foraging habitat. Travelling along consistently different departure angles and to consistently different foraging

locations to other individuals suggests spatial partitioning of foraging habitat; perhaps aimed at reducing con-specific competition (Bolnick *et al.* 2002; Pettex *et al.* 2010; Patrick *et al.* 2014). However, consistent and repeated flight routes taken from colonies could result in increased (or decreased) risk of mortality from collision if individuals repeatedly fly through wind farm areas or through areas away from wind farms. Trip duration and the distances covered during foraging trips may lack consistency owing to the influence of weather conditions, which may affect the route taken to foraging areas (Patrick *et al.* 2014). For example, for some species (e.g. albatross and shearwaters), wind conditions have been shown to affect the route taken to and from foraging locations as individuals target favourable wind conditions (such as tail-winds) to increase flight speeds while reducing energetic costs (Weimerskirch *et al.* 1993; González-Solís *et al.* 2009; Wakefield *et al.* 2009). In addition, the time taken to capture prey is likely to vary, which may be particularly true for birds scavenging on discards having to wait until catches are hauled and discarding occurs (Hudson & Furness 1989).

My results indicate breeding birds travel faster than failed breeders (Fig. 4.2). This could be driven by the greater number of trips breeding birds are required to undertake in order to respond to the additional nutritional requirements of chicks and to minimise time spent away from the nest (Chapter 3). These results suggest that breeding birds have a more conservative foraging strategy than failed breeders, and expend more energy during foraging trips. As such, breeding birds may be less able to alter their foraging strategy, which could make breeding birds more vulnerable to negative effects of human activities at sea. For example, breeding birds may be less able to shift their foraging strategy or foraging location if disturbed by MRED activity. The implications of displacing breeding birds from

foraging areas, or reducing prey abundance through a discard ban (as a result of changes to the EU CFP), include the potential for reduced productivity and associated population declines (Bicknell *et al.* 2013). Additionally, a greater number of great skuas could begin to prey on other declining seabird species, which could lead to population declines of targeted prey species (Furness 1997; Votier, Heubeck & Furness 2008b; Horswill *et al.* 2014).

Repeatability in the foraging destinations of breeding great skuas could be driven by individuals targeting spatially predictable and persistent marine habitat. Breeding great skuas have a strong, significant, negative relationship with seabed slope gradient, with birds travelling more slowly over areas of greater seabed slope (Fig. 4.6 A). This is in contrast to birds that have failed in their breeding attempt that show no relationship with seabed slope (Fig. 4.6 B). Seabed slopes are static features known to create areas of elevated productivity when deeper, cooler, nutrient-rich water is upwelled to the surface (Weimerskirch 2007; Ainley *et al.* 2009; Scales *et al.* 2014b). This bathymetry-driven process generates a spatially predictable resource that breeding birds may exploit for foraging when constrained to regularly return to the colony to provision chicks. Conversely, failed breeders that are less constrained in their movements may have a reduced requirement for spatially predictable features. In addition to seabed slope, both breeders and failed breeders show a strong positive relationship with ocean fronts (Fig. 4.5 A and B). Birds reduce their flight speeds when closer to fronts, which suggest that these marine features are important locations for great skua foraging. My results provide evidence that spatially predictable, persistent and productive features play a key role in influencing great skua movements at sea and are important foraging habitat (Scales *et al.* 2014b; a). Based on this evidence, great skuas may be negatively affected by developments in the marine environment, such as MREDs, if they

are located in areas where persistent fronts and bathymetry-induced upwellings are located. Avoiding development in these locations could reduce negative impacts on great skuas, and other marine species potentially targeting these areas, particularly less flexible breeding birds. Frontal systems and upwelling areas could also be candidate locations for protection through conservation measures (see also Scales *et al.* 2014b).

Overall, the majority of population-level great skua relationships with environmental variables tend to be consistent for both breeding birds and those that have failed in their breeding attempt (Figs. 4.5 and 4.6). This suggests that great skua foraging behaviour and movements at sea may alter based on breeding status but that great skuas target similar resources and habitat throughout time. Consistent relationships throughout the breeding season were observed with proximity to front (Fig. 4.5 A and B), front density (Fig. 4.5 C and D) and distance from the coast (Fig. 4.6 E and F). Birds were found to reduce their flight speeds over areas of lower front density, indicating that great skuas may target areas of more distinct front systems, where foraging seabirds and prey may be aggregated in a concentrated area. As great skuas kleptoparasitise prey from other seabirds, an interesting area for further research would be to investigate the potential for correlation between great skua movements and other species' foraging distributions. This currently unexplored avenue would improve our understanding of what attracts great skuas to particular environmental conditions. Conversely, I have found evidence for different relationships between great skua foraging and chl-*a* production according to breeding status (Fig. 4.5 G and H). Chl-*a* production is a proxy measure of surface phytoplankton concentration and is directly related to rates of primary productivity (Scheffer, Rinaldi & Kuznetsov 2000). Chl-*a* production has also been shown to be highly correlated with zooplankton abundance (secondary

productivity), which is a key food source for fish preyed upon by seabirds, such as sandeels. Given the evidence that biomass of fish stocks are strongly linked to primary productivity, chl-a production is an appropriate proxy to identify areas at sea where bottom-up ecosystem processes drive increased seabird prey abundance (Ware & Thomson 2005; Frederiksen *et al.* 2006). However, I found that during breeding, birds appear to reduce flight speeds over areas of low chl-*a* production, which was unexpected as previous research has shown that some seabird species increase their foraging activity in response to greater chl-*a* abundance (Pinaud & Weimerskirch 2005). Conversely, once great skuas had failed in their breeding attempt, birds travelled more slowly over areas of greater chl-*a* production. This could reflect changes in diet during the breeding season or could be the result of natural seasonal fluctuations in chl-*a* abundance. For example, for chl-*a* to drive foraging behaviour, a threshold of chl-*a* productivity may need to be attained, above which primary productivity is converted into fish biomass. The required threshold of chl-*a* may only be attained during the later summer period, which could coincide with when the majority of great skuas fail in their breeding attempts. Alternatively, there may be a time lag between high levels of primary productivity and the presence of seabird prey.

As has been suggested in other studies of great skuas (Votier *et al.* 2008a), I have found evidence of intra-population variation; with individuals exhibiting strongly consistent foraging behaviours, and in many cases targeting consistently different environmental conditions. These consistent differences between individuals suggest the existence of individual foraging and/or dietary strategies. For example, it is known from extensive dietary studies that great skuas forage extensively on fisheries discards; primarily in the form of discarded demersal whitefish (Votier *et al.* 2008a). Based on this knowledge it was

surprising to find only a marginally significant relationship between flight speed and demersal fishing activity when analysing all data (Fig. 4.4 G and H), and non-significant relationships when analysing the two datasets split according to breeding status (Fig. 4.6 I and J). However, these weak relationships can be attributed to the broad individual variation observed in great skua relationships with fishing activity (Fig. 4.4 I; Fig. 4.6 K and L). For example, the majority of individuals reduce their speed of travel when over areas of more intense demersal fishing activity, which likely indicates individuals foraging on discards. Alternatively, some individuals continue to travel quickly over areas of intense fishing activity, therefore appearing not to target discards. These results imply the presence of individual foraging strategies and differences in prey targeted within great skua populations. It is important to bear in mind that the fisheries data incorporated in analyses may influence the significance of the relationship observed between great skuas and fisheries activity, as only data from large (>15m) vessels are included. Interactions with inshore creel boats and other small vessels that may represent foraging opportunities for great skuas are not considered here as very little data is available on their activity. Intra-population variation was also observed in great skua relationships with seabed slope and distance from the coast (Fig. 4.4 C and F; Fig. 4.6 C, D, G and H). Distance from the coast was included in analyses as dietary studies indicate that within what is considered to be a generalist foraging species, a minority of great skua individuals are seabird prey specialists. These specialist foragers target coastal seabird colonies to prey on eggs, chicks and adult seabirds rather than foraging out at sea (Votier *et al.* 2004a, 2006). My results suggest that it may be possible to identify these dietary specialists by assessing individual slope relationships with distance from coast; with seabird specialists potentially identified as individuals not travelling far from the shore. Alternatively, it may be that these individuals are not targeting seabirds as

prey but could be foraging on discards from inshore creel boats (H. Wade pers. obs.). Overall, my results provide evidence for consistent individual variation in foraging strategies and diet. I also suggest that it may be important to reconsider weakly significant or non-significant population-level relationships when identifying important environmental variables, as individual variation may affect model outcomes.

In this study I have shown that individuals consistently differ in their foraging behaviour and movements at sea along environmental gradients, and that this intra-population variation can result in the repeated use of different foraging locations; especially in relation to predictable and persistent habitat features. My results indicate that although individuals are flexible in their foraging tactics, individuals also show a degree of foraging site fidelity and potential dietary consistency (Votier *et al.* 2010). An implication of this study is that subsections of great skua populations may be differently affected by human activities in the marine environment. This study illustrates the importance of improving our understanding of both population- and individual-level responses to the marine environment to more fully understand species behaviour and accurately predict potential effects of human activities at sea. Consideration of population-level responses to environmental variables are key to developing SDMs for incorporation in marine spatial planning and the designation of marine protected areas. However, as these population-level studies reveal little about individual strategies, I suggest that intra-population variation is also important to consider to better inform marine management decisions.

Chapter 5:

Life in the fast lane: how seabirds use a high velocity current tidal channel

Abstract

Marine renewable energy developments (MREDs) are becoming an increasing feature of the marine environment. Compared with wind and tidal-range energy technologies, tidal-stream energy generation is a relatively new industry, with devices still in the development and testing phase. Owing to the small number of deployed tidal-stream energy generating devices, the absence of operating arrays and the technological challenges of investigating seabird use of the water column in high flow habitats, the potential effects of tidal-stream energy developments (TEDs) on seabirds are not fully known. I used land-based vantage-point surveys to investigate seabird behaviour in a high velocity current tidal channel that is leased for a TED. Understanding how seabirds use high velocity current flows will enable more accurate predictions of which species may be most vulnerable to TEDs. I found that different species of seabird were present in the tidal channel throughout the year, but that populations of black guillemots and shags were present all year. Not all species observed in the tidal channel used the site for foraging and only low numbers of birds were found to dive in current velocities optimal for commercially generating energy from tidal-stream devices. These findings suggest that high-energy tidal-streams are generally not an attractive foraging habitat for most species of seabirds, and imply that a low proportion of birds are likely to be at greatest risk of collision or entanglement with devices. Conversely, most birds tended to dive facing into the oncoming current, which may suggest that diving species are less able to detect devices if approaching facing away from them; therefore placing them at greater risk of entanglement and collision. Overall, I found that effects of TEDs are species-specific. I

suggest increased research, and methodological developments may be required to more reliably predict how seabirds may be affected by TEDs.

5.1 Introduction

Marine renewable energy developments (MREDs) are becoming an increasing feature of the marine environment. Scotland has developed an advanced marine plan to capitalise on its renewable energy resources and the Crown Estate is working in partnership with the Scottish Government to realise their aim of generating 100% of electricity through renewable sources by 2020 (Scottish Government 2011b). Resources for generating energy from tidal currents (known as tidal-stream energy) occur at a number of discrete sites around the UK (The Crown Estate 2012). Compared with wind and tidal range energy technologies, tidal-stream energy generation is a relatively new industry, with devices still in the development and testing phase. The Pentland Firth and Orkney waters in north Scotland is the first area to be made available for commercial scale tidal-stream developments (The Crown Estate 2011) and represents the greatest density of leased and proposed tidal-stream energy development (TED) sites in the UK (The Crown Estate 2012). The Inner Sound tidal-stream development, located in the Pentland Firth, is the largest planned tidal-stream energy project in the world, with aims to deploy up to 398 tidal turbines equating to 398MW of tidal-stream energy (Meygen Ltd. 2014).

Commercial tidal-stream energy generation requires mean spring tide current velocities greater than 2 ms^{-1} (Fraenkel 2006). Current speeds of this magnitude are caused by tidal currents interacting with coastal topography and bathymetry. Tidal current flow is constricted, either by being forced around shallow headlands or through channels, which results in the acceleration of current velocities. The resulting dynamic hydrography

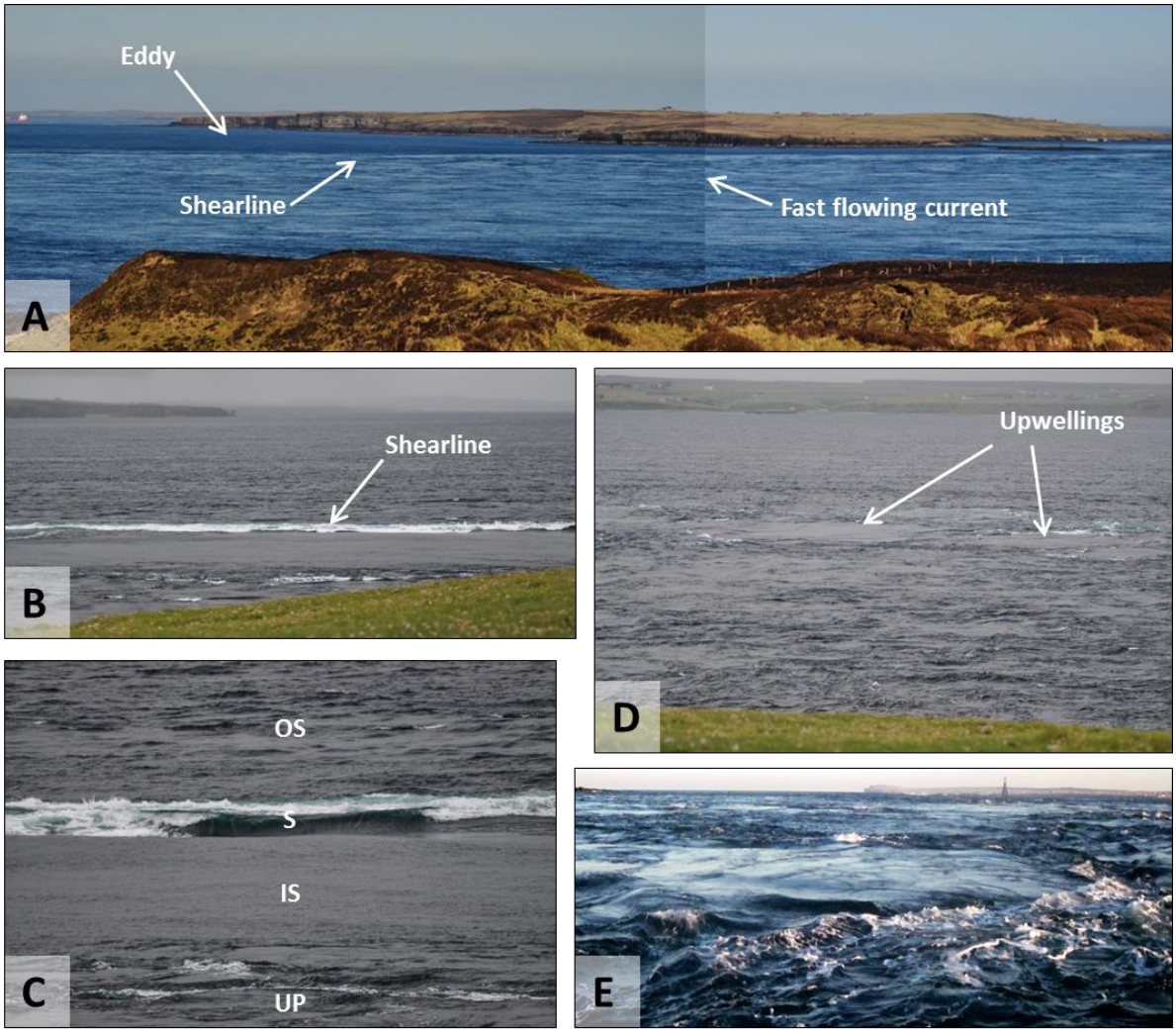

Figure 5.1: A) View from VP1 during ebb tide illustrating the presence of an eddy to the left of Stroma island, a shear line running right to left in front of the island and then the faster flowing current indicated by the lighter blue water. B) and C) View from VP2 showing the turbulent shearline (S), smooth water of the inner shearline (IS), choppy water of the outer shearline (OS) and the upwellings (UP) on the inside of the shearline near the Stroma eddy. D) and E) Upwellings and kolk boils.

generates localised oceanographic features, including upwellings and kolk boils, shearlines and fronts, and eddies (Wolanski & Hamner 1988; Bailey & Thompson 2006; Johnston & Read 2007). These fine-scale (10s to 100s m) oceanographic features manifest on the sea surface as visibly slick areas indicating upwelling water, turbulence and standing waves along

shearlines, and slick laminar flows where the current velocity is greatest (Fig. 5.1). The hydrodynamic processes generating these features influence the availability of prey items by retaining and aggregating plankton and fish through biophysical coupling (Alldredge & Hamner 1980; Wolanski & Hamner 1988; St. John *et al.* 1992). For example, high current speeds transport large volumes of water containing plankton and fish. These prey items are then advected and concentrated in slow circulating eddies that form in the lee of headlands or islands (Wolanski *et al.* 1996). In addition, distinct water masses flowing in differing directions, and at different velocities, create turbulence and bubbles where they meet. The resulting barrier throughout the water column manifests as a turbulent shearline on the surface, along which prey items may aggregate through convergence (St. John *et al.* 1992; Wolanski *et al.* 1996; Johnston, Thorne & Read 2005a). The boundary feature, present throughout the water column, may also trap plankton and fish unable, or unwilling, to cross this functional barrier (Johnston *et al.* 2005a).

Biophysical coupling and subsequent bottom-up biomass accumulation across trophic levels can provide predictable spatial and temporal foraging hotspots in a heterogeneous environment with patchy resources (Boyd 1996; Johnston & Read 2007). Evidence presented in Chapter 4, and in other published studies, indicate that seabirds repeatedly forage in areas of predictable prey resources to maximise their energetic gain, whilst minimising foraging effort (Hamer *et al.* 2001, 2007). The importance of meso-scale oceanographic features (10s to 100s km), such as fronts, in influencing seabird foraging distributions is well known (Hunt *et al.* 1998; Weimerskirch 2007; Scales *et al.* 2014a). However, the influence of fine-scale oceanographic features is poorly understood (Johnston, Westgate & Read 2005b; Cox, Scott & Camphuysen 2013; Drew, Piatt & Hill 2013); particularly in high current velocity habitats (see Chapter 2). This is owing to the difficulties

associated with sampling abiotic and biotic variables in ephemeral and physically dynamic environments (Embling *et al.* 2012). Fine-scale surface features generated by tidal currents are likely to provide visible cues for foraging seabirds; particularly for diving species whose prey is not always visible from the air (Decker & Hunt 1996) and may be particularly important during energetically demanding periods (for example, during the breeding season: Hunt & Harrison 1990). Several studies have found that seabirds target tidally generated surface features, with some species adjusting their foraging trips to coincide with tide phases (Slater 1976; Irons 1998; Hunt *et al.* 1998; Durazo, Harrison & Hill 1998; Ladd *et al.* 2005; Robbins 2012a; b). Despite evidence for seabird use of locations with tidally driven hydrodynamic regimes, Chapter 2 indicates that there is a lack of knowledge regarding how seabirds use high-velocity current habitats, and how current speeds influence seabird foraging behaviour (Heath, Gilchrist & Ydenberg 2006; Furness *et al.* 2012). There is some evidence that seabirds may vary their foraging behaviour in relation to current velocity. For example, Heath & Gilchrist (2010) propose that increased energy costs deter common eiders from foraging in polynyas when current speeds reach greater than 1 ms^{-1}. In addition, studies of Magellanic penguins suggest individuals avoid greater energy costs associated with swimming in areas of high current velocity by timing the direction of their movements to match tidal flow (Rey *et al.* 2010), by swimming closer to shore where current velocities are less, or by walking on land where possible (Wilson *et al.* 2001). More research in this area is required; particularly given the likelihood that these dynamic environments will be targeted for deployment of TEDs.

Owing to the small number of deployed tidal-stream energy generating devices, the absence of operating arrays, and the technological challenges of investigating seabird use of the water column in high flow habitats, the potential effects of TEDs on seabirds are not fully

known (Cook *et al.* 2012a; McCluskie *et al.* 2013; Broadhurst, Barr & Orme 2014; Copping *et al.* 2014). This is despite several leased and proposed sites being located within, or in close proximity to, existing special protection areas (SPAs) and draft marine SPAs designated specifically to safeguard breeding populations of internationally important seabird species (JNCC 2014a; Scottish Natural Heritage *et al.* 2014). SPA legislation (EC Birds Directive 2009/147/EC) requires that developments do not damage the integrity of protected seabird populations (Scottish Natural Heritage 2014). Evidence from the offshore wind energy industry suggests that potential effects of tidal-stream energy developments (TEDs) on seabirds could include: 1) disturbance and displacement from foraging and loafing areas; 2) risk of entanglement and collision with devices; and 3) altered foraging opportunities if devices modify current flows (Furness *et al.* 2012). Results presented in Chapter 2 (Table 2.4), and in recently published research, suggest that seabird species most likely to be vulnerable to potential effects of TEDs include diving auks, cormorants and diver species (Furness *et al.* 2012; Waggitt & Scott 2014).

Disturbance and displacement of seabirds could occur as a result of increased vessel traffic and activity in development sites during construction, maintenance and monitoring, and during decommissioning of devices (Table 2.4; Furness *et al.* 2012; McCluskie *et al.* 2013). Evidence indicates that seabird responses to vessel traffic are species-specific, with diver species identified as the most sensitive to disturbance (Schwemmer *et al.* 2011; Mendel *et al.* 2014).

Collision with devices will only occur if species spatially overlap with TEDs, therefore diving species using high current flow (>2 ms^{-1}) habitats will be most vulnerable (Table 2.4; Furness *et al.* 2012; Waggitt & Scott 2014). Risk of collision and entanglement are therefore species-

specific and will likely vary by age and life-stage of the individual. TEDs will also alter hydrodynamic processes by removing energy from tidal currents, transform current flow patterns caused by introduced obstacles in the form of turbines, and modify sediment transport processes (Gill 2005; Langhamer *et al.* 2010; Shields *et al.* 2011; McCluskie *et al.* 2013; Scott *et al.* 2014). These effects on current flow could alter the occurrence or location of visible surface cues targeted by seabirds as predictable foraging locations. Increased sediment suspension in the water column could affect prey detection and/or reduce diving species' ability to detect and avoid collision with devices (Shields *et al.* 2011; Scott *et al.* 2014). Potential positive effects of TEDs on seabirds include increased foraging opportunities if devices act as fish aggregation devices (Wilhelmsson *et al.* 2006; Langhamer & Wilhelmsson 2009), artificial reefs (Langhamer *et al.* 2009; Reubens *et al.* 2013a; b) or *de facto* marine protected areas (Inger *et al.* 2009; Grecian *et al.* 2010).

Owing to the rapid development of the tidal-stream renewable energy industry it is imperative that potential risks to seabirds are identified and quantified. Based on knowledge that potential effects on seabirds will be dictated by spatial overlap with TED sites and species-specific behaviour, I aim to investigate seabird vulnerability to TEDs by observing seabird distribution and behaviour over 1 year in a tidal channel that has been leased for a TED. To establish which species might be vulnerable to disturbance or displacement I aim to 1) identify species present in the tidal channel and 2) identify which species forage in the site. To identify which species may be at risk of collision or entanglement with devices I aim to 3) identify the range of current speeds encountered by diving species during foraging and 4) investigate diving behaviour in relation to current flow. To investigate which species could be vulnerable to modified current flows I aim to 5) identify species associating with visible fine-scale surface features.

5.2 Materials and methods

Owing to the infancy of the tidal-stream industry, no standardised or established method exists for surveying seabirds in the complex and dynamic high current flow habitats in which TEDs will be located, or for monitoring seabird associations with fine-scale visible surface features. Therefore, the methods presented here are novel and have been adapted from methods in Jackson and Whitfield (2011) and Robinson and Cook (2011), and methods developed by A. Robbins (pers. comm.). The methods used were developed specifically for the site surveyed. Appropriate survey methods for other TED sites are likely to be site-specific.

5.2.1 Study site

Shore-based surveys were undertaken from a fixed vantage point location on the north coast of mainland Scotland (hereafter known as VP1) located to the west of Gill's Bay (58°39'N, 3°7'W), and from Mell Head on the island of Stroma (hereafter known as VP2) (58°40'N, 3°8'W) (Fig. 5.2 A and B). Both vantage points overlooked the Inner Sound of the Pentland Firth and included overlap with a 398MW capacity leased tidal-stream development site (Meygen Ltd. 2014). The study sites had a 90° viewshed and were divided into a grid (Fig. 5.2 C and D). Owing to differences in vantage point elevation (VP1 elevation: 31.5 m above mean sea level (AMSL); VP2 elevation: 10 m AMSL) the VP1 study site had a viewshed radius of 2 km and the VP2 study site a viewshed radius of 1.5km (Fig. 5.2 C and D). This resulted in a small overlap of these two viewsheds in the middle of the Inner Sound. The study sites were characterized by a range of current flows with maximal velocities amongst the highest in UK waters; reaching approximately 4.5 ms^{-1} (Easton *et al.* 2011) (Fig. 5.3). The current flows interacted with bathymetry and coastal topography to create visible fine-scale

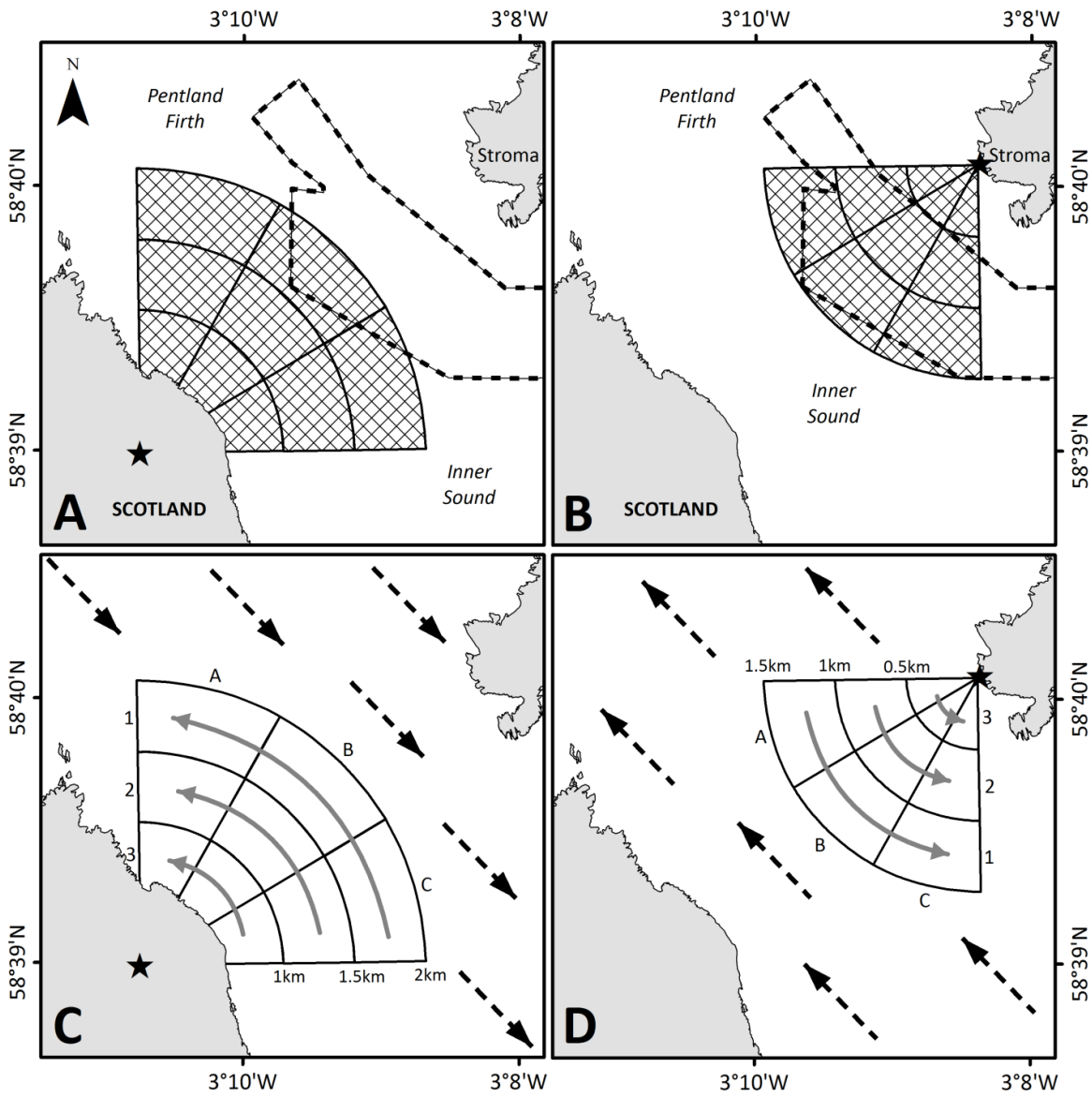

Figure 5.2: Vantage point locations (stars) on the mainland (VP1) (A) and on Stroma (VP2) (B) overlooking the gridded study sites (hatched) located in the Inner Sound of the Pentland Firth. A leased 398MW capacity tidal-stream development site is also shown (dashed line). (C) and (D) illustrate the method of scanning the study site in the opposite direction (grey curved arrows) to the current flow (black dashed arrows) depending on stage of tide.

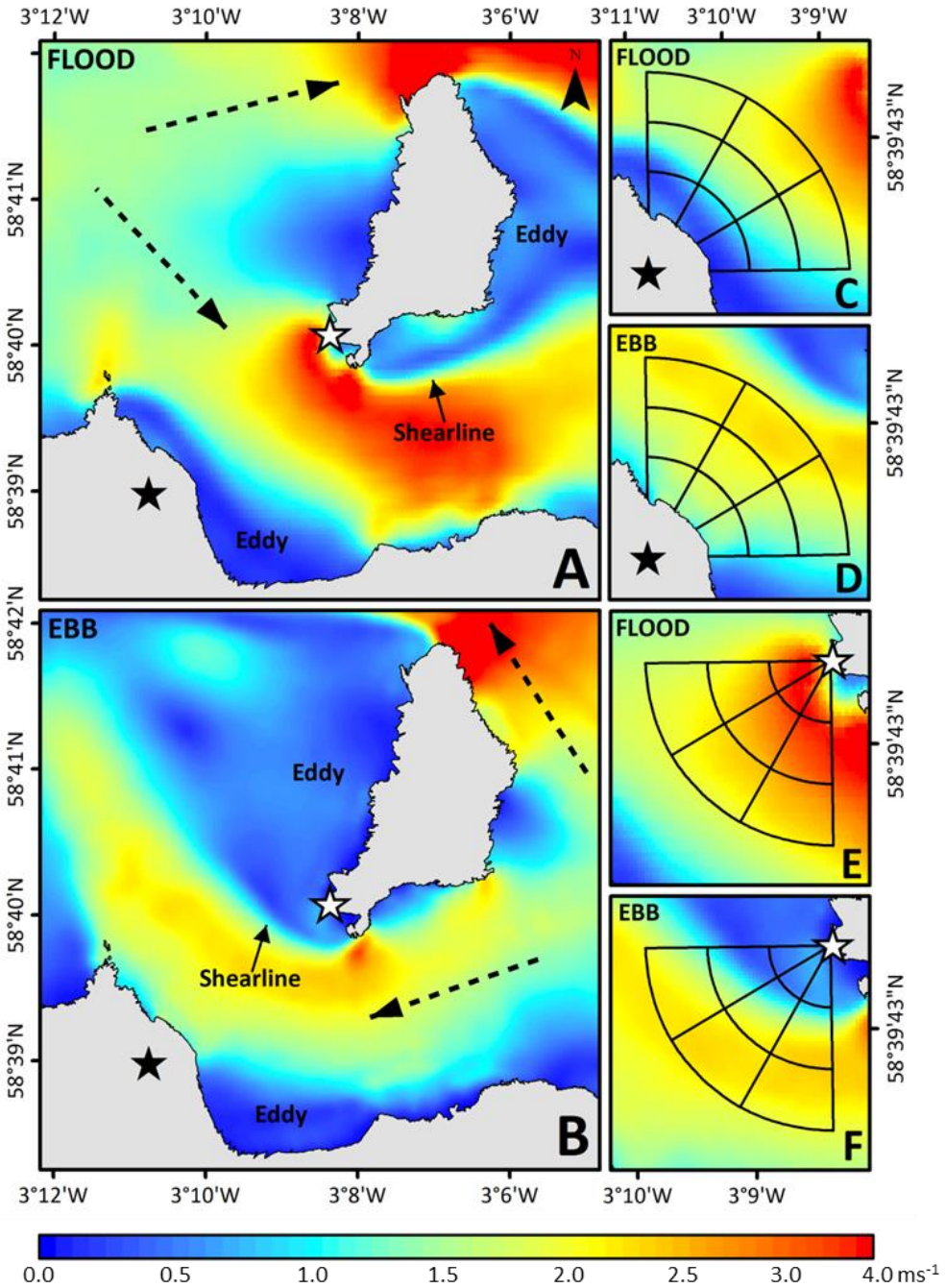

Figure 5.3: Examples of current speeds during flood (A, C and E) and ebb tide (B, D and F) during a peak spring tide. The direction of the main current flow is indicated by dashed arrows. Examples of current velocities in the VP1 study site during flood (C) and ebb (D) tide, and in the VP2 study site during flood (E) and ebb (F) tide. VP1 indicated by black star and VP2 indicated by white star.

oceanographic features, including eddies, upwellings, fast laminar flows, choppy fast flows and turbulent shearlines (Fig. 5.1).

5.2.2 Survey methodology

From VP1, four blocks of surveys were undertaken over a twelve month period to ensure ecologically important phases in the seabird life cycle were incorporated. In addition, one survey block in summer was undertaken from VP2. Each block of surveys was designed to comprise fourteen consecutive days of a spring to neap tidal cycle. However, consecutive days of surveys were often not possible owing to poor weather conditions. Surveys were not conducted in weather conditions greater than sea state 4 on the Beaufort scale, or if wind speeds were greater than 32 kmh^{-1} (equivalent to 20 mih^{-1}). Wind speeds greater than 32 kmh^{-1} caused instability of the telescope, which reduced the observer's ability to detect and identify birds. To account for poor weather conditions and to ensure that survey blocks remained within distinct seasons, a period of 30 days was allocated in which to complete the 14 days of surveying. Surveys targeted a range of tidal phases (ebb, flood and slack) throughout the spring to neap cycle, to account for associated differences in current velocity and direction. The study sites were split into 9 grid cells (Fig. 5.2) and surveys consisted of scanning the site using a telescope (Swarovski ATS HD 80mm angled), fitted with a variable zoom lens (20-60x), to record the distribution and behaviour of birds interacting with the water. The aim of the method was to record every bird interacting with the water in the site (excluding birds only seen in flight) during each survey. As such, the furthest distance band was scanned first as it required more survey effort to minimise the number of undetected birds; the middle and nearest band were scanned second and third respectively. The nearest distance band was scanned at x20 zoom, whilst the middle and furthest distance bands were scanned at x30 zoom. No distance-correction methods were applied to the observation data as an environmental gradient (in the form of varying current velocities and associated visible surface features) across the study site likely violated the assumption of equal distribution of birds. Bearings to chosen features on the horizon (i.e. landmarks on

islands) and range finders (JNCC 2011) (to identify distance bands), were used to define the study site grid. These bearings were used to make permanent reference marks using waterproof paint at 0°, 30°, 60° and 90° on the swivel head of the spotting scope to indicate the boundaries between cell segments. These marks were validated for VP1 before data collection commenced by using a boat as a marker in the study area at fixed GPS locations corresponding to the intersections between cell segments and distance bands in the grid. Photographs of the boat were taken through the scope to record its location in the field of view, which were subsequently used in the field as a reference guide to assigning birds to the appropriate distance bands. The permanent reference marks on the tripod and photo reference guides ensured that during periods of reduced visibility, when the horizon could not be seen, observations could continue. In addition they provided a rapid reference tool to quickly assign birds to their appropriate grid cell. Marker buoys to mark the study site could not be used as they would have presented a shipping hazard and would likely have been pulled under the surface by currents during peak flows. Permanent ground fixings were used to hold the telescope tripod in place and the tripod was leveled prior to each survey to ensure a consistent field of view. Calibrating the telescope and tripod in these ways ensured repeatability and consistency in the field of view, and efficient assignment of birds to the appropriate grid cell. The sites were scanned in the opposite direction to the current flow to reduce the risk of birds being missed or double-counted (A. Robbins pers. comm.) (Fig. 5.2 C and D). The observer's scanning rate aimed to be of a speed that minimized the possibility of missing diving birds but balanced the need to complete the scan as quickly as possible in order to replicate a 'snapshot' of bird distribution and abundance. Rapid completion of surveys also ensured minimal differences in environmental conditions between each distance band scan. Surveys were separated by at least 1 hour to ensure distinct periods were surveyed and to reduce observer fatigue. During surveys, the observer

recorded the species of detected birds, the grid cell location of the bird, its behaviour, and if the bird was associating with any visible surface features. Adapting definitions given in the methods of Holm and Burger (2002), foraging behaviour was defined as holding food, diving from sitting on the water, plunge diving, surface pecking whilst sat on the water, head-dipping, eating, and surfacing from a dive. Non-foraging behaviour was defined as loafing, flapping, landing, taking off, swimming, interacting with other individuals, calling and preening. The direction in which the bird was moving was recorded, as was the direction the bird was facing and whether the bird was moving with or against the tide. Environmental variables were recorded at the beginning of each survey and if conditions changed during the survey. Counts of birds in each grid cell were converted to a measure of birds observed per km^2. Scaling the counts to an equal area measure allowed comparison between grid cells and surveys.

5.2.3 Assessing potential vulnerability to disturbance or displacement

In order to assess potential vulnerability to disturbance and displacement, species observed foraging and species observed not foraging activities (e.g. loafing or preening) in the study sites were recorded. For each season, a percentage was calculated for each species, which indicates the percentage that species comprised of the total number of birds observed. The number of birds per km^2 observed foraging and not foraging in each study site, and in each season, was calculated for each species.

5.2.4 Assessing potential vulnerability to collision and entanglement

In order to assess potential vulnerability to collision and entanglement with tidal-stream energy devices, mean current velocities birds were observed foraging in were calculated. As current speeds of 2 ms^{-1} are required to generate commercially viable energy (Fraenkel

2006), only species foraging in current velocities of 2 ms^{-1} or greater are likely to be most vulnerable to collision and entanglement. Measurements of current velocities recorded concurrently with surveys was not possible, therefore current velocities in the VP1 study site were extracted from a two dimensional hydrodynamic (HD) model of the Pentland Firth (Easton, Woolf & Bowyer 2012). The HD model was calibrated using tide gauge data from the British Oceanographic Data Centre (BODC) and acoustic Doppler current profilers (ADCPs) located in the Pentland Firth. A comprehensive description of the model can be found in Easton et al. (2012). Current velocity data were not extracted for the VP2 study site as model accuracy was not reliable in the areas of intricate coastline and bathymetry characteristic of the VP2 study site (M. Easton pers. comm.).

HD model data covered a neap to spring tidal cycle in 2011. Tide elevation data were extracted from the nearest BODC tide gauge (Wick: 58°26'N, 3°5'W) that covered the mainland vantage point survey period (April 2012 to January 2013). The model data were matched to the tide gauge data (Fig. 5.4 A). My focus was on matching the timings of the peaks and troughs across the two datasets as they indicate the phasing of the tidal cycle, which is proposed to be the most important relationship in replicating current flow (Fig. 5.4 B). Elevation of the tidal cycle was deemed less important in matching the two datasets as the HD model sometimes overestimated the tidal amplitude (Easton *et al.* 2012). Survey start times were rounded to the nearest 15 min (for example, 12:00, 12:15, 12:30 or 12:45) in order to match the tide gauge sampling intervals. These were then marked at the relevant point on the plotted tide cycle (Fig. 5.4). Survey start times were matched to the nearest 10 min time-step in the model (for example 12:00, 12:10, 12:20, 12:30 etc.). Current velocity data for the survey site were extracted from the HD model at the matched survey start time-step, and for subsequent time-steps falling within the duration of the survey (Fig.

5.4 B). This process was repeated for all survey bouts. Current flow data for the survey site were sampled from the model at points corresponding to the underlying model construction (Fig. 5.5). Mean current velocity of each grid cell during a survey was calculated by averaging the current velocity values from all points within each cell, across all time-steps extracted for that survey (Fig. 5.5). These data were combined with bird counts in grid squares to assess vulnerability to collision and entanglement.

Tendency for individuals to dive and surface with or against the current flow was assessed using binomial tests. All dives from sitting on the surface of the water and all observations of surfacing in both study sites were included.

5.2.5 Vulnerability to modified current flows

During surveys of the VP1 study site, birds were recorded as foraging in association with choppy fast flowing water (CFF), eddies (ED), shearlines (S), slick fast flowing water (SFF), upwellings and kolk boils (UP), and areas with no visible surface feature (NF). In some cases, association with features was not recorded during surveys (NR). Due to the greater proximity of VP2 to faster flowing water and the associated shearline, birds in the VP2 study site were categorised as foraging in association with choppy fast flowing water (CFF), eddies (ED), the shearline (S), the inner shearline (the area directly next to the turbulent shearline closest to the observer) (IS), the outer shearline (the area directly next to the turbulent shearline furthest from the observer) (OS) or with no visible surface feature (NF). In each

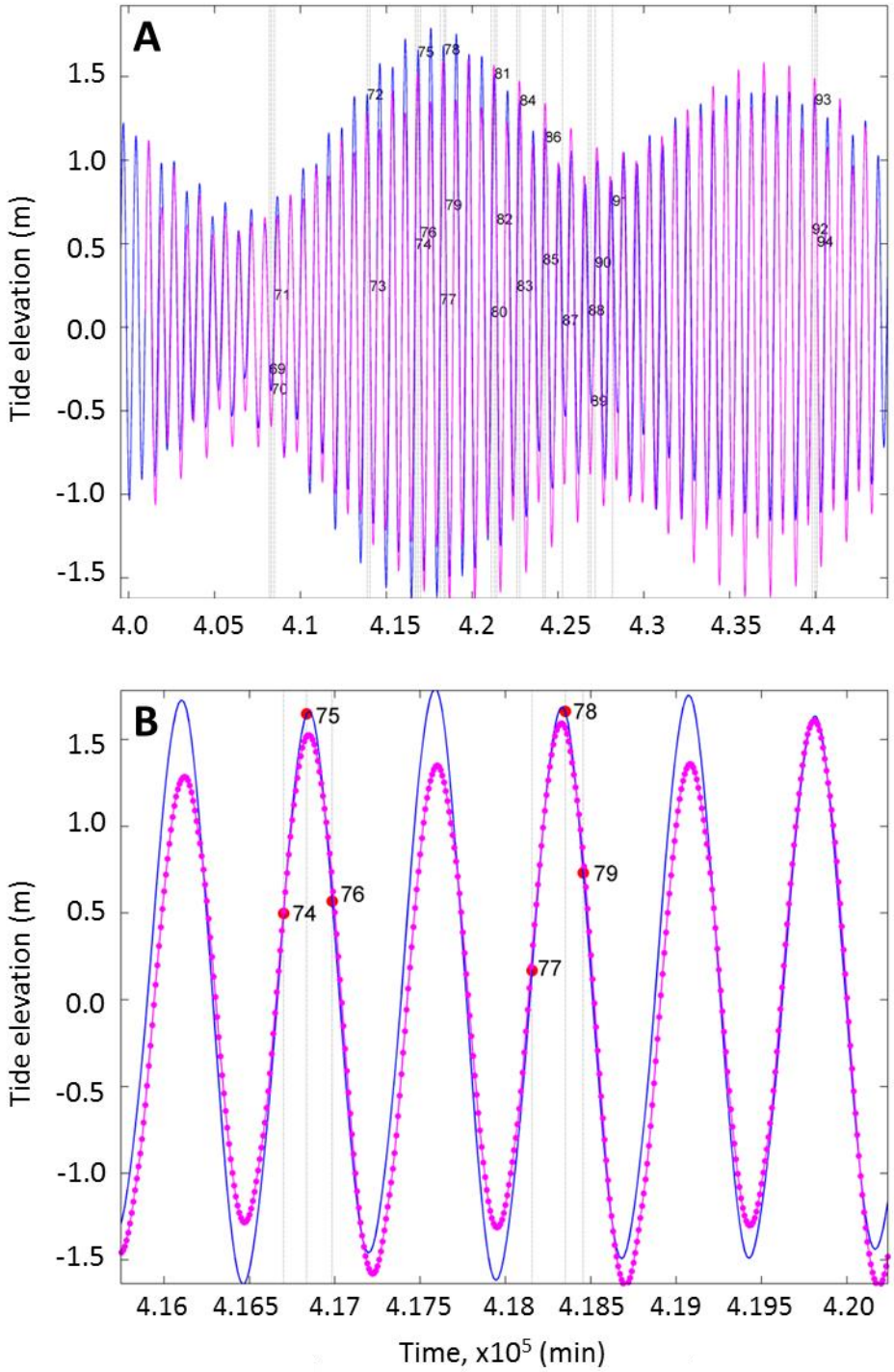

Figure 5.4: Matching hydrodynamic model data (pink line) and tide gauge data (blue line). Surveys during the tidal cycle (blue line) are indicated by black numbers and/or red dots. Numbers indicate the survey ID. Pink dots represent 10 min time-steps in the model data (pink line).

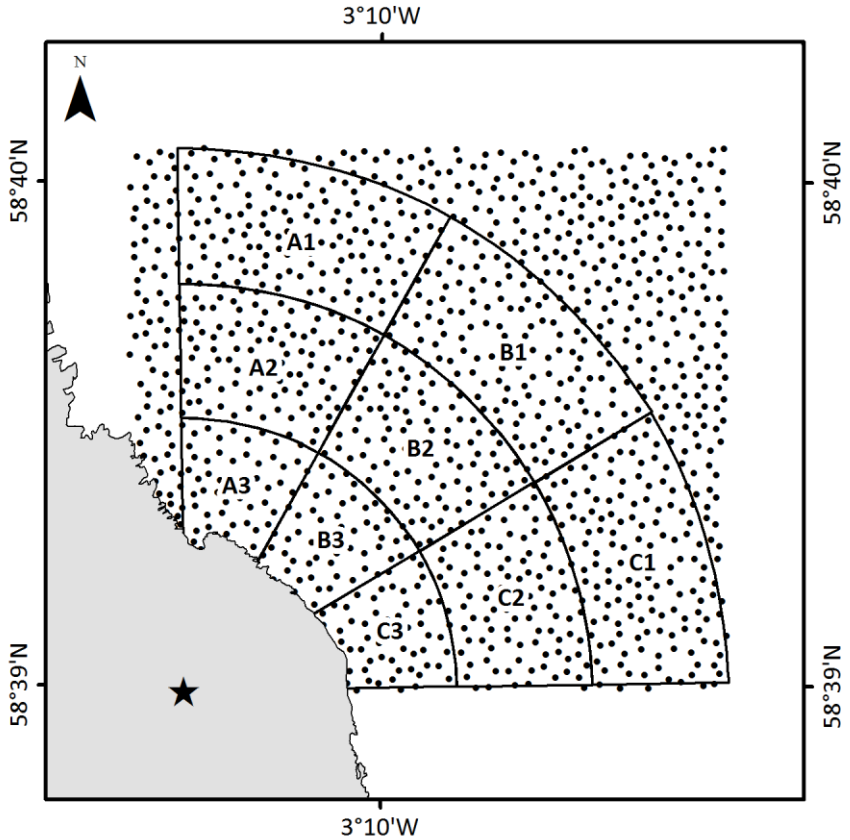

Figure 5.5: Current velocity sampling points used to calculate mean current velocities in each grid cell. Black lines indicate the grid of the VP1 survey site. Black dots indicate the current velocity sampling locations in the hydrodynamic model corresponding to locations within the survey grid. The black star indicates the location of VP1.

study site and for each species observed foraging, the percentage of individuals associating with each surface feature was calculated. Chi-squared contingency tests were conducted to assess differences between species associations with visible surface features. To account for small sample sizes, *p*-values were calculated using Monte Carlo simulations with 2000 replicates (Hope 1968).

5.3 Results

105 surveys were completed from VP1 over the four survey periods. 32 surveys were completed in spring (April to May 2012), 36 in summer (July to August 2012), 26 in autumn (October to November 2012) and 11 in winter (January 2013). 3010 birds were detected in the VP1 study site during this time, with 20 species identified over the four seasons (Table 5.1). Some individuals were identified only to the family level or were unidentified (n = 7). Owing to variable weather conditions between surveys and varying numbers of birds in the study site, survey duration ranged from 14 – 58 min, with a mean of 32 min. 30 surveys were completed from the VP2 vantage point during the summer (July 2012), with 1353 birds detected in the VP2 study site. 12 species were identified (Table 5.1), with some individuals identified only to the family level. Variable weather conditions between surveys and varying numbers of birds in the study site caused survey durations to range from 11 – 36 min, with a mean of 21 min.

5.3.1 Vulnerability to disturbance or displacement

5.3.1.1 Species observed in the study sites

Species presence in VP1 varied by season. During spring, both fulmar and shag amounted to 27% of all birds observed in the VP1 study site. Black guillemot amounted to 18%, whilst eider amounted to 9% of birds observed. In summer, fulmar, puffin and black guillemot comprised 37%, 24% and 14% respectively, of all birds observed in the VP1. Whilst in autumn, gannets and black guillemots amounted to 19% and 9% of all birds recorded, with shags constituting 49% of all birds observed in that season. During winter, shags and fulmars amounted to 39% and 35% of all birds recorded in VP1, whilst great northern divers comprised 8% of all birds observed (Fig. 5.6).

Table 5.1: Species detected in VP1 and VP2 study sites.

Family	Species	Scientific name
Auk spp.*	Atlantic puffin*	*Fratercula arctica*
	Black guillemot*	*Cepphus grylle*
	Common guillemot*	*Uria aalge*
	Razorbill*	*Alca torda*
	Common guillemot or razorbill*	
Diver spp.	Great northern diver	*Gavia immer*
	Red-throated diver	*Gavia stellata*
Diver/Cormorant spp.	European shag*	*Phalacrocorax aristotelis*
	Great cormorant	*Phalacrocorax carbo*
Gull spp.	Black-legged kittiwake*	*Rissa tridactyla*
	Common gull*	*Larus canus*
	Great black-backed gull*	*Larus marinus*
	Herring gull	*Larus argentatus*
Skua spp.	Arctic skua	*Stercorarius parasiticus*
	Great skua*	*Stercorarius skua*
Tern spp.†		
	Common eider	*Somateria mollissima*
	Greylag goose	*Anser anser*
	Long-tailed duck	*Clangula hyemalis*
	Northern fulmar*	*Fulmarus glacialis*
	Northern gannet*	*Morus bassanus*

* Detected in both the VP1 and VP2 study sites

† Detected only in the VP2 study site

Despite overlapping to some extent with the VP1 study site, different species assemblages were observed in the VP2 study site during summer compared to VP1; with common guillemots and razorbills constituting 35% and 9% of species observed from VP2 despite being largely absent in VP1. This was in addition to black guillemots and puffins that were recorded in both sites but at VP2 constituted 20% and 14% respectively of species observed (Fig.5.6).

5.3.1.2 Species observed foraging in the study sites

The greatest densities of birds were observed in the VP1 study site during spring. Eiders occurred at a density of 233 km^{-2}, gull species at 193 km^{-2}, black guillemots at 157 km^{-2} and shags at 142 km^{-2}. Of those eiders present, 69% were foraging. 99% of gull species interacting with the water were foraging. Of the black guillemots present, 24% were foraging. Fulmar density equaled shag density in spring but only 0.5% of fulmars were foraging compared to 15% of shags. In summer, puffins represented the greatest single-species density in the study site but only 3% were foraging. Black guillemots comprised the second greatest density, with 21% of birds foraging. 62% of shags and 70% of kittiwakes were foraging during summer. In autumn, shags represented the greatest density of birds at 922 km^{-2}, with 11% foraging. Gannets on the water (i.e. not including birds flying through the site) comprised a density of 346 km^{-2}, but only 1.5% were foraging. In winter, shags represented the greatest density of birds at 496 km^{-2}, with 28% foraging. Of the great northern divers present, 38% were foraging. 29% of black guillemots and 37% of eiders were also observed to be foraging (Table 5.2).

The greatest densities of birds observed in the VP2 site during summer were common guillemots, puffins, razorbills and black guillemots (Table 5.3).

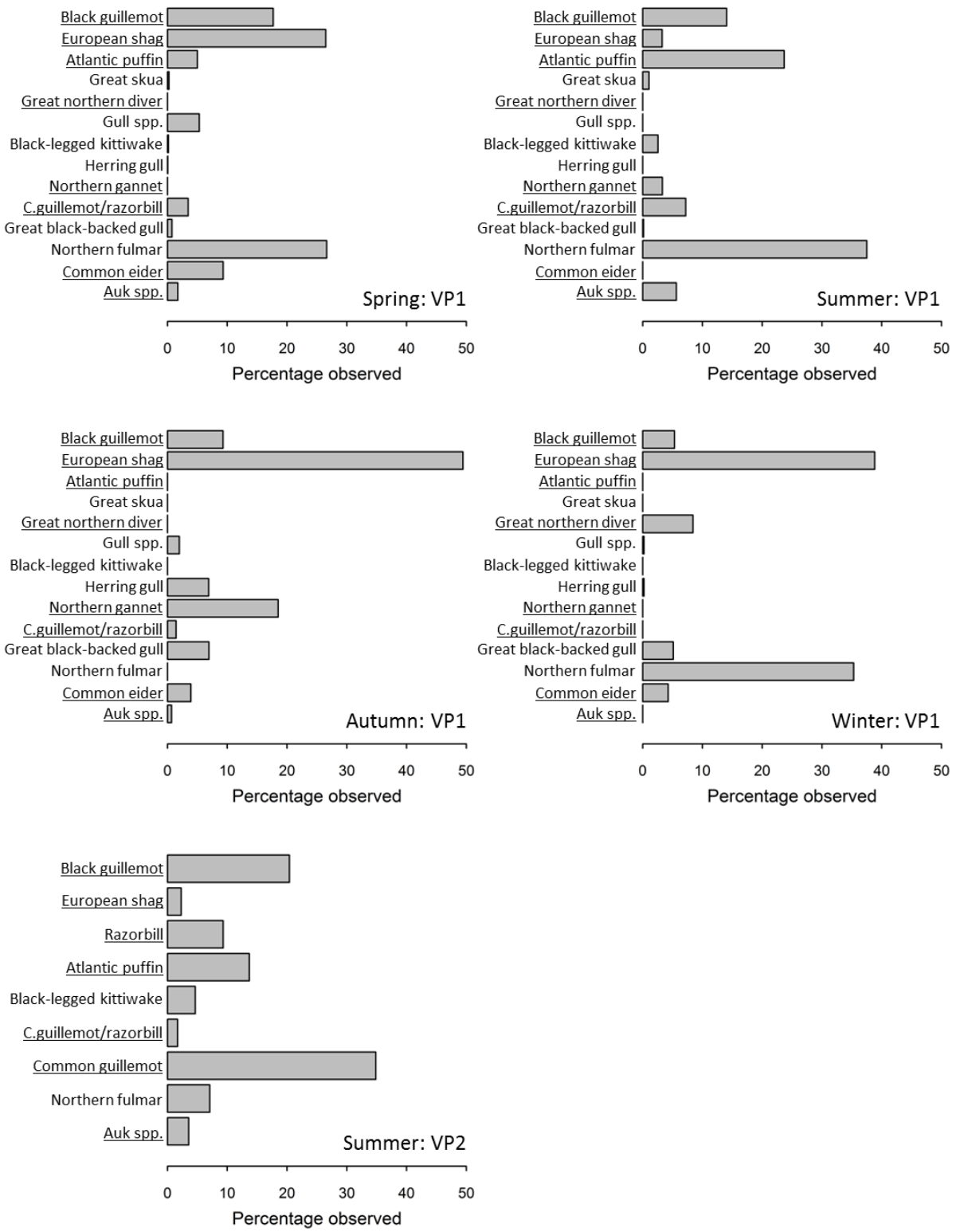

Figure 5.6: Percentage of each species observed in each season (only showing species that constituted >1% of birds in at least one season). Species capable of diving to depths overlapping with tidal-stream turbines are underlined. C. guillemot refers to common guillemot.

5.3.2 Vulnerability to collision and entanglement

5.3.2.1 Current speeds encountered during foraging

The average percentage of the VP1 study area that was subject to mean current speeds >2 ms^{-1} was 9% (range 0% to 64%). Of the number of shags observed foraging in the VP1 study site only 6% foraged in current speeds >2 ms^{-1}. 3% of black guillemots and 10% of puffins foraged in current speeds >2 ms^{-1}. Auk species identified at the family level foraging in current speeds >2 ms^{-1} amounted to 25% of birds. No great northern divers or eiders were observed foraging in mean current speeds greater than 2 ms^{-1} (Fig. 5.7).

Table 5.2: Total density of birds km^{-2} per species; density of foraging birds km^{-2} per species; and percentage of foraging birds per species observed in the VP1 study site in each season (showing only species where birds were observed foraging).

Species	Spring			Summer			Autumn			Winter		
	Density of foraging birds km^{-2}	Total density of birds km^{-2}	Percent. foraging birds	Density of foraging birds km^{-2}	Total density of birds km^{-2}	Percent. foraging birds	Density of foraging birds km^{-2}	Total density of birds km^{-2}	Percent. foraging birds	Density of foraging birds km^{-2}	Total density of birds km^{-2}	Percent. foraging birds
Atlantic puffin	22.5	182.8	12.3	21.4	728.5	2.9	-	-	-	-	-	-
Black guillemot	156.5	648.0	24.2	92.3	433.1	21.3	5.1	173.6	2.9	19.7	68.0	29.0
Black-legged kittiwake	6.2	6.2	100.0	55.2	78.6	70.2	-	-	-	-	-	-
Common guillemot/ razorbill	15	126.6	11.8	-	-	-	-	-	-	-	-	-
Common eider	233.3	340	68.6	-	-	-	9.3	72.6	12.8	20.3	54.5	37.2
Common gull	30.4	30.4	100.0	-	-	-	-	-	-	-	-	-
Cormorant spp.	-	-	-	2.2	2.2	100.0	2.2	2.2	100.0	5.1	5.1	100.0
Great black-backed gull	-	-	-	-	-	-	3.1	129.7	2.4	-	-	-
Great cormorant	30.4	30.4	100.0	-	-	-	-	-	-	-	-	-
Great northern diver	-	-	-	-	-	-	-	-	-	40.3	107.6	37.5
Great skua	3.1	9.7	32.0	-	-	-	-	-	-	-	-	-

Gull spp.	192.7	194.9	98.9	-	-	-	-	-	-	3.1	3.1	100.0
Herring gull	2.2	2.2	100.0	-	-	-	-	-	-	-	-	-
Northern fulmar	4.4	973.6	0.5	-	-	-	-	-	-	-	-	-
Northern gannet	-	-	-	-	-	-	5.1	345.8	1.5	-	-	-
Razorbill	5.1	32.2	15.8	-	-	-	-	-	-	-	-	-
Shag	142.4	969.5	14.7	61.8	100.4	61.6	97.6	922.3	10.6	140.9	495.9	28.4

Table 5.3: Total density of birds km^{-2} per species; density of foraging birds km^{-2} per species; and percentage of foraging birds per species observed in the VP2 study site in summer (showing only species observed foraging).

Species	Summer		
	Density of foraging birds km^{-2}	Total density of birds km^{-2}	Percent. foraging birds
Atlantic puffin	148.3	793.9	18.7
Auk spp.	167.1	205.9	81.1
Black guillemot	361.3	1182.5	30.6
Black-legged kittiwake	266.0	269.0	98.9
Common guillemot	6.2	98.0	6.3
Common guillemot/razorbill	6.2	98.0	6.3
Common gull	45.8	56.0	81.7
Great black-backed gull	0.0	8.2	0.0
Great skua	3.1	33.8	9.2
Northern fulmar	6.1	409.7	1.5
Northern gannet	0.0	29.6	0.0
Razorbill	73.4	540.5	13.6
Shag	15.2	133.1	11.4
Tern spp.	361.3	1182.5	30.6

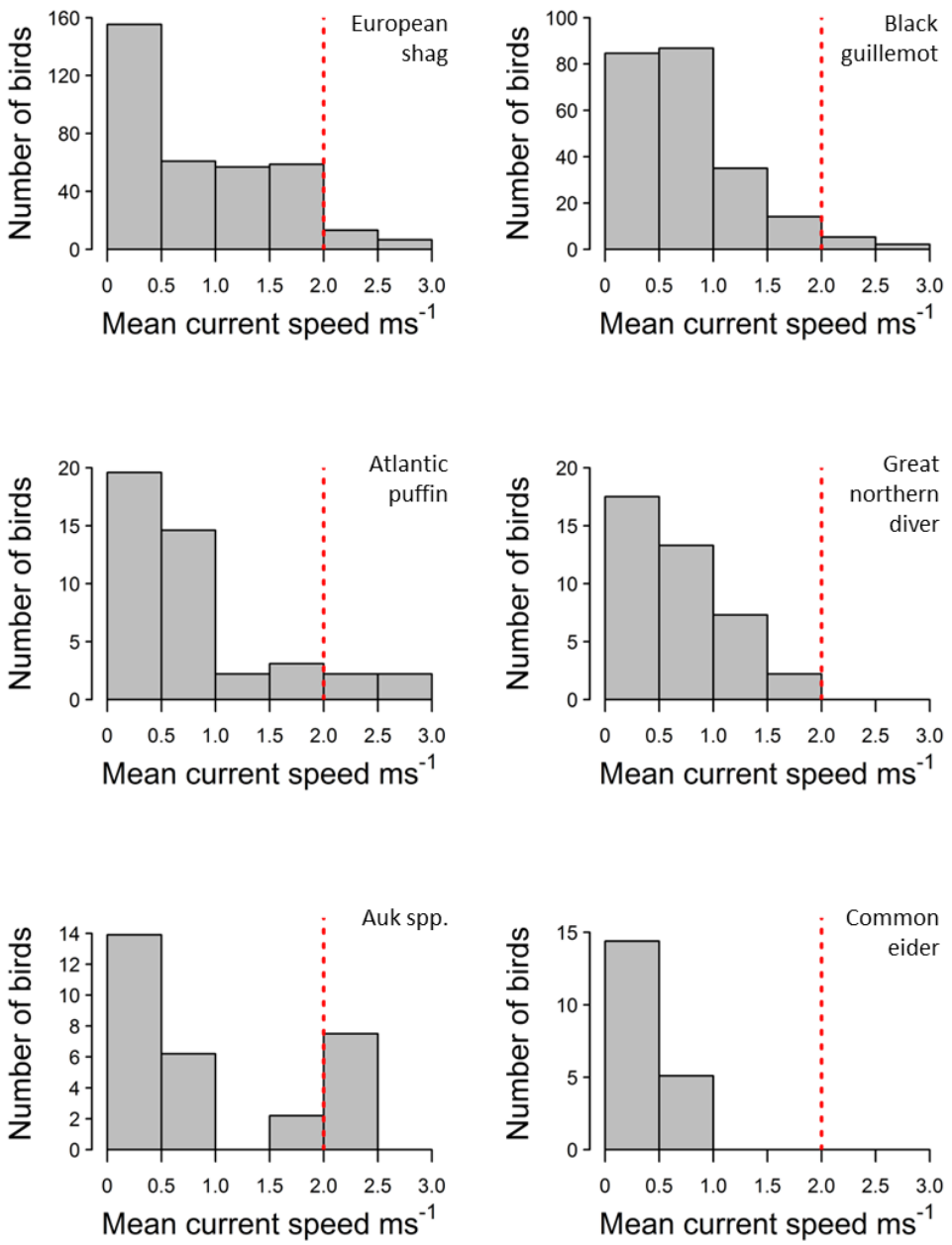

Figure 5.7: Mean current speeds of grid cells in the VP1 study site where foraging diving birds were observed. Red dashed line indicates current speeds suitable for commercial tidal-stream energy extraction (Fraenkel 2006).

In the VP1 study site, 3% of the total number of foraging shags foraged in current speeds >2 ms^{-1} in spring and autumn. 1% of shags foraged in current speeds >2 ms^{-1} in summer and no shags were recorded foraging in current speeds >2 ms^{-1} in winter. Of the total number of black guillemots recorded as foraging, 1% and 2% foraged in current speeds >2 ms^{-1} in spring

and summer respectively. No black guillemots were observed foraging in current speeds >2 ms^{-1} in autumn and winter. Foraging puffins were only observed in spring and summer. In summer 10% of the total number of foraging puffins were recorded as foraging in current speeds >2 ms^{-1}, whilst none were observed in current speeds >2 ms^{-1} during spring. Auk species identified to family level were only observed foraging in spring, summer and autumn. Of the total numbers of auks identified to family level, no birds were observed foraging in current speeds >2 ms^{-1} during spring and autumn, whilst in summer 25% of birds were recorded foraging in current speeds >2 ms^{-1} (Fig. 5.8).

5.3.2.2 Diving behaviour in relation to current flows

70 dives from sitting on the surface of the water recorded in the VP1 study site were included in binomial tests, and 27 dives from the VP2 study site. Dives were recorded for auk species recorded only to family level, cormorants, eiders, diver and cormorant species recorded only to family level, common guillemots, great northern divers, puffins, razorbills, shags and black guillemots. Diving birds observed in the VP1 study site were found to be significantly more likely to dive against the direction of current flow than with the direction of current flow (Binomial test: $p<0.01$, n = 70, dives against = 48). The same result was found when including additional dives from the VP2 study site (Binomial test: $p<0.001$, n = 97, dives against = 66). In each of the two tests, birds were found to dive against the direction of current flow 68% of the time. When analysing data by species, shags (n = 39, dives against = 25) and black guillemots (n = 20, dives against = 14) showed no clear tendency to dive with or against the direction of the current flow. However, puffins were significantly more likely to dive against the current (Binomial test: $p<0.05$, n = 18, dives against = 14). Analysis of individuals surfacing in both study sites indicated that birds showed no tendency to surface facing into, or away, from the oncoming direction of the current (n = 14, dives against = 5).

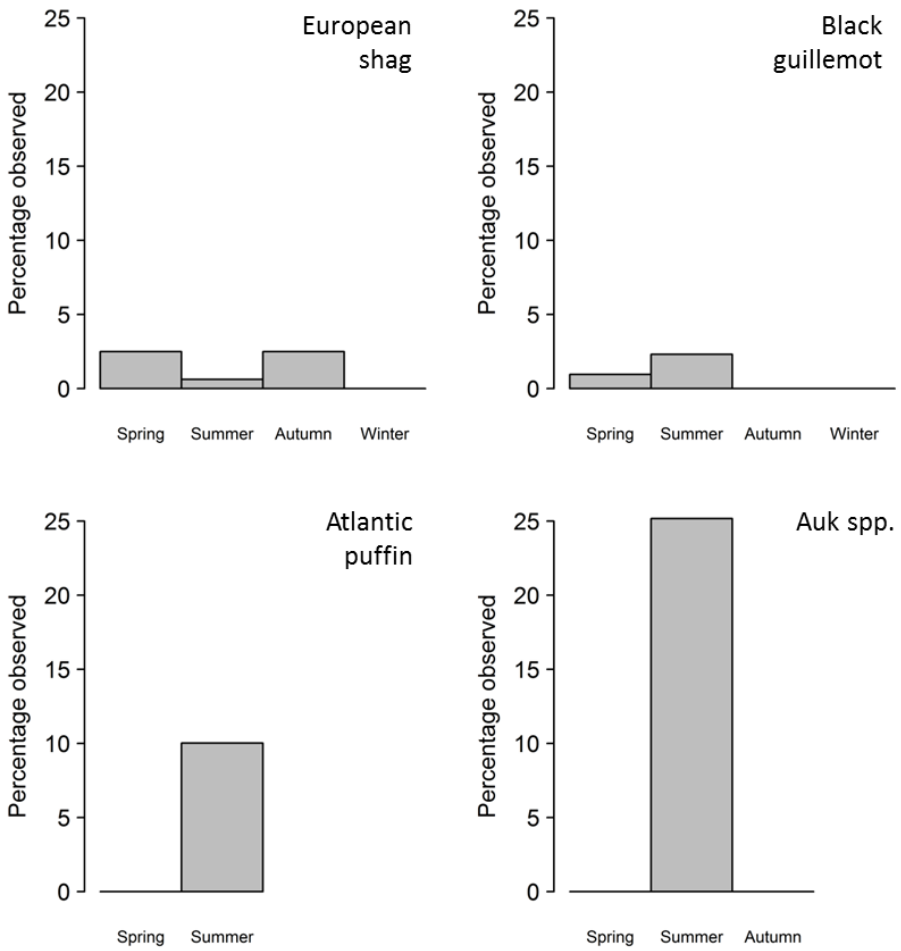

Figure 5.8: Percentage of total numbers of birds observed foraging in current speeds >2ms^{-1} according to season. Where no y-axis labels and axis line is present, no birds were recorded as foraging in that season.

5.3.3 Association with visible fine-scale oceanographic features

In the VP1 study site, 95% of foraging great northern divers, 82% of kittiwakes, 41% of shags, 29% of puffins and 23% of black guillemots showed no association with visible surface features. Of the auk species identified to family level, 25% foraged in association with areas of slick, fast flowing water, 25% showed no association with surface features and 17% associated with eddies (Fig. 5.9). Owing to small sample sizes, differences in associations with visible surface features between species observed in the VP1 study site were unable to be calculated.

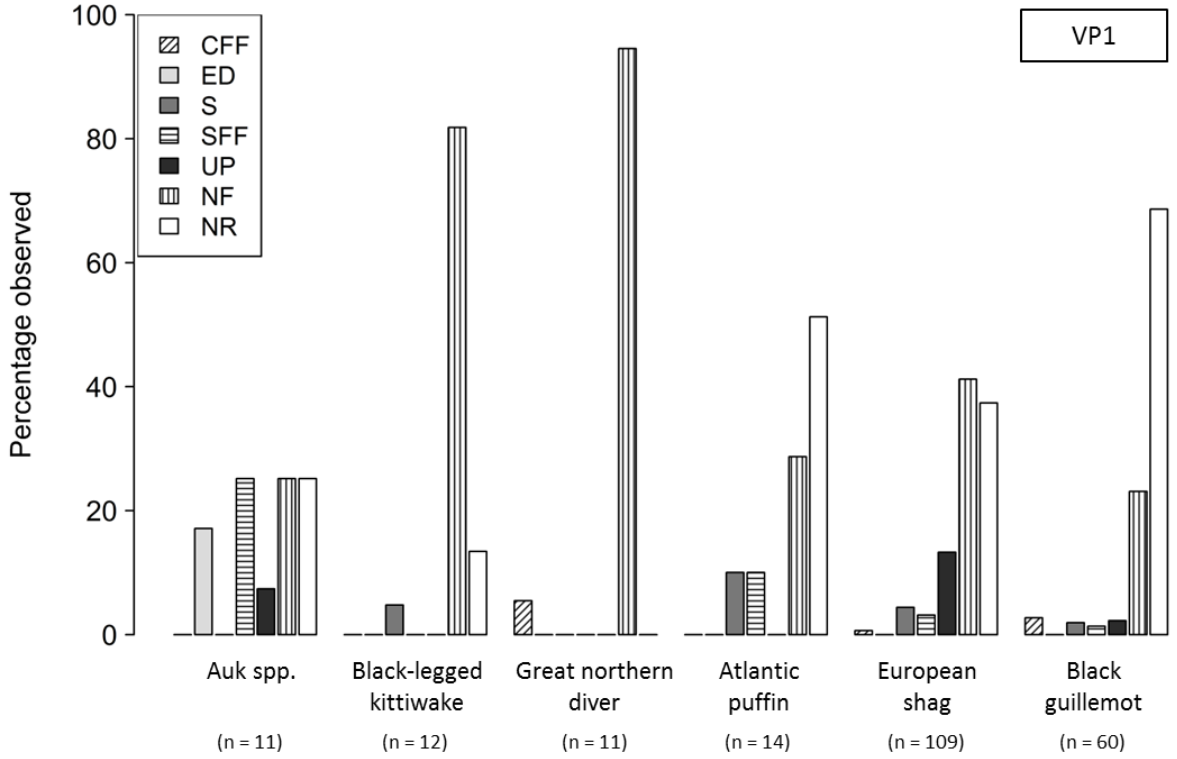

Figure 5.9: Percentage of foraging birds associating with visible surface features in the VP1 study site. Visible surface features are defined as: choppy fast flowing water (CFF), eddies (ED), shearlines (S), slick fast flowing water (SFF), upwellings and kolk boils (UP), no visible feature (NF) and no association was recorded (NR).

In the VP2 study site, 96% of auk species identified to the family level, 83% of kittiwakes, 47% of common guillemots and 32% of black guillemots showed no association with visible surface features when foraging. Of the common guillemots observed foraging, 37% associated with the outer edge of the shearline, as did 21% of foraging razorbills. During foraging, 52% of black guillemots and 24% of puffins associated with eddy features whilst 32% of puffins associated with the inner edge of the shearline. Of the razorbills observed foraging in the study site, 46% associated with the shearline, as did 17% of kittiwakes.

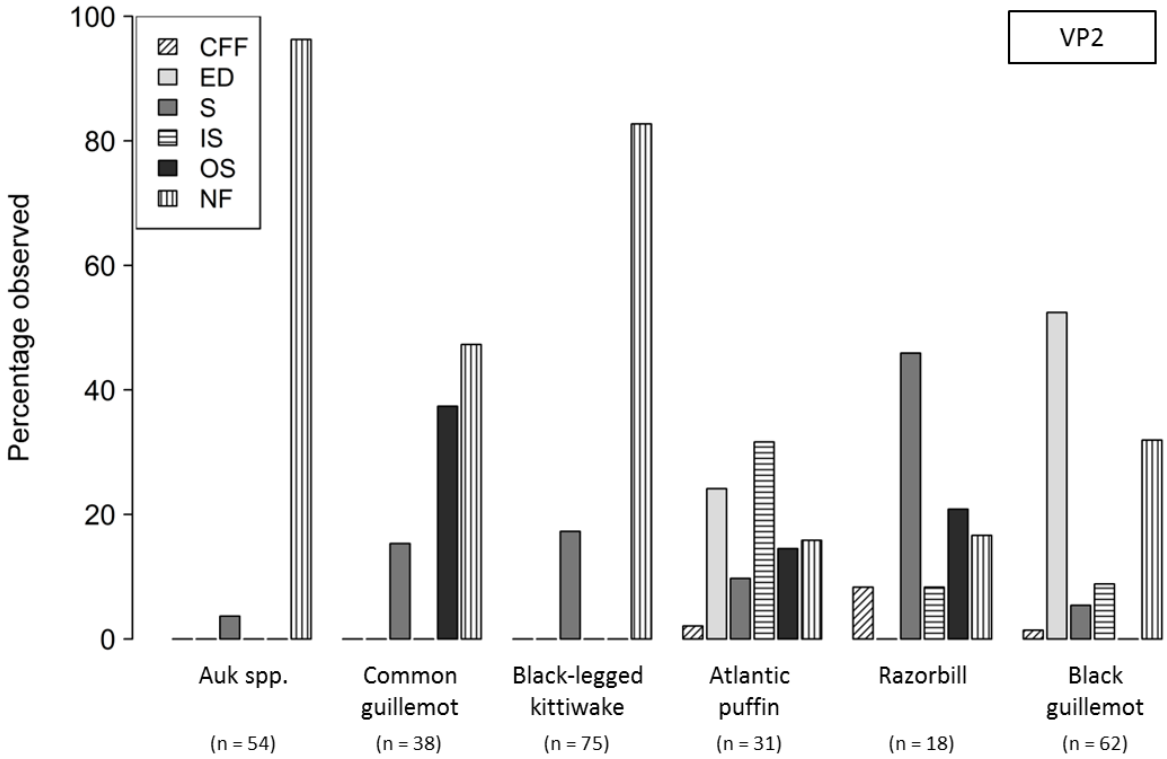

Figure 5.10: Percentage of foraging birds associating with visible surface features in the VP2 study site. Visible surface features are defined as: choppy fast flowing water (CFF), eddies (ED), the shearline (S), the inner shearline (area directly next to the turbulent shearline closest to the observer) (IS), the outer shearline (area directly next to the turbulent shearline furthest to the observer) (OS) and no visible feature (NF).

Tests to investigate differences between species in their associations with six visible surface features (choppy fast flowing water, eddies, the shearline, the inner shearline (area directly next to the turbulent shearline closest to the observer), the outer shearline (area directly next to the turbulent shearline furthest to the observer) and no visible feature) indicate that puffins and razorbills foraged in association with similar features (X^2=8.27, p>0.05). Common guillemots and black guillemots differed from puffins and razorbills in the surface features they associated with during foraging (common guillemot and puffin: X^2=26.12,

$p<0.001$; black guillemot and puffin: $X^2=15.26$, $p<0.01$; black guillemot and razorbill: $X^2=30.53$, $p<0.001$). In addition, common guillemots and black guillemots differed in their association with surface features ($X^2=43.23$, $p<0.001$). Furthermore, kittiwakes were found to differ to puffins in their association with surface features ($X^2=66.88$, $p<0.001$). It was only possible to test for a difference in association with surface features between kittiwakes and black guillemots when the six surface feature categories were consolidated into 4 categories; consisting of 1) features associated with the shearline (including the shearline, the inner shearline and the outer shearline), 2) areas of choppy fast flowing water, 3) slow flowing water in the form of eddies, and 4) areas with no visible surface features. It was found that kittiwakes and black guillemots differ in their associations with surface features when foraging ($X^2=56.67$, $p<0.001$).

5.4 Discussion

This is one of the first studies to investigate seabird use of a high velocity current channel and a proposed tidal-stream development site. By observing the distribution, abundance and behaviour of seabirds, I aimed to gain a better understanding of how seabirds use areas of high velocity current flows, which was an area identified in Chapter 2 as lacking in information (Table 2.13). I found seasonally different seabird assemblages present in the tidal channel but that diving species were common and populations of black guillemots and shags were present throughout the year. I found that of the species observed, not all were foraging. By assessing the current flows experienced by diving species during foraging in the VP1 study area, I found that low numbers of birds foraged in current velocities greater than 2 ms^{-1}, which is the threshold for commercially generating energy from tidal-stream devices (Fraenkel 2006). However, this type of analysis was not possible for the VP2 study area where greater numbers of diving species (in particular common guillemot and razorbill) were

present and where the study area overlaps to a greater extent with the proposed tidal-stream development site. In addition, I present evidence to suggest that birds rarely dive passively with the current flow, which could have implications for entanglement and collision risk. My results indicate that few species specifically target visible fine-scale surface oceanographic features but that association with surface features does occur. Overall, I conclude that effects of TEDs will be species-specific but that shags and auk species, namely black guillemots, puffins and potentially razorbills, are likely to be most vulnerable.

5.4.1 Survey limitations

Attempts to survey the study area as quickly as possible to gain a 'snapshot' of bird activity, resulted in an average survey duration for VP1 of 32 min and for VP2 of 21 min. Taking the example of VP1 and averaging time equally across each grid cell, each cell would take ca. 200 sec to survey. Within a 200 sec time period, the water in that cell could move 800m downstream, or it could move 0 m depending on the tide phase and the location of the cell. This is a limitation of the VP method when applied to marine environments consisting of fast flowing water, as the total number of birds observed in the site will be overestimated as birds are transported on currents into the study area during surveys. This bias is further complicated as different cells in the study area were subject to different current speeds depending on their location and the tide phase. An example, of when this bias should be taken into consideration is when comparing site usage according to tide phase. For example, it could be concluded that fewer seabirds use tidal-stream sites during slack tide. However, this conclusion could be the result of biased data caused by more birds being flushed through the site during flood and ebb tides, compared to during slack tide when there is little to no current flow. Currently there are no quantitative methods to deal with this bias but it should be borne in mind when interpreting survey results.

Surveying across an environmental gradient, in the form of varying current velocities and associated visible fine-scale surface features, prevented distance-correction methods being applied to the survey data presented here. This was because the assumption of an equal distribution of birds across the survey area was violated by the presence of an environmental gradient hypothesised to affect seabird distribution. As such, surveys may underestimate the number of birds in the furthest distance bands owing to a lower detection probability. Greater current velocities tended to occur in the furthest distance band; where the rate of travel of birds through the site was increased, which could additionally result in an overestimate of counts in that band as birds are flushed through the site on fast current flows. However, turbulence associated with the fastest flowing water is also likely to have further reduced detectability of birds in these areas. This is recognised as a problem in shore-based surveys in TED sites (Waggitt & Scott 2014) but for which a quantitative solution is yet to be achieved. Reduced detectability potentially explains the large numbers of auks identified only to family level in the VP1 study site; particularly given that a relatively high proportion of these birds were observed in areas with mean current velocities >2ms^{-1} (Fig. 5.7). An additional independent observer working concurrently at the same vantage point location, or on the opposite side of the channel, would improve these survey methods by establishing if the same birds were detected and enabling a measure of reliability. Reducing the distance over which observers are expected to record birds would reduce the issue of poor detectability in the furthest distance bands and is something I would implement if conducting these surveys again. In my opinion, a viewshed extending out to 2 km is too far to reliably detect birds, particularly if the furthest distance band overlaps with fast-flowing, turbulent water. In the case of VP1, I would recommend a viewshed extending to a maximum of 1.5 km.

Observations of birds are likely to have been biased towards larger, more conspicuous species as they are easier to detect. In addition, detectability of some species is likely to have been influenced by environmental conditions and plumage colour. For example, observations of some species (e.g. black guillemot) may have declined once birds had moulted into non-breeding plumage. Black guillemots moult into a mix of white and grey plumage, which amongst chop and foam on wave crests is very difficult to detect. This could explain the decline in black guillemot observations from spring and summer to autumn and winter, which were confounding as this species does not migrate and is likely to be resident in the area (Mitchell *et al.* 2004). By considering plumage detectability, it may be that the decline in numbers during autumn and winter reflects the reduced detectability of individuals rather than a decline in numbers using the site. Choppy water with even small white caps (beginning at sea state 3) made it more difficult to detect birds with a mix of black and white plumage, as it was difficult to differentiate a bird with white patches amongst masses of white foam. This is of particular concern for detecting birds in high current speeds, as turbulence is usually associated with areas of the fastest flowing water and could bias the number of birds detected. Additionally, dark coloured birds were more difficult to detect during strong sunlight as the sea surface appeared a dark blue colour, which reduced the contrast between the dark bird and the sea surface. I found that problems detecting birds affected by environmental conditions were largely resolved by reducing the distance over which the viewshed extended (e.g. maximum of 1.5 km), and where possible, siting the vantage point close to the fastest flowing water and turbulent areas. These conclusions were reached following comparisons of surveying from VP1 (viewshed extent 2 km, with fastest flowing water in the furthest distance band) with surveying from VP2 (viewshed extent 1.5km, with fastest flowing water present in distance bands 1 and 2).

5.4.2 Disturbance and displacement

All sixteen species of seabird observed using the tidal channel of the Inner Sound could be vulnerable to disturbance or displacement as a result of increased vessel traffic during installation, maintenance, monitoring and decommissioning of TEDs. Large numbers of shags, fulmars, black guillemots, puffins and gannets were observed from VP1 during the year but species recorded tended to vary by season; with fulmars absent in autumn, great northern diver only present in winter and shags much more common outside the summer season. Despite overlapping with VP1 to some extent, results from VP2 indicate that in addition to puffins and black guillemots, relatively large numbers of additional species not recorded to a great extent in VP1 were observed. These included common guillemots and razorbills. Conversely, low numbers of fulmars were recorded in VP2 compared to those recorded in VP1 (Fig. 5.6; Tables 5.2 and 5.3). The proximity of the survey sites to numerous SPAs designated specifically to safeguard a variety of seabird species (JNCC 2014a) has likely influenced the range of species observed in the Inner Sound and highlights the importance of the area to seabird populations.

Reaction to vessel traffic associated with installation, monitoring and maintenance, and decommissioning of TEDs is likely to be species-specific (Appendix 1, Table 4; Furness *et al.* 2012). Great northern divers observed in VP1 are likely to be vulnerable to TEDs as they are known to be extremely sensitive to disturbance by vessel traffic and have been observed to be flushed from boats at distances of 500 m as well as being displaced from areas of high vessel activity (Garthe & Hüppop 2004; Schwemmer *et al.* 2011; Mendel *et al.* 2014). Shags and black guillemots present throughout the year and observed in both study sites (Fig. 5.6) are also known to be sensitive to disturbance by vessel traffic (Ronconi & Clair 2002; Velando & Munilla 2011). As such, these species could be vulnerable to disturbance and

displacement during the development of the tidal-stream energy site in the Inner Sound. Shags and black guillemots may be most at risk because they are resident throughout the year (Mitchell *et al.* 2004; Grist *et al.* 2014). As such, disturbance of these species' may be unavoidable. Other species known to be sensitive to disturbance by vessel traffic include other auk species and eider (Bellefleur *et al.* 2009; Schwemmer *et al.* 2011). As such, the high numbers of puffins and eider observed (Fig. 5.6) could be vulnerable to the TED planned for the Inner Sound. Mitigation against disturbance and displacement caused by vessel traffic could be achieved by adopting measures proposed by (Ronconi & Clair 2002), who suggest reducing vessel speeds and avoiding close approaches to shore. Conversely, fulmars and gannets are unlikely to be at high risk of disturbance and displacement as a result of vessel activity owing to their tendency to associate with vessels at sea; particularly fishing vessels (Furness, Edwards & Oro 2007; Votier *et al.* 2013). These species were also rarely observed foraging in the study sites (Tables 5.2 and 5.3) and so are unlikely to be affected by interrupted foraging and disturbance of prey caused by vessels (Schwemmer *et al.* 2011). However, it is worth bearing in mind that the Inner Sound is already subject to vessel traffic in the form of fishing and recreational vessels, and the passage of a large vehicle ferry three or four times per day. As such, some seabird populations may have developed a level of habituation to vessel traffic, although a magnitude of increase in vessel activity would likely represent a cause for concern.

5.4.3 Collision and entanglement

5.4.3.1 Current speeds encountered during foraging

Collison and entanglement with tidal-stream energy generating devices represents a greater risk to seabird populations than displacement and disturbance, as adult mortality in k-selected species has greater population level effects than negative impacts on breeding

productivity and fitness (Sandvik *et al.* 2012; Satterthwaite *et al.* 2012). Seabirds will only be at risk of collision or entanglement with devices if they spatially overlap with TEDs. Of the species observed foraging in the Inner Sound study sites (Tables 5.2 and 5.3), diving species were common; with all species except fulmars and kittiwakes capable of diving to depths likely to overlap with tidal-stream energy devices (Langton *et al.* 2011; Furness *et al.* 2012). However, previous studies indicate that some species actively avoid foraging and travelling in the high current velocities likely to characterise TED sites, owing to associated increased energy expenditure (Wilson *et al.* 2001; Heath & Gilchrist 2010; Rey *et al.* 2010). Results presented here indicate that most diving species observed in the VP1 study site did not forage in current velocities greater than the 2 ms^{-1} (Fig. 5.7). Given that 2 ms^{-1} is the threshold required to generate commercially viable energy from tidal-stream resources (Fraenkel 2006), these results suggest that the majority of diving individuals will not overlap with areas optimal for tidal-stream energy generation. For example, eider were never observed foraging in mean speeds above 1 ms^{-1} (Fig. 5.7), which correlates with observations presented in (Heath *et al.* 2006). Furthermore, great northern divers did not forage in mean current speeds greater than 2 ms^{-1} (Fig. 5.7). The results indicate that these species are unlikely to overlap with areas where tidal-stream energy generating devices are located and are therefore unlikely to be highly vulnerable to collision or entanglement. However, a low percentage of the populations of foraging shags and black guillemots were observed diving in current velocities >2 ms^{-1} (Fig. 5.7). In addition, 10% of puffins and a quarter of auk species only able to be identified to family level (encompassing black guillemot, common guillemot, Atlantic puffin and razorbill) were observed foraging in current speeds >2 ms^{-1} (Fig. 5.7). These results suggest that auk species and puffins are likely to be at greater risk of collision or entanglement with tidal-stream energy devices than shags and black guillemots. My results also suggest that species tend to forage to a greater extent in current velocities

>2 ms^{-1} during the summer (Fig. 5.8). As suggested in Chapter 4, these findings indicate that tidally-generated predictable foraging locations may be of greater importance to seabirds during the breeding season. Results presented here also suggest that adult seabirds may increase their energy expenditure during breeding by foraging in energetically costly environments. This may be a response to the increased prey requirements of provisioning chicks and reflects evidence in Chapter 4 indicating that breeding great skuas expend more energy when foraging by increasing flight speeds; potentially to achieve minimal time between provisioning bouts. Breeding birds also undertake more foraging trips per day during chick-rearing than during the non-breeding season (Chapter 3; Grant, Trinder & Harding 2014). In conjunction with evidence that individuals dive repeatedly during foraging bouts (Holm & Burger 2002; Rodger 2014; Robbins *et al.* 2014a), it is likely that breeding birds may be at greater risk of collision with devices and displacement from foraging sites. In addition, collision or entanglement risk is likely to vary by age; with naïve and inexperienced juvenile birds more vulnerable than adult birds (Murray *et al.* 1994; Žydelis *et al.* 2009; Furness *et al.* 2012). It is worth bearing in mind that analysis of association with current speeds was only possible for VP1 and that relatively large numbers of diving seabirds (including puffins, black guillemots, common guillemots and razorbills) were observed in the VP2 study area. This study area was designed to incorporate a greater area subject to high-velocity currents and a greater overlap with the area leased for tidal-stream development. As such, results suggesting that the majority of diving individuals do not overlap with areas of current speeds optimal for energy generation may be a conservative estimate and is an area that requires further research.

5.4.3.2 Diving behaviour in relation to current flow

Vulnerability to TEDs is likely to be influenced by species-specific diving behaviour. Owing to the technical challenges of observing seabirds underwater, there is limited knowledge of species' behaviour in relation to prey, and avoidance responses to underwater obstacles. It has been hypothesised that seabirds diving in high velocity currents move passively with the flow, which is proposed to reduce the risk of collision with devices as birds are entrained in the flow of water as it sweeps around obstacles (Fraenkel 2006). Results presented here, and in Rodger (2014), suggest that birds tend to dive against the direction of current flow, which requires active propulsion rather than passive movement. Although these results are based on observations of behaviour on the surface, they suggest that birds dive through the water column facing into the current. Given the streamlined fusiform body shape of diving seabirds, it seems feasible that individuals would prefer to face into the flow of water. This is also observed in birds sitting on nests and when taking off, where individuals tend to face into the oncoming wind. This behaviour could have implications for collision risk as diving birds transported downstream in high current speeds may be facing away from approaching tidal-stream devices, which could compromise their ability to detect and avoid collisions. Conversely, diving birds showed no tendency to surface from dives in a particular direction, which could be because birds switch their direction of movement underwater or could be the result of passive ascents caused by positive buoyancy (Watanuki *et al.* 2006; Elliott, Davoren & Gaston 2007). During passive ascent, individuals may have little control over the direction in which they surface, which has implications for collision risk, as birds may find it challenging to avoid obstacles when fighting against positive buoyancy. An additional explanation could be that like human divers, birds entering a dive from the surface are able to orientate themselves relative to the coast and the oncoming current. However, once underwater and in the flow of water, the direction of the current flow is not distinguishable.

As such, birds could surface in any direction as they would be unable to orientate themselves according to the current flow. Overall, an improved understanding of how diving seabirds behave underwater is imperative to improve collision risk predictions.

5.4.4 Association with visible fine-scale oceanographic features

Species associating with visible fine-scale oceanographic features could be affected by TEDs if TEDs modify hydrographic regimes by reducing current flow and/or altering the occurrence of visible fine-scale surface features used by seabirds as foraging cues (Shields *et al.* 2011; Copping *et al.* 2014). Razorbills showed a clear association with the shearline (S, OS and IS) and choppy, turbulent areas of fast flowing water (CFF) (Fig. 5.10) (Holm & Burger 2002; Ladd *et al.* 2005) but the species was never seen to forage in eddies (ED), although some individuals were observed foraging in areas with no visible feature (NF). Half of the razorbills observed foraging were accompanied by chicks that followed diving adults into the turbulent areas of the shearline; although the chicks did not dive. Despite the turbulence and presumed poor visibility below the water surface, adults with chicks were rarely separated; with diving adults usually surfacing within visual detection distance of the chick/adult. On the rare occasion that adults surfaced beyond the visual detection distance of the chicks/adults, adults and chicks appeared to call to locate each other. Puffins and common guillemots were also observed associating with the main shearline and the choppy, turbulent areas of the outer shearline; although the majority of common guillemots foraged in areas with no visible surface feature (Figs. 5.9 and 5.10). Puffins had the most varied relationship with visible surface features; they were observed diving in the slick, laminar flows associated with fast flowing water (SFF) but also foraging in eddies and areas with no visible surface features (Figs. 5.9 and 5.10). Auk species identified to the family level exhibited a varied relationship with surface features but this is likely caused by the potential range of species

and associated differences in behaviour included in this category (Figs. 5.9 and 5.10). Based on this evidence, all of these species could be vulnerable if tidal-stream devices extract enough energy from current flows to result in a reduced occurrence of the physical processes that generate visible surface features.

Of the non-diving species, kittiwakes generally did not target areas with visible surface features, although foraging birds sometimes targeted the shearline (Figs. 5.9 and 5.10). This could be influenced by turbulence along the shearline driving prey close to the surface (Zamon 2003; Schwemmer *et al.* 2009). Diving species may also target shearlines owing to the elevated productivity caused by large volumes of water containing prey items passing rapidly through an area and the prey barrier shearlines represent. Black guillemots and puffins were the only species to show clear association with eddy features (Figs. 5.9 and 5.10), which are created by fast currents flowing around islands and headlands (Wolanski & Hamner 1988). These species may be targeting these productive locations where plankton and fish in the free stream region are advected and aggregated in the eddy (Wolanski, Imberger & Heron 1984; Johnston *et al.* 2005a). Black guillemots and puffins also associated with the inner shear area between the shearline and eddy. It is possible that to increase foraging success these species use the shearline as a functional barrier against which to chase prey concentrated in the eddy region (Zamon 2003; Schwemmer *et al.* 2009). These differing associations observed between species and visible surface features were also evident in analyses investigating potential differences in the use of features between species. Puffins and razorbills were similar in their associations with visible surface features during foraging, compared to common guillemots and black guillemots that differed in their behaviour. Additionally, common guillemots differed to black guillemots in their association with surface features, and kittiwakes appeared to associate with different surface features

to all other seabird species. These differences in habitat use during foraging in tidal channels may indicate fine-scale habitat segregation adopted to reduce inter-specific competition for prey resources. In summary, surface features represent visible cues that often indicate elevated nutrient levels, and these features represent efficient foraging locations for seabirds. Should TEDs modify the development of visible surface features by altering current flows, this may affect the occurrence of predictable and productive resources important for foraging seabirds.

Conclusions

The use of high current flow environments varied according to species. As such, seabird vulnerability to TEDs is dependent on species' behaviour and is likely to be site-specific. Breeding species, and species present throughout the year, are likely to be more vulnerable to TEDs. Low numbers of birds were observed foraging in current speeds associated with optimal locations of tidal-stream energy generating devices, which could indicate that the majority of birds will not be at high risk of collision. However, it is worth bearing in mind that these estimates may be conservative, as analysis of association with current speeds was not possible for VP2, which encompassed a greater area subject to high-velocity current flows and numerous diving species were observed there. Visible surface features associated with dynamic tidal habitats were targeted by some seabird species during foraging. As such, these species could be affected if TEDs alter current regimes and the development of surface features. Key areas requiring further research include improving our understanding of diving seabird behaviour underwater and further investigation into the association of seabirds with high-velocity current flows. Technological limitations associated with monitoring seabirds in the challenging environment of TED sites continue to challenge data collection and improving our understanding of seabird behaviour in these dynamic

environments may require the development of novel methodological approaches. For example, multiple-sensor marine monitoring platforms, such as the FLOWBEC platform (Williamson *et al.* 2014), and marine radar (Bell *et al.* 2014). Data presented here could inform future monitoring to target particular habitats and seasons in order to focus on species of concern; for example, to improve our understanding of razorbill use of tidal-races, it would be appropriate to focus monitoring on shearlines during the summer. Overall, this is a research area with exciting opportunities for further research to improve our understanding of how seabirds use high-velocity current environments and to address knowledge gaps limiting predictions of how seabirds may be affected by TEDs.

Chapter 6:

General Discussion

This thesis has aimed to address gaps in knowledge regarding how marine renewable energy developments (MREDs) may affect seabird populations. In this thesis I have considered potential effects of MREDs at a range of spatial scales, in different habitats and across various life stages. My intent has been to contribute information to aid our understanding of those seabird species that may be most vulnerable in a Scottish context. In this discussion I summarise my key findings from each of the four preceding data chapters, consider some of the challenges encountered during my PhD research, set my findings in a wider context and suggest directions for future research.

The aim of **Chapter 2** was to update and expand on two previously published papers that estimated the effects of offshore wind, wave and tidal-stream renewable energy developments on seabird populations in Scotland (Furness *et al.* 2012, 2013). The chapter details the development of existing vulnerability indices and the generation of indices indicating the level of confidence inherent in data used in calculating vulnerability indices. The indices predict marine bird vulnerability to MREDs and are intended to provide relevant, up-to-date information that can be used by developers and regulators to design monitoring programmes to fulfil environmental impact assessment (EIA) requirements. By additionally generating confidence indices, my intention was to identify where caution in interpreting the results of vulnerability indices may be required and to highlight areas where more research and monitoring could be beneficial.

I found that newly available literature altered the rankings of only a few species, compared to the previously published indices, and that overall species rankings remained broadly the same. This indicates that the method incorporating existing knowledge combined with expert moderation to generate vulnerability indices employed in Chapter 2, and used previously by Furness *et al.* (2012, 2013), is robust in estimating species vulnerability to MREDs. I have found a greater overall confidence in data on how marine birds may react to MRED structures and what percentage of flight overlaps with turbine blade altitudes. However, within these two vulnerability factors several species (including great skua) are lacking in data and these species should be a focus for additional monitoring and research. I also recommend that accurate flight heights of seabird species are collected at different times of year to account for differences in behaviour during different life stages. In addition, estimates of macro- and micro-avoidance rates are urgently required to generate reliable collision risk models. I found an overall lower confidence in data regarding how marine birds may react to vessel and/or helicopter traffic associated with MREDs, and how marine bird species use tidal races. I propose additional monitoring and research is required to improve our understanding of seabird behaviour in relation to these two vulnerability factors. I begin to address this shortage of data in Chapter 5 by investigating how marine birds use tidal races.

I recommend that the vulnerability and confidence indices presented in Chapter 2 are viewed as a work in progress and are not used as a substitute for site-specific monitoring. I suggest that the vulnerability indices are regularly updated as it is likely that some species may alter their response to MREDs in the longer term and that our knowledge of species' responses to MREDs may change. The vulnerability and confidence indices in Chapter 2 are set in a Scottish context; however, I suggest the methods are transferable to other countries

and regions. I recommend the use of vulnerability and confidence indices elsewhere to gain a rapid estimation of the potential effects of MREDs on seabird populations, and to quickly identify gaps in knowledge that can be addressed in compulsory EIA seabird monitoring programmes and research.

The great skua is an internationally important seabird species breeding in Scotland and is identified as potentially vulnerable to MREDs in Chapter 2. In **Chapter 3**, I focused on improving understanding of great skua *Stercorarius skua* movements at sea in order to better estimate the potential effects of MREDs on this species in Scottish waters. As over 60% of the global great skua population breeds in Scotland (Mitchell *et al.* 2004), often in special protection areas (SPAs), I deemed it imperative to improve understanding of how MREDs could affect this species. My aim in conducting the research included in Chapter 3 was to provide information on great skua movements that could be incorporated in marine spatial planning decisions.

This study was the first to track individual great skuas throughout the breeding season using GPS technology. I found that the movements of great skuas varied throughout the breeding season, with actively breeding birds remaining closer to the nest and spending less time away from the colony than birds that had failed in their breeding attempt. I also found that movements differed according to breeding colony. These differences in behaviour suggest that MREDs could differently affect subsections of great skua populations. Overlap with MREDs was found to be low, but historical data indicated that great skua movements have changed over time. Breeding birds now undertake longer trips and spend more time away from the nest. Great skuas currently appear to be at an energetic maximum in terms of effort available to invest in foraging, as despite an increase in foraging effort over time,

productivity has failed to improve and large great skua colonies, such as those studied in Foula and Hoy, are continuing to decline. Based on this evidence, Chapter 3 indicates that the great skua is a species under pressure and therefore vulnerable to any additional effects of MREDs. The changes I have found in great skua movements over time highlight a challenge in using short-term tracking studies in isolation to predict the effects of MREDs on seabird species and incorporating them as a basis for marine spatial planning. Based on comparisons of tracking data with historical data, I suggest that although short-term tracking of seabirds can be useful in marine spatial planning and EIAs wherever possible, tracking studies should be conducted over several years. Ideally, seabird tracking should be undertaken in conjunction with monitoring of populations and distributions at site-specific (in the case of MREDs) and colony levels, and throughout different life stages to fully assess effects.

Chapter 4 continued to focus on great skua overlap with MREDs by investigating in more detail the differences in behaviour and foraging strategy of birds throughout the breeding season. This chapter was developed based on evidence found in Chapter 3 suggesting that differences in behaviour could lead to subsections of great skua populations being differently affected by MREDs. This chapter also considered the impact on great skuas of changes to the EU Common Fisheries Policy, in the form of a discard ban. My aim in this chapter was to consider the consequences of individual variation in behaviour and dietary strategies for population-level effects of human activities in the marine environment. I also aimed to identify environmental conditions influencing foraging behaviour. This information is intended to be used to more accurately predict the effects of human activities on great skuas and inform marine spatial planning decisions.

I found that individual great skuas consistently differed in their foraging behaviour and movements at sea along environmental gradients, and that this intra-population variation resulted in the repeated use of different foraging locations by individuals. Evidence for consistent foraging behaviour in Chapter 4 was often linked to predictable and persistent habitat features, such as upwellings caused by bathymetry. I also found that breeding birds have a more conservative and constrained foraging strategy than failed breeders but that they expend more energy during foraging trips. These findings suggests that breeding birds may be less able to alter their foraging strategy, which could make breeding birds more vulnerable to negative effects of human activities at sea. My results also corroborate evidence presented in Chapter 3 that suggests different subsections of great skua populations may be differently affected by human activities in the marine environment. My findings indicate that although individuals may be flexible in their foraging tactics, individuals show a degree of foraging site fidelity and potential dietary consistency.

Findings in Chapter 4 illustrate the importance of improving our understanding of both population- and individual-level responses to the marine environment in order to more fully understand the behaviour of seabird species. I suggest that although consideration of population-level responses to environmental variables are key to developing species' distribution models that can be incorporated in marine spatial planning, these studies reveal little about individual strategies. As individual strategies, and differences in behaviour according to breeding status, could lead to subsections of populations being differently affected by human activities, I recommend an increased consideration of intra-population variation in seabird research. This will facilitate a greater understanding of seabird behaviour and better inform marine management decisions. I also recommend that future research focusses on investigating the fine-scale behaviour at sea of juvenile seabirds, non-

breeding individuals and migrating/overwintering birds in order to gain a comprehensive understanding of seabird behaviour and how human activities may affect subsections of seabird populations. However, I recognise that achieving this recommendation is currently constrained by technological limitations.

I encountered four main challenges when analysing the great skua tracking data presented in **Chapters 3 and 4**. As tracking technology appears to have advanced more rapidly than associated analytical techniques, I highlight the challenges I encountered to promote discussion regarding potential development of existing and/or alternative analytical techniques:

Firstly, I found that there is no standardised or recommended method for calculating the utilisation distribution (UDs) of GPS tracking data. This is partly because techniques developed to estimate UDs were intended for application to VHF telemetry tracking data, where the number and accuracy of location fixes is less than that recorded by current GPS data technology. This is potentially an issue for marine spatial planners required to base marine spatial planning decisions on UD data. I recommend discussions amongst seabird ecologists and spatial statisticians aimed at developing appropriate UD methods suitable for use with GPS data. I also suggest that a standardised approach to calculating UDs may be beneficial to marine spatial planners.

Secondly, when investigating great skua movements I had intended to use first passage time (FPT) analysis (Fauchald & Tveraa 2003) to identify foraging locations. However, this method proved to be relatively labour-intensive, with a graph of each trip requiring inspection to identify the scale at which foraging behaviour was occurring. With 2039 trips to assess, this

method was not appropriate for use with the great skua dataset. As with UD methods, this is probably because FPT was developed for application with earlier deployments of tracking devices, which were unable to collect large quantities of data. I suggest that it may be beneficial to develop this popular method to enable more efficient analysis of large datasets.

Thirdly, during initial data exploration I applied FPT to subsections of the great skua dataset. However, I found it difficult to identify clear scales at which foraging behaviour was occurring. I also encountered the same problem when attempting to calculate the tortuosity of movement tracks when analysing the relationships between foraging behaviour and environmental variables presented in Chapter 4. Owing to problems in deciding on the scale at which great skuas forage I decided to only use speed of travel as an indication of foraging behaviour. Based on results in Chapter 4, I suggest that difficulties in identifying a common scale at which foraging may occur could be confounded by the varying foraging and dietary strategies found in great skua populations. I suggest that whilst identifying common scales at which foraging may occur can be useful in some studies of seabird foraging, these methods may not be appropriate for all species owing to intra-population variation in foraging behaviour. Similar issues have also been raised by others (e.g. Warwick-Evans *et al.* in review).

Fourthly, published studies such as Bodey *et al.* (2014) investigate the influence of contemporaneous fishing activity on seabird behaviour. I had hoped to conduct similar analyses as part of my PhD research as it is known from extensive dietary studies that great skuas scavenge at fishing boats for discarded fish. However, issues of confidentiality affect the availability of vessel monitoring system (VMS) tracking data, which prevented access to VMS data collected over the same time period as great skuas were tracked. Although

evidence suggests that fishing hotspots are consistent over time (Witt & Godley 2007), contemporaneous vessel tracking data would have enabled a more accurate understanding of the relationships between great skuas and fishing vessels. In light of recent changes to the discard policy in the EU CFP (Bicknell *et al.* 2013) and the ecosystem-management approach promoted in the EU Marine Strategy Framework Directive, I suggest more effort is made to make VMS tracking data available to researchers, whilst still maintaining a level of confidentiality required by vessel owners.

In **Chapter 5** I focused specifically on investigating the use of a high current flow environment by seabird species, which was identified as an area lacking in information in Chapter 2. This chapter used land-based vantage-point surveys to investigate seabird use of high velocity currents, and to establish interactions with fine-scale habitat in a leased tidal-stream renewable energy site in Scotland. The chapter aimed to more accurately predict which species may be most vulnerable to tidal-stream energy developments (TEDs).

This study is one of the first to investigate the use of high-velocity current habitat and a proposed tidal-stream development site by seabirds. I found that different species of seabird were present in the channel throughout the year, but that populations of black guillemots and shags were present all year. These results suggest that black guillemots and shags are likely to be most vulnerable to TEDs and so should be a focus for monitoring and research. Findings in Chapter 5 also indicate that of the species observed in the site, not all used the site for foraging. This indicates that measures of seabird presence and abundance do not equate to foraging birds and suggests species differ in their use of high velocity current habitats, such as tidal channels. I also observed low numbers of birds foraging in current velocities optimal for commercially generating energy from tidal-stream devices

(Fraenkel 2006), suggesting that high-energy tidal streams are generally not an attractive foraging habitat for most species of seabirds. This finding suggests a low proportion of birds are likely to be at greatest risk of collision or entanglement with devices. I also found evidence that birds rarely dive passively with the current flow, and tend to dive facing into the oncoming current. These findings suggest that diving birds may be less able to detect devices if approaching facing away from them, therefore placing them at greater risk of entanglement and collision. However, it should be considered that all conclusions regarding seabird diving behaviour were drawn from observations of birds above the water surface. As such, more research is required to investigate the diving behaviour of seabirds; particularly in high velocity current flows, and to assess detection and avoidance responses of seabirds to underwater obstacles; particularly in varying levels of visibility. Findings in Chapter 5 also indicate that few species specifically target visible fine-scale surface oceanographic features but that association with surface features does occur. This suggests that seabirds may be at relatively low risk of adverse effects if TEDs alter current flows and associated surface features. However, it should be borne in mind that sample sizes in this chapter were relatively small; therefore results should potentially be treated with caution. The short-term nature of this study also highlights the importance of long-term monitoring studies for generating robust conclusions.

Overall, I conclude that effects of TEDs will be species-specific but that auk species and shags are most likely to be at risk. I recommend that investigating high velocity current flow habitat use is a focus for research and compulsory EIA monitoring. Ideally, monitoring and research will consider underwater dive behaviour of species. I also suggest the development of novel methods to collect seabird observation data, in conjunction with contemporaneous hydrodynamic information, to enable relationships between behaviour and habitat use to be

quantified (e.g. (Williamson *et al.* 2014). Increased research and methodological developments are imperative to enable more reliable predictions of how seabirds may be affected by TEDs.

Future directions

There are several avenues for future research that have developed in my mind as a result of the PhD research I have undertaken. I briefly outline these ideas below:

1. Much more is known about the movement behaviour and ecology of adult seabirds than immature birds, which limits our understanding of how seabirds may be affected by human activities in the marine environment to a sub-section of the population. As the immature period of seabird life-history is proposed to be a developmental and learning phase, it would be expected that immature birds will behave differently to adult birds. As evidence from Chapters 4 and 5 suggests, differences in behaviour have the potential to influence the impacts of human activities on sub-sections of seabird populations. Given the technological advances in tracking devices, it is now possible to track immature birds effectively. This is an important area for future research, not only from a general ecological and behavioural perspective, but with the particular aim of improving predictions of how seabird populations may be affected by MREDs.

2. Given the relatively long time-series of data collected during the tracking of great skuas, it is potentially possible to estimate how many tracks are required to characterise relationships with environmental conditions. This information would be useful to seabird biologists to optimise tracking studies and has already been

investigated in relation to foraging ranges (Soanes *et al.* 2013a). This is an aspect of research that I began during my PhD but did not have the time to complete. My preliminary results suggest that analysing only the first four trips per bird to investigate relationships with environmental conditions gives different results than when the whole dataset is included. This is of relevance because commonly used tracking devices (e.g. iGotU tags) that rely on battery power rather than solar power generally collect only a few trips per deployment. As data from these tracking devices is often used in conjunction with information on environmental conditions to inform conservation measures and marine spatial planning it would be useful to deduce whether enough data is being collected to accurately predict habitat use.

3. Predictable and productive features, such as upwellings and fronts, are increasingly being shown to be key foraging habitats targeted by seabirds and could be candidates for effective offshore marine protected areas (Chapter 4, Scales *et al.* 2014a, b). Most studies focus on single-species use of these habitat types but given the large number of tracking studies now being undertaken globally, a multi-species assessment of use seems a logical future step forward to more accurately assess the importance of upwellings and fronts to seabirds. Assessment of multi-species relationships with environmental conditions could also prove useful in establishing how scavenging seabirds might be affected by a discard ban.

4. Establishing the importance of predictable and productive features is not only important at the meso- (10s – 100s km) and sub-mesoscale (ca. 1 km) but also for fine-scale (10s – 100s m) features, such as those described in Chapter 5. This is particularly true for tidal-races that are currently being targeted for development by

the renewable energy industry. Results in Chapter 5 tended to prompt more questions than provide answers and it became clear that a key area for future research is establishing a better understanding of seabird behaviour underwater; particularly with regards to the direction that birds face in relation to current flows. This knowledge gap could be addressed through targeted tracking studies incorporating devices such as compasses or accelerometers combined with GPS loggers. This data could be analysed in conjunction with hydrodynamic models to assess diving seabird movements in relation to current direction. In addition, opportunities exist to use tidal-stream devices to monitor seabird behaviour through mounted cameras; although the success of this technique depends on visibility in the water column, which could be problematic in such a turbulent and dynamic environment. Sonar as used on the FLOWBEC platform (Williamson *et al.* 2014) could represent an alternative method to investigate seabird movements underwater in relation to tidal-stream renewable energy devices. Understanding seabird diving behaviour in relation to current direction, and their ability to detect and avoid collisions with devices is a key area of future research regarding the effects of tidal-stream developments on seabirds.

Final conclusions

Global increases in the use of the marine environment by humans, for example in the form of MREDs and fisheries activity, are likely to have numerous impacts on marine species. Negative effects need to be recognised and limited if sustainable use of marine ecosystems is to be achieved. This thesis indicates that seabird responses to MREDs are likely to be species-specific and will vary dependent on the development location and design of the energy generating technology. My research also indicates that effects will differ dependent

on individual foraging strategies, age and life stage of individuals. These findings imply that MREDs will differently affect subsections of seabird populations. Although tools such as vulnerability indices can contribute to designing EIA monitoring and predicting potential effects of MREDs on seabird populations, they should not become a substitute for long-term, *in-situ* monitoring of MRED sites and detailed species-specific research studies.

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Appendix 1:

Contents

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Supplementary Table 2 - Flight altitudes: Table of data contributing to calculations of confidence levels for each species.

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Supplementary Table 4 - Disturbance and displacement caused by vessels and/or helicopter traffic: Table of data contributing to calculations of confidence levels for each species.

Supplementary Table 5 - Use of tidal races: Table of data contributing to calculations of confidence levels for each species.

References

Appendix 1
Supplementary Table 1: Common and scientific names

Common and scientific names of the species assessed for vulnerability to marine renewable energy developments.

Species	
Arctic skua	*Stercorarius parasiticus*
Arctic tern	*Sterna paradisaea*
Atlantic puffin	*Fratercula arctica*
Black guillemot	*Cepphus grylle*
Black-headed gull	*Chroicocephalus ridibundus*
Black-legged kittiwake	*Rissa tridactyla*
Black-throated diver	*Gavia arctica*
Common eider	*Somateria mollissima*
Common goldeneye	*Bucephala clangula*
Common guillemot	*Uria aalge*
Common gull	*Larus canus*
Common scoter	*Melanitta nigra*
Common tern	*Sterna hirundo*
European shag	*Phalacrocorax aristotelis*
European storm-petrel	*Hydrobates pelagicus*
Great black-backed gull	*Larus marinus*
Great cormorant	*Phalacrocorax carbo*
Great northern diver	*Gavia immer*
Great skua	*Stercorarius skua*
Great-crested grebe	*Podiceps cristatus*
Greater scaup	*Aythya marila*
Herring gull	*Larus argentatus*
Leach's storm-petrel	*Oceanodroma leucorhoa*
Lesser black-backed gull	*Larus fuscus*
Little auk	*Alle alle*
Little tern	*Sternula albifrons*
Long-tailed duck	*Clangula hyemalis*
Manx shearwater	*Puffinus puffinus*
Northern fulmar	*Fulmarus glacialis*
Northern gannet	*Morus bassanus*
Razorbill	*Alca torda*
Red-throated diver	*Gavia stellata*
Roseate tern	*Sterna dougallii*
Sandwich tern	*Thalasseus sandvicensis*
Slavonian grebe	*Podiceps auritus*
Sooty shearwater	*Puffinus griseus*
Velvet scoter	*Melanitta fusca*
White-tailed eagle	*Haliaeetus albicilla*

Appendix 1

Supplementary Table 2: Flight altitudes (turbine swept area *ca.* 20-150m altitude unless otherwise stated)

Tables of data contributing to confidence levels for each species

Obs = observation
Unk = unknown

1. Greater scaup

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Walls *et al.* 2013)	Greater scaup	Boat obs	Indirect 1	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	160 (operation)	2	0% (26-200m)
(Walls *et al.* 2013)	Greater scaup	Boat obs	Indirect 2	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	358 (pre-construction +construction)	7	0% (26-200m)

2. Common eider

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Genesis 2012)	Eider	Boat obs	Indirect 2	1	1	Aberdeen, UK	98	2	1% (25-200m)
(Johnston *et al.* 2014)	Eider	Boat obs and modelling	Combined	11	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham	34513	Unk	0.3% (CI = 0.035-0.558%)

						Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.			
(Garthe & Hüppop 2004)	Eider	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 1 (median <5m)
(Day *et al.* 2003)	Eider	Land obs	Indirect 2	1	1	St Lawrence Island	17	1	0%
(Leopold *et al.* 2004)	Eider	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	235	1	1.3% between 25-200m
(Sadoti *et al.* 2005a)	Eider	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	84	3 (fall)	0%
(Paton *et al.* 2010)	Eider	Land obs	Indirect 2	1	1	Rhode Island	24195	1	0.2% between 25-125m
(Paton *et al.* 2010)	Eider	Boat obs	Indirect 2	1	1	Rhode Island	294	1	0% between 25-125m
(Rothery, Newton & Little 2009)	Eider	Land obs	Indirect 1	1	1	Blyth, NE England	1282	6	0%
(Pettersson 2005)	Eider	Radar	Direct	1	1	Utgrunden, Kalmar Sound, Sweden	57	4	0%
(Pettersson 2005)	Eider	Radar	Direct	1	1	Yttre Stengrund, Kalmar Sound, Sweden	2044	4	20%
(Krüger & Garthe 2001)	Eider	Land obs	Indirect 2	1	1	Wangerooge Island, Germany	14405	1	2%
(Petersen *et al.* 2006)	Eider	Radar	Direct	1	1	Horns Rev, Denmark	193	3	16%
(Larsen & Guillemette 2007)	Eider	Platform obs	Indirect 1	1	1	Tunø Knob, Denmark	1277	1	2%

3. Long-tailed duck

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Day *et al.* 2003)	Long-tailed duck	Land obs	Indirect 2	1	1	St Lawrence Island	108	1	0%
(Paton *et al.* 2010)	Long-tailed duck	Land or boat obs	Indirect 2	1	1	Rhode Island	280	1	0% between 25-125m
(Cook *et al.* 2012)	Long-tailed duck	Boat obs	Combined	3	3	Burbo Bank, St. Lawrence Island in Alaska and Nantucket Sound in Massachusetts	114	Unk	0%

4. Common scoter

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Genesis 2012)	Common scoter	Boat obs	Indirect 2	1	1	Aberdeen, UK	77	2	3% (25-200m)
(Gill *et al.* 2008)	Common scoter	Boat obs	Indirect 1	1	1	Kentish Flats, UK	66	2	0% (>20m)
(Johnston *et al.* 2014)	Common scoter	Boat obs and modelling	Combined	18	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of	30847	Unk	0.019% (CI = 0.001-0.109%)

						Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.			
(Krijgsveld *et al.* 2011)	Common scoter	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond,Netherlands	Unk	3	40%
(NWP Offshore Ltd 2006) p102	Common scoter	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	91	Unk	0% >20m
(NWP Offshore Ltd 2008)	Common scoter	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	10052	Unk	2% (>20m)
(RSK Environmental Ltd 2012)	Common scoter	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	8	2	13%
(Vanermen *et al.* 2013)	Common scoter	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	1453	4	2% (30-150m)
(Walls *et al.* 2013)	Common scoter	Boat obs	Indirect 1	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	1992 (operation)	2	0% (26-200m)
(Walls *et al.* 2013)	Common scoter	Boat obs	Indirect 2	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	31173 (pre-construction +construction)	7	0% (26-200m)
(Garthe & Hüppop 2004)	Common scoter	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 1 (median <5m)
(Leopold *et al.* 2004)	Common scoter	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	2258	1	0.8% between 25-200m
(Sadoti *et al.* 2005a)	Common scoter	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	96	3 (fall)	43%
(Rothery *et al.* 2009)	Common scoter	Land obs	Indirect 1 (although in presence of WF)	1	1	Blyth, NE England	341	6	0%
(Paton *et al.* 2010)	Common scoter	Land obs	Indirect 2	1	1	Rhode Island	4756	1	0.2% between 25-125m

(Paton *et al.* 2010)	Common scoter	Boat obs	Indirect 2	1	1	Rhode Island	277	1	0% between 25-125m
(Cook *et al.* 2012)	Common scoter	Boat obs	Combined	18	22	Barrow, Blyth, Burbo Bank, Docking Shoal, Dogger Bank, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Kentish Flats, London Array, Lynn & Inner Dowsing, North Hoyle, Rampion West of Duddon Sands, Horns Rev in Denmark, Thorntonbank in Belgium and Wangerooge in Germany	30847	Unk	1% (<0.1 – 17)
(Krüger & Garthe 2001)	Common scoter	Land obs	Indirect 2	1	1	Wangerooge Island, Germany	6754	1	3%

5. Velvet scoter

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Garthe & Hüppop 2004)	Velvet scoter	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 1 (median <5m)
(Day *et al.* 2003)	Velvet scoter	Land obs	Indirect 2	1	1	St Lawrence Island	5	1	0%
(Sadoti *et al.* 2005a)	Velvet scoter	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	88	3 (fall)	3%
(Paton *et al.* 2010)	Velvet scoter	Land obs	Indirect 2	1	1	Rhode Island	2973	1	7% between 25-125m
(Paton *et al.* 2010)	Velvet scoter	Boat obs	Indirect 2	1	1	Rhode Island	161	1	0% between 25-125m
(Cook *et al.* 2012)	Velvet scoter	Boat obs	Combined	3	4	Gunfleet Sands, Gwynt Y Mor and Weybourne	20	Unk	0%

6. Common goldeneye

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Paton *et al.* 2010)	Common goldeneye	Land obs	Indirect 2	1	1	Rhode Island	336	1	11.3% between 25-125m

7. Red-throated diver

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Genesis 2012)	Red-throated diver	Boat obs	Indirect 2	1	1	Aberdeen, UK	191	2	5% (25-200m)
(Gill *et al.* 2008)	Red-throated diver	Boat obs	Indirect 1	1	1	Kentish Flats, UK	300	2	2% (>20m)
(Johnston *et al.* 2014)	Red-throated diver	Boat obs and modelling	Combined	18	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	9686	Unk	0.062% (CI = 0.015-0.323%)

(NWP Offshore Ltd 2006) p102	Red-throated diver	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	9	Unk	0% >20m
(NWP Offshore Ltd 2008)	Red-throated diver	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	48	Unk	2% (>20m)
(Vanermen *et al.* 2013)	Red-throated diver	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	506	4	4% (30-150m)
(Walls *et al.* 2013)	Red-throated diver	Boat obs	Indirect 1	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	232 (operation)	2	17% (26-200m)
(Walls *et al.* 2013)	Red-throated diver	Boat obs	Indirect 2	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	415 (pre-construction +construction)	7	4% (26-200m)
(Garthe & Hüppop 2004)	Red-throated diver	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 2 (median 5-10m)
(Leopold *et al.* 2004)	Red-throated diver	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	284	1	8.5% between 25-200m
(Krijgsveld *et al.* 2005)	Red-throated diver	Radar	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond,Netherlands	103	2	24%
(Paton *et al.* 2010)	Red-throated diver	Land obs	Indirect 2	1	1	Rhode Island	1226	1	7.1% between 25-125m
(Paton *et al.* 2010)	Red-throated diver	Boat obs	Indirect 2	1	1	Rhode Island	106	1	28.3% between 25-125m
(Sadoti *et al.* 2005a)	Red-throated diver	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	28	3 (fall)	3%

(Cook *et al.* 2012)	Red-throated diver	Boat obs	Combined	18	22	Barrow, Burbo Bank, Docking Shoal, Dogger Bank, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Kentish Flats, Lincs, London Array, Lynn & Inner Dowsing, North Hoyle, West of Duddon Sands, Horns Rev in Denmark, Thorntonbank in Belgium, Egmond aan Zee wind farm in the Netherlands and Wangerooge in Germany.	9715	Unk	2% (<0.1 - 22)
(Krüger & Garthe 2001)	Red-throated diver	Land obs	Indirect 2	1	1	Wangerooge Island, Germany	247	1	0%

8. Black-throated diver

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Gill *et al.* 2008)	Black-throated diver	Boat obs	Indirect 1	1	1	Kentish Flats, UK	22	2	9% (>20m)
(Johnston *et al.* 2014)	Black-throated diver	Boat obs and modelling	Combined	6	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race	126	Unk	0.081% (CI = 0.068-0.331%)

						Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.			
(NWP Offshore Ltd 2008)	Black-throated diver	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	2	Unk	50% (>20m)
(Garthe & Hüppop 2004)	Black-throated diver	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 2 (median 5-10m)
(Cook *et al.* 2012)	Black-throated diver	Boat obs	Combined	6	7	Dogger Bank, Gunfleet Sands, Gwynt Y Mor, Kentish Flats, London Array and North Hoyle.	126	Unk	0.1% (<0.1 - 30)

9. Great northern diver

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(NWP Offshore Ltd 2006) p102	Great northern diver	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	1	Unk	0% >20m
(Paton *et al.* 2010)	Great northern diver	Land obs	Indirect 2	1	1	Rhode Island	2762	1	22.8% between 25-125m
(Paton *et al.* 2010)	Great northern diver	Boat obs	Indirect 2	1	1	Rhode Island	292	1	5.8% between 25-125m
(Sadoti *et al.* 2005a)	Great northern diver	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	27	3 (fall)	4%
(Cook *et al.* 2012)	Great northern diver	Boat obs	Combined	4	5	Argyll Array, Humber Gateway, Gwynt Y Mor and Burbo Bank	14	Unk	0%

10. Great-crested grebe

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Gill *et al.* 2008)	Great crested grebe	Boat obs	Indirect 1	1	1	Kentish Flats, UK	3	2	0% (>20m)
(NWP Offshore Ltd 2008)	Great crested grebe	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	1	Unk	0% (>20m)
(Vanermen *et al.* 2013)	Great crested grebe	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	175	4	0% (30-150m)
(Garthe & Hüppop 2004)	Great crested grebe	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 2 (median 5-10m)
(Cook *et al.* 2012)	Great crested grebe	Boat obs	Combined	4	4	Gunfleet Sands, Gwynt Y Mor, Kentish Flats and Egmond aan Zee	82	Unk	0%
(Leopold *et al.* 2004)	Great crested grebe	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	32	1	0% between 25-200m

11. Slavonian grebe

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(NWP Offshore Ltd 2008)	Related spp. – red-necked grebe	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	1	Unk	0% (>20m)
(Paton *et al.* 2010)	Slavonian grebe	Land obs	Indirect 2	1	1	Rhode Island	85	1	23.5% between 25-125m
(Cook *et al.* 2012)	Related spp. – red-necked grebe	Boat obs	Combined	1	1	North Hoyle	1	Unk	0%

12. Northern fulmar

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Driessen 2013)	Fulmar	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	776	2	0% (>20m)
(Genesis 2012)	Fulmar	Boat obs	Indirect 2	1	1	Aberdeen, UK	919	2	1% (25-200m)
(Gill *et al.* 2008)	Fulmar	Boat obs	Indirect 1	1	1	Kentish Flats, UK	3	2	0% (>20m)
(Johnston *et al.* 2014)	Fulmar	Boat obs and modelling	Combined	22	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	29168	Unk	0.010% (CI = 0.000-0.092%)
(Moray Offshore Renewables Ltd 2012)	Fulmar	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	3978	2	0% (20-200m)
(NWP Offshore Ltd 2006) p102	Fulmar	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	1	Unk	0% >20m

(NWP Offshore Ltd 2008)	Fulmar	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	6	Unk	0% (>20m)
(RSK Environmental Ltd 2012)	Fulmar	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	677	2	0%
(Vanermen *et al.* 2013)	Fulmar	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	810	4	0% (30-150m)
(Garthe & Hüppop 2004)	Fulmar	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 1 (median <5m)
(Cook *et al.* 2012)	Fulmar	Boat obs	Combined	21	25	Argyll Array, Barrow, Docking Shoal, Dudgeon, Dogger Bank, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Islay, Kentish Flats, London Array, Lynn & Inner Dowsing, Moray Firth, Neart na Gaoithe, North Hoyle, Race Back, Rampion, West of Duddon Sands and Meetpost Nordwijk and Egmond aan Zee wind farm in the Netherlands	29168	Unk	0.2% (<0.1 – 22)
(Leopold *et al.* 2004)	Fulmar	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	178	1	0% between 25-200m
(Krijgsveld *et al.* 2005)	Fulmar	Radar	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond,Netherlands	10	2	0%

13. Sooty shearwater

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Driessen 2013)	Sooty shearwater	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	59	2	0% (>20m)

(Moray Offshore Renewables Ltd 2012)	Sooty shearwater	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	48	2	0% (20-200m)
(Cook *et al.* 2012)	Sooty shearwater	Boat obs	Combined	2	2	Humber Gateway and Weybourne.	2	Unk	0%
(Paton *et al.* 2010)	Sooty shearwater	Land obs	Indirect 2	1	1	Rhode Island	5	1	0% between 25-125m
(Paton *et al.* 2010)	Sooty shearwater	Boat obs	Indirect 2	1	1	Rhode Island	16	1	0% between 25-125m

14. Manx shearwater

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Driessen 2013)	Manx shearwater	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	106	2	0% (>20m)
(Genesis 2012)	Manx shearwater	Boat obs	Indirect 2	1	1	Aberdeen, UK	13	2	0% (25-200m)
(Johnston *et al.* 2014)	Manx shearwater	Boat obs and modelling	Combined	10	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	6801	Unk	0.000% (CI = 0.000-0.000%)

(Moray Offshore Renewables Ltd 2012)	Manx shearwater	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	11	2	0% (20-200m)
(NWP Offshore Ltd 2008)	Manx shearwater	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	23	Unk	0% (>20m)
(RSK Environmental Ltd 2012)	Manx shearwater	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	11	2	0%
(Walls *et al.* 2013)	Manx shearwater	Boat obs	Indirect 1	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	103 (operation)	2	0% (26-200m)
(Walls *et al.* 2013)	Manx shearwater	Boat obs	Indirect 2	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	2131 (pre-construction +construction)	7	0% (26-200m)
(Cook *et al.* 2012)	Manx shearwater	Boat obs	Combined	10	10	Argyll Array, Dogger Bank, Gwynt Y Mor, Humber Gateway, Islay, Moray Firth, Neart na Gaoithe, North Hoyle, Rampion and West of Duddon Sands.	6957	Unk	0.04% (<0.01 – 10)

15. European storm-petrel

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Moray Offshore Renewables Ltd 2012)	European storm-petrel	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	45	2	0% (20-200m)
(Cook *et al.* 2012)	European storm-petrel	Boat obs	Combined	2	2	Gwynt Y Mor and West of Duddon Sands	52	Unk	2%

16. Leach's storm-petrel

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Moray Offshore Renewables Ltd 2012)	Leach's storm-petrel	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	1	2	0% (20-200m)
(Cook *et al.* 2012)	Related spp. - European storm-petrel	Boat obs	Combined	2	2	Gwynt Y Mor and West of Duddon Sands	52	Unk	2%

17. Northern gannet

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Cook *et al.* 2012)	Gannet	Boat obs	Combined	27	32	Argyll Array, Barrow, Blyth, Docking Shoal, Dogger Bank, Dudgeon, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Islay, Kentish Flats, Lincs, London Array, Lynn & Inner Dowsing, Moray Firth, Neart na Gaoithe, North Hoyle, Race Bank, Rampion, Sheringham Shoal, West of Duddon Sands, Westernmost Rough, Horns Rev in Denmark, Meetpost Nordwijk and Egmond aan Zee wind farm in the Netherlands and Thorntonbank in Belgium.	44851	Unk	9.6 (<0.1 – 19.9)
(NWP Offshore Ltd 2006) p102	Gannet	Boat obs	Indirect 1	1	1	North Hoyle, NW Wales, UK	29	Unk	0% >20m

(NWP Offshore Ltd 2008)	Gannet	Boat obs	Indirect 1	1	1	North Hoyle, NW Wales, UK	162	Unk	7% (12/162 >20m)
(Krijgsveld *et al.* 2011)	Gannet	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	Unk	3	40%
(Krijgsveld *et al.* 2005)	Gannet	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	143	2	44%
(Mendel *et al.* 2014)	Gannet	Boat obs (laser height finders)	Indirect 2	1	1	Alpha Ventus WF	24	2	40% Median height 15-18m
(Rothery *et al.* 2009)	Gannet	Land obs	Indirect 1	1	1	Blyth, NE England	414	6	13%
(Garthe & Hüppop 2004)	Gannet	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 3 (median 10-20m, few >50m)
(Sadoti *et al.* 2005b)	Gannet	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	85	2 (breed)	0% (2003 mean = 5m, 2004 mean = 7m)
(Paton *et al.* 2010)	Gannet	Land obs	Indirect 2	1	1	Rhode Island	8560	1	10% between 25-125m
(Paton *et al.* 2010)	Gannet	Boat obs	Indirect 2	1	1	Rhode Island	1278	1	7% between 25-125m
(Leopold *et al.* 2004)	Gannet	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	803	1	13% between 25-200m
(Moray Offshore Renewables Ltd 2012)	Gannet	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	608	2	12% (20-200m)
(Driessen 2013)	Gannet	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	11846	2	16% (>20m)
(RSK Environmental Ltd 2012)	Gannet	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	1554	2	13%

(Triton Knoll Offshore Wind Farm Ltd 2011)	Gannet	Boat obs	Indirect 2	1	1	Triton Knoll, East England, UK	2709	2	16%
(Genesis 2012)	Gannet	Boat obs	Indirect 2	1	1	Aberdeen, UK	1403	2	9% (25-200m)
(Walls *et al.* 2013)	Gannet	Boat obs	Indirect 2	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	951 (pre-construction +construction)	7	8% (26-200m)
(Walls *et al.* 2013)	Gannet	Boat obs	Indirect 1	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	113 (operation)	2	8% (26-200m)
(Vanermen *et al.* 2013)	Gannet	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	3713	4	5% (30-150m)
(Johnston *et al.* 2014)	Gannet	Boat obs and modelling	Combined	27	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	44851	Unk	13% (CI = 6-20%)
(Gill *et al.* 2008)	Gannet	Boat obs	Indirect 1	1	1	Kentish Flats, UK	59	2	7% (>20m)

18. Great cormorant

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Genesis 2012)	Great cormorant	Boat obs	Indirect 2	1	1	Aberdeen, UK	160	2	2% (25-200m)
(Gill *et al.* 2008)	Great cormorant	Boat obs	Indirect 1	1	1	Kentish Flats, UK	99	2	4% (>20m)
(Johnston *et al.* 2014)	Great cormorant	Boat obs and modelling	Combined	14	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	20227	Unk	0.017% (CI = 0.008-0.271%)
(Krijgsveld *et al.* 2011)	Great cormorant	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond,Netherlands	Unk	3	25%
(NWP Offshore Ltd 2006) p102	Great cormorant	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	104	Unk	0% >20m
(NWP Offshore Ltd 2008)	Great cormorant	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	425	Unk	19% (>20m)

(Vanermen *et al.* 2013)	Great cormorant	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	1216	4	8% (30-150m)
(Walls *et al.* 2013)	Great cormorant	Boat obs	Indirect 1	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	681 (operation)	2	12% (26-200m)
(Walls *et al.* 2013)	Great cormorant	Boat obs	Indirect 2	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	1756 (pre-construction +construction)	7	3% (26-200m)
(Garthe & Hüppop 2004)	Great cormorant	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 1 (median <5m)
(Rothery *et al.* 2009)	Great cormorant	Land obs	Indirect 1	1	1	Blyth, NE England	352	6	13%
(Leopold *et al.* 2004)	Great cormorant	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	929	1	7.5% between 25-200m
(Paton *et al.* 2010)	Great cormorant	Land obs	Indirect 2	1	1	Rhode Island	2014	1	7.3% between 25-125m
(Paton *et al.* 2010)	Great cormorant	Boat obs	Indirect 2	1	1	Rhode Island	15	1	0% between 25-125m
(Krijgsveld *et al.* 2005)	Great cormorant	Radar	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond,Netherlands	Unk	2	28%

19. Shag

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Driessen 2013)	Shag	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	15	2	0% (>20m)
(Genesis 2012)	Shag	Boat obs	Indirect 2	1	1	Aberdeen, UK	126	2	0% (25-200m)
(Johnston *et al.* 2014)	Shag	Boat obs and modelling	Combined	4	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London	233	Unk	0.126% (CI = 0.020-0.643%)

						Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.			
(Moray Offshore Renewables Ltd 2012)	Shag	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	8	2	0% (20-200m)
(NWP Offshore Ltd 2006) p102	Shag	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	5	Unk	0% >20m
(NWP Offshore Ltd 2008)	Shag	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	23	Unk	0% (>20m)
(Cook *et al.* 2012)	Shag	Boat obs	Combined	4	5	Barrow, Dogger Bank Gwynt Y Mor and North Hoyle	233	Unk	12.4 (1.9 – 60%)

20. White-tailed eagle

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Nygård *et al.* 2010)	White-tailed eagle	GPS	Direct	1	1	Smøla, western Norway	34	5	24% (hub height = 70 m, blade radius = 38-41 m)
(Kahlert *et al.* 2011)	White-tailed eagle	Land obs	Indirect 1	1	1	Rodsand Lagoon, Denmark	9	1	4/9 birds at rotor height (inshore local flights) (22-115m)

21. Arctic skua

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Cook *et al.* 2012)	Arctic skua	Boat obs	Combined	12	14	Argyll Array, Barrow, Dogger Bank, Greater Gabbard, Gwynt Y Mor, Humber Gateway, Islay, Kentish Flats, Lynn & Inner Dowsing, Moray Firth, North Hoyle and Rampion	331	Unk	3.8 (<0.1 – 5.7)
(NWP Offshore Ltd 2008)	Arctic skua	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	3	Unk	0% (0/3 >20m)
(Garthe & Hüppop 2004)	Arctic skua	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 3 (median 10-20m, few >50m)
(Paton *et al.* 2010)	Arctic skua	Land obs	Indirect 2	1	1	Rhode Island	19	1	21% between 25-125m
(Paton *et al.* 2010)	Arctic skua	Boat obs	Indirect 2	1	1	Rhode Island	1	1	0% between 25-125m
(Moray Offshore Renewables Ltd 2012)	Arctic skua	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	28	2	0% (20-200m)
(Driessen 2013)	Arctic skua	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	21	2	5% (>25m)
(RSK Environmental Ltd 2012)	Arctic skua	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	3	2	0%
(Triton Knoll Offshore Wind Farm Ltd 2011)	Arctic skua	Boat obs	Indirect 2	1	1	Triton Knoll, East England, UK	110	2	5%
(Genesis 2012)	Arctic skua	Boat obs	Indirect 2	1	1	Aberdeen, UK	56	2	16% (25-200m)

(Johnston *et al.* 2014)	Arctic skua	Boat obs and modelling	Combined	12	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	331	Unk	3% (CI = 2-10%)
(Gill *et al.* 2008)	Arctic skua	Boat obs	Indirect 1	1	1	Kentish Flats, UK	3	2	0% (>20m)

22. Great skua

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Cook *et al.* 2012)	Great skua	Boat obs	Combined	11	14	Argyll Array, Dogger Bank, Greater Gabbard, Gwynt Y Mor, Humber Gateway, Islay, Lynn & Inner Dowsing, Moray Firth, Neart na Gaoithe, North Hoyle and Rampion.	1202	Unk	4.3 (1.2 – 28.4)
(NWP Offshore Ltd 2008)	Great skua	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	2	Unk	0% (0/2 >20m)
(Garthe & Hüppop 2004)	Great skua	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 3 (median 10-20m, few >50m)

(Moray Offshore Renewables Ltd 2012)	Great skua	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	110	2	1% (20-200m)
(Driessen 2013)	Great skua	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	39	2	9% (>25m)
(RSK Environmental Ltd 2012)	Great skua	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	61	2	16%
(Genesis 2012)	Great skua	Boat obs	Indirect 2	1	1	Aberdeen, UK	30	2	13% (25-200m)
(Vanermen *et al.* 2013)	Great skua	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	192	4	5% (30-150m)
(Johnston *et al.* 2014)	Great skua	Boat obs and modelling	Combined	12	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	1202	Unk	6% (CI = 4-18%)

23. Black-headed gull

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Cook *et al.* 2012)	Black-headed gull	Boat obs	Combined	17	20	Barrow, Blyth, Dogger Bank, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Kentish Flats, London Array, Lynn & Inner Dowsing, Neart na Gaoithe, North Hoyle, Rampion, West of Duddon Sands, Thorntonbank in Belgium and two sites in the Netherlands.	4490	Unk	7.9 (0.4 – 50.1)
(NWP Offshore Ltd 2006) p102	Black-headed gull	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	4	Unk	0% >20m
(NWP Offshore Ltd 2008)	Black-headed gull	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	7	Unk	14% (1/7 >20m)
(Krijgsveld *et al.* 2011)	Black-headed gull	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond,Netherlands	Unk	3	40%
(Krijgsveld et al. 2005)	Black-headed gull	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond,Netherlands	334	2	25%
(Bergh, van den Spaans & van Swelm 2002)	Black-headed gull	Land obs	Indirect 2	1	Unk	Slufterdam	82	Unk	4%
(Bergh *et al.* 2002)	Black-headed gull	Land obs	Indirect 2	1	Unk	Slag Dobbelsteen	41	Unk	78%
(Garthe & Hüppop 2004)	Black-headed gull	ESAS	Indirect 2	Unk		Unk	Unk	Unk	Score 4 (median height 10-20m, 10% above 100m)
(Rothery *et al.* 2009)	Black-headed gull	Land obs	Indirect 1	1	1	Blyth, NE England	978	6	4%

(Vanermen *et al.* 2013)	Black-headed gull	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	2413	4	6% (30-150m)
(Johnston *et al.* 2014)	Black-headed gull	Boat obs and modelling	Combined	16	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	4436	Unk	14% (CI = 6-26%)
(Gill *et al.* 2008)	Black-headed gull	Boat obs	Indirect 1	1	1	Kentish Flats, UK	112	2	9% (>20m)

24. Common gull

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Bergh *et al.* 2002)	Common gull	Land obs	Indirect 2	Unk	Unk	Slag Dobbelsteen	120	Unk	58
(Garthe & Hüppop 2004)	Little gull	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 3
(Krijgsveld et al. 2005)	Common gull	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	1517	2	48

(NWP Offshore Ltd 2006) p102	Common gull	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	93	2	0
(NWP Offshore Ltd 2008)	Common gull	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	125	2	22
(Cook *et al.* 2012)	Common gull	Boat obs	Combined	19	23	Barrow, Dogger Bank, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Islay, Kentish Flats, Lincs, London Array, Lynn & Inner Dowsing, Moray Firth, Neart na Gaoithe, North Hoyle, Rampion, West of Duddon Sands, Westernmost Rough, Thorntonbank in Belgium and Meetpost Nordwijk in the Netherlands.	10190	Unk	22.9 (CI = 8.5-47)
(Krijgsveld *et al.* 2011)	Common gull	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond,Netherlands	Unk	3	45%
(Driessen 2013)	Common gull	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	20	2	60% (>20m)
(RSK Environmental Ltd 2012)	Common gull	Boat obs	Indirect 2	1	1	Rampion WF, UK	317	2	15%
(Genesis 2012)	Common gull	Boat obs	Indirect 2	1	1	Aberdeen, UK	761	2	31% (25-200m)
(Vanermen *et al.* 2013)	Common gull	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	8648	4	15% (30-150m)
(Johnston *et al.* 2014)	Common gull	Boat obs and modelling	Combined	20	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater	10190	Unk	22% (CI = 19-30%)

						Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.			
(Gill *et al.* 2008)	Common gull	Boat obs	Indirect 1	1	1	Kentish Flats, UK	341	2	24% (>20m)

25. Lesser black-backed gull

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Cook *et al.* 2012)	Lesser black-backed gull	Boat obs	Combined	23	29	Barrow, Docking Shoal, Dogger Bank, Dudgeon, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Islay, Kentish Flats, Lincs, London Array, Lynn & Inner Dowsing, Moray Firth, Neart na Gaoithe, North Hoyle, Race Bank, Rampion, Sheringham Shoal, West of Duddon Sands, Westernmost Rough Thorntonbank in Belgium, and two sites in the Netherlands.	35114	Unk	25.2 (7.8 – 51.6)
(NWP Offshore Ltd 2006) p102	Lesser black-backed gull	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	28	Unk	11% (3/28 >20m)

(NWP Offshore Ltd 2008)	Lesser black-backed gull	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	63	Unk	37% (23/63 >20m)
(Bergh *et al.* 2002)	Lesser black-backed gull	Land obs	Indirect 2	1	Unk	Slufterdam	92	Unk	34%
(Bergh *et al.* 2002)	Lesser black-backed gull	Land obs	Indirect 2	1	Unk	Slag Dobbelsteen	1828	Unk	90%
(Garthe & Hüppop 2004)	Lesser black-backed gull	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 4 (median 10-20m, 10% abv 50m)
(Krijgsveld et al. 2005)	Lesser black-backed gull	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	2470	2	55%
(Krijgsveld *et al.* 2011)	Lesser black-backed gull	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	Unk	3	60%
(Mendel *et al.* 2014)	Lesser black-backed gull	Boat obs (laser height finders)	Indirect 2	1	1	Alpha Ventus WF	637	2	65% Median height 25-30m
(Corman & Garthe 2014)	Lesser black-backed gull	GPS	Direct	2	1	Germany	8	1	10% 20-200m
(Moray Offshore Renewables Ltd 2012)	Lesser black-backed gull	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	11	2	27% (20-200m)
(Driessen 2013)	Lesser black-backed gull	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	43	2	60% (>20m)
(RSK Environmental Ltd 2012)	Lesser black-backed gull	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	160	2	25%
(Vanermen *et al.* 2013)	Lesser black-backed gull	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	14919	4	22% (30-150m)

(Johnston *et al.* 2014)	Lesser black-backed gull	Boat obs and modelling	Combined	23	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	35045	Unk	28% (CI = 20-43%)
(Gill *et al.* 2008)	Lesser black-backed gull	Boat obs	Indirect 1	1	1	Kentish Flats, UK	375	2	22% (>20m)

26. Herring gull

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Cook *et al.* 2012)	Herring gull	Boat obs	Combined	19	24	Barrow, Blyth, Dogger Bank, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Islay, Kentish Flats, London Array, Lynn & Inner Dowsing, Moray Firth, Neart na Gaoithe, North Hoyle, Rampion, West of Duddon Sands, Westernmost Rough, Meetpost Nordwijk in the Netherlands and Thorntonbank in the Belgium.	25252	Unk	28.4 (15.9 – 48.1)

(NWP Offshore Ltd 2006) p102	Herring gull	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	87	Unk	2% (2/87 >20m)
(NWP Offshore Ltd 2008)	Herring gull	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	308	Unk	15% (46/308 >20m)
(Krijgsveld *et al.* 2011)	Herring gull	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	Unk	3	60%
(Krijgsveld et al. 2005)	Herring gull	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	2223	2	50%
(Mendel *et al.* 2014)	Herring gull	Boat obs (laser height finders)	Indirect 2	1	1	Alpha Ventus WF	25	2	50% Median height 30-35m
(Rothery *et al.* 2009)	Herring gull	Land obs	Indirect 1	1	1	Blyth, NE England	1408	6	33%
(Bergh *et al.* 2002)	Herring gull	Land obs	Indirect 2	1	Unk	Slufterdam	71	Unk	33%
(Bergh *et al.* 2002)	Herring gull	Land obs	Indirect 2	1	Unk	Slag Dobbelsteen	7327	Unk	84%
(Garthe & Hüppop 2004)	Herring gull	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 4 (median 10-20m, 10% >50m)
(Sadoti *et al.* 2005b)	Herring gull	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	63	2 (breed)	22% (2003 mean = 15m, 2004 mean = 8m)
(Sadoti *et al.* 2005a)	Herring gull	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	66	3 (fall)	5% (2002 mean = 8m, 2003 mean = 6m, 2004 mean = 8m)
(Paton *et al.* 2010)	Herring gull	Land obs	Indirect 2	1	1	Rhode Island	51036	1	15% between 25-125m
(Paton *et al.* 2010)	Herring gull	Boat obs	Indirect 2	1	1	Rhode Island	1652	1	13% between 25-125m

(Moray Offshore Renewables Ltd 2012)	Herring gull	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	313	2	33.5% (20-200m)
(Driessen 2013)	Herring gull	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	310	2	52% (>20m)
(RSK Environmental Ltd 2012)	Herring gull	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	1451	2	26%
(Genesis 2012)	Herring gull	Boat obs	Indirect 2	1	1	Aberdeen, UK	1197	2	32% (25-200m)
(Vanermen *et al.* 2013)	Herring gull	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	4518	4	15% (30-150m)
(Johnston *et al.* 2014)	Herring gull	Boat obs and modelling	Combined	20	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	25253	Unk	31% (CI = 25-41%)
(Gill *et al.* 2008)	Herring gull	Boat obs	Indirect 1	1	1	Kentish Flats, UK	793	2	21% (>20m)

27. Great black-backed gull

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Cook *et al.* 2012)	Great black-backed gull	Boat obs	Combined	19	24	Barrow, Blyth, Dogger Bank, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Islay, Kentish Flats, London Array, Lynn & Inner Dowsing, Moray Firth, Neart na Gaoithe, North Hoyle, Rampion, West of Duddon Sands, Westernmost Rough, Meetpost Nordwijk in the Netherlands and Thorntonbank in Belgium.	8911	Unk	33.1 (18.2 – 57.1)
(NWP Offshore Ltd 2006) p102	Great black-backed gull	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	47	Unk	0 >20m
(NWP Offshore Ltd 2008)	Great black-backed gull	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	66	Unk	39% (26/66 >20m)
(Krijgsveld *et al.* 2011)	Great black-backed gull	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	Unk	3	60%
(Krijgsveld et al. 2005)	Great black-backed gull	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	143	2	57%
(Mendel *et al.* 2014)	Great black-backed gull	Boat obs (laser height finders)	Indirect 2	1	1	Alpha Ventus WF	25	2	75% Median height 30-35m
(Rothery *et al.* 2009)	Great black-backed gull	Land obs	Indirect 1	1	1	Blyth, NE England	564	6	44%
(Garthe & Hüppop 2004)	Great black-backed gull	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 3 (median 10-20m, few >50m)

(Sadoti *et al.* 2005b)	Great black-backed gull	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	163	2 (breed)	28% (2003 mean = 16m, 2004 mean = 13m)
(Sadoti *et al.* 2005a)	Great black-backed gull	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	38	3 (fall)	5% (2002 mean = 0m, 2003 mean = 6m, 2004 mean = 6m)
(Paton *et al.* 2010)	Great black-backed gull	Land obs	Indirect 2	1	1	Rhode Island	8610	1	8% between 25-125m
(Paton *et al.* 2010)	Great black-backed gull	Boat obs	Indirect 2	1	1	Rhode Island	1001	1	8% between 25-125m
(Moray Offshore Renewables Ltd 2012)	Great black-backed gull	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	207	2	30% (20-200m)
(Driessen 2013)	Great black-backed gull	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	259	2	61% (>20m)
(RSK Environmental Ltd 2012)	Great black-backed gull	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	748	2	36%
(Triton Knoll Offshore Wind Farm Ltd 2011)	Great black-backed gull	Boat obs	Indirect 2	1	1	Triton Knoll, East England, UK	1197	2	41%
(Genesis 2012)	Great black-backed gull	Boat obs	Indirect 2	1	1	Aberdeen, UK	328	2	41% (25-200m)
(Vanermen *et al.* 2013)	Great black-backed gull	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	2330	4	20% (30-150m)
(Johnston *et al.* 2014)	Great black-backed gull	Boat obs and modelling	Combined	19	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London	8911	Unk	33% (CI = 29-43%)

						Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.			
(Gill *et al.* 2008)	Great black-backed gull	Boat obs	Indirect 1	1	1	Kentish Flats, UK	62	2	13% (>20m)

28. Black-legged kittiwake

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Cook *et al.* 2012)	Kittiwake	Boat obs	Combined	25	29	Argyll Array, Barrow, Blyth, Docking Shoal, Dogger Bank, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Islay, Kentish Flats, London Array, Lynn & Inner Dowsing, Moray Firth, Neart na Gaoithe, North Hoyle, Race Bank, Rampion, West of Duddon Sands, Westernmost Rough, Weybourne, Meetpost Nordwijk and Egmond aan Zee wind farm in the Netherlands, Thorntonbank in Belgium and St. Lawrence Island in Alaska.	62975	Unk	15.7 (7.9 - 23.6)
(NWP Offshore Ltd 2006) p102	Kittiwake	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	113	Unk	0% >20m

(NWP Offshore Ltd 2008)	Kittiwake	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	32	Unk	9% (32/353 >20m)
(Krijgsveld *et al.* 2011)	Kittiwake	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	Unk	3	50%
(Krijgsveld et al. 2005)	Kittiwake	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	2459	2	38%
(Mendel *et al.* 2014)	Kittiwake	Boat obs (laser height finders)	Indirect 2	1	1	Alpha Ventus WF	36	2	40% Median height 15-18m
(Rothery *et al.* 2009)	Kittiwake	Land obs	Indirect 1	1	1	Blyth, NE England	1350	6	11%
(Garthe & Hüppop 2004)	Kittiwake	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 2 (median 5-10m)
(Paton *et al.* 2010)	Kittiwake	Land obs	Indirect 2	1	1	Rhode Island	56	1	0% between 25-125m
(Paton *et al.* 2010)	Kittiwake	Boat obs	Indirect 2	1	1	Rhode Island	55	1	11% between 25-125m
(Leopold *et al.* 2004)	Kittiwake	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	637	1	9% between 25-200m
(Chamberlain *et al.* 2005)	Kittiwake	Boat obs	Indirect 2	1	1	Unk	2036	1	4%
(Day *et al.* 2003)	Kittiwake	Land obs	Indirect 2	1	1	St Lawrence Island	36	1	10%
(Moray Offshore Renewables Ltd 2012)	Kittiwake	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	2123	2	4.6% (20-200m)
(Driessen 2013)	Kittiwake	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	2381	2	25% (>20m)

(RSK Environmental Ltd 2012)	Kittiwake	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	1008	2	14%
(Triton Knoll Offshore Wind Farm Ltd 2011)	Kittiwake	Boat obs	Indirect 2	1	1	Triton Knoll, East England, UK	4051	2	15%
(Genesis 2012)	Kittiwake	Boat obs	Indirect 2	1	1	Aberdeen, UK	2791	2	19% (25-200m)
(Walls *et al.* 2013)	Kittiwake	Boat obs	Indirect 2	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	1431 (pre-construction +construction)	7	3% (26-200m)
(Walls *et al.* 2013)	Kittiwake	Boat obs	Indirect 1	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	192 (operation)	2	7% (26-200m)
(Vanermen *et al.* 2013)	Kittiwake	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	9310	4	9% (30-150m)
(Johnston *et al.* 2014)	Kittiwake	Boat obs and modelling	Combined	24	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	62939	Unk	15% (CI = 12-17%)

(Gill *et al.* 2008)	Kittiwake	Boat obs	Indirect 1	1	1	Kentish Flats, UK	76	2	1% (>20m)

29. Little tern

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Everaert & Stienen 2007)	Little tern	Land obs	Indirect 1	1	1	Zeebrugge, Belgium	2124	2	23%

30. Sandwich tern

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Genesis 2012)	Sandwich tern	Boat obs	Indirect 2	1	1	Aberdeen, UK	104	2	6% (25-200m)
(Gill *et al.* 2008)	Sandwich tern	Boat obs	Indirect 1	1	1	Kentish Flats, UK	74	2	1% (>20m)
(Johnston *et al.* 2014)	Sandwich tern	Boat obs and modelling	Combined	19	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	33982	Unk	0.070% (CI = 0.061-0.149%)

(Krijgsveld *et al.* 2011)	Sandwich tern	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	Unk	3	55%
(NWP Offshore Ltd 2006) p102	Sandwich tern	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	56	Unk	0% >20m
(NWP Offshore Ltd 2008)	Sandwich tern	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	24	Unk	4% (>20m)
(RSK Environmental Ltd 2012)	Sandwich tern	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	13	2	8%
(Triton Knoll Offshore Wind Farm Ltd 2011)	Sandwich tern	Boat obs	Indirect 2	1	1	Triton Knoll, East England, UK	Unk	2	41%
(Vanermen *et al.* 2013)	Sandwich tern	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	2111	4	2% (30-150m)
(Garthe & Hüppop 2004)	Sandwich tern	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 3 (median 10-20m, few abv 50m)
(Leopold *et al.* 2004)	Sandwich tern	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	419	1	4.5% between 25-200m
(Krijgsveld et al. 2005)	Sandwich tern	Sea obs (fixed platform)	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	236	2	56%
(Rothery *et al.* 2009)	Sandwich tern	Land obs	Indirect 1	1	1	Blyth, NE England	2137	6	3%
(Cook *et al.* 2012)	Sandwich tern	Boat obs	Combined	21	24	Barrow, Blyth, Docking Shoal, Dogger Bank, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Kentish Flats, London Array, Lynn & Inner Dowsing, North Hoyle, Race Bank,	33392	Unk	3.6% (0.7 – 35)

						Rampion, Sheringham Shoal, West of Duddon Sands, Westernmost Rough, Zeebrugge and Thorntonbank in Belgium, Egmond aan Zee wind farm in the Netherlands and Wangerooge in Germany			
(Krüger & Garthe 2001)	Sandwich tern	Land obs	Indirect 2	1	1	Wangerooge Island, Germany	959	1	0%
(Everaert & Stienen 2007)	Sandwich tern	Land obs	Indirect 1	1	1	Zeebrugge, Belgium	27571	2	10%
(Perrow, Skeate & Gilroy 2011)	Sandwich tern	Boat obs	Indirect 2	1	1	Norfolk, UK	Unk	3	48%

31. Common tern

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Driessen 2013)	Common tern	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	9	2	0% (>20m)
(Genesis 2012)	Common tern	Boat obs	Indirect 2	1	1	Aberdeen, UK	108	2	3% (25-200m)
(Gill *et al.* 2008)	Common tern	Boat obs	Indirect 1	1	1	Kentish Flats, UK	203	2	1% (>20m)
(Johnston *et al.* 2014)	Common tern	Boat obs and modelling	Combined	18	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22	19329	Unk	0.074% (CI = 0.044-0.099%)

						North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.			
(Moray Offshore Renewables Ltd 2012)	Common tern	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	1	2	0% (20-200m)
(NWP Offshore Ltd 2006) p102	Common tern	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	2	Unk	0% >20m
(NWP Offshore Ltd 2008)	Common tern	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	1	Unk	0% (>20m)
(RSK Environmental Ltd 2012)	Common tern	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	14	2	0%
(Triton Knoll Offshore Wind Farm Ltd 2011)	Common tern	Boat obs	Indirect 2	1	1	Triton Knoll, East England, UK	Unk	2	2%
(Vanermen *et al.* 2013)	Common tern	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	4112	4	1% (30-150m)
(Garthe & Hüppop 2004)	Common tern	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 2 (median 5-10m)
(Cook *et al.* 2012)	Common tern	Boat obs	Combined	19	23	Dogger Bank, Dudgeon, Greater Gabbard, Gwynt Y Mor, Humber Gateway, Kentish Flats, Lincs, London Array, Lynn & Inner Dowsing, Moray Firth, North Hoyle, Race Bank, Rampion, Sheringham	19332	Unk	12.7% (6 - 19)

						Shoal, West of Duddon Sands, Westernmost Rough, Weybourne and Thorntonbank and Zeebrugge in Belgium.			
(Paton *et al.* 2010)	Common tern	Land obs	Indirect 2	1	1	Rhode Island	3644	1	3.8% between 25-125m
(Paton *et al.* 2010)	Common tern	Boat obs	Indirect 2	1	1	Rhode Island	61	1	11.5% between 25-125m
(Leopold *et al.* 2004)	Common tern	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	1011	1	0.6% between 25-200m
(Sadoti *et al.* 2005b)	Common tern	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	29	2 (breed)	4%
(Sadoti *et al.* 2005a)	Common tern	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	193	3 (fall)	9%
(Bergh *et al.* 2002)	Common tern	Land obs	Indirect 2	1	Unk	Slufterdam	113	Unk	8%
(Krüger & Garthe 2001)	Common tern	Land obs	Indirect 2	1	1	Wangerooge Island, Germany	271	1	0%
(Everaert & Stienen 2007)	Common tern	Land obs	Indirect 1	1	1	Zeebrugge, Belgium	14491	2	13%

32. Roseate tern

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Paton *et al.* 2010)	Roseate tern	Land obs	Indirect 2	1	1	Rhode Island	125	1	0% between 25-125m
(Paton *et al.* 2010)	Roseate tern	Boat obs	Indirect 2	1	1	Rhode Island	8	1	12.5% between 25-125m

33. Arctic tern

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Driessen 2013)	Arctic tern	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	125	2	0% (>20m)

(Genesis 2012)	Arctic tern	Boat obs	Indirect 2	1	1	Aberdeen, UK	56	2	16% (25-200m)
(Johnston *et al.* 2014)	Arctic tern	Boat obs and modelling	Combined	10	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	2571	Unk	0.040% (CI = 0.006-0.143%)
(Moray Offshore Renewables Ltd 2012)	Arctic tern	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	520	2	3.5% (20-200m)
(RSK Environmental Ltd 2012)	Arctic tern	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	4	2	0%
(Garthe & Hüppop 2004)	Arctic tern	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 1 (median <5m)
(Cook *et al.* 2012)	Arctic tern	Boat obs	Combined	9	11	Barrow, Docking Shoal, Dogger Bank, Gwynt Y Mor, Humber Gateway, Moray Firth, Neart na Gaoithe, Rampion and West of Duddon Sands.	2571	Unk	2.8% (<0.1 – 23)

(Alerstam & Gudmundsson 1999)	Arctic tern	Radar	Direct (migration)	15	1	Northeast Passage, Arctic	Unk	1	Mainly migrate at 200-2000m
(Gudmundsson *et al.* 2002)	Arctic tern	Radar	Direct (migration)	23	1	Northwest Tundra, Arctic	Unk	1	Mean migration height 800m

34. Common guillemot

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Driessen 2013)	Common guillemot	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	2032	2	0% (>20m)
(Genesis 2012)	Common guillemot	Boat obs	Indirect 2	1	1	Aberdeen, UK	1631	2	1% (25-200m)
(Gill *et al.* 2008)	Common guillemot	Boat obs	Indirect 1	1	1	Kentish Flats, UK	8	2	0% (>20m)
(Johnston *et al.* 2014)	Common guillmot	Boat obs and modelling	Combined	22	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	36256	Unk	0.004% (CI = 0.000-0.102%)

(Moray Offshore Renewables Ltd 2012)	Common guillemot	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	3098	2	0% (20-200m)
(NWP Offshore Ltd 2006) p102	Common guillmot	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	29	Unk	0% >20m
(NWP Offshore Ltd 2008)	Common guillemot	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	116	Unk	0% (>20m)
(Vanermen *et al.* 2013)	Common guillemot	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	995	4	0% (30-150m)
(RSK Environmental Ltd 2012)	Common guillemot	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	3517	2	1%
(Walls *et al.* 2013)	Common guillemot	Boat obs	Indirect 1	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	277 (operation)	2	0% (26-200m)
(Walls *et al.* 2013)	Common guillemot	Boat obs	Indirect 2	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	1067 (pre-construction +construction)	7	1% (26-200m)
(Garthe & Hüppop 2004)	Common guillemot	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 1 (median <5m)
(Leopold *et al.* 2004)	Common guillemot	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	316	1	0.6% between 25-200m
(Krijgsveld et al. 2005)	Common guillemot	Radar	Indirect 2	1	1	Meetpost Noordwijk, Nr Egmond, Netherlands	440	2	3%
(Paton *et al.* 2010)	Common guillemot	Boat obs	Indirect 2	1	1	Rhode Island	131	1	0% between 25-125m
(Cook *et al.* 2012)	Common guillemot	Boat obs	Combined	22	26	Barrow, Burbo Bank, Dogger Bank, Greater Gabbard, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Kentish Flats, Lincs, London Array, Lynn & Inner Dowsing, Moray Firth, Neart na Gaoithe, North Hoyle,	36260	Unk	0.01% (<0.01 – 3.9)

						Race Bank, Rampion, Sheringham Shoal, West of Duddon Sands, Westernmost Rough, Weybourne, Thorntonbank in Belgium and Egmond aan Zee Windfarm in the Netherlands.			
(Day *et al.* 2003)	Common guillemot	Land obs	Indirect 2	1	1	St Lawrence Island	172	1	0%

35. Razorbill

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Driessen 2013)	Razorbill	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	1377	2	1% (>20m)
(Genesis 2012)	Razorbill	Boat obs	Indirect 2	1	1	Aberdeen, UK	668	2	0% (25-200m)
(Johnston *et al.* 2014)	Razorbill	Boat obs and modelling	Combined	19	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.	13172	Unk	0.027% (CI = 0.000-0.137%)

(Moray Offshore Renewables Ltd 2012)	Razorbill	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	796	2	0% (20-200m)
(NWP Offshore Ltd 2006) p102	Razorbill	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	7	Unk	0% >20m
(NWP Offshore Ltd 2008)	Razorbill	Boat survey	Indirect 1	1	1	North Hoyle, NW Wales, UK	72	Unk	0% (>20m)
(RSK Environmental Ltd 2012)	Razorbill	Boat obs	Indirect 2	1	1	Rampion WF, Sussex, UK	436	2	0%
(Triton Knoll Offshore Wind Farm Ltd 2011)	Razorbill	Boat obs	Indirect 2	1	1	Triton Knoll, East England, UK	Unk	2	0%
(Vanermen *et al.* 2013)	Razorbill	Boat obs	Indirect 1	3	1	North Sea, Belgium (incl. Thorntonbank)	937	4	0% (30-150m)
(Walls *et al.* 2013)	Razorbill	Boat obs	Indirect 1	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	30 (operation)	2	0% (26-200m)
(Walls *et al.* 2013)	Razorbill	Boat obs	Indirect 2	1	1	Robin Rigg WF, Solway Firth, Scotland, UK	1202 (pre-construction+construction)	7	0% (26-200m)
(Garthe & Hüppop 2004)	Razorbill	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 1 (median <5m)
(Leopold *et al.* 2004)	Razorbill	Boat obs	Indirect 2	1	1	East North Sea – nr Dutch coast	33	1	0% between 25-200m
(Sadoti *et al.* 2005a)	Razorbill	Boat obs	Indirect 2	1	1	Horseshoe Shoal, Nuntucket Sound, USA	3	3 (fall)	0%
(Paton *et al.* 2010)	Razorbill	Land obs	Indirect 2	1	1	Rhode Island	135	1	0% between 25-125m
(Paton *et al.* 2010)	Razorbill	Boat obs	Indirect 2	1	1	Rhode Island	93	1	0% between 25-125m

(Cook *et al.* 2012)	Razorbill	Boat obs	Combined	18	21	Barrow, Burbo Bank, Dogger Bank, Gunfleet Sands, Gwynt Y Mor, Humber Gateway, Islay, London Array, Lynn & Inner 35 Dowsing, Moray Firth, Neart na Gaoithe, North Hoyle, Race Bank, Rampion, Sheringham Shoal, Westernmost Rough, Thorntonbank in Belgium and Egmond aan Zee wind farm in the Netherlands.	13171	Unk	0.4% (<0.1 – 25)

36. Black guillemot

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Paton *et al.* 2010)	Black guillemot	Land obs	Indirect 2	1	1	Rhode Island	1	1	0% between 25-125m

37. Little auk

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Driessen 2013)	Little auk	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	152	2	0% (>20m)
(Johnston *et al.* 2014)	Little auk	Boat obs and modelling	Combined	4	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham	1287	Unk	0.036% (CI = 0.000-0.050%)

						Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.			
(Moray Offshore Renewables Ltd 2012)	Little auk	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	33	2	0% (20-200m)
(Paton *et al.* 2010)	Little auk	Boat obs	Indirect 2	1	1	Rhode Island	125	1	0% between 25-125m
(Cook *et al.* 2012)	Little auk	Boat obs	Combined	4	5	Dogger Bank, Gwynt Y Mor, Islay and Moray Firth	1287	Unk	0.03% (<0.01 – 15)

38. Atlantic puffin

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	No. of years	% turbine flight
(Driessen 2013)	Puffin	Boat obs	Indirect 2	1	1	Inch Cape, nr Angus, Scotland	1061	2	0% (>20m)
(Genesis 2012)	Puffin	Boat obs	Indirect 2	1	1	Aberdeen, UK	169	2	0% (25-200m)
(Johnston *et al.* 2014)	Puffin	Boat obs and modelling	Combined	8	Unk	1 Argyll Array, 2 Barrow, 3 Blyth, 4 Burbo Bank, 5 Docking Shoal, 6 Dogger Bank, 7 Dudgeon, 8 Egmond ann Zee, 9 Greater Gabbard, 10 Gunfleet Sands, 11 Gwynt y Mor, 12 Horns Rev, 13 Humber Gateway, 14 Islay, 15 Kentish Flats, 16 Lincs, 17 London Array, 18 Lynn & Inner Dowsing, 19 Meetpost Noordwijk, 20 Moray Firth, 21 Neart na Gaoithe, 22 North Hoyle, 23 Nysted, 24 Race Bank, 25 Rampion, 26 Sheringham Shoal, 27 Thorntonbank, 28 Tuno Knob, 29 Wangerooge, 30 West of	5979	Unk	0.000% (CI = 0.000-0.068%)

						Duddon Sands, 31 Westermost Rough, 32 Zeebrugge.			
(Moray Offshore Renewables Ltd 2012)	Puffin	Boat obs	Indirect 2	1	1	Moray Firth, Scotland	397	2	0% (20-200m)
(Triton Knoll Offshore Wind Farm Ltd 2011)	Puffin	Boat obs	Indirect 2	1	1	Triton Knoll, East England, UK	Unk	2	0%
(Garthe & Hüppop 2004)	Puffin	ESAS	Indirect 2	Unk	Unk	Unk	Unk	Unk	Score 1 (median <5m)
(Paton *et al.* 2010)	Puffin	Boat obs	Indirect 2	1	1	Rhode Island	5	1	0% between 25-125m
(Cook *et al.* 2012)	Puffin	Boat obs	Combined	8	20	Dogger Bank, Humber Gateway, Islay, Moray Firth, Neart na Gaoithe, West of Duddon Sands, Westernmost Rough and Weybourne.	5981	Unk	0.1% (<0.1 – 8)

Appendix 1
Supplementary Table 2: Disturbance and displacement caused by structures
Tables of data contributing to confidence levels for each species

Obs = observation
Unk = unknown

1. Greater scaup

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Poulton, Lovvorn & Takekawa 2002)	Greater scaup	Human activity	Land obs	Anecdotal	3	1	California, USA	Unk	na	Jan-Mar	1	Absent/lower numbers in sites when humans/hunters present.
(Forrester & Andrews 2007)	Greater scaup	Structures	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	May avoid structures, preferring open water.

2. Common eider

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(ORPC Maine 2012)	Common eider	Pile drive noise	Land survey	Speculative/ anecdotal	1	1	Cobscook Bay, US	Unk	Tidal	Mar	1	No response to vibratory or diesel impact pile drive hammer.
(Inger, Harrison & Bearhop 2010)	Common eider	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	343	Tidal	All	4	No effect of SeaGen device found. Some evidence that mean distance of eider to device was greater during deployment than during operation. Unknown if biologically significant.
(Petersen & Fox 2007)	Common eider	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	Jan-Apr	1	Not seen in WF area – mostly close to shore in shallow water.

(NIRAS Consulting Engineers and Planners A/S 2008)	Common eider	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Forrester & Andrews 2007)	Common eider	Structures	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Seems to be more attracted to structures, such as discharge pipes, than repelled.

3. Long-tailed duck

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Petersen *et al.* 2013)	Long-tailed duck	Structures	Aerial obs	BACI	2	1	Nysted and Rodsand, Denmark	220-894	Wind	All	9	Reduction in abundance in and around WFs, with apparent redistribution away from WF to shallower waters. No sign of habituation.
(Forrester & Andrews 2007)	Long-tailed duck	Structures	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	May avoid structures, preferring open water.

4. Common scoter

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Petersen & Fox 2007)	Common scoter	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	All	9	Avoidance of WF area for 5 years following construction. After 5yrs no difference in numbers of birds inside and outside WF. Possible explanations include habituation and/or time-lagged increase in benthic prey.

(NIRAS Consulting Engineers and Planners A/S 2008)	Common scoter	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Petersen *et al.* 2006)	Common scoter	Structures	Radar	Observation	1	1	Horns Rev, Denmark	2379	Wind	All	3	Some flew into WF (33/288 tracks). Most pass around WF.
(Walls *et al.* 2013)	Common scoter	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	178117	Wind	All	11	Decline in numbers on the water between pre-construction and construction, and in flight between pre-construction and operation.
(Krijgsveld *et al.* 2011)	Common scoter	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Strong avoidance - rarely observed in WF.
(Leopold, Dijkman & Teal 2011)	Common scoter	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Never seen in WF but potentially rarely ventured far enough from near-shore waters to be in vicinity of WF.
(NWP Offshore Ltd 2008)	Common scoter	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	No evidence of barrier effect. Some observed flying through the WF – some are potentially 'escape' flights caused by survey vessel.
(Gill *et al.* 2008)	Common scoter	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	Some seen flying in WF.
(Forrester & Andrews 2007)	Common scoter	Structures	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	May avoid structures, preferring open water.

(Christensen *et al.* 2004)	Common scoter	Structures	Platform obs Radar	Observation	1	1	Horns Rev, Denmark	70	Wind	All	2	Few entered WF, all deflected flight routes around WF. Some birds exhibited avoidance behaviour 300-1000m from WF.

5. Velvet scoter

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Forrester & Andrews 2007)	Velvet scoter	Structures	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	May avoid structures, preferring open water.

6. Common goldeneye

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Forrester & Andrews 2007)	Common goldeneye	Structures	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Seems to be more attracted to structures, such as discharge pipes, than repelled.

7. Red-throated diver

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(NWP Offshore Ltd 2008)	Diver spp.	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Some evidence of avoidance once WF in operation.

(NIRAS Consulting Engineers and Planners A/S 2008)	Diver spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Leopold *et al.* 2011)	Red-throated diver	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Rarely seen in WF but potentially rarely ventured far enough from near-shore waters to be in vicinity of WF. Some seen inside WF area and one seen foraging.
(Gill *et al.* 2008)	Red-throated diver	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	Evidence of avoidance – lower numbers in WF during operation than pre-construction and construction phases.
(Petersen & Fox 2007)	Red-throated diver	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	All	9	Did not habituate to offshore wind turbines in Danish waters more than five years after construction. Closest diver seen at 1.6km from WF. Also avoided 4km buffer zone around WF.
(Walls *et al.* 2013)	Red-throated diver	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	1618	Wind	All	11	Decline in numbers on the water during construction and evidence of decrease in WF area during operation.
(Christensen *et al.* 2004)	Red-throated diver	Structures	Platform obs Radar	Observation	1	1	Horns Rev, Denmark	70	Wind	All	2	None entered WF, all deflected flight routes around WF. Observed foraging at least 100-800m from nearest turbine.

(Roycroft, Kelly & Lewis 2004)	Red-throated diver	Structures	Land obs	Observation	1	1	Bantry Bay, Ireland	Unk	na	All	2	Not found in the vicinity of mussel suspension structures – hypothesised that mechanical harvesting action and human disturbance kept them from using habitat.

8. Black-throated diver

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Petersen & Fox 2007)	Black-throated diver	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	All	9	Did not habituate to offshore wind turbines in Danish waters more than five years after construction. Closest diver seen at 1.6km from WF. Also avoided 4km buffer zone around WF.
(NIRAS Consulting Engineers and Planners A/S 2008)	Diver spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Leopold *et al.* 2011)	Black-throated diver	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Never seen in WF but potentially rarely ventured far enough from near-shore waters to be in vicinity of WF.
(NWP Offshore Ltd 2008)	Diver spp.	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Some evidence of avoidance once WF in operation.
(Gill *et al.* 2008)	Black-throated diver	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	Evidence of avoidance – lower numbers in WF during operation than pre-construction and construction phases.

(Christensen *et al.* 2004)	Black-throated diver	Structures	Platform obs Radar	Observation	1	1	Horns Rev, Denmark	70	Wind	All	2	None entered WF, all deflected flight routes around WF. Observed foraging at least 100-800m from nearest turbine.

9. Great northern diver

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(NWP Offshore Ltd 2008)	Diver spp.	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Some evidence of avoidance once WF in operation.
(NIRAS Consulting Engineers and Planners A/S 2008)	Diver spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(ORPC Maine 2012)	Great northern diver	Pile drive noise	Land survey	Speculative/ anecdotal	1	1	Cobscook Bay, US	3	Tidal	Mar	1	Brief decline (3 to 1) during impact hammer activity but not sure if usual fluctuations in movements – returned to normal within 15min
(Roycroft *et al.* 2004)	Great northern diver	Structures	Land obs	Observation	1	1	Bantry Bay, Ireland	Unk	na	All	2	Mean number of birds found near mussel suspension sites was non-significantly greater than in control sites (low sample size) – implies this spp. less disturbed by human structures/activity than other diver spp.

10. Great-crested grebe

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Gill *et al.* 2008)	Great crested grebe	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	Small numbers seen – coastal species not expected as far from coast to be seen in WF regularly.
(Forrester & Andrews 2007)	Great crested grebe	Structures	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	May avoid structures, preferring open water.
(Christensen *et al.* 2004)	Related spp. – Red-necked grebe	Structures	Platform obs	Observation	1	1	Horns Rev, Denmark	3	Wind	All	2	Birds showed signs of panic between the 1st and 2nd row of turbines – they spread out before joining again and continued to fly out of WF

11. Slavonian grebe

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Gill *et al.* 2008)	Related spp. – great crested grebe	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	Small numbers seen – coastal species not expected as far from coast to be seen in WF regularly.
(Forrester & Andrews 2007)	Slavonian grebe	Structures	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	May avoid structures, preferring open water.
(Christensen *et al.* 2004)	Related spp. – Red-necked grebe	Structures	Platform obs	Observation	1	1	Horns Rev, Denmark	3	Wind	All	2	Birds showed signs of panic between the 1st and 2nd row of turbines – they spread out before joining again and continued to fly out of WF

12. Northern fulmar

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Jackson 2014)	Northern fulmar	Structures	Photo obs	Observation	1	1	Pelamis device testing, Orkney, UK	Unk	Wave	Feb Mar May Jun	1	Occasionally rested on structure– no avoidance.
(Leopold *et al.* 2011)	Fulmar	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Never seen in WF but low numbers of birds in area.
(Baird 1990)	Fulmar	Structures	Boat obs	Observation	1	1	Bering Sea	Unk	na	Breed	1	Slightly higher numbers found at oil rig than in control area – suggests some attraction to structures.

13. Sooty shearwater

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Baird 1990)	Shearwater spp.	Structures	Boat obs	Observation	1	1	Bering Sea	Unk	na	Breed	1	Higher numbers found at oil rig than in control area – suggests some attraction to structures.
(Wiese *et al.* 2001)	Shearwater spp.	Structures Lighting	Boat obs Platform obs	Observation	1	1	Grand Banks	Unk	na	Breed	2	Attracted to lights on offshore structures, such as oil rigs. Authors suggest that circling structures as a result of light attraction can lead to death from exhaustion.

14. Manx shearwater

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Baird 1990)	Shearwater spp.	Structures	Boat obs	Observation	1	1	Bering Sea	Unk	na	Breed	1	Higher numbers found at oil rig than in control area – suggests some attraction to structures.
(Wiese *et al.* 2001)	Shearwater spp.	Structures Lighting	Boat obs Platform obs	Observation	1	1	Grand Banks	Unk	na	Breed	2	Attracted to lights on offshore structures, such as oil rigs. Authors suggest that circling structures as a result of light attraction can lead to death from exhaustion.

15. European storm-petrel

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Baird 1990)	Storm-petrel spp.	Structures	Boat obs	Observation	1	1	Bering Sea	Unk	na	Breed	1	Higher numbers found at oil rig than in control area – suggests some attraction to structures.
(Wiese *et al.* 2001)	Storm-petrel spp.	Structures Lighting	Boat obs Platform obs	Observation	1	1	Grand Banks	Unk	na	Breed	2	Attracted to lights on offshore structures, such as oil rigs. Authors suggest that circling structures as a result of light attraction can lead to death from exhaustion.

16. Leach's storm-petrel

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Baird 1990)	Storm-petrel spp.	Structures	Boat obs	Observation	1	1	Bering Sea	Unk	na	Breed	1	Higher numbers found at oil rig than in control area – suggests some attraction to structures.

(Wiese *et al.* 2001)	Storm-petrel spp.	Structures Lighting	Boat obs Platform obs	Observation	1	1	Grand Banks	Unk	na	Breed	2	Attracted to lights on offshore structures, such as oil rigs. Authors suggest that circling structures as a result of light attraction can lead to death from exhaustion.

17. Northern gannet

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Inger *et al.* 2010)	Northern gannet	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	344	Tidal	All	4	No clear effect but possibly a weak displacement effect during deployment rather than during operation – increase of 83m from device. But mean of 28m closer during operation. Unknown if biologically significant.
(Vanermen *et al.* 2013)	Northern gannet	Structures and construction activity (vessels not specifically mentioned)	Boat obs	BACI	2	1	Thorntonbank and Blighbank, Belgium	Unk	Wind	All	6	Avoided WF during operation – at least 3km from nearest turbines.
(Petersen *et al.* 2006)	Gannet	Structures	Aerial obs	BACI	2	1	Horns Rev and Nysted, Denmark	Unk	Wind	All	7	Increased avoidance of WF area, incl. 4km buffer, after turbine erection.
(Petersen *et al.* 2006)	Gannet	Structures	Radar	Observation	1	1	Horns Rev, Denmark	268	Wind	All	3	Never seen flying into WF. Few individuals that came close made marked turns to avoid WF or individual turbines.
(Walls *et al.* 2013)	Gannet	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	1453	Wind	All	11	Decline in numbers on the water during operation compared to pre-construction levels. Some avoidance behaviour.

(Krijgsveld *et al.* 2011)	Gannet	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Strong avoidance. Some flying inside WF or at edge.
(Leopold *et al.* 2011)	Gannet	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Strong avoidance – potential displacement from foraging areas.
(NWP Offshore Ltd 2008)	Gannet	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Some flying into the WF area but lower numbers than before – some avoidance but not exclusion.
(Gill *et al.* 2008)	Gannet	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	No significant evidence of change but increase in numbers inside the WF during operation.
(Skov *et al.* 2012)	Gannet	Structures	Radar Platform obs	BACI	2	1	Horns Rev 1 and 2, Denmark	Unk	Wind	All	5	Seen in HR2 WF.
(Christensen *et al.* 2004)	Gannet	Structures	Platform obs Radar	Observation	1	1	Horns Rev, Denmark	70	Wind	All	2	Few entered WF, others deflected flight routes around WF.

18. Great cormorant

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Keenan *et al.* 2011)	Cormorant spp.	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	63146	Tidal	All	4	25% decline in cormorant sightings when in operation compared to periods of non-operation.
(Inger *et al.* 2010)	Cormorant spp.	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	63146	Tidal	All	4	Minor displacement from SeaGen device in 1t half of year – max mean 20m but confounded by being closer to device in latter part of year. Unknown if biologically significant.

(Petersen *et al.* 2006)	Cormorant	Structures	Boat obs	Observation	1	1	Horns Rev, Denmark	185	Wind	All	3	14% seen inside WF. Seen perching and resting on turbines and meteorological masts.
(Petersen *et al.* 2006)	Cormorant	Structures	Radar	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Some flocks (6 and 40 birds) did not show marked differences in behaviour when entering WF (some over turbine height, some at turbine height). 2 flocks (6 and 13 birds) reacted at 200-200m from WF, stalled and scattered before reforming and entering WF. Some seen making 'panic' descents before entering.
(Petersen *et al.* 2006)	Cormorant spp.	Structures	Boat obs	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Resting on turbines – mostly at edge of WF.
(Walls *et al.* 2013)	Cormorant	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	4945	Wind	All	11	Increase in numbers in flight and on the water in the WF area from pre-construction to operation. Regularly observed sitting on turbines.
(Krijgsveld *et al.* 2011)	Cormorant spp.	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	No avoidance but not attracted.
(Leopold *et al.* 2011)	Cormorant	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Attraction to WF from near-shore areas – few observed at WF distance from shore prior to construction. Rest on structures and forage. Fly between turbines with no visible hesitation.
(NWP Offshore Ltd 2008)	Cormorant	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Observed on structures in and around the WF. Observed foraging near turbines – hypothesised to be reef effect of foundations.

(Gill *et al.* 2008)	Cormorant	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	No evidence of change.

19. Shag

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Jackson 2014)	Shag	Structures	Photo obs	Observation	1	1	Pelamis device testing, Orkney, UK	Unk	Wave	Feb Mar May Jun	1	No evidence of use.
(Keenan *et al.* 2011)	Cormorant spp.	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	63146	Tidal	All	4	25% decline in cormorant sightings when in operation compared to periods of non-operation.
(Inger *et al.* 2010)	Cormorant spp.	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	63146	Tidal	All	4	Minor displacement from SeaGen device in 1st half of year – max mean 20m but confounded by being closer to device in latter part of year. Unknown if biologically significant.
(Petersen *et al.* 2006)	Shag	Structures	Boat obs	Observation	1	1	Horns Rev, Denmark	1	Wind	All	3	Seen inside the WF. 2-3 also seen perching and resting on turbines and meteorological masts. Some foraging close to turbines and around transformer station.
(Petersen *et al.* 2006)	Shag	Structures	Radar	Observation	1	1	Horns Rev, Denmark	1	Wind	All	3	Seen flying between turbines and foraging.
(Petersen *et al.* 2006)	Cormorant spp.	Structures	Boat obs	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Resting on turbines – mostly at edge of WF.
(Krijgsveld *et al.* 2011)	Cormorant spp.	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	No avoidance but not attracted.

(NWP Offshore Ltd 2008)	Shag	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Some seen within WF – some evidence of increased use since operational.

20. White-tailed eagle

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Nygård *et al.* 2010)	White-tailed eagle	Structures	GPS	Observation	1	1	Smøla, western Norway	34	Wind	All	5	No clear avoidance of wind turbines. 4/36 GPS tagged birds killed in collision with turbines. Observers seen birds circle close to and around turbines.
(Hoel 2009)	White-tailed eagle	Structures	Unk	Unk	Unk	Unk	Smøla, western Norway	Unk	Wind	Unk	Unk	Apparently similar use of air space inside and outside of WF.
(Dahl *et al.* 2012)	White-tailed eagle	Structures	Land obs	BACI	1	1	Smøla, western Norway	94	Wind	All	12	Desertion of breeding territories – result of displacement or collision of breeding birds with turbines. 28 birds killed (2005-2009). Some evidence for displacement from area. Increased human activity also likely to cause disturbance.
(Dahl *et al.* 2013)	White-tailed eagle	Structures	Land obs	Observation	1	1	Smøla, western Norway	Unk	Wind	Breed	1	No clear avoidance responses to turbines. No significant difference in flight activity within or outside WF. Sub-adults observed in WF more than adults.

21. Arctic skua

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Petersen *et al.* 2006)	Skua spp.	Structures	Boat obs	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Some seen between turbines. Parasitic chases with terns sometimes performed at altitudes overlapping with turbine rotors.
(Christensen *et al.* 2004)	Arctic skua	Structures	Platform obs Radar	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	2	Observed no reaction to WF.

22. Great skua

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Vanermen *et al.* 2013)	Great skua	Structures and construction activity (vessels not specifically mentioned)	Boat obs	BACI	2	1	Thorntonbank and Blighbank, Belgium	Unk	Wind	All	6	Attracted to area during operation and construction.
(Petersen *et al.* 2006)	Skua spp.	Structures	Boat obs	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Some seen between turbines. Parasitic chases with terns sometimes performed at altitudes overlapping with turbine rotors.

23. Black-headed gull

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Petersen *et al.* 2006)	Gull spp.	Structures	Radar	Observation	1	1	Horns Rev, Denmark	1254	Wind	All	3	Some flew into and inside WF (109/461 tracks).
(Petersen *et al.* 2006)	Gull spp.	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Attraction to WF during construction (during increase in vessel activity in area)
(NIRAS Consulting Engineers and Planners A/S 2008)	Gull spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Jackson 2014)	Black-headed gull	Structures	Photo obs	Observation	1	1	Pelamis device testing, Orkney, UK	Unk	Wave	Feb Mar May Jun	1	Regularly rested on structure– no avoidance.
(Krijgsveld *et al.* 2011)	Black-headed gull	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Relatively high avoidance – few flying inside WF.
(Leopold *et al.* 2011)	Black-headed gull	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Never seen in WF but potentially rarely ventured far enough from near-shore waters to be in vicinity of WF.

24. Common gull

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Petersen *et al.* 2006)	Gull spp.	Structures	Radar	Observation	1	1	Horns Rev, Denmark	1254	Wind	All	3	Some flew into and inside WF (109/461 tracks).

(Petersen *et al.* 2006)	Gull spp.	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Attraction to WF during construction (during increase in vessel activity in area)
(NIRAS Consulting Engineers and Planners A/S 2008)	Gull spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Vanermen *et al.* 2013)	Common gull	Structures and construction activity (vessels not specifically mentioned)	Boat obs	BACI	2	1	Thorntonbank and Blighbank, Belgium	Unk	Wind	All	6	Attracted to area during operation and construction. Commonly seen in WF between turbines and often rest on jacket foundations.
(Krijgsveld *et al.* 2011)	Common gull	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Evidence of avoidance.
(Leopold *et al.* 2011)	Common gull	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	No evidence of effect. Gull spp. seen flying in WF and resting on structures.
(NWP Offshore Ltd 2008)	Common gull	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Some evidence for increased numbers of birds in WF site.

25. Lesser black-backed gull

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Petersen *et al.* 2006)	Gull spp.	Structures	Radar	Observation	1	1	Horns Rev, Denmark	1254	Wind	All	3	Some flew into and inside WF (109/461 tracks).
(Petersen *et al.* 2006)	Gull spp.	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Attraction to WF during construction (during increase in vessel activity in area)

(Walls *et al.* 2013)	Large gull spp. (herring, lesser and greater black-backed)	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	26528	Wind	All	11	Increase in birds on the water. Birds in flight increased during construction but declined during operation. Species-specific effects: herring gull declined from pre-construction whilst greater black-backed gulls increased.
(NIRAS Consulting Engineers and Planners A/S 2008)	Gull spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Vanermen *et al.* 2013)	Lesser black-backed gull	Structures and construction activity (vessels not specifically mentioned)	Boat obs	BACI	2	1	Thorntonbank and Blighbank, Belgium	Unk	Wind	All	6	Avoidance of WF during operation and construction. Still seen in WF site during construction and often rest on jacket foundations. Sometimes seen in WF during operation. Foraged near jacket foundations.
(Petersen *et al.* 2006)	Lesser black-backed gull	Structures	Boat obs	Observation	1	1	Horns Rev, Denmark	1	Wind	All	3	Resting on turbines - mostly at edge of WF (some obs made during time when all turbines were inactive and most nacelles dismounted for renovation).
(Krijgsveld *et al.* 2011)	Lesser black-backed gull	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Some evidence of avoidance.
(Leopold *et al.* 2011)	Lesser black-backed gull	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	No evidence of effect. Gull spp. seen flying in WF and resting on structures.
(NWP Offshore Ltd 2008)	Lesser black-backed gull	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	No evidence of change.

(Gill *et al.* 2008)	Lesser black-backed gull	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	Evidence of lower numbers in WF since operational in winter (Oct-Mar).

26. Herring gull

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Petersen *et al.* 2006)	Gull spp.	Structures	Radar	Observation	1	1	Horns Rev, Denmark	1254	Wind	All	3	Some flew into and inside WF (109/461 tracks).
(Petersen *et al.* 2006)	Herring gull	Structures	Boat obs	Observation	1	1	Horns Rev, Denmark	8	Wind	All	3	Resting on turbines - mostly at edge of WF (some obs made during time when all turbines were inactive and most nacelles dismounted for renovation).
(Petersen *et al.* 2006)	Herring gull	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	Jan-Apr	1	Attraction to WF during construction (during increase in vessel activity in area) but avoided area when in operation. Some seen in WF but in lower numbers than in surrounding area.
(Walls *et al.* 2013)	Large gull spp. (herring, lesser and greater black-backed)	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	26528	Wind	All	11	Increase in birds on the water. Birds in flight increased during construction but declined during operation. Species-specific effects: herring gull declined from pre-construction whilst greater black-backed gulls increased.
(NIRAS Consulting Engineers and Planners A/S 2008)	Gull spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.

(Jackson 2014)	Herring gull	Structures	Photo obs	Observation	1	1	Pelamis device testing, Orkney, UK	Unk	Wave	Feb Mar May Jun	1	Occasionally rested on structure– no avoidance.
(Vanermen *et al.* 2013)	Herring gull	Structures and construction activity (vessels not specifically mentioned)	Boat obs	BACI	2	1	Thorntonbank and Blighbank, Belgium	Unk	Wind	All	6	Attracted to area during operation and construction. At Blighbank numbers increased after construction. Commonly seen in WF and often rest on jacket foundations.
(Krijgsveld *et al.* 2011)	Herring gull	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Evidence of avoidance.
(Leopold *et al.* 2011)	Herring gull	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	No evidence of effect. Gull spp. seen flying in WF and resting on structures.
(NWP Offshore Ltd 2008)	Herring gull	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	No evidence of change.
(Gill *et al.* 2008)	Herring gull	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	No evidence of change.
(Christensen *et al.* 2004)	Herring gull	Structures	Platform obs Radar	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	2	Frequently entered WF.

27. Great black-backed gull

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Jackson 2014)	Great black-backed gull	Structures	Photo obs	Observation	1	1	Pelamis device testing, Orkney, UK	Unk	Wave	Feb Mar May Jun	1	Occasionally rested on structure– no avoidance.

(Vanermen *et al.* 2013)	Great black-backed gull	Structures and construction activity (vessels not specifically mentioned)	Boat obs	BACI	2	1	Thorntonbank and Blighbank, Belgium	Unk	Wind	All	6	Attracted to area – higher numbers inside WF area during construction and operation. Commonly seen in the wind farm and often rested on monopole jackets during construction
(Petersen *et al.* 2006)	Great black-backed gull	Structures	Boat obs	Observation	1	1	Horns Rev, Denmark	22	Wind	All	3	Resting on turbines - mostly at edge of WF (some obs made during time when all turbines were inactive and most nacelles dismounted for renovation).
(Petersen *et al.* 2006)	Gull spp.	Structures	Radar	Observation	1	1	Horns Rev, Denmark	1254	Wind	All	3	Some flew into and inside WF (109/461 tracks).
(Petersen *et al.* 2006)	Gull spp.	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Attraction to WF during construction (during increase in vessel activity in area)
(Walls *et al.* 2013)	Large gull spp. (herring, lesser and greater black-backed)	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	26528	Wind	All	11	Increase in birds on the water. Birds in flight increased during construction but declined during operation. Species-specific effects: herring gull declined from pre-construction whilst greater black-backed gulls increased.
(NIRAS Consulting Engineers and Planners A/S 2008)	Gull spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Krijgsveld *et al.* 2011)	Great black-backed gull	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Some evidence of avoidance.
(Leopold *et al.* 2011)	Great black-backed gull	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	No evidence of effect. Gull spp. seen flying in WF and resting on structures.

(NWP Offshore Ltd 2008)	Great black-backed gull	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	No evidence of change.
(Gill *et al.* 2008)	Great black-backed gull	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	No evidence of change.
(Christensen *et al.* 2004)	Great black-backed gull	Structures	Platform obs Radar	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	2	Frequently entered WF.

28. Black-legged kittiwake

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Petersen *et al.* 2006)	Gull spp.	Structures	Radar	Observation	1	1	Horns Rev, Denmark	1254	Wind	All	3	Some flew into and inside WF (109/461 tracks).
(Petersen *et al.* 2006)	Gull spp.	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Attraction to WF during construction (during increase in vessel activity in area)
(Walls *et al.* 2013)	Black-legged kittiwake	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	3002	Wind	All	11	Decline in numbers on water during construction and into operation year 1 – but could reflect inter-annual variation.
(NIRAS Consulting Engineers and Planners A/S 2008)	Gull spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Krijgsveld *et al.* 2011)	Black-legged kittiwake	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Observed inside WF – evidence of attraction

(Jackson 2014)	Black-legged kittiwake	Structures	Photo obs	Observation	1	1	Pelamis device testing, Orkney, UK	Unk	Wave	Feb Mar May Jun	1	Regularly rested on structure– no avoidance.
(Vanermen *et al.* 2013)	Black-legged kittiwake	Structures and construction activity (vessels not specifically mentioned)	Boat obs	BACI	2	1	Thorntonbank and Blighbank, Belgium	Unk	Wind	All	6	Attracted to area during operation but some decrease during construction. Commonly seen in WF between turbines. Seen foraging in seemingly greater numbers inside the WF than control area during operation - possible lagged reef/FAD effect?
(Leopold *et al.* 2011)	Black-legged kittiwake	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	No evidence of effect. Gull spp. seen flying in WF and resting on structures.
(NWP Offshore Ltd 2008)	Black-legged kittiwake	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Some evidence of avoidance.
(Baird 1990)	Black-legged kittiwake	Structures	Boat obs	Observation	1	1	Bering Sea	Unk	na	Breed	1	Higher numbers found at oil rig than in control area – suggests some attraction to structures.

29. Little tern

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Keenan *et al.* 2011)	Tern spp.	Structures Human activity on the structure	Land obs	BACI	1	1	Strangford Lough, Ireland	1893	Tidal	All	4	Lowest numbers during operation – human activity on the pile and vessel activity may have contributed. Tern abundance increased over survey period.

(Inger *et al.* 2010)	Tern spp.	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	1893	Tidal	All	4	Significant effect of SeaGen device on distance of terns to the device – terns moved small distance closer to the site during operation – max change in mean distance of 6m. Unknown if biologically significant.
(Petersen *et al.* 2006)	Tern spp.	Structures	Radar	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Some inside WF (44/144 tracks).
(Krijgsveld *et al.* 2011)	Tern spp. (mostly sandwich tern)	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Relatively scarce but seen flying and foraging inside wind farm – mostly flying near edge. No clear avoidance.
(Perrow *et al.* 2011)	Little tern	Structures Pile noise	Land obs Trawls	BACI	1	1	Scroby Sands, UK	Unk	Wind	Breed	5	Displacement by structures – loss of foraging habitat as pile drive noise hypothesised to be responsible for low herring availability.
(Paiva *et al.* 2008)	Little tern	Structures	Land obs Boat obs	Observation	1	1	Algarve, Portugal	Unk	na	Breed	3	Avoids areas with human constructions.

30. Sandwich tern

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Keenan *et al.* 2011)	Tern spp.	Structures Human activity on the structure	Land obs	BACI	1	1	Strangford Lough, Ireland	1893	Tidal	All	4	Lowest numbers during operation – human activity on the pile and vessel activity may have contributed. Tern abundance increased over survey period.

(Inger *et al.* 2010)	Tern spp.	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	1893	Tidal	All	4	Significant effect of SeaGen device on distance of terns to the device – terns moved small distance closer to the site during operation – max change in mean distance of 6m. Unknown if biologically significant.
(Vanermen *et al.* 2013)	Sandwich tern	Structures and construction activity (vessels not specifically mentioned)	Boat obs	BACI	2	1	Thorntonbank and Blighbank, Belgium	Unk	Wind	All	6	Attracted to area – higher numbers inside WF area during construction and operation. Regularly seen in the wind farm
(Petersen *et al.* 2006)	Tern spp.	Structures	Radar	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Some inside WF (44/144 tracks).
(Krijgsveld *et al.* 2011)	Tern spp. (mostly sandwich tern)	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Relatively scarce but seen flying and foraging inside wind farm – mostly flying near edge. No clear avoidance.
(Leopold *et al.* 2011)	Sandwich tern	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Rarely seen in WF – seem to prefer to fly around WF rather than enter.
(NWP Offshore Ltd 2008)	Sandwich tern	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Observed in WF.
(Gill *et al.* 2008)	Sandwich tern	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	Fewer flying through WF compared to same area during pre-construction – evidence of barrier effect.
(Christensen *et al.* 2004)	Sandwich tern	Structures	Platform obs	Observation	1	1	Horns Rev, Denmark	3	Wind	All	2	Frequently entered WF with no reaction.

31. Common tern

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Keenan *et al.* 2011)	Tern spp.	Structures Human activity on the structure	Land obs	BACI	1	1	Strangford Lough, Ireland	1893	Tidal	All	4	Lowest numbers during operation – human activity on the pile and vessel activity may have contributed. Tern abundance increased over survey period.
(Inger *et al.* 2010)	Tern spp.	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	1893	Tidal	All	4	Significant effect of SeaGen device on distance of terns to the device – terns moved small distance closer to the site during operation – max change in mean distance of 6m. Unknown if biologically significant.
(Vanermen *et al.* 2013)	Common tern	Structures and construction activity (vessels not specifically mentioned)	Boat obs	BACI	2	1	Thorntonbank and Blighbank, Belgium	Unk	Wind	All	6	Attracted to area – higher numbers inside WF area during operation.
(Petersen *et al.* 2006)	Common tern	Structures	Aerial obs	BACI	2	1	Horns Rev and Nysted, Denmark	Unk	Wind	All	7	Absence in WF area post-construction.
(Krijgsveld *et al.* 2011)	Tern spp. (mostly sandwich tern)	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Relatively scarce but seen flying and foraging inside wind farm – mostly flying near edge. No clear avoidance.
(NWP Offshore Ltd 2008)	Common tern	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Observed in WF.

(Gill *et al.* 2008)	Common tern	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	No significant evidence of change but fewer flying through WF compared to same area during pre-construction – evidence of barrier effect.
(Christensen *et al.* 2004)	Common tern	Structures	Platform obs	Observation	1	1	Horns Rev, Denmark	3	Wind	All	2	Frequently entered WF but often returned out of area after passing 100-200m beyond first row of turbines.

32. Roseate tern

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Keenan *et al.* 2011)	Tern spp.	Structures Human activity on the structure	Land obs	BACI	1	1	Strangford Lough, Ireland	1893	Tidal	All	4	Lowest numbers during operation – human activity on the pile and vessel activity may have contributed. Tern abundance increased over survey period.
(Inger *et al.* 2010)	Tern spp.	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	1893	Tidal	All	4	Significant effect of SeaGen device on distance of terns to the device – terns moved small distance closer to the site during operation – max change in mean distance of 6m. Unknown if biologically significant.
(Petersen *et al.* 2006)	Tern spp.	Structures	Radar	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Some inside WF (44/144 tracks).
(Krijgsveld *et al.* 2011)	Tern spp. (mostly sandwich tern)	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Relatively scarce but seen flying and foraging inside wind farm – mostly flying near edge. No clear avoidance.

33. Arctic tern

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Jackson 2014)	Arctic tern	Structures	Photo obs	Observation	1	1	Pelamis device testing, Orkney, UK	Unk	Wave	Feb Mar May Jun	1	Large numbers frequently rested on structure– no avoidance.
(Keenan *et al.* 2011)	Tern spp.	Structures Human activity on the structure	Land obs	BACI	1	1	Strangford Lough, Ireland	1893	Tidal	All	4	Lowest numbers during operation – human activity on the pile and vessel activity may have contributed. Tern abundance increased over survey period.
(Inger *et al.* 2010)	Tern spp.	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	1893	Tidal	All	4	Significant effect of SeaGen device on distance of terns to the device – terns moved small distance closer to the site during operation – max change in mean distance of 6m. Unknown if biologically significant.
(Petersen *et al.* 2006)	Arctic and common tern	Structures	Aerial obs	BACI	2	1	Horns Rev and Nysted, Denmark	Unk	Wind	All	7	Absence in WF area post-construction.
(Petersen *et al.* 2006)	Tern spp.	Structures	Radar	Observation	1	1	Horns Rev, Denmark	Unk	Wind	All	3	Some inside WF (44/144 tracks).
(Krijgsveld *et al.* 2011)	Tern spp. (mostly sandwich tern)	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Relatively scarce but seen flying and foraging inside wind farm – mostly flying near edge. No clear avoidance.
(Christensen *et al.* 2004)	Arctic tern	Structures	Platform obs	Observation	1	1	Horns Rev, Denmark	3	Wind	All	2	Frequently entered WF but often returned out of area after passing 100-200m beyond first row of turbines..

34. Common guillemot

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Inger *et al.* 2010)	Auk spp. (common guillemot and razorbill)	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	6945	Tidal	All	4	Mixed effects – sometimes a displacement max mean of 42m. Unknown if biologically significant.
(Vanermen *et al.* 2013)	Common guillemot	Structures and construction activity (vessels not specifically mentioned)	Boat obs	BACI	2	1	Thorntonbank and Blighbank, Belgium	Unk	Wind	All	6	Avoided WF area during operation and construction, although regularly seen in WF area – possible lagged reef/FAD effect?
(Petersen & Fox 2007)	Razorbill/ Common guillemot	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	Jan-Apr	1	Avoidance of WF with none seen in the WF area.
(Walls *et al.* 2013)	Common guillemot	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	11733	Wind	All	11	Decline in numbers on the water between pre-construction and construction, with an increase in operation year 1 – small increase observed in turbine area – indicates potential avoidance.
(NIRAS Consulting Engineers and Planners A/S 2008)	Auk spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Krijgsveld *et al.* 2011)	Auk spp.	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Rarely observed in WF.
(Leopold *et al.* 2011)	Common guillemot	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Significant avoidance of WF but still seen inside WF area.

(NWP Offshore Ltd 2008)	Common guillemot	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Evidence of attraction – significantly greater numbers in WF once operational.
(Thuringer & Reidy 2006)	Related spp. - Pigeon guillemot	Pile drive noise	Land suvey	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Not displaced by drilling.
(Gill *et al.* 2008)	Common guillemot	Structures	Boat obs	BACI	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	Lower numbers in Jan-Mar since WF operational.
(Baird 1990)	Common guillemot	Structures	Boat obs	Observation	1	1	Bering Sea	Unk	na	Breed	1	Slightly higher numbers found at oil rig than in control area – suggests some attraction to structures.

35. Razorbill

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Inger *et al.* 2010)	Auk spp. (common guillemot and razorbill)	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	6945	Tidal	All	4	Mixed effects – sometimes a displacement max mean of 42m. Unknown if biologically significant.
(Vanermen *et al.* 2013)	Razorbill	Structures and construction activity (vessels not specifically mentioned)	Boat obs	BACI	2	1	Thorntonbank and Blighbank, Belgium	Unk	Wind	All	6	Avoided Thorntonbank WF area during construction. Attracted during operation and regularly seen in WF area. Avoided Blighbank during construction and operation, although still seen entering area – possible lagged reef/FAD effect?
(Petersen & Fox 2007)	Razorbill/ Common guillemot	Structures	Aerial obs	BACI	1	1	Horns Rev, Denmark	Unk	Wind	Jan-Apr	1	Avoidance of WF with none seen in the WF area.

(Walls *et al.* 2013)	Razorbill	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	5763	Wind	All	11	Decline in numbers on the water and in flight during construction, with some increase during operation although at lower levels in WF area than surrounding area.
(Walls *et al.* 2013)	Auk spp.	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	19922	Wind	All	11	Decline in numbers on the water between pre-construction and construction, with an increase during operation.
(NIRAS Consulting Engineers and Planners A/S 2008)	Auk spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Krijgsveld *et al.* 2011)	Auk spp.	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Rarely observed in WF.
(Leopold *et al.* 2011)	Razorbill	Structures	Boat obs	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Significant avoidance of WF but still seen inside WF area.
(NWP Offshore Ltd 2008)	Razorbill	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	No evidence of change.
(Thuringer & Reidy 2006)	Related spp. - Pigeon guillemot	Pile drive noise	Land suvey	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Not displaced by drilling.

36. Black guillemot

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Related spp. - Pigeon guillemot	Pile drive noise	Land suvey	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Not displaced by drilling.
(Jackson 2014)	Black guillemot	Structures	Photo obs	Observation	1	1	Pelamis device testing, Orkney, UK	Unk	Wave	Feb Mar May Jun	1	Small numbers regularly rested on structure– no avoidance.
(Inger *et al.* 2010)	Black guillemot	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	10437	Tidal	All	4	Weak significant effect of SeaGen on distribution. Max mean displacement 100m. Unknown if biologically significant.
(NWP Offshore Ltd 2008)	Auk spp.	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Seen in the WF site – most on water.

37. Little auk

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Inger *et al.* 2010)	Auk spp. (common guillemot and razorbill)	Structures during operation	Land obs	BACI	1	1	Strangford Lough, Ireland	6945	Tidal	All	4	Mixed effects – sometimes a displacement max mean of 42m. Unknown if biologically significant.
(Walls *et al.* 2013)	Auk spp.	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	19922	Wind	All	11	Decline in numbers on the water between pre-construction and construction, with an increase during operation.

(NIRAS Consulting Engineers and Planners A/S 2008)	Auk spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(Krijgsveld *et al.* 2011)	Auk spp.	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Rarely observed in WF.
(NWP Offshore Ltd 2008)	Auk spp.	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Seen in the WF site – most on water.
(Wiese *et al.* 2001)	Little auk	Structures Lighting	Boat obs Platform obs	Observation	1	1	Grand Banks	Unk	na	Breed	2	Attracted to lights on offshore structures, such as oil rigs. Authors suggest that circling structures as a result of light attraction can lead to death from exhaustion.
(Bradstreet & Brown 1985)	Little auk	Structures	Unk	Unk	Unk	Unk	Unk	Unk	na	Unk	Unk	Occurs widely in pelagic habitat - unlikely to occur in large numbers where wave devices will be deployed.

38. Atlantic puffin

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Related spp. - Pigeon guillemot	Pile drive noise	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Not displaced by drilling.
(Krijgsveld *et al.* 2011)	Auk spp.	Structures	Sea obs (fixed platform)	BACI	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	3	Rarely observed in WF.

(Walls *et al.* 2013)	Auk spp.	Structures	Boat obs	BACI	1	1	Robin Rigg, UK	1992 2	Wind	All	11	Decline in numbers on the water between pre-construction and construction, with an increase during operation.
(NIRAS Consulting Engineers and Planners A/S 2008)	Auk spp.	Structures	Boat obs Aerial obs	BACI	1	1	Barrow WF, UK	Unk	Wind	All	3	No indication of effect.
(NWP Offshore Ltd 2008)	Auk spp.	Structures	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Seen in the WF site – most on water.
(Baird 1990)	Atlantic puffin	Structures	Boat obs	Observation	1	1	Bering Sea	Unk	na	Breed	1	Slightly higher numbers found at oil rig than in control area – suggests some attraction to structures.

Appendix 1

Supplementary Table 3: Disturbance and displacement caused by vessels and/or helicopter traffic

Tables of data contributing to confidence levels for each species

Obs = observation
Unk = unknown
BACI = Before-After-Control-Impact
ESAS = European Seabirds at Sea

1. Greater scaup

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Platteeuw & Beekman 1994)	Greater scaup	Vessels	Unk	Unk	Unk	Unk	IJsseloog and Ketelmeer, Netherlands	Unk	Vessel	Unk	Unk	Fly from boats up to 400m away.
(Poulton *et al.* 2002)	Greater scaup	Vessels	Land obs	Anecdotal	3	1	California, USA	Unk	na	Jan-Mar	1	Small watercraft flushed birds.

2. Common eider

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(ORPC Maine 2012)	Eider	Vessels Pile drive noise	Land survey	Speculative/ anecdotal	1	1	Cobscook Bay, US	Unk	Tidal	Mar	1	Displaced by fishing vessel but resettled in same area to continue foraging. No response to vibratory or diesel impact pile drive hammer. No response to barge present for pile drive activities.
(Garthe & Hüppop 2004)	Eider	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 3

(Schwemmer *et al.* 2011)	Eider	Vessels	Boat obs	Observation (with analysis)	1	1	Baltic Sea	154	na	Oct - Apr	1	208m median flush distance from boats. No reaction from some flocks.
(Schwemmer & Garthe 2006)	Eider	Vessels	Boat obs	Observation (with analysis)	1	1	Baltic Sea	na	na	Oct - Apr	1	Flushed for the shortest distances and returned to pre-disturbance levels after ca. 2hrs. Flush distances increased with flock size.

3. Long-tailed duck

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Schwemmer *et al.* 2011)	Long-tailed duck	Vessels	Boat obs	Observation (with analysis)	1	1	Baltic Sea	123	na	Oct - Apr	1	293m median flush distance from boats.
(Schwemmer & Garthe 2006)	Long-tailed duck	Vessels	Boat obs	Observation (with analysis)	1	1	Baltic Sea	na	na	Oct - Apr	1	Flushed for the shortest distances and returned to pre-disturbance levels after ca. 2hrs. Flush distances increased with flock size.

4. Common scoter

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Petersen *et al.* 2006)	Common scoter	Helicopter Vessels	Anecdotal	Observation	1	1	Horns Rev, Belgium	Unk	Wind	All	3	Massive flush in response to helicopter activity at a turbine. Birds took to the air (but did not enter WF) but returned once helicopter left. Some less conspicuous responses to smaller boats operating at outer turbine row or just outside WF.

(NWP Offshore Ltd 2008)	Common scoter	Vessels	Boat obs Aerial obs	BACI	1	1	North Hoyle, UK	Unk	Wind	All	5	Several large flocks flushed by survey vessels (ca. 750 up to 1200 birds).
(Gill *et al.* 2008)	Common scoter	Vessels	Boat obs	Observation	1	1	Kentish Flats WF, UK	Unk	Wind	All	6	Large flocks (1000-2000 birds) flushed by survey vessel. Smaller flock (8 birds) flushed at 400m from vessel.
(Schwemmer *et al.* 2011)	Common scoter	Vessels	Boat obs	Observation (with analysis)	1	1	Baltic Sea	210	na	Oct - Apr	1	Median flush distance was 804m from ships. Maximum flush distance was 3.2km.
(Garthe & Hüppop 2004)	Common scoter	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 5
(Kaiser *et al.* 2006)	Common scoter	Vessels	Boat obs Boat radar	Observation	1	1	East England and north Wales, UK	Unk	na	Oct-Mar	1	Large flocks flushed at 1000-2000m; smaller flocks flushed at distances <1000m. Evidence of avoidance of shipping routes – low likelihood of habituation.
(Schwemmer & Garthe 2006)	Common scoter	Vessels	Boat obs	Observation (with analysis)	1	1	Baltic Sea	na	na	Oct - Apr	1	Flushed at greatest distances and returned to pre-disturbance levels after > ca.4hrs. Flush distances increased with flock size.

5. Velvet scoter

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Garthe & Hüppop 2004)	Velvet scoter	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 5

(Schwemmer & Garthe 2006)	Velvet scoter	Vessels	Boat obs	Observation (with analysis)	1	1	Baltic Sea	na	na	Oct - Apr	1	Flushed at medium distances (compared to eider, long-tailed duck and common scoter) and returned to pre-disturbance levels after ca.3hrs. Flush distances increased with flock size.
(Schwemmer *et al.* 2011)	Velvet scoter	Vessels	Boat obs	Observation (with analysis)	1	1	Baltic Sea	59	na	Oct - Apr	1	Fly from boats over 400m away.

6. Common goldeneye

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Platteeuw & Beekman 1994)	Common goldeneye	Vessels	Unk	Unk	Unk	Unk	IJsseloog and Ketelmeer, Netherlands	Unk	Vessel	Unk	Unk	Fly from ships passing at 500-1000m away.
(Batten 1977)	Common goldeneye	Vessels	Land obs	Observation	1	1	Brent Reservoir, London, UK	Unk	na	Unk	Unk	Usually leave reservoir at the onset of vessel activity. Said to be particularly sensitive and flush up to 700m from motor vessels.

7. Red-throated diver

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Schwemmer *et al.* 2011)	Red-throated diver	Vessels	Aerial obs	Observation (with analysis)	1	1	German North Sea	Unk	na	Mar Apr	2	Fly from boats over 1000m away. Avoidance of shipping lanes – suggests no habituation to regular vessel traffic.
(Garthe & Hüppop 2004)	Red-throated diver	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 4

8. Black-throated diver

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Schwemmer *et al.* 2011)	Black-throated diver	Vessels	Aerial obs	Observation (with analysis)	1	1	German North Sea	Unk	na	Mar Apr	2	Fly from boats over 1000m away. Avoidance of shipping lanes – suggests no habituation to regular vessel traffic.
(Garthe & Hüppop 2004)	Black-throated diver	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 4

9. Great northern diver

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Forrester & Andrews 2007)	Great northern diver	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Apparently less sensitive to vessel traffic than other diver spp.

10. Great-crested grebe

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Garthe & Hüppop 2004)	Great-crested grebe	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 3
(Batten 1977)	Great-crested grebe	Vessels	Land obs	Observation	1	1	Brent Reservoir, London, UK	Unk	na	Unk	Unk	Single birds or pairs dive to escape. Small flocks fly off and leave the area. Constant vessel disturbance close to breeding sites displaces birds.

11. Slavonian grebe

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Garthe & Hüppop 2004)	Related spp. - Great-crested grebe	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 3
(Forrester & Andrews 2007)	Slavonian grebe	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Similar to other grebes?
(Batten 1977)	Related spp. - Great-crested grebe	Vessels	Land obs	Observation	1	1	Brent Reservoir, London, UK	Unk	na	Unk	Unk	Single birds or pairs dive to escape. Small flocks fly off and leave the area. Constant vessel disturbance close to breeding sites displaces birds.

12. Northern fulmar

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Garthe & Hüppop 2004)	Northern fulmar	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 1
(Votier *et al.* 2013)	Northern fulmar	Vessels	Bird-borne cameras	Observation	1	1	Grassholm Island, Wales, UK	10	na	Breed	1	Regularly associate with fishing vessels
(Hudson & Furness 1988)	Northern fulmar	Vessels	Boat obs	Observation	1	1	Shetland, UK	Unk	na	Apr-Aug	2	Associate with fishing vessels.

13. Sooty shearwater

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Cramp & Simmons 1977)	Sooty shearwater	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response
(del Hoyo, Elliott & Sargatal 1992)	Sooty shearwater	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response
(Forrester & Andrews 2007)	Sooty shearwater	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response

14. Manx shearwater

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Cramp & Simmons 1977)	Manx shearwater	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response
(del Hoyo *et al.* 1992)	Manx shearwater	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response
(Forrester & Andrews 2007)	Manx shearwater	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response

15. European storm-petrel

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Cramp & Simmons 1977)	European storm-petrel	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response
(del Hoyo *et al.* 1992)	European storm-petrel	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response
(Forrester & Andrews 2007)	European storm-petrel	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response

16. Leach's storm-petrel

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Cramp & Simmons 1977)	Leach's storm-petrel	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response
(del Hoyo *et al.* 1992)	Leach's storm-petrel	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response
(Forrester & Andrews 2007)	Leach's storm-petrel	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response

17. Northern gannet

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Garthe & Hüppop 2004)	Northern gannet	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 2
(Votier *et al.* 2013)	Northern gannet	Vessels	Bird-borne cameras	Observation	1	1	Grassholm Island, Wales, UK	10	na	Breed	1	Regularly associate with fishing vessels
(Hudson & Furness 1988)	Northern gannet	Vessels	Boat obs	Observation	1	1	Shetland, UK	Unk	na	Apr-Aug	2	Associate with fishing vessels.

18. Great cormorant

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Garthe & Hüppop 2004)	Great cormorant	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 4.
(Rodgers & Schwikert 2003)	Related spp. – Double crested cormorant	Vessels	Boat obs	Observation	5	1	Florida, USA	95	na	Mar-Aug	3	Flushed by airboats at mean distances of 152 ± 43m, and by out-board motorboats at 43 ± 20m.

19. Shag

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Cramp & Simmons 1977)	Shag	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response

(del Hoyo *et al.* 1992)	Shag	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response
(Velando & Munilla 2011)	Shag	Vessels	Land obs	Observation	1	1	Illas Cíes, Iberian Peninsula	Unk	na	Jun-Aug	1	Birds became alert and fled (98% of obs) from recreational boats when they came within 500m. Boats reduced foraging and altered areas used. Excluded from areas of high vessel traffic.

20. White-tailed eagle

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Forrester & Andrews 2007)	White-tailed eagle	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Little response
(Rodgers & Schwikert 2003)	Related spp. – Bald eagle	Vessels	Boat obs	Observation	5	1	Florida, USA	46	na	Mar-Aug	3	Flushed by airboats at mean distances of 173 ± 66m.

21. Arctic skua

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Garthe & Hüppop 2004)	Arctic skua	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 1

22. Great skua

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Garthe & Hüppop 2004)	Great skua	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 1
(Hudson & Furness 1988)	Great skua	Vessels	Boat obs	Observation	1	1	Shetland, UK	Unk	na	Apr-Aug	2	Associate with fishing vessels.

23. Black-headed gull

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Gull spp.	Vessels	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Not disturbed from rocks during cable laying/construction.
(Petersen *et al.* 2006)	Gull spp.	Vessels	Aerial obs	BACI	1	1	Horns Rev, Belgium	Unk	Wind	All	3	Attraction to WF during construction (during increase in vessel activity in area)
(Garthe & Hüppop 2004)	Black-headed gull	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 2

24. Common gull

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Gull spp.	Vessels	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Not disturbed from rocks during cable laying/construction.

(Petersen *et al.* 2006)	Gull spp.	Vessels	Aerial obs	BACI	1	1	Horns Rev, Belgium	Unk	Wind	All	3	Attraction to WF during construction (during increase in vessel activity in area)
(Garthe & Hüppop 2004)	Common gull	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 2

25. Lesser black-backed gull

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Gull spp.	Vessels	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Not disturbed from rocks during cable laying/construction.
(Petersen *et al.* 2006)	Gull spp.	Vessels	Aerial obs	BACI	1	1	Horns Rev, Belgium	Unk	Wind	All	3	Attraction to WF during construction (during increase in vessel activity in area)
(Garthe & Hüppop 2004)	Lesser black-backed gull	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 2
(Hudson & Furness 1988)	Lesser black-backed gull	Vessels	Boat obs	Observation	1	1	Shetland, UK	Unk	na	Apr-Aug	2	Associate with fishing vessels.

26. Herring gull

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Gull spp.	Vessels	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Not disturbed from rocks during cable laying/construction.
(Petersen *et al.* 2006)	Herring gull	Vessels	Aerial obs	BACI	1	1	Horns Rev, Belgium	Unk	Wind	Jan-Apr	1	Attraction to WF during construction (during increase in vessel activity in area) but avoided area when in operation.

(Garthe & Hüppop 2004)	Herring gull	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 2
(Hudson & Furness 1988)	Herring gull	Vessels	Boat obs	Observation	1	1	Shetland, UK	Unk	na	Apr-Aug	2	Associate with fishing vessels.

27. Great black-backed gull

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Gull spp.	Vessels	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Not disturbed from rocks during cable laying/construction.
(Petersen *et al.* 2006)	Gull spp.	Vessels	Aerial obs	BACI	1	1	Horns Rev, Belgium	Unk	Wind	All	3	Attraction to WF during construction (during increase in vessel activity in area)
(Garthe & Hüppop 2004)	Great black-backed gull	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 2
(Hudson & Furness 1988)	Great black-backed gull	Vessels	Boat obs	Observation	1	1	Shetland, UK	Unk	na	Apr-Aug	2	Associate with fishing vessels.

28. Black-legged kittiwake

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Gull spp.	Vessels	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Not disturbed from rocks during cable laying/construction.
(Petersen *et al.* 2006)	Gull spp.	Vessels	Aerial obs	BACI	1	1	Horns Rev, Belgium	Unk	Wind	All	3	Attraction to WF during construction (during increase in vessel activity in area)

(Garthe & Hüppop 2004)	Black-legged kittiwake	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 2

29. Little tern

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Perrow *et al.* 2006)	Little tern	Helicopter	Observation	Anecdotal	1	1	Scroby Sands, UK	Unk	Wind	Breed	1	Prevented colony formation near to WF activity.

30. Sandwich tern

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Garthe & Hüppop 2004)	Sandwich tern	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 2
(Perrow, Skeate & Gilroy 2011)	Sandwich tern	Vessels	Boat tracking	Observation	2	1	Norfolk and Anglesey, UK	117	na	Breed	3	No evidence of avoiding tracking boat

31. Common tern

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Garthe & Hüppop 2004)	Common tern	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 2
(Perrow *et al.* 2011)	Common tern	Vessels	Boat tracking	Observation	2	1	Norfolk and Anglesey, UK	25	na	Breed	1	No evidence of avoiding tracking boat

32. Roseate tern

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Perrow *et al.* 2011)	Related spp. – Sandwich, common and arctic tern	Vessels	Boat tracking	Observation	2	1	Norfolk and Anglesey, UK	117	na	Breed	3	No evidence of avoiding tracking boat

33. Arctic tern

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Garthe & Hüppop 2004)	Arctic tern	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 2
(Perrow *et al.* 2011)	Arctic tern	Vessels	Boat tracking	Observation	1	1	Anglesey, UK	7	na	Breed	1	No evidence of avoiding tracking boat

34. Common guillemot

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Related spp. – Pigeon guillemot	Vessels Pile drive noise	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Took flight when approached by boats – resettled quickly. Not displaced by drilling.
(Leopold, Dijkman & Teal 2011)	Common guillemot	Vessels	Boat obs	Observation	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Wary of approaching vessels – dive to escape.

(Garthe & Hüppop 2004)	Common guillemot	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 3
(Ronconi & Clair 2002)	Related spp. - Black guillemot	Vessels	Land obs	BACI	1	1	Grand Manan Island, Canada	Unk	na	Jul Aug	1	Increased flush response the further from shore birds were. Large boats and those travelling slowly elicited a lower flush response than small, fast boats.
(Bellefleur, Lee & Ronconi 2009)	Related spp. – Marbled murrelet	Vessels	Boat obs	BACI/ Experimental	1	1	Pacific Rim National Park Reserve, Canada	7500	Unk	Aug	1	Strong flush response at 30-40m distance from boat. More often dived than flew. If birds dived, they were more likely to stay in area. If flew, birds tended to leave the area.
(Barrett & Vader 1984)	Auk spp.	Vessels	Unk	Unk	Unk	Unk	Norway	Unk	na	Unk	Unk	Moderate avoidance at short range
(Evans & Nettleship 1985)	Auk spp.	Vessels	Unk	Unk	Unk	Unk	Unk	Unk	na	Unk	Unk	Moderate avoidance at short range
(Carney & Sydeman 1999)	Common guillemot	Aircraft	Literature review	Unk	Unk	Unk	Scotland	Unk	na	Unk	Unk	Virtually no reaction to light aircraft within 100m of breeding colony
(Rojek *et al.* 2007)	Common guillemot	Helicopter Aircraft Vessels	Land obs	Observation	3	1	California, USA	Unk	na	Breed	3	Aircraft disturbance at breeding colonies greatest when craft <ca. 300m above sea level. Helicopters caused more disturbance than fixed-wing aircraft – likely because of greater noise. Low flyovers flushed birds. Most vessel disturbance when within 50m of colony and if present for extended periods.

35. Razorbill

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Related spp. – Pigeon guillemot	Vessels Pile drive noise	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Took flight when approached by boats – resettled quickly. Not displaced by drilling.
(Leopold *et al.* 2011)	Auk spp.	Vessels	Boat obs	Observation	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Wary of approaching vessels – dive to escape.
(Garthe & Hüppop 2004)	Razorbill	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 3
(Ronconi & Clair 2002)	Related spp. - Black guillemot	Vessels	Land obs	BACI	1	1	Grand Manan Island, Canada	Unk	na	Jul Aug	1	Increased flush response the further from shore birds were. Large boats and those travelling slowly elicited a lower flush response than small, fast boats.
(Bellefleur *et al.* 2009)	Related spp. – Marbled murrelet	Vessels	Boat obs	BACI/ Experimental	1	1	Pacific Rim National Park Reserve, Canada	7500	Unk	Aug	1	Strong flush response at 30-40m distance from boat. More often dived than flew. If birds dived, they were more likely to stay in area. If flew, birds tended to leave the area.
(Barrett & Vader 1984)	Auk spp.	Vessels	Unk	Unk	Unk	Unk	Norway	Unk	na	Unk	Unk	Moderate avoidance at short range
(Evans & Nettleship 1985)	Auk spp.	Vessels	Unk	Unk	Unk	Unk	Unk	Unk	na	Unk	Unk	Moderate avoidance at short range

(Carney & Sydeman 1999)	Razorbill	Aircraft	Literature review	Unk	Unk	Unk	Scotland	Unk	na	Unk	Unk	Virtually no reaction to light aircraft within 100m of breeding colony
(Rojek *et al.* 2007)	Related spp. – common guillemot	Helicopter Aircraft Vessels	Land obs	Observation	3	1	California, USA	Unk	na	Breed	3	Aircraft disturbance at breeding colonies greatest when craft <ca. 300m above sea level. Helicopters caused more disturbance than fixed-wing aircraft – likely because of greater noise. Low flyovers flushed birds. Most vessel disturbance when within 50m of colony and if present for extended periods.

36. Black guillemot

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Related spp. – Pigeon guillemot	Vessels Pile drive noise	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Took flight when approached by boats – resettled quickly. Not displaced by drilling.
(Leopold *et al.* 2011)	Auk spp.	Vessels	Boat obs	Observation	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Wary of approaching vessels – dive to escape.
(Ronconi & Clair 2002)	Black guillemot	Vessels	Land obs	BACI/ Experimental	1	1	Grand Manan Island, Canada	Unk	na	Jul Aug	1	Increased flush response the further from shore birds were. Large boats and those travelling slowly elicited a lower flush response than small, fast boats.

(Bellefleur *et al.* 2009)	Related spp. – Marbled murrelet	Vessels	Boat obs	BACI/ Experimental	1	1	Pacific Rim National Park Reserve, Canada	7500	na	Aug	1	Strong flush response at 30-40m distance from boat. More often dived than flew. If birds dived, they were more likely to stay in area. If flew, birds tended to leave the area.
(Barrett & Vader 1984)	Auk spp.	Vessels	Unk	Unk	Unk	Unk	Norway	Unk	na	Unk	Unk	Moderate avoidance at short range
(Evans & Nettleship 1985)	Auk spp.	Vessels	Unk	Unk	Unk	Unk	Unk	Unk	na	Unk	Unk	Moderate avoidance at short range
(Carney & Sydeman 1999)	Related spp. – common guillemot	Aircraft	Literature review	Unk	Unk	Unk	Scotland	Unk	na	Unk	Unk	Virtually no reaction to light aircraft within 100m of breeding colony
(Rojek *et al.* 2007)	Related spp. – common guillemot	Helicopter Aircraft Vessels	Land obs	Observation	3	1	California, USA	Unk	na	Breed	3	Aircraft disturbance at breeding colonies greatest when craft <ca. 300m above sea level. Helicopters caused more disturbance than fixed-wing aircraft – likely because of greater noise. Low flyovers flushed birds. Most vessel disturbance when within 50m of colony and if present for extended periods.
(Garthe & Hüppop 2004)	Related spp. – common guillemot	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 3

37. Little auk

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Related spp. – Pigeon guillemot	Vessels Pile drive noise	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Took flight when approached by boats – resettled quickly. Not displaced by drilling.
(Leopold *et al.* 2011)	Auk spp.	Vessels	Boat obs	Observation	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Wary of approaching vessels – dive to escape.
(Cramp & Simmons 1980)	Little auk	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Low response
(del Hoyo, Elliott & Sargatal 1996)	Little auk	Vessels	Unk	Method not specified	Unk	Unk	Unk	Unk	na	Unk	Unk	Low response
(Ronconi & Clair 2002)	Related spp. - Black guillemot	Vessels	Land obs	BACI	1	1	Grand Manan Island, Canada	Unk	na	Jul Aug	1	Increased flush response the further from shore birds were. Large boats and those travelling slowly elicited a lower flush response than small, fast boats.
(Bellefleur *et al.* 2009)	Related spp. – Marbled murrelet	Vessels	Boat obs	BACI/ Experimental	1	1	Pacific Rim National Park Reserve, Canada	7500	Unk	Aug	1	Strong flush response at 30-40m distance from boat. More often dived than flew. If birds dived, they were more likely to stay in area. If flew, birds tended to leave the area.
(Evans & Nettleship 1985)	Auk spp.	Vessels	Unk	Unk	Unk	Unk	Unk	Unk	na	Unk	Unk	Slight avoidance at short range

38. Atlantic puffin

Study	Species	Measure of	Measured	Measure type	No. sites	No. studies	Study location	n	MRED type	Time of year	No. of years	Response
(Thuringer & Reidy 2006)	Related spp. – Pigeon guillemot	Vessels Pile drive noise	Land obs	Speculative/ anecdotal	1	1	Race Rocks, nr Vancouver Island, Canada	Unk	Tidal	Jul	1	Took flight when approached by boats – resettled quickly. Not displaced by drilling.
(Leopold *et al.* 2011)	Auk spp.	Vessels	Boat obs	Observation	1	1	OWEZ (Egmond) WF, Netherlands	Unk	Wind	All	8	Wary of approaching vessels – dive to escape.
(Garthe & Hüppop 2004)	Atlantic puffin	Vessels Helicopter	ESAS	Observation	Unk	Unk	Unk	Unk	na	Unk	Unk	Score 2
(Ronconi & Clair 2002)	Related spp. - Black guillemot	Vessels	Land obs	BACI	1	1	Grand Manan Island, Canada	Unk	na	Jul Aug	1	Increased flush response the further from shore birds were. Large boats and those travelling slowly elicited a lower flush response than small, fast boats.
(Bellefleur *et al.* 2009)	Related spp. – Marbled murrelet	Vessels	Boat obs	BACI/ Experimental	1	1	Pacific Rim National Park Reserve, Canada	7500	Unk	Aug	1	Strong flush response at 30-40m distance from boat. More often dived than flew. If birds dived, they were more likely to stay in area. If flew, birds tended to leave the area.
(Barrett & Vader 1984)	Auk spp.	Vessels	Unk	Unk	Unk	Unk	Norway	Unk	na	Unk	Unk	Slight avoidance at short range
(Evans & Nettleship 1985)	Auk spp.	Vessels	Unk	Unk	Unk	Unk	Unk	Unk	na	Unk	Unk	Slight avoidance at short range
(Carney & Sydeman 1999)	Atlantic puffin	Aircraft	Literature review	Unk	Unk	Unk	Scotland	Unk	na	Unk	Unk	Virtually no reaction to light aircraft within 100m of breeding colony

(Rojek *et al.* 2007)	Related spp. – common guillemot	Helicopter Aircraft Vessels	Land obs	Observation	3	1	California, USA	Unk	na	Breed	3	Aircraft disturbance at breeding colonies greatest when craft <ca. 300m above sea level. Helicopters caused more disturbance than fixed-wing aircraft – likely because of greater noise. Low flyovers flushed birds. Most vessel disturbance when within 50m of colony and if present for extended periods.

Appendix 1
Supplementary Table 4: Use of tidal races
Tables of data contributing to confidence levels for each species

Obs = observation
Unk = unknown

1. Greater scaup

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Holm & Burger 2002)	Related spp. – Velvet scoter	Land obs Boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Fast current flows avoided – mainly in shallow slack water but used faster currents than common goldeneye related spp.
(Jones & Drobney 1986)	Greater scaup	Sediment samples and stomach analysis	Observation	1	1	Detroit River, Michigan, USA	47	All	1	Favours areas of low current owing to presence of sediment type for foraging.

2. Common eider

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Robbins 2012)	Common eider	Land obs	Observation	1	1	Fall of Warness, Shetland, UK	66254		6	Usually close to shore
(Inger, Harrison & Bearhop 2010)	Common eider	Land obs	Observation	1	1	Strangford Lough, Ireland	343	All	4	Seen in the tidal channel
(Wade – see Chapter 5)	Common eider	Land obs	Observation with modelled tidal data	1	1	Inner Sound, Scotland, UK	5	All	1	In tidal channel but never dived in mean current flow speeds >1ms^{-1}

(Holm & Burger 2002)	Related spp. – Velvet scoter	Land and boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Fast current flows avoided – mainly in shallow slack water but used faster currents more than common goldeneye related spp. but less than long-tailed duck

3. Long-tailed duck

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Holm & Burger 2002)	Long-tailed duck	Land obs Boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Uses slack, eddy and main flow habitats. Used faster currents more than common goldeneye related spp. but not turbulent areas.

4. Common scoter

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Holm & Burger 2002)	Related spp. – Velvet scoter	Land and boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Fast current flows avoided – mainly in shallow slack water but used faster currents more than common goldeneye related spp. but less than long-tailed duck
(Kaiser *et al.* 2006)	Common scoter	Land, aerial and boat obs	Observation	1	1	Liverpool Bay, UK	Unk	All	3	Mean surface current where observed: 0.29 ± 0.15 ms^{-1} and not >0.6 ms^{-1}

5. Velvet scoter

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Holm & Burger 2002)	Velvet scoter	Land obs Boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Fast current flows avoided – mainly in shallow slack water but used faster currents more than common goldeneye related spp. but less than long-tailed duck

6. Common goldeneye

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Holm & Burger 2002)	Related spp. – Bufflehead	Land obs Boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Fast current flows and turbulence avoided – mainly in shallow slack water
(Jones & Drobney 1986)	Related spp. - Greater scaup	Sediment samples and stomach analysis	Observation	1	1	Detroit River, Michigan, USA	39	All	1	Favours areas of low current owing to presence of sediment type for foraging.

7. Red-throated diver

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Robbins 2012)	Diver spp.	Land obs	Observation	1	1	Fall of Warness, Shetland, UK	4638	All	6	Dive in areas of depths 21-31m – could overlap
(Wade – see Chapter 5)	Diver spp.	Land obs	Observation with modelled tidal data	1	1	Inner Sound, Scotland, UK	11	All	1	In tidal channel but never dived in mean current flow speeds >2ms^{-1}
(Rodger 2014)	Diver spp.	Land obs	Observation with modelled tidal data	1	1	Bluemull Sound, Shetland, UK	Unk	Non-breeding season – Jan, Feb, Mar	1	Abundance significantly higher in areas with visibly fast, unidirectional current associated with conditions suitable for tidal energy generation
(Wade – see Chapter 5)	Diver spp.	Land obs	Observation	1	1	Inner Sound, Scotland, UK	11	All	1	Mostly associated with areas of no visible surface feature when foraging in tidal channel.
(Holm & Burger 2002)	Related spp. – great northern diver	Land and boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Habitat slightly avoided. Slack and main flow areas used.

(Gill *et al.* 2008)	Diver spp.	Aerial obs.	Observation	1	1	Kentish Flats WF, UK	Unk	Feb	6	Large accumulation of diver spp. near the outflow of an estuary – hypothesised this is because of strength of tidal currents and huge volume of water rushing in and out of estuary with each tide.

8. Black-throated diver

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Robbins 2012)	Diver spp.	Land obs	Observation	1	1	Fall of Warness, Shetland, UK	4638	All	6	Dive in areas of depths 21-31m – could overlap
(Wade – see Chapter 5)	Diver spp.	Land obs	Observation with modelled tidal data	1	1	Inner Sound, Scotland, UK	11	All	1	In tidal channel but never dived in mean current flow speeds >2ms^{-1}
(Rodger 2014)	Diver spp.	Land obs	Observation with modelled tidal data	1	1	Bluemull Sound, Shetland, UK	Unk	Non-breeding season – Jan, Feb, Mar	1	Abundance significantly higher in areas with visibly fast, unidirectional current associated with conditions suitable for tidal energy generation
(Wade – see Chapter 5)	Diver spp.	Land obs	Observation	1	1	Inner Sound, Scotland, UK	11	All	1	Mostly associated with areas of no visible surface feature when foraging in tidal channel.
(Holm & Burger 2002)	Related spp. – great northern diver	Land and boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Habitat slightly avoided. Slack and main flow areas used.
(Gill *et al.* 2008)	Diver spp.	Aerial obs.	Observation	1	1	Kentish Flats WF, UK	Unk	Feb	6	Large accumulation of diver spp. near the outflow of an estuary – hypothesised this is because of strength of tidal currents and huge volume of water rushing in and out of estuary with each tide.

9. Great northern diver

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Robbins 2012)	Diver spp.	Land obs	Observation	1	1	Fall of Warness, Shetland, UK	4638	All	6	Dive in areas of depths 21-31m – could overlap
(Rodger 2014)	Great northern diver	Land obs	Observation with modelled tidal data	1	1	Bluemull Sound, Shetland, UK	Unk	Non-breeding season – Jan, Feb, Mar	1	Abundance significantly higher in areas with visibly fast, unidirectional current associated with conditions suitable for tidal energy generation
(Wade – see Chapter 5)	Great northern diver	Land obs	Observation with modelled tidal data	1	1	Inner Sound, Scotland, UK	11	All	1	In tidal channel but never dived in mean current flow speeds >2ms^{-1}
(Wade – see Chapter 5)	Great northern diver	Land obs	Observation	1	1	Inner Sound, Scotland, UK	11	All	1	Mostly associated with areas of no visible surface feature when foraging in tidal channel.
(Holm & Burger 2002)	Great northern diver	Land and boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Habitat slightly avoided. Slack and main flow areas used.
(Winiarski *et al.* 2013)	Great northern diver	Aerial obs	Observation	1	1	Rhode Island, USA	951	Dec, Jan, Feb	2	Primarily found in shallow nearshore water <35m deep and with high long term (10yr mean) chl a surface concentrations – high chl a concentrations unlikely to occur in turbulent mixed areas of water associated with tidal races
(Gill *et al.* 2008)	Diver spp.	Aerial obs.	Observation	1	1	Kentish Flats WF, UK	Unk	Feb	6	Large accumulation of diver spp. near the outflow of an estuary – hypothesised this is because of strength of tidal currents and huge volume of water rushing in and out of estuary with each tide.

10. Great-crested grebe

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Holm & Burger 2002)	Related spp. – Red-necked grebe	Land obs Boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Mainly in slack water and main flow areas

11. Slavonian grebe

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Holm & Burger 2002)	Related spp. – Red-necked grebe	Land obs Boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Mainly in slack water and main flow areas

12. Northern fulmar

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Ladd *et al.* 2005)	Fulmar	Boat obs	Observation	8	1	Aleutian Islands, USA	Unk	May, Jun	2	Foraged over slicks associated with convergence zones and tidally driven eddies.

13. Sooty shearwater

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Fenwick 1978)	Sooty shearwater	Boat obs	Anecdotal	1	1	Snares Islands	Unk	Unk	2	Possible use of turbulent areas.

14. Manx shearwater

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Fenwick 1978)	Related spp. - Sooty shearwater	Boat obs	Anecdotal	1	1	Snares Islands	Unk	Unk	2	Possible use of turbulent areas.

15. European storm-petrel

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
No data										

16. Leach's storm-petrel

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
No data										

17. Northern gannet

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Robbins 2012)	Northern gannet	Land obs	Observation	1	1	Fall of Warness, Shetland, UK	10623	All	6	More frequent close to shore and majority dive in depths 21-31m
(Inger *et al.* 2010)	Northern gannet	Land obs	Observation	1	1	Strangford Lough, Ireland	344	All	4	Seen in the tidal channel

18. Great cormorant

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Robbins 2012)	Cormorant spp.	Land obs	Observation	1	1	Fall of Warness, Shetland, UK	145613	All	6	Dive in depths 1-31m. More frequent close to shore.

(Inger *et al.* 2010)	Cormorant	Land obs	Observation	1	1	Strangford Lough, Ireland	63146 (cormorant and shag combined)	All	4	Seen in the tidal channel
(Holm & Burger 2002)	Related spp. – Brandt's and Pelagic cormorant	Land and boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Turbulent water used, as well as slack and main flow areas. Increased diving with increasing current speeds.

19. Shag

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Robbins 2012)	Cormorant spp.	Land obs	Observation	1	1	Fall of Warness, Shetland, UK	145613	All	6	Dive in depths 1-31m. More frequent close to shore.
(Inger *et al.* 2010)	Shag	Land obs	Observation	1	1	Strangford Lough, Ireland	63146 (cormorant and shag combined)	All	4	Seen in the tidal channel
(Waggitt *et al.* in prep.)	Shag	Land obs	Observation with modelled tidal data	5	5	Bluemull Sound, Fall of Warness, Rousay Sound, Sound of Islay, Inner Sound – all UK	Unk	May, Jun, Jul	1-2	More often found in non-turbine habitat (areas with current speeds <2ms^{-1}) than turbine habitat (4 out of 5 studies)

(Rodger 2014)	Shag	Land obs	Observation with modelled tidal data	1	1	Bluemull Sound, Shetland, UK	Unk	Non-breeding season – Jan, Feb, Mar	1	Favour fast unidirectional flow likely to be associated with tidal-stream development sites and a greater density of birds found in current speeds >2ms^{-1} but with no clear preference for depth – range from shallow to deeper waters. But more likely to dive in slow current speed areas than fast flow areas. Tidal conveyor behaviour observed.
(Wade – see Chapter 5)	Shag	Land obs	Observation with modelled tidal data	1	1	Inner Sound, Scotland, UK	109	All	1	Some diving in mean current flow speeds >2ms^{-1} – not in winter.
(Wade – see Chapter 5)	Shag	Land obs	Anecdotal	1	1	Inner Sound, Scotland, UK	Unk	All	1	Tidal conveyor behaviour observed.
(Wade – see Chapter 5)	Shag	Land obs	Observation	1	1	Inner Sound, Scotland, UK	109	All	1	Mostly associated with areas of no visible surface feature when foraging in tidal channel. Some association with upwellings.
(Holm & Burger 2002)	Related spp. – Brandt's and Pelagic cormorant	Land and boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	Turbulent water used, as well as slack and main flow areas. Increased diving with increasing current speeds.
(Langston 2010)	Shag	Unk	Unk	Unk	Unk	Unk	Unk	Unk	Unk	Areas of tidal flow identified as key habitat
(Wanless, Harris & Morris 1991)	Shag	Radio-telemetry	Observation	1	1	Isle of May, Scotland, UK	31	Jun, Jul	3	Feeding in areas subject to stronger tidal flows than common around study site.

20. White-tailed eagle

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
No data										

21. Arctic skua

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
No data										

22. Great skua

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
No data										

23. Black-headed gull

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
No data										

24. Common gull

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Wade – see Chapter 5)	Common gull	Land obs	Anecdotal	1	1	Inner Sound, Scotland, UK	Unk	All	1	Several birds seen plunge diving into turbulent shearline – moved repeatedly down the shearline similar to the conveyor foraging seen in black guillemots and shags.

25. Lesser black-backed gull

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
No data										

26. Herring gull

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
No data										

27. Great black-backed gull

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
No data										

28. Black-legged kittiwake

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Wade – see Chapter 5)	Black-legged kittiwake	Land obs	Observation	1	1	Inner Sound, Scotland, UK	87	All	1	Mostly associated with areas of no visible surface feature when foraging in tidal channel. Some association with shearlines
(Irons 1998)	Black-legged kittiwake	Radio-telemetry	Observation	1	1	Prince William Sound, Alaska	26	Breed	1	Foraged in areas of potential tidal upwellings or eddies.
(Langston 2010)	Black-legged kittiwake	Unk	Unk	Unk	Unk	Unk	Unk	Unk	Unk	Use of tidal upwellings and eddies.

29. Little tern

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Keenan *et al.* 2011)	Tern spp	Land obs	Observation	1	1	Strangford Lough, Ireland	1893	All	4	Seen using tidal channel. Highest numbers during flood and ebb, lowest at slack. Say this could be turbulence of wake bringing food to surface.
(Perrow *et al.* 2006)	Little tern	Radio-telemetry	Observation	1	1	Scroby Sands, UK	Unk	Breed	1	Fed mainly close to shore, rarely more than 800m from coast. No obvious association with water current or stage of tide.
(Paiva *et al.* 2008)	Little tern	Land obs	Observation	6	1	South Portugal	Unk	Apr - Jul	3	Prefers sheltered lagoons to the sea – fed in lagoons more at low tide. Fed in channels of lagoons more during flowing tide, perhaps because fish availability increases with current flow.

30. Sandwich tern

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Keenan *et al.* 2011)	Tern spp.	Land obs	Observation	1	1	Strangford Lough, Ireland	1893	All	4	Seen using tidal channel. Highest numbers during flood and ebb, lowest at slack. Say this could be turbulence of wake bringing food to surface.

31. Common tern

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Keenan *et al.* 2011)	Tern spp	Land obs	Observation	1	1	Strangford Lough, Ireland	1893	All	4	Seen using tidal channel. Highest numbers during flood and ebb, lowest at slack. Say this could be turbulence of wake bringing food to surface.
(Perrow, Skeate & Gilroy 2011)	Common tern	Boat obs	Observation	1	1	Norfolk and Anglesey, UK	25	Breed	1	Repeated foraging flights along tidal front features.
(Safina 1990)	Common tern	Land obs	Observation	2	1	Western Block Island Sound and Long Island, NY, USA	Unk	Breed	4	Birds often foraged over abrupt topographic changes and where tidal currents were strong (current velocity not specified).

32. Roseate tern

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Keenan *et al.* 2011)	Tern spp	Land obs	Observation	1	1	Strangford Lough, Ireland	1893	All	4	Seen using tidal channel. Highest numbers during flood and ebb, lowest at slack. Say this could be turbulence of wake bringing food to surface.
(Safina 1990)	Roseate tern	Land obs	Observation	2	1	Western Block Island Sound and Long Island, NY, USA	Unk	Breed	4	Birds often foraged over tide rips and shoals in water <8m deep.

33. Arctic tern

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Keenan *et al.* 2011)	Tern spp	Land obs	Observation	1	1	Strangford Lough, Ireland	1893	All	4	Seen using tidal channel. Highest numbers during flood and ebb, lowest at slack. Say this could be turbulence of wake bringing food to surface.
(Langston 2010)	Arctic tern	Unk	Unk	Unk	Unk	Unk	Unk	Unk	Unk	Use of upwellings.

34. Common guillemot

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Robbins 2012)	Common guillemot	Land obs	Observation	1	1	Fall of Warness, Shetland, UK	54771	All	6	>90% obs in water 20m or greater depth. More often in flood rather than slack tides.
(Inger *et al.* 2010)	Common guillemot	Land obs	Observation	1	1	Strangford Lough, Ireland	6945 (common guillemot and razorbill combined)	All	4	Seen in the tidal channel
(Wade – see Chapter 5)	Common guillemot	Land obs	Observation	1	1	Inner Sound, Scotland, UK	38	All	1	Mostly associated with areas of no visible surface feature and shearlines when foraging in tidal channel.
(Holm & Burger 2002)	Common guillemot	Boat obs	Observation	2	1	Vancouver Island, Canada	21	Winter	1	Use areas of main tidal flow.

(Wanless, Harris & Morris 1990)	Common guillemot	Radio-telemetry	Observation	1	1	Isle of May, Scotland, UK	6	Breeding season – May-Jul	1	Frequent diving in a tide rip 0.5-1.0km from the Isle of May.
(Slater 1976)	Common guillemot	Anecdotal	Anecdotal	1	1	Copinsay, Orkney, UK	Unk	Breeding season – Apr, May	1	Suggests guillemots may use complex tidal flows to increase foraging success on sandeels and fish – turbulence would drive them to the surface
(Coyle *et al.* 1992)	Common guillemot	Ship obs	Observation	1	1	St. George Island, Bering Sea	Unk	Aug	1	Foraging in 'strong' flows of 0.5m s^{-1} – more often on an ebb tide and likely associated with tidal advection of prey due to ridge topography

35. Razorbill

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Robbins 2012)	Razorbill	Land obs	Observation	1	1	Fall of Warness, Shetland, UK	1699	All	6	More often away from shore rather than close to shore. More often during flooding tides.
(Inger *et al.* 2010)	Razorbill	Land obs	Observation	1	1	Strangford Lough, Ireland	6945 (common guillemot and razorbill combined)	All	4	Seen in the tidal channel
(Wade – see Chapter 5)	Razorbill	Land obs	Observation	1	1	Inner Sound, Scotland, UK	18	All	1	Mostly associated with shearlines when foraging in tidal channel.

(Holm & Burger 2002)	Related spp. – piscivorous auks	Land and boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	May use edge of high flow areas. Habitats used: slack, eddy and main flow.
(Wanless *et al.* 1990)	Razorbill	Radio-telemetry	Observation	1	1	Isle of May, Scotland, UK	3	Breeding season – May-Jul	1	Rare use of a tide rip 0.5-1.0km from the Isle of May.

36. Black guillemot

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Robbins 2012)	Black guillemot	Land obs	Observation	1	1	Fall of Warness, Shetland, UK	78071	All	6	More often close to shore and during ebb and slack tides. 82% in water depths 1-31m
(Inger *et al.* 2010)	Black guillemot	Land obs	Observation	1	1	Strangford Lough, Ireland	10437	All	4	Seen in the tidal channel
(Waggitt *et al.* in prep.)	Black guillemot	Land obs	Observation with modelled tidal data	5	5	Bluemull Sound, Fall of Warness, Rousay Sound, Sound of Islay, Inner Sound – all UK	Unk	May, Jun, Jul	1-2	More often found in turbine habitat (areas with current speeds >2ms^{-1}) than non-turbine habitat (3 out of 5 studies)
(Robbins *et al.* 2014)	Black guillemot	Land obs	Observation with modelled tidal data	1	1	Bluemull Sound, Shetland, UK	Unk	Breeding season and Jan	2	Feed in 'conveyor belt' of tidal stream – short flights upstream against the current, drift downstream on surface or during dive, then upstream flight

(Rodger 2014)	Black guillemot	Land obs	Observation with modelled tidal data	1	1	Bluemull Sound, Shetland, UK	Unk	Non-breeding season – Jan, Feb, Mar	1	Favour fast unidirectional flow likely to be associated with tidal-stream development sites and a greater density of birds found in current speeds 1.5-2ms^{-1} and in depths >20m. Declines in speeds >2ms^{-1}. But more likely to dive in slow current speed areas than fast flow areas. Tidal conveyor behaviour observed.
(Wade – see Chapter 5)	Black guillemot	Land obs	Observation with modelled tidal data	1	1	Inner Sound, Scotland, UK	60	All	1	Some diving in mean current flow speeds >2ms^{-1} – not in autumn or winter.
(Wade – see Chapter 5)	Black guillemot	Land obs	Anecdotal	1	1	Inner Sound, Scotland, UK	Unk	All	1	Tidal conveyor behaviour observed.
(Wade – see Chapter 5)	Black guillemot	Land obs	Observation	2	1	Inner Sound, Scotland, UK	122	All	1	Mostly associated with areas of no visible surface feature and eddies when foraging in tidal channel.
(Bradstreet & Brown 1985)	Black guillemot	Unk	Unk	Unk	Unk	Unk	Unk	Unk	Unk	Tidal channels/races preferred habitat
(Holm & Burger 2002)	Related spp. – pigeon guillemot	Land and boat obs	Observation	2	1	Vancouver Island, Canada	214	Winter	1	Uses areas of max tidal flow and eddy areas. Used slack water less than other piscivores – significantly higher abundances at times of max flow. Tidal conveyor behaviour observed. Selected water types formed by currents.

37. Little auk

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Holm & Burger 2002)	Related spp. – piscivorous auks	Land and boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	May use edge of high flow areas. Habitats used: slack, eddy and main flow.

38. Atlantic puffin

Study	Species	Measured	Measure type	No. sites	No. studies	Study location	n	Time of year	No. of years	Response
(Robbins 2012)	Atlantic puffin	Land obs	Observation	1	1	Fall of Warness, Shetland, UK	7882	All	6	More often away from shore rather than close to shore.
(Wade – see Chapter 5)	Atlantic puffin	Land obs	Observation with modelled tidal data	1	1	Inner Sound, Scotland, UK	14	All	1	Some diving in mean current flow speeds >2ms^{-1} – only in summer.
(Wade – see Chapter 5)	Atlantic puffin	Land obs	Observation	1	1	Inner Sound, Scotland, UK	45	All	1	Varied association with surface features when foraging in tidal channel. Some association with areas of no visible feature, shearlines and eddies.
(Wanless *et al.* 1990)	Atlantic puffin	Radio-telemetry	Observation	1	1	Isle of May, Scotland, UK	1	Breeding season – May-Jul	1	Frequent diving in a tide rip 0.5-1.0km from the Isle of May.
(Ladd *et al.* 2005)	Related species, tufted puffin – Atlantic puffin	Boat obs	Observation	8	1	Aleutian Islands, USA	Unk	May, Jun	2	Foraged in turbulent, well mixed areas
(Holm & Burger 2002)	Related spp. – piscivorous auks	Land and boat obs	Observation	2	1	Vancouver Island, Canada	Unk	Winter	1	May use edge of high flow areas. Habitats used: slack, eddy and main flow.

Appendix 1
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Supplementary Table 1: Locations and size (defined by Apparently Occupied Territories: AOT) of great skua colonies in Orkney, Shetland and north Scotland.

Colony	Location	Count year	AOT	Reference
Foula	Shetland	2007	1657	(JNCC 2014)
South Mainland†	Shetland	2000	1454	(Mitchell *et al.* 2004)
Unst	Shetland	2000	1385	(Mitchell *et al.* 2004)
Hoy & South Walls	Orkney	2010	1346	(Meek *et al.* 2011)
Fetlar	Shetland	2000	593	(Mitchell *et al.* 2004)
Yell	Shetland	2000	384	(Mitchell *et al.* 2004)
North Mainland†	Shetland	2000	293	(Mitchell *et al.* 2004)
Fair Isle	Shetland	2013	266	(JNCC 2014)
Handa	Sutherland	2013	135	(JNCC 2014)
West Mainland†	Shetland	2000	135	(Mitchell *et al.* 2004)
East Mainland†	Shetland	2000	103	(Mitchell *et al.* 2004)
Rousay	Orkney	2010	85	(Meek *et al.* 2011)
Mainland	Orkney	2010	81	(Meek *et al.* 2011)
Papa Stour	Shetland	2000	48	(Mitchell *et al.* 2004)
Eday	Orkney	2010	37	(Meek *et al.* 2011)
Flotta	Orkney	2010	29	(Meek *et al.* 2011)
Stronsay	Orkney	2010	26	(Meek *et al.* 2011)
Papa Westray	Orkney	2013	24	(JNCC 2014)
Westray	Orkney	2010	19	(Meek *et al.* 2011)
Fara	Orkney	2012	17	(JNCC 2014)
Whalsay	Shetland	2000	15	(Mitchell *et al.* 2004)
Calf of Eday	Orkney	2000	10	(Mitchell *et al.* 2004)
Eynhallow	Orkney	2010	9	(Meek *et al.* 2011)
Gairsay	Orkney	2010	7	(Meek *et al.* 2011)
South Ronaldsay	Orkney	2010	7	(Meek *et al.* 2011)
Copinsay	Orkney	2012	4	(JNCC 2014)
Swona	Orkney	2011	4	(JNCC 2014)
Sule Skerry	Orkney	2011	3	(JNCC 2014)
Biel of Duncansby	Caithness	2000	2	(Mitchell *et al.* 2004)
North Ronaldsay	Orkney	2012	2	(JNCC 2014)
Sanday	Orkney	2010	2	(Meek *et al.* 2011)
Cava	Orkney	2010	2	(Meek *et al.* 2011)
Muckle Skerry	Orkney	2010	2	(Meek *et al.* 2011)
Stroma	Caithness	2000	1	(Mitchell *et al.* 2004)
Faray & Holm	Orkney	2010	1	(Meek *et al.* 2011)
Auskerry	Orkney	2010	1	(Meek *et al.* 2011)
Shapinsay	Orkney	2012	1	(JNCC 2014)
Helliar Holm	Orkney	2010	1	(Meek *et al.* 2011)
Burray (Hunda)	Orkney	2010	1	(Meek *et al.* 2011)

† Consists of several area counts amalgamated. The AOT figure does not represent one colony. See (Mitchell *et al.* 2004) for figure showing breakdown of counts included in these areas.

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Supplementary Table 2: Comparison of trip characteristics and area used by birds from Foula and Hoy during breeding (incubating eggs and chick-rearing) and following a failed breeding attempt in the period 03 June 2011 to 15 July 2011. 25th and 75th refer to the 25th and 75th percentile of the interquartile range.

Colony and breeding status	Trip duration (h)			Trip length (km)			Max. distance from colony (km)			50% UD area (km2)
	25th	Median	75th	25th	Median	75th	25th	Median	75th	
Foula:										
Breeding (n=10; trips=330)	2.76	5.03	8.92	66.50	116.9	218.5	29.38	51.27	94.94	4049
Failed (n=4; trips=36)	4.07	7.34	17.87	83.05	174.20	267.4	33.09	70.44	102.7	8192
Hoy:										
Breeding (n=7; trips=193)	1.03	2.48	4.95	8.16	54.73	117.5	3.11	23.06	43.79	2519
Failed (n=5; trips=92)	1.53	4.85	12.72	8.51	90.18	231.8	4.03	35.77	93.40	17754

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Supplementary Table 3: Comparison of the number of trips undertaken per day by birds from Foula and Hoy during breeding (incubating eggs and chick-rearing) and following a failed breeding attempt in the period 03 June 2011 to 15 July 2011. 25th and 75th refer to the 25th and 75th percentile of the interquartile range.

Colony and breeding status	Trips per day			Frequency of the number of trips undertaken per day					
	25th	Median	75th	0	1	2	3	4	5
Foula:									
Breeding (n=10; trips=330)	1	1	1.75	68	161	64	12	0	1
Failed (n=4; trips=36)	0	0	1	35	31	1	1	0	0
Hoy:									
Breeding (n=7; trips=193)	1	2	2	11	39	32	18	4	4
Failed (n=5; trips=92)	0	1	1	31	43	16	4	0	1

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Supplementary Table 4: Comparison of trip characteristics and area used by birds from Hoy during incubation of eggs, chick-rearing, following successful fledging of chicks and following a failed breeding attempt in the period 11 June 2011 to 15 August 2011. 25th and 75th refer to the 25th and 75th percentile of the interquartile range.

Breeding status (Hoy)	Trip duration (h)			Trip length (km)			Max. distance from colony (km)			50% UD area (km2)
	25th	Median	75th	25th	Median	75th	25th	Median	75th	
Incubating (n=6; trips=17)	1.50	3.27	5.13	7.07	59.74	92.99	3.53	25.41	43.17	3742
Chick-rearing (n=4; trips=283)	1.20	2.87	5.72	8.04	52.41	104.87	3.06	21.38	34.25	1852
Fledged (n=2; trips=32)	3.20	7.07	9.07	18.01	84.01	113.79	8.08	29.98	31.34	1501
Failed (n=5; trips=215)	1.53	3.95	10.08	5.68	37.55	155.09	2.42	15.98	59.66	13112

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Supplementary Table 5: Comparison of the number of trips undertaken per day by birds from Hoy during incubation of eggs, chick-rearing, following successful fledging of chicks and following a failed breeding attempt in the period 11 June 2011 to 15 August 2011. 25th and 75th refer to the 25th and 75th percentile of the interquartile range.

Breeding status (Hoy)	Trips per day			Frequency of the number of trips undertaken per day						
	25th	Median	75th	0	1	2	3	4	5	6
Incubating (n=6; trips=17)	1	1	1	4	11	3	0	0	0	0
Chick-rearing (n=4; trips=283)	1	2	3	3	32	44	31	10	6	0
Fledged (n=2; trips=32)	1	2	3	1	6	3	4	2	0	0
Failed (n=5; trips=215)	0	1	2	44	70	32	16	3	3	1

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Appendix 3:

Published Articles

Article 1: Furness, R. W., H. M. Wade, A. M. C. Robbins, and E. A. Masden. 2012. Assessing the sensitivity of seabird populations to adverse effects from tidal stream turbines and wave energy devices. ICES Journal of Marine Science 69: 1466–1479.

Article 2: Furness, R. W., H. M. Wade, and E. A. Masden. 2013. Assessing vulnerability of marine bird populations to offshore wind farms. Journal of Environmental Management 119: 56–66.

Article 3: Wade, H. M., E. A. Masden, A. C. Jackson, C. B. Thaxter, N. H. K. Burton, W. Bouten, and R. W. Furness. 2014. Great skua (Stercorarius skua) movements at sea in relation to marine renewable energy developments. Marine Environmental Research 101:69–80.

ICES Journal of
Marine Science

ICES CIEM International Council for the Exploration of the Sea Conseil International pour l'Exploration de la Mer

ICES Journal of Marine Science (2012), 69(8), 1466–1479. doi:10.1093/icesjms/fss131

Assessing the sensitivity of seabird populations to adverse effects from tidal stream turbines and wave energy devices

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Furness, R. W., Wade, H. M., Robbins, A. M. C., and Masden, E. A. 2012. Assessing the sensitivity of seabird populations to adverse effects from tidal stream turbines and wave energy devices. – ICES Journal of Marine Science, 69: 1466–1479.

Received 4 May 2012; accepted 22 June 2012.

Tidal turbines and wave energy devices may affect seabird populations through collision mortality, disturbance and habitat loss. Given the pressures to harness tidal and wave energy, especially in Scottish waters, there is an urgent need to assess population-level impacts on seabird species. With a lack of deployed devices to monitor in areas of importance for seabirds, our approach uses data from scientific literature on seabird ecology and conservation importance likely to influence population vulnerability to "wet renewables" in Scottish waters. At this stage however, we can only infer likely interactions with tidal and wave devices. We identify black guillemot, razorbill, European shag, common guillemot, great cormorant, divers and Atlantic puffin as the species most vulnerable to adverse effects from tidal turbines in Scottish waters. We identify divers as the species most vulnerable to adverse effects from wave energy devices in Scottish waters. Wave energy devices seem likely to represent a lesser hazard to seabirds than tidal turbines, and both forms of energy capture seem likely to represent a lower hazard to seabirds than offshore wind farms (wind-power plants). The indices developed here for Scottish seabird populations could be applied to populations elsewhere. This approach will help in identifying likely impacts of tidal and wave energy deployments on seabirds, and in optimizing deployment of resources for compulsory environmental monitoring.

Keywords: Conservation, marine renewables, population vulnerability, Special Protection Areas.

Introduction

To meet targets for a reduction in greenhouse gas emissions, the Scottish and UK Governments are encouraging the rapid development of new marine technologies in Scottish waters to generate electricity from tidal and wave power ("wet renewables") (HM Government, 2010). Many tidal stream turbines resemble wind turbines but designs also include vertical or horizontal axis cross-flow turbines, flow augmented turbines (within a duct or shroud), oscillating devices and venture effect devices (see en.wikipedia.org/wiki/Tidal_stream_generator). Most tidal turbines are designed to be pinned to the sea bed well below depths used by shipping. Wave energy devices are usually floating structures that capture wave energy from the bending of joints between linked modules. However, a wide variety of devices may capture wave energy, including devices on the seabed (see en.wikipedia.org/wiki/Wave_power). Tidal and wave energy devices are likely to be deployed in large arrays, though occupying much smaller areas of sea than taken up by offshore wind farms (ICES, 2010). Scotland has some of the best natural resources in the world in terms of tidal flows and wave climate (Shields *et al.*, 2009), however, the development of wet renewables may potentially impact seabird populations through the effects of collision, disturbance, and/or habitat loss (ICES, 2010; Langton *et al.*, 2011). Since seabirds are typically long-lived and produce only small numbers of offspring that have a low probability of surviving to breeding age, direct mortality of adult seabirds has a far more pronounced impact on population dynamics than effects influencing breeding success (Furness and Monaghan, 1987). As a result, the hazards of additional mortality to adult seabirds greatly outweigh indirect influences resulting from disturbance or habitat loss affecting breeding productivity.

As well as holding some of the best tidal and wave energy resources, Scotland also holds internationally important populations of many seabird species (Mitchell *et al.*, 2004; Forrester

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et al., 2007). Many seabirds in Scotland breed within Special Protection Areas (SPAs) and so are protected by law (Scottish Habitats Regulations), potentially creating a conflict between energy generation and seabird conservation (a figure showing SPAs in Scotland and details of designated species can be found at SNH Sitelink http://gateway.snh.gov.uk/sitelink/ but note that there are over 40 sites already designated as SPAs for seabirds around Scotland, many of which are close to the key sites for wave and tidal stream energy development). There is a need to assess which seabird species are most likely to be affected by wet renewables, so that research and monitoring can focus on those species in locations where their populations overlap with wet renewable deployments (ICES, 2010). It is also important that species unlikely to be affected by interactions with wet renewables can be excluded, in an objective and scientifically appropriate manner, by regulators, developers and consultants. Impacts of tidal turbines and wave energy devices on seabirds are however largely unknown at present because there have not yet been enough deployments to study (Witt *et al.*, 2012). Nevertheless, an assessment can be made on the basis of knowledge of seabird ecology and inference of the effects of these devices on the physical environment and on lower trophic level organisms i.e. prey of seabirds (WWT Consulting, 2010; ABP Marine Environmental Research, 2011). OSPAR (Oslo and Paris Convention; the mechanism by which 15 European governments with western coasts and catchments protect the marine environment) recently requested guidance on this issue from the International Council for the Exploration of the Sea (ICES), and the ICES response was summarized as "impacts of wave, and particularly tidal stream devices, have the potential to be significant for some groups of organisms. It is important that the results of thorough monitoring of early deployments of wave and tidal stream devices are published and used to guide the management of subsequent developments" (ICES, 2010). However, assessment of potential impacts is required imminently to assist in the legal process of Environmental Impact Assessment (EIA) in relation to proposed development, especially where such developments may affect seabird populations in SPAs.

Fraenkel (2006) suggested that diving seabirds may normally be swept through the blades of a tidal turbine unharmed due to entrainment in the flow of water. However, this assumes that seabirds flow passively in the water rather than moving across or against the current. It is very unlikely that seabirds simply move passively with flowing water, and all the available evidence indicates that diving seabirds move actively through the water, and often swim against the current (e.g. Lovvorn and Jones, 1991; Holm and Burger, 2002; Daunt *et al.*, 2003; Heath *et al.*, 2006; Watanuki *et al.*, 2006; Harding *et al.*, 2009; Langston, 2010). For example, common eiders *Somateria mollissima* always dive against the current and normally surface upstream of their entry point (Heath *et al.*, 2006). Seabirds that forage on the sea floor (i.e. on benthic prey) may be particularly vulnerable to collision with tidal turbine blades, especially where birds have travelled some distance across the sea floor before they ascend to the surface. This may bring them, unexpectedly, into the swept area of a turbine blade, from below. It is evident that seabirds able to dive deep are more likely to come into contact with tidal turbines, as are those seabirds that make particular use of tidal races as foraging habitat. Seabirds that are less able to avoid underwater hazards are more likely to be killed by such structures, and seabirds that forage over small areas of sea close to the coast are more likely to be affected by tidal turbines than seabirds that forage over large areas of pelagic habitat. It has also been suggested that the slow turbine speeds relative to the manoeuvrability of diving bird species would make the risk of seabird mortality very low (Awatea, 2008). However, a typical seabird swimming speed is of the order of 1.5 ms^{-1} (Wanless *et al.*, 1988, 1993, 1997, 1998; Halsey *et al.*, 2006a, b; Heath *et al.*, 2006; Ribak *et al.*, 2008; Thaxter *et al.* 2010) and a tidal turbine blade tip turning at 15 r.p.m. would be moving faster than this (ICES, 2010) and so potentially be difficult for a bird to avoid. ICES (2010) also highlight that "alterations in patterns of turbulence may affect the feeding behaviour of some seabirds, particularly terns". In addition, wave energy convertors have the potential to alter water column and sea bed habitats, and by changing the wave climate/environment may cause changes some distance from the installation, however due to the infancy of the industry, empirical evidence is lacking. It is against this background of a lack of scientific evidence from research studies into the impacts of wave and tidal devices on seabirds, that there is a need to evaluate likely impacts of these new technologies on seabirds. Wherever possible it should be based on the detailed knowledge of relevant aspects of seabird ecology and behaviour and mindful of the absence of detailed information on the devices and their deployment. Here we review evidence for likely impacts on seabirds, and construct indices assessing the relative vulnerability of seabird species' populations to impacts of tidal turbines and of wave energy devices.

Methods and Results

This review firstly considers tidal turbines and impacts they may have on seabirds, and secondly considers wave energy devices and their impacts. Desholm (2009) argued that in order to prioritize bird species for assessment of the impact of mortality at wind farms, it is possible to consider just two criteria; proportion of the population at risk and demographic elasticity (essentially represented by adult survival rate). Birds with high proportions of their populations coming into contact with devices and with high adult survival rates will be more severely impacted than birds with small proportions of their populations coming into contact with devices and with low natural survival rates. While that approach has the benefit of great simplicity, it does not take into account the fact that some kinds of birds are more, or less, likely to be affected as a consequence of their species-specific ecology or behaviour. This review therefore follows the approach developed by Furness and Tasker (2000), and successfully implemented for offshore wind farm hazards to seabirds in the southern North Sea by Garthe and Hüppop (2004). The method scores a number of factors considered to influence risk at the population level and wherever possible, the scores allocated to species for each factor are evidence-based, with relevant references cited where appropriate. Where the evidence base is especially limited however, the set of scoring criteria and provisional scores for seabird species, which are based on the available evidence taken from the reviewed literature, have been circulated to a group of appropriate experts for moderation (details in Acknowledgements). This was intended to ensure that the final criteria and scores have wide consensus support from stakeholders, including seabird ecologists, and conservationists. The list of seabird species includes true seabirds, wintering sea ducks and grebes, and the white-tailed eagle *Haliaetus albicilla*. Seabirds excluded from the list were the few species considered not to be likely to interact with wet renewables in Scottish waters either because

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their distribution was entirely coastal (such as red-breasted mergansers *Mergus serrator*), or because their numbers in Scottish waters were very low or limited to occurring over very short time periods.

Conservation status factors

We have scored similar factors to those presented by Garthe and Hüppop (2004), but we have adjusted the factors to reflect conservation importance of seabird populations in a Scottish rather than southern North Sea context. The index developed by Garthe and Hüppop (2004) included nine factors, of which three represented conservation status of the species and six represented aspects of the hazard that devices were considered to represent, based on aspects of the ecology of each species. A similar approach has been followed here though four factors are used as measures of conservation status: status in relation to the Birds Directive, percentage of the biogeographic population that occurs in Scotland, adult survival rate, and UK threat status.

Birds Directive status

The Birds Directive is a European Union directive adopted in 2009 to protect European wild birds and their habitats, in particular through the designation of Special Protection Areas (SPAs). The Birds Directive gives extremely powerful protection to birds, particularly those listed in Annex 1. Species listed in Annex 1 of the Birds Directive were given a score of 5 while species qualifying as "Migratory species" but not on Annex 1 were scored 3. Remaining species score 1. These scores (Supplementary Table 1) reflect aspects of conservation importance, but are not necessarily optimal for guidance regarding consenting risk.

Percentage of the biogeographic population in Scotland

The percentage of the biogeographic population (usually based on continuous distribution of the relevant subspecies, taken as the North Atlantic or European or Palearctic population as seems appropriate for particular species) occurring in Scotland was assessed from Forrester *et al.* (2007), or by comparing the population estimate in Forrester *et al.* (2007) with the biogeographic population estimates given in del Hoyo *et al.* (1992, 1996). Scores were allocated as: 1 ($<$1%), 2 (1–4%), 3 (5–9%), 4 (10–19%), or 5 ($\geq$ 20%) (Supplementary Table 1). Since this metric may vary seasonally, we used the highest seasonal score for each species.

Adult survival rate

Published data on adult survival rate (which are robust data for most seabird species) were included as a factor to reflect the vulnerability of species to any increase in mortality (due to "wet renewables") above natural mortality. Species with low rates of adult survival will tend to be less vulnerable to additional mortality than species with high annual survival. Data were taken from the scientific literature (del Hoyo *et al.* 1992, 1996; Garthe and Hüppop, 2004; Glutz von Blotzheim and Bauer,1982; Saether, 1989), from individual species studies, or estimated from data for closely related species. Where several estimates were available, preference was given to more recent studies, and studies in the UK, since survival rates in populations of the same species may sometimes differ between geographical regions. Adult survival rates were classified on a scale from 1 to 5 according to Garthe and Hüppop (2004): 1 (adult survival $<$ 0.749), 2 (adult survival 0.75–0.799), 3 (0.80–0.849), 4 (0.85–0.899), 5 (adult survival $>$ 0.90) (Supplementary Table 1).

UK threat status

This factor reflects both threat and conservation status of the species in the UK, as given by Eaton *et al.* (2009) in "Birds of Conservation Concern 3" (BOCC3). For some species, the classification in BOCC3 differs from that in the previous assessment (BOCC2), and these changes are also taken into account here, given the implications of changes in status. Scores were allocated as follows: 1 (green in BOCC2 and BOCC3), 2 (amber in BOCC2 and green in BOCC3), 3 (green in BOCC2 and amber in BOCC3), 4 (amber in BOCC3 and BOCC2), 5 (red in BOCC3) (Supplementary Table 1).

Vulnerability factors for tidal turbines

Scores for the various vulnerability component factors are arranged on a scale of 1–5, where 5 is a strong anticipated negative impact. It is assumed that these individual factor scores can then be summed to give a total for each species that ranks species according to the likely conservation concern with regard to these developments. Seven factors are scored, representing negative effects of tidal turbines on seabirds or sensitivities of the ecology of seabird species: drowning risk, mean and maximum diving depth, benthic foraging, use of tidal races for foraging, feeding range, disturbance by ship traffic, and habitat specialization. Other factors, such as impacts of anti-fouling paints on structures or chemical spillages associated with the structures, were considered not to represent a significant threat to seabirds.

Drowning risk

Seabird species vary in their risk of drowning. For example some species appear to be particularly prone to getting stuck in nets and traps, while others avoid such hazards more successfully. Differences among species are likely to be caused by a range of features, including species' morphology, feeding ecology, and behaviour. It is well known that juvenile birds are often more prone to such mortality than adults. For example, ring recoveries of juvenile shags *Phalacrocorax aristotelis* occur not infrequently in lobster pots, whereas adult shags are rarely trapped in this way (Galbraith *et al.*, 1981). Scoring species on this factor is difficult, since it is impossible to obtain quantitative data on such risk across species, but there are published studies reviewing the causes of mortality of seabirds which identify drowning through entanglement with underwater structures and its prevalence in certain birds (such as sea ducks and divers; Zydelis *et al.*, 2009). For surface-feeding seabirds the risk of encountering tidal turbines is clearly very low. For diving seabirds we have reviewed literature to identify species most at risk, and have used the peer review process (expert judgement) to moderate scores. Risk is scored from extremely low (score 1) to moderate (score 5) (Supplementary Table 2).

Mean and maximum diving depth

Seabirds capable of diving to depths where tidal turbines will be deployed, and regularly doing so, will presumably be at greater risk of colliding with these structures. Depth deployment of tidal turbines is uncertain at present, and varies with design but is typically 30–50 m below sea surface (Aquatera, 2010). Mean and maximum diving depths of seabirds have been recorded for many species, predominantly by deployment of data loggers on breeding seabirds. Scores were allocated as follows: score 1 (surface feeders with maximum diving depth no more than 1 m); score 2 (regularly dive to 2 or 3 m but have a maximum

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diving depth of no more than 5 m); score 3 (regularly dive to 5 m but rarely below 20 m); score 4 (regularly dive to 20 m but rarely below 40 m); score 5 (regularly dive to 30 m and deeper). These depth categories were chosen to spread the species out among the five categories, and to clearly separate species likely to attain depths where tidal turbines are deployed from those unable to access such depths. Where data differ between studies, we gave greater weighting to more recent studies which tend to use instrumentation rather than inference i.e. inferred depth from time spent (Supplementary Table 3).

Benthic foraging

Benthic foraging seabirds are more likely to interact with tidal turbines than seabirds that do not forage on benthic prey. This score ranges from 1 (<5% benthic foraging); 2 (5% ≤ benthic foraging < 20%); 3 (20% ≤ benthic foraging < 40%); 4 (40% ≤ benthic foraging < 70%); 5 (70% ≤ benthic foraging < 100%). Again, these categories were selected to spread species out over the five scores (Supplementary Table 4).

Use of tidal races for foraging

There have been very few studies of the use of high tidal flow areas by foraging seabirds, and most of the few published studies relate to seabirds in the North Pacific. However, Slater (1976) showed that common guillemots *Uria aalge,* in Orkney displayed a tidal rhythm in foraging activity during the early part of the breeding season, but not during chick-rearing. He suggested that this tidal rhythm indicated a higher prey capture rate during low tide periods or possibly during the rising tide, and that this might be a feature particularly prominent in Orkney due to the high flow through the archipelago and the Pentland Firth. Arctic terns *Sterna paradisaea* and common terns *S. hirundo* in the Wadden Sea also forage selectively at stages of the tide and in geographical locations with relatively faster flowing (1 m/sec) shallow (<10 m) water (Schwemmer *et al*., 2009), because this apparently increases their prey capture rates (Schwemmer *et al*., 2009). However, the tidal flow rates reported by Schwemmer *et al.* (2009) are relatively low in comparison to areas under consideration for deployment of tidal turbines (usually in excess of 4 m/sec). Scores are based on these cited references and others wherever possible, moderated by peer assessment, and are assigned to spread species out over the five categories. The scores ranged from 1 (habitat strongly avoided) to 5 (preferred habitat) (Supplementary Table 5).

Feeding range

Breeding seabirds are "central place foragers" and so are constrained to return to the central place (the nest site). In winter and during migration periods, seabirds are considerably less constrained and some species of seabirds may travel over enormous distances, but some seabirds even in winter are limited by the need to return to a safe nocturnal roost site, or to spend the night out of water (e.g. cormorants). Although the distribution of predictable feeding hotspots may influence habitat quality for seabirds, species with short feeding ranges will be more likely to be affected by the placement of renewable energy devices than seabirds with greater foraging ranges. The latter species can pass by areas with tidal turbines with minimal loss of foraging opportunity. Seabirds with short foraging ranges may lose higher proportions of their available foraging area if displaced by the presence of devices. We have classified scores as: score 1 (mean range generally over 90 km, and maximum generally over 150 km); score 2 (mean range 20–90 km and maximum around 100–150 km); score 3 (mean range 10–20 km and maximum around 50–80 km); score 4 (mean range 5–10 km and maximum around 20–50 km); score 5 (mean range less than 5 km and maximum generally less than 20 km). Where the maximum and the mean range values reported in the literature fell into different scores based on these criteria, we gave higher weight to the mean values. Where several estimates were available in the literature and these fell into different score bands, we gave less weighting to estimates that were in review publications rather than based on original data, less weighting to estimates reported in unpublished reports or reports that appeared not to have been peer reviewed, less weighting to older literature, and less weighting to estimates derived indirectly from data such as time spent away from the nest combined with estimated flight speed (and more weighting to data derived from direct measures such as deployment of GPS loggers on breeding birds). Where no data could be found for particular species, scores were estimated from the scores allocated to closely related species with similar ecology (Supplementary Table 6).

Disturbance by ship traffic

Seabird species differ in their reaction to the ship traffic that occurs during deployment and maintenance of tidal turbines. This behaviour relates in part to the general responsiveness of species to disturbance, and in part to specific responses to ships, although the latter can vary as a result of habituation. In the context of offshore wind farms, Garthe and Hüppop (2004) presented a similar vulnerability factor based on both ships and helicopters as disturbances. For tidal turbine arrays however, it is likely that ships will be used for maintenance work rather than helicopters, so the index here is only in relation to ship disturbance. It is known that alcids can be disturbed by boats hundreds of metres away (Ronconi and Clair, 2002; Bellefleur *et al*., 2009), that divers are especially sensitive to approaching boats more than 1 km away (Schwemmer *et al*., 2011), and amongst the sea ducks, scoters are particularly vulnerable to disturbance by boats (Kaiser *et al*., 2006; Schwemmer *et al*., 2011). Scores were allocated as: 1 (little or no response of birds to vessels passing close by), 2 (slight avoidance at short range), 3 (low to moderate flush distance, or moderate avoidance at short range), 4 (moderate to high flush distance), 5 (birds often flushed at long range (1000 m or more) and moderate to high short-term loss of foraging opportunity due to disturbance) (Supplementary Table 7).

Habitat specialization

Seabirds vary in the range of habitats they use, for example relating to water masses and frontal systems and whether they use these as specialists or generalists. This score classifies species into categories from 1 (tend to forage over large marine areas with little known association with particular marine features) to 5 (tend to feed on very specific habitat features, such as shallow banks with bivalve communities, or kelp beds). Where available, scores presented by Garthe and Hüppop (2004) were used. Scores for other species were based on foraging ecology described in single species studies in the literature, or from standard handbook descriptions. Literature indicates many cases of species showing limited flexibility in feeding habitat. For example, common eiders, long-tailed ducks *Clangula hyemalis* and common scoters *Melanitta nigra* are dependent on shallow feeding grounds with shellfish banks (Garthe, 2006). Species scoring 4 or 5 are more

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likely to be adversely affected by loss of habitat if tidal turbines or wave energy devices are placed within areas that they would otherwise use for foraging (Supplementary Table 8).

Assessing an overall score for species' vulnerability to tidal turbines

Garthe and Hüppop (2004) computed a risk index that summed four related factors and divided the sum by four, and multiplied that by the sum of two other factors divided by two. This recognized that the last two factors related to different aspects from the first four. They then multiplied for each species the answer from this formula by the conservation importance score for the species. Ranking species by the final index value provides a simple way to identify those populations that appear to be most vulnerable. A similar procedure is used here.

Of the various risk factors scored, the use of tidal races as a foraging habitat seems to be especially important in determining overall vulnerability. The diving ability of species appears also to be highly important, while the other five factors may be placed as of similar, but lower importance. To recognize this, the overall index value for each species has been computed as:

Use of tidal races score × diving depth score × mean score for other five factors × conservation importance score, divided by 100 (an arbitrary value).

$$Overall\,score = \frac{Tidal\,race \times diving\,depth \times \frac{1}{5}\sum_{i=1}^{5} score_i \times conservation\,(summed)}{100}$$

The index value could therefore range up to 25. We divided index scores into 5 categories: very low vulnerability (score ≤ 1), low vulnerability ($1 <$ score ≤ 2), moderate vulnerability ($2 <$ score ≤ 5), high vulnerability ($5 <$ score ≤ 10), and very high vulnerability ($10 <$ score ≤ 25). These categories are arbitrary, but are designed to be precautionary (in setting a very wide range of scores in the top category of very high vulnerability and very narrow ranges of scores in the two lowest categories) and to discriminate between high and low groups of species (Table 1).

Table 1. Species vulnerability index for tidal turbine impacts on seabirds (ranked by species score).

Species	Vulnerability index	Descriptor on 5-score scale
Black guillemot	9.9	4: high vulnerability
Razorbill	9.6	4: high vulnerability
Shag	9.6	4: high vulnerability
Common guillemot	9.0	4: high vulnerability
Great cormorant	7.0	4: high vulnerability
Great northern diver	4.1	3: moderate vulnerability
Red-throated diver	3.8	3: moderate vulnerability
Atlantic puffin	3.8	3: moderate vulnerability
Black-throated diver	3.6	3: moderate vulnerability
Little auk	2.2	3: moderate vulnerability
Slavonian grebe	2.0	2: low vulnerability
Arctic tern	1.9	2: low vulnerability
Common eider	1.5	2: low vulnerability
Common scoter	1.5	2: low vulnerability
Manx shearwater	1.5	2: low vulnerability
Velvet scoter	1.4	2: low vulnerability
Northern gannet	1.4	2: low vulnerability
Common goldeneye	1.1	2: low vulnerability
Great-crested grebe	1.1	2: low vulnerability
Sooty shearwater	1.1	2: low vulnerability
Sandwich tern	1.1	2: low vulnerability
Greater scaup	1.0	1: very low vulnerability
Long-tailed duck	1.0	1: very low vulnerability
Great black-backed gull	1.0	1: very low vulnerability
Roseate tern	1.0	1: very low vulnerability
Black-legged kittiwake	0.9	1: very low vulnerability
Herring gull	0.8	1: very low vulnerability
Great skua	0.7	1: very low vulnerability
Common gull	0.7	1: very low vulnerability
Lesser black-backed gull	0.7	1: very low vulnerability
Little tern	0.7	1: very low vulnerability
White-tailed eagle	0.6	1: very low vulnerability
Arctic skua	0.6	1: very low vulnerability
Common tern	0.6	1: very low vulnerability
Black-headed gull	0.6	1: very low vulnerability
Northern fulmar	0.5	1: very low vulnerability
European storm-petrel	0.5	1: very low vulnerability
Leach's storm-petrel	0.5	1: very low vulnerability

Vulnerability factors for wave energy devices

We have included seven factors. Of these, five are scored on a 1 to 5 scale (risk of collision mortality due to structures, exclusion from foraging habitat, disturbance by structures, disturbance by ship traffic, and flexibility in habitat use), with 5 representing high vulnerability or impact. The final two factors (benefit from roost platform, and benefit from fish attraction device effects or biofouling) are scored on a negative scale from 0 to −2 (0 representing no effect and −2 representing a small benefit for birds of that species). Other factors, such as impacts of anti-fouling paints on structures or chemical spillages associated with the structures, were considered not to represent a significant threat to seabirds so are not considered here.

Risk of collision mortality due to structures

Some seabirds may be at risk of injury or death from colliding with wave energy devices, either in flight or while swimming or diving. This score classifies species into categories from 1 (minimal risk of mortality) to 5 (moderate risk of mortality) (Supplementary Table 9). Given the nature of wave energy devices, even a score of 5 on this factor would probably represent a relatively low risk compared to risks such as entanglement in netting.

Exclusion from foraging habitat due to behavioural constraints

Wave energy devices might prevent some seabirds from foraging in important habitat. This may be because the seabirds are unable to land or take off readily where devices are present in the water, because other birds have been attracted into the area and affect their foraging, or because they need to spend time avoiding the devices rather than searching for food. The impact may be trivial however for seabirds which have long foraging ranges and a wide diversity of habitats in which they can feed. For example, divers need open water for landing and taking off, and may be unable to land in areas where devices block a descent flight onto the water. The score classifies species into categories from 1 (minimal exclusion) to 5 (moderate exclusion from foraging habitat) (Supplementary Table 10).

Benefit from roost platform

Under relatively calm sea conditions, wave energy devices may provide some seabirds with a resting platform. For some seabirds

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such an opportunity could extend their potential foraging area. For example, great cormorants *Phalacrocorax carbo* and shags need to return to shore to dry their plumage after a foraging bout. Having resting sites at sea could allow these birds to exploit areas further from shore that would otherwise be uneconomical for birds to commute to from the shore. Such situations have been reported, for example in the case of cormorants in the Baltic Sea which now extend their foraging distribution further offshore by roosting on the structures of the offshore wind farm at Nysted. The score for this factor classifies species into categories from 0 (no significant benefit likely) to −2 (moderate likelihood of species gaining benefit from opportunity to roost on structures) (Supplementary Table 11). The choice of −2 as the extreme and the use of only three categories recognizes that this effect is likely to be relatively minor at a population level.

Benefit from Fish Attraction Device (FAD) effect or biofouling

Wave energy devices will probably provide shelter for small fishes and so are likely to act as a fish attraction device (FAD) (Rountree, 1990; Castro *et al.*, 2001; Løkkeborg *et al.*, 2002; Wilhelmsson *et al.*, 2006; Langhamer and Wilhelmsson, 2009). They may also represent surfaces onto which biofouling organisms will settle. Aggregations of fish under wave devices, and attached animals could attract foraging seabirds by providing locally high densities of prey. This may explain the reported attraction of some seabirds to structures such as oil platforms (Baird, 1990; Wiese *et al.*, 2001). The score for this factor classifies species into categories from 0 (no significant benefit likely) to −2 (moderate likelihood of species gaining benefit from opportunity to feed on a locally high density of fish attracted to devices) (Table S12).

Disturbance by structures

Seabird species differ in their reaction to structures. This behaviour relates in part to the general responsiveness of species to disturbance, and in part to their perception of the hazards represented by structures. This score classifies species into categories from 1 (minimal risk of disturbance) to 5 (moderate risk of disturbance) (Table S13).

Disturbance by ship traffic

Disturbance generated through deployment and/or maintenance of wave energy arrays was considered to be the same as the level of disturbance generated from tidal turbine deployment and/or maintenance. We applied the same scores and score allocations detailed in the tidal turbine assessment.

Habitat specialization

Species that use a wide range of habitats or forage over large areas may not be constrained by reduced availability of a small area around a wave device. In contrast, seabirds that have highly specialized and restricted foraging habitat in small areas where wave devices may be deployed could be prevented from using key habitat for foraging. This score classifies species into categories from 1 (use a wide range of habitats over a large area) to 5 (specialize in using a very limited and predominantly inshore habitat). The same scores are used as in the tidal turbine assessment.

Assessing an overall score for species' vulnerability to wave energy devices

It is not obvious that any of the seven effect factors considered and scored above is more important than any other, although two factors act in a positive way while all others have negative impacts. So for this overall score of species vulnerability in relation to wave energy devices we have simply summed the scores for the seven factors, and multiplied the total by the species conservation concern score.

$$Overall\ species\ score = \sum_{i=1}^{7} score_i \times conservation$$

Scores were then grouped into five categories: very high vulnerability (scores above 400), high vulnerability (scores 301–400), moderate vulnerability (scores 201–300), low vulnerability (scores 101–200), and very low vulnerability (scores 0–100) (Table 2). These categories are arbitrary, but are designed to permit species to be labelled into groups that are convenient for developers and regulators.

Key Results

The key results from this assessment are ranked species lists in Table 1 (species ranked by population vulnerability in relation to

Table 2. Species vulnerability index for wave energy device impacts on seabirds (ranked by species score).

Species	Score	Descriptor on 5-score scale
Red-throated diver	288	3: moderate vulnerability
Black-throated diver	288	3: moderate vulnerability
Great northern diver	270	3: moderate vulnerability
Razorbill	192	2: low vulnerability
Common scoter	180	2: low vulnerability
Common guillemot	176	2: low vulnerability
Black guillemot	169	2: low vulnerability
Slavonian grebe	169	2: low vulnerability
Shag	165	2: low vulnerability
Atlantic puffin	160	2: low vulnerability
Little tern	156	2: low vulnerability
Greater scaup	154	2: low vulnerability
Velvet scoter	154	2: low vulnerability
Arctic tern	153	2: low vulnerability
Common goldeneye	144	2: low vulnerability
Northern gannet	136	2: low vulnerability
Roseate tern	135	2: low vulnerability
Common eider	130	2: low vulnerability
Common tern	126	2: low vulnerability
Sandwich tern	125	2: low vulnerability
Great cormorant	110	2: low vulnerability
Manx shearwater	102	2: low vulnerability
Black-legged kittiwake	98	1: very low vulnerability
Long-tailed duck	96	1: very low vulnerability
Great skua	96	1: very low vulnerability
Great-crested grebe	91	1: very low vulnerability
Arctic skua	84	1: very low vulnerability
Little auk	81	1: very low vulnerability
Northern fulmar	80	1: very low vulnerability
Great black-backed gull	75	1: very low vulnerability
Sooty shearwater	72	1: very low vulnerability
White-tailed eagle	72	1: very low vulnerability
European storm-petrel	68	1: very low vulnerability
Common gull	65	1: very low vulnerability
Lesser black-backed gull	64	1: very low vulnerability
Leach's storm-petrel	64	1: very low vulnerability
Black-headed gull	60	1: very low vulnerability
Herring gull	48	1: very low vulnerability

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tidal turbine impacts) and in Table 2 (species ranked by population vulnerability in relation to wave energy device impacts). For tidal turbine impacts, no species fell into the very high vulnerability category (Table 1). Five species fell into the high vulnerability category: black guillemot *Cepphus grylle*, razorbill *Alca torda*, European shag, common guillemot, and great cormorant. Five species fell into the moderate vulnerability category: great northern diver *Gavia immer*, red-throated diver *G. stellata*, Atlantic puffin *Fratercula arctica*, black-throated diver *Gavia arctica*, and little auk *Alle alle*. The remaining 28 species fell into categories low or very low vulnerability, suggesting that their populations are unlikely to be affected by tidal turbine development. For wave energy devices, no species fell into the categories of very high or high vulnerability (Table 2). Three species (red-throated diver, black-throated diver and great northern diver) fell into the category of moderate vulnerability. A total of 19 species fell into the category of low vulnerability and 16 into very low vulnerability.

Discussion

With rapid development of wet renewables in UK and Scottish waters, there is an urgent need to assess any effects these developments may have on the environment. At present, however, there is a lack of information on the impacts that tidal turbines and wave energy devices may have on seabird populations, which hinders the consenting process for the early developments. Here we have presented an index to assess the vulnerability of seabird populations based on appraisal of detailed scientific literature on aspects of seabird ecology and conservation. The ten species that were identified as moderately or very highly vulnerable to tidal stream turbines should be a focus for monitoring and for research into effects of tidal turbines on seabirds in Scottish waters. The results for wave energy devices suggest that they represent a lesser hazard to seabirds than do tidal turbines, and both types of wet renewables appear to represent a lesser hazard than offshore wind, though to a different set of species (offshore wind collision hazard seems likely to be especially high for large gulls and northern gannets (Garthe and Hüppop, 2004; Furness and Wade, 2012), species that are predicted to be at low risk from tidal turbines and wave energy devices). From this study it is possible to highlight species such as black guillemot and divers as most likely to be affected by "wet renewable" developments. Although in this paper the focus is on seabird species, in some cases different populations of a species are present at different times of year. Breeding numbers of seabirds in Scotland are well known, but numbers and populations present outside the breeding season may be less certain. Nevertheless, in theory at least, hazard can be assessed separately for breeding populations and for migrants visiting Scottish waters.

During this assessment there were limitations due to the available literature. One such limitation was that data in the BirdLife online database were not clearly attributed to sources and appeared to exaggerate diving depths, especially mean dive depth, compared to most published studies. We also found that some studies of foraging range report mean foraging distance whilst others report the maximum. Where possible, we gave higher importance to reported mean foraging distances than to maximum values. This is partly because maximum values tend to increase with sample size whereas means are not biased in this way. However, maximum foraging ranges of seabird species have been reported in the literature for various species, locations and time periods, more often than mean values have been reported (Thaxter *et al.*, 2012). It must be remembered that mean and maximum ranges can vary considerably according to environmental conditions. In particular, foraging ranges tend to be short in years with good food supply, but much longer when food is scarce (e.g. Hamer *et al.*, 1993). Ranges can also be much greater when seabirds are nesting in very large colonies (Hamer *et al.*, 2001). Also, while it is clear that breeding seabirds have a well-defined foraging range from their colony, wintering seabirds and sea ducks have less clearly defined foraging ranges from roosting areas to feeding grounds.

While we have taken the approach of defining risk factors and scoring seabird species according to chosen risk factors, risk levels are likely to change in relation to a wide range of additional variables. Risk will vary seasonally, in relation to weather conditions, age, experience and breeding status of individual birds, and probably for a variety of other reasons, some of which are understood by ecologists and some of which are probably not. Scores allocated are averages for each species, but within-species variation should not be ignored. For example, seabird at-sea activity levels tend to be considerably higher in summer when birds are breeding than they are during winter (Markones *et al.*, 2010). High energetic demands of breeding and increased travelling activity and foraging will potentially increase risk of interactions with tidal turbines and wave energy devices. On the other hand, while breeding, seabirds are central-place foragers, tied to areas in the vicinity of their breeding colonies. This could result in rates of interaction with tidal turbines and wave energy devices varying seasonally depending on whether these devices are close to colonies (in which case interactions may predominantly occur during the breeding season) or in areas distant from colonies (in which case interactions may occur predominantly during seabird migration periods, or in winter). At-sea distributions and behaviours of breeding seabirds are becoming quite well known, meaning we have a detailed knowledge of the distributions and sizes of seabird colonies, and deployment of data loggers is starting to provide information on the foraging behaviour of breeding adults. The distributions and behaviour of non-breeding (immature) seabirds are however poorly known, and the youngest seabirds may be most vulnerable to hazards because they lack experience. Juvenile European shags occasionally drown by becoming trapped in lobster pots; adult shags hardly ever die in this way (Galbraith *et al.*, 1981). A similar age-related variation in risk applies in the context of drowning in fishing nets (Murray *et al.*, 1994). These differences may relate in part to the distributions of these age classes of birds, but is more likely a consequence of the inexperience of juvenile birds. Similar differences in relation to hazards posed by tidal turbines and wave energy devices are likely to affect juvenile seabirds in particular.

Suitability of a Vulnerability Index approach

Desholm (2009) identified two weaknesses in the approach we have used. The first is the fact that different factors are measured on different scales. It does have to be acknowledged that it is impossible to add up a mortality rate and a percentage of biogeographic population in a common currency. It is because factors are measured on different scales that the index classifies species into five levels (scores) for each factor, and then combines scores. Because we are trying to compare "apples and oranges" these comparisons and scores are using a small number of categories and essentially ranking species in order to identify those with extreme characteristics on each factor. This clearly does need to

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be done with caution given that different factors are measured in different units. The assumption being made is that all of these factors influence risk, and therefore that a species scoring high on all factors will be at highest risk and a species scoring low on all factors will be at lowest risk. It would be unwise to assume that species A, scoring high on the first factor but low on a second, had exactly the same risk as species B, scoring low on the first factor but high on the second. Scores for each species should be seen as an indication of the likely overall picture, and not as an exact quantitative measure on a scale. The issue of correlations between factors was also raised by Desholm (2009) who pointed out that there were correlations between nearly one-third of pairwise comparisons of factors in their study of seabirds and offshore wind farm risk. Similar correlations may exist between some factors scored in this study. For example, it would be surprising if the percentage of benthic food in the diet did not correlate with diving depth, simply because seabirds that cannot dive have little or no access to benthic prey. Such correlations need not be a problem. For example, in using biometrics to discriminate between males and females of a particular seabird species with limited sexual dimorphism, it is usually more successful to combine several linear measurements (head, bill depth, tarsus length etc.) rather than use a single metric. This would be particularly true if each measurement differed slightly between the sexes, but with considerable individual variation and measurement error. Then combining several measures reduces the effects of measurement error. For our scoring, there is considerable uncertainty in the allocation of scores for individual species for each factor, so combining several factors to give an overall index may be better than relying on a measurement for a single factor, especially if that measurement is inaccurate.

Limitations of data

A draft version of the factors and species-scores was sent to seabird experts for comment. Most reviewers suggested no change to the factors used, and no change to most scores. Most of the reviewers felt that between one and four out of the 456 scores for risk factors might be altered slightly, but were content with over 99% of the allocated scores. Therefore the scores presented in this paper have broad agreement from a diverse group of relevant experts in seabird ecology and conservation, selected for their particular interest in wet renewables. None of the reviewers questioned scoring in the assessment of conservation importance of species. That reflects the high quality of data on Scottish seabird population sizes and conservation status, and the strongly quantitative and evidence-based approach used in scoring. Nevertheless, within the scoring factor "Birds Directive status" we felt that it would be useful also to consider the proportion of the Scottish population of each species that is protected within the SPA network, a statistic of direct relevance to consenting risk. Unfortunately, such data are not readily available for Scotland. In the SPA review (Stroud *et al.*, 2001) such a statistic is considered at a Great British and biogeographic level, but those data are not calculated at a Scottish level, and are now also somewhat out of date. This score could therefore be improved if a database of current numbers of seabirds in SPAs in Scotland could be established, and kept up to date.

The evidence base for scoring species risk factors is much less secure, and the relatively consistent consensus view of seabird ecologists is based on detailed knowledge of seabird ecology, but on a lack of evidence from deployed tidal and wave energy devices. It would not be too surprising, therefore, if at least a few of these consensus views turn out in future to be inaccurate. It is, therefore, important to treat the results of this project with caution, and it should be an objective to continuously update the assessments made in this report with data that will eventually become available as the wet renewables industry develops. This view is closely in agreement with the conclusions reached by ICES (2010) in their advice on wet renewables environmental impacts given to OSPAR.

Applications of findings to impact assessments

Despite the relatively high uncertainty over species-specific responses to tidal devices and to wave energy devices, there is a very strong consensus in the published literature that these wet renewables technologies are unlikely to represent as great a hazard to seabirds as posed by offshore wind farms (Awatea, 2008; Inger *et al.*, 2009; Aquatera, 2010; ICES, 2010; Grecian, 2010; Langton *et al.*, 2011; Witt *et al.*, 2012). The relatively low risk to seabirds from wet renewables also contrasts strongly with the high impact on seabird populations resulting from depletion of fish stocks by global fisheries (Karpouzi *et al.*, 2007; Cury *et al.*, 2011), and potentially from climate change (Mackas *et al.*, 2007; Beaugrand *et al.*, 2009; Quillfeldt *et al.*, 2010). In this context of changing numbers of seabirds as a result of changes in fish stocks and fisheries, zooplankton populations, climate change and other pressures, the influences of wet renewables on seabird populations are inevitably going to be very difficult to measure. This review represents a small step towards identifying the seabird species in Scottish waters that should be the main focus of concern for the regulator (Marine Scotland) the statutory advisor (Scottish Natural Heritage), developers and their consultants, in relation to deployments of tidal turbines and wave energy devices. We strongly recommend that this assessment be updated as more studies provide relevant information, and that monitoring and research particularly targets the species in the top parts of the lists in Tables 1 and 2. This review does not inform on site-specific issues, and in the context of EIA should be seen as assisting with scoping and with cumulative impacts assessment. Clearly each individual development will need to consider site-specific issues and the local community of seabirds in that area.

Supplementary data

Supplementary material is available at the *ICESJMS* online version of the manuscript.

Acknowledgements

We are very grateful to a large number of seabird ecologists who provided advice on the factors and the species scores for factors, and suggested useful literature to review:Prof. Stuart Bearhop (University of Exeter), Dr John Calladine (BTO Scotland), Dr Keith Hamer (University of Leeds), Prof. Mike Harris (CEH Edinburgh), Martin Heubeck (SOTEAG, Shetland), Dr Liz Humphreys (BTO Scotland), Dr Beth Scott (University of Aberdeen), Dr Chris Thaxter (BTO), Prof. Paul Thompson (University of Aberdeen), Dr Steve Votier (University of Plymouth), Prof. Sarah Wanless (CEH Edinburgh), Dr Steve Newton (Birdwatch Ireland), and Dr James Grecian (University of Glasgow). We thank Dr Roel May and an anonymous reviewer for helpful comments on an earlier draft.

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Funding

This study was jointly funded by Scottish Natural Heritage and MacArthur Green Ltd.

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Journal of Environmental Management 119 (2013) 56–66

Contents lists available at SciVerse ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Assessing vulnerability of marine bird populations to offshore wind farms

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ARTICLE INFO

Article history:
Received 11 June 2012
Received in revised form
17 January 2013
Accepted 27 January 2013
Available online 27 February 2013

Keywords:
Marine birds
Offshore wind farms
Adverse effects
Renewable energy
Population vulnerability
Collision mortality
Displacement index

ABSTRACT

Offshore wind farms may affect bird populations through collision mortality and displacement. Given the pressures to develop offshore wind farms, there is an urgent need to assess population-level impacts on protected marine birds. Here we refine an approach to assess aspects of their ecology that influence population vulnerability to wind farm impacts, also taking into account the conservation importance of each species. Flight height appears to be a key factor influencing collision mortality risk but improved data on flight heights of marine birds are needed. Collision index calculations identify populations of gulls, white-tailed eagles, northern gannets and skuas as of particularly high concern in Scottish waters. Displacement index calculations identify populations of divers and common scoters as most vulnerable to population-level impacts of displacement, but these are likely to be less evident than impacts of collision mortality. The collision and displacement indices developed here for Scottish marine bird populations could be applied to populations elsewhere, and this approach will help in identifying likely impacts of future offshore wind farms on marine birds and prioritising monitoring programmes, at least until data on macro-avoidance rates become available.

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1. Introduction

To meet targets for a reduction in greenhouse gas emissions, many Governments, especially within the European Union but also in North America, are encouraging the development of renewable energy generation such as offshore wind farms. There is the potential however for such developments to have adverse effects on the environment and particularly on marine birds through collision mortality, disturbance, or foraging habitat loss. Collision is more likely to occur if seabirds fail to avoid wind farms, whereas displacement from foraging habitat is more likely to occur if seabirds do avoid wind farms. Thus displacement may be a consequence of avoidance behaviour. Disturbance, caused by structures or by human activity associated with wind farms, may also cause displacement if birds move away from wind farms due to associated disturbance. Here we define displacement as a reduced number of birds occurring within or immediately adjacent to offshore wind farms, and we define disturbance as birds spending extra time and/or energy to avoid structures or human activity associated with offshore wind farms. Birds may show two kinds of avoidance behaviour at offshore wind farms, often termed 'macro-avoidance' and 'micro-avoidance'. Macro-avoidance occurs when birds alter their flight path to keep clear of the whole wind farm (Desholm and Kahlert, 2005), whereas micro-avoidance occurs when birds enter the wind farm but take evasive action to avoid individual turbines (Band, 2011; Cook et al., 2012). If species-specific rates of macro-avoidance and micro-avoidance were known, it would be easy to assess vulnerability of different species' populations. However, data on macro-avoidance rates are very limited and inconsistent, while data on micro-avoidance by marine birds at offshore wind farms are extremely scarce (Cook et al., 2012). For example, it has been stated that northern gannets *Morus bassanus* show higher macro-avoidance (64% Krijgsveld et al., 2011) than shown by gulls. However, while Krijgsveld et al. (2011) report an 18% macro-avoidance rate by gulls, Petersen et al. (2006) report a 76% macro-avoidance rate by gulls, higher than that reported for gannets. Until avoidance rates are better quantified we have assumed that avoidance rates are not significantly different among taxa, since existing data fail to contradict that simplification.

The increase in electricity generation from offshore wind creates an urgent need to assess effects on marine birds, and potential impacts on their populations. The most detailed studies of the effects of wind farms on marine birds have been at the Nysted and Horns Rev offshore wind farms in Danish waters, where

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0301-4797/$ – see front matter © 2013 Elsevier Ltd. All rights reserved.
http://dx.doi.org/10.1016/j.jenvman.2013.01.025

research has demonstrated a low risk of collision, though for a limited set of species, notably common eiders *Somateria mollissima* (Desholm and Kahlert, 2005). Fox et al. (2006a,b) however emphasised the high variability in behavioural responses and thus vulnerability amongst marine bird species to offshore wind farms. Due to this variability in behavioural responses, it is vital to identify species' populations most likely to be at risk. In Europe, many marine birds breed within Special Protection Areas (SPAs) and so are protected by European law, in particular the Birds Directive (2009/147/EC). This legal protection has the potential to constrain development of offshore wind farms so there is a strong incentive to improve understanding of the risks and to develop better directed monitoring. Offshore wind in the United States similarly needs to consider possible impacts on marine bird populations in the context of protection afforded by legislation such as the Endangered Species Act.

A traditional method in conservation for setting priorities is to develop lists of at-risk species (Gardali et al., 2012) but a means by which to identify which species are most vulnerable is therefore necessary. A well-established approach has been to use indices of sensitivity or population vulnerability to particular hazards (Furness and Tasker, 2000; Gardali et al., 2012; Garthe and Hüppop, 2004; Sonntag et al., 2012). Garthe and Hüppop (2004) developed an index of marine bird population vulnerability to offshore wind farms, based on scores of conservation importance of different species' populations and perceived behaviour-related risks of collision and displacement, combined into a single index. They applied this index to marine birds in the southern North Sea. Here we develop and refine the approach advocated by Garthe and Hüppop (2004), extending the species list and incorporating new data from recent research on the flight behaviour of marine birds. An important innovation in our approach is the separation of assessments of risk due to collision and risk due to avoidance/displacement. This separation is particularly useful where the species most at risk of collision differ from those most at risk of displacement, as is the case here. Our assessment focuses on the example of marine birds in Scotland, but the approach could equally be applied elsewhere.

2. Methods

We followed the approach of Garthe and Hüppop (2004) scoring broadly similar factors in the assessment of vulnerability of marine bird populations to offshore wind farms. Where we used the same factors as Garthe and Hüppop (2004), scores allocated by them were reviewed individually. In many cases those scores appeared appropriate but where more recent data were available, scores were modified. As far as possible, the scoring criteria for each factor and the respective provisional scores for each marine bird species were evidence-based, with data taken from the reviewed literature. Scoring criteria and provisional scores were subsequently circulated to a group of appropriate experts for review, in order to ensure consensus support for the final criteria and scorings from a range of stakeholders; including seabird ecologists and conservationists (see details in Acknowledgements). The list of marine bird species assessed includes true seabirds, wintering sea ducks and grebes, and white-tailed eagle *Haliaeetus albicilla*. This last species was included because it feeds extensively over the sea, and there are detailed studies of its collisions at coastal terrestrial wind farms. Both breeding and nonbreeding (wintering or passage) marine birds were scored. A few marine bird species of conservation importance that occur in Scotland were omitted, on the basis that they occur very infrequently, if ever, in Scottish marine areas identified as suitable for offshore wind farms (Marine Scotland, 2011). These included red-breasted merganser *Mergus serrator* and little gull *Hydrocoloeus minutus*.

The method considers factors representing conservation importance and factors representing aspects of sensitivity to wind farm impacts. In some regions, a single factor may be adequate to rank and assign scores for conservation importance, but often there are multiple rankings available, based on slightly different criteria. In such cases an average of several rankings may provide a better scoring system than relying on any one individual factor. In our case study, four factors represent the conservation importance of the species in a Scottish context; status in relation to the Birds Directive, percentage of the biogeographic population that occurs in Scotland, adult survival rate, and UK threat status. Status in relation to the Birds Directive gives emphasis to species considered by the European Commission to be in particular need of conservation protection through European legislation. However, some of these species show increasing populations. UK threat status emphasises species showing population decline, but may give a low score to populations of high endemism if these are not in decline. Percentage of biogeographic population in the focal area emphasises endemism but does not take account of population trend. Considering adult survival rate recognises that added mortality of adult birds with high natural survival rates (and corresponding low productivity) has a greater impact on population dynamics than added mortality to populations with low survival rates. Combining these factors appears to provide a better ranking of conservation importance than achieved by any component factor (for details of these factors and scores see Furness et al., 2012).

Six factors represent aspects of species' behaviour that contribute to their potential vulnerability to wind farms (flight altitude, flight manoeuvrability, percentage of time flying, nocturnal flight activity, disturbance by wind farm structures, ship and helicopter traffic, and habitat specialisation). Scores were assigned on a scale of 1–5 for almost all factors, where 5 is a strong anticipated negative impact. It was felt more appropriate to score the factor assessing flight altitude as a percentage value of a species' flight altitude spent at turbine blade height, rather than on a five point scale. Individual factor scores were combined to give a total for each species that ranked species according to their vulnerability to offshore wind farm developments separately in terms of collision risk and habitat loss through avoidance. The factors assessed and calculations used to combine factor scores are outlined below.

2.1. Flight altitude

This factor is widely considered to be of overwhelming importance in determining the risk of collision of marine birds with offshore wind turbines (Band, 2011; Cook et al., 2012). Birds that only fly very low over the water will be below the area swept by turbine blades, whereas marine birds that habitually fly at greater heights may experience a greater risk of collision with blades if flight heights coincide with rotor swept areas of a wind farm. Flight altitude scores were initially taken from Garthe and Hüppop (2004), and Cook et al. (2012) but we present flight altitude as percentages of birds flying at blade height (Table 1), as opposed to collapsing such data into a 5 point scale (Garthe and Hüppop, 2004). When there was a lack of consensus between Garthe and Hüppop (2004) and Cook et al. (2012), we estimated a flight altitude that was consistent with these two reviews and was consistent with data from closely related species, or with other published data such as Rothery et al. (2009), and Krijgsveld et al. (2011), trying to account for very different sample sizes in different studies, different methods of measurement, and trying to retain consistency across related species. Flight altitude includes birds in all activities (such as foraging, commuting, migrating). It may vary seasonally, but there are too few data available at present to test this possibility.

Table 1
Estimated percent of flight at turbine blade height (ca. 20–150 m asl).

Species	Reference	Estimated % at blade height
Greater scaup	Dirksen et al., 1998 (10%, no sample size quoted, radar at night).	3
	Assumed similar to other ducks.	
Common eider	Krüger and Garthe, 2001 (2%, $n = 14{,}405$, land-based obs)	3
	Day et al., 2003 (0%, $n = 17$, land-based obs)	
	Garthe and Hüppop, 2004 (score of 1, median height <5 m).	
	Leopold et al., 2004 (1.3%, $n = 235$, ship-based obs)	
	Petterson, 2005 (Utgrunden, 0%, $n = 57$, radar)	
	Petterson, 2005 (Yttre Stengrund, 20%, $n = 2044$, radar)	
	Sadoti et al., 2005a (0%, $n = 84$, boat-based obs)	
	Petersen et al., 2006 (16%, $n = 193$ flocks mostly eiders, radar)	
	Larsen and Guillemette, 2007 (2%, $n = 1277$ platform-based obs)	
	Rothery et al., 2009 (0%, $n = 1282$, land-based obs)	
	Paton et al., 2010 (0.2%, $n = 24{,}195$ land-based obs)	
	Paton et al., 2010 (0%, $n = 294$ ship-based obs)	
Long-tailed duck	Day et al., 2003 (0%, $n = 108$, land-based obs)	3
	Paton et al., 2010 (0%, $n = 280$ land- or ship-based obs)	
	Cook et al., 2012 (0%, $n = 114$ ship-based obs)	
	Our estimate assumes this species is similar to other ducks.	
Common scoter	Krüger and Garthe, 2001 (3%, $n = 6754$, land-based obs)	3
	Garthe and Hüppop, 2004 (score of 1, median height <5 m)	
	Leopold et al., 2004 (0.8%, $n = 2258$, ship-based obs)	
	Sadoti et al., 2005a (43%, $n = 96$, boat-based obs)	
	Npower renewables, 2006 (0.2%, $n = 1274$, boat-based obs)	
	Rothery et al., 2009 (0%, $n = 341$, land-based obs)	
	Paton et al., 2010 (0.2%, $n = 4756$ land-based obs)	
	Paton et al., 2010 (0%, $n = 277$ ship-based obs)	
	Cook et al., 2012 (1%, 95% ci <0.1–17%, $n = 30{,}847$ ship-based obs)	
	Garthe et al., 2012b (24%, no sample size, boat-based obs)	
Velvet scoter	Day et al., 2003 (0%, $n = 5$, land-based obs)	3
	Garthe and Hüppop, 2004 (score of 1, median height <5 m);	
	Sadoti et al., 2005a (3%, $n = 88$, boat-based obs)	
	Paton et al., 2010 (7.0%, $n = 2973$ land-based obs)	
	Paton et al., 2010 (0%, $n = 161$ ship-based obs)	
	Cook et al., 2012 (0%, $n = 20$ ship-based obs)	
Common goldeneye	Dirksen et al., 1998 (5%, no sample size quoted, radar at night)	3
	Paton et al., 2010 (11.3%, $n = 336$ land-based obs)	
	Assumed similar to other ducks	
Red-throated diver	Krüger and Garthe, 2001 (0%, $n = 247$, land-based obs)	5
	Garthe and Hüppop, 2004 (score of 2, median height 5–10 m)	
	Leopold et al., 2004 (8.5%, $n = 284$, ship-based obs, may include a few black-throated divers)	
	Krijgsveld et al., 2005 (24%, $n = 103$, radar, may include a few black-throated divers)	
	Sadoti et al., 2005a (3%, $n = 28$, boat-based obs)	
	Npower renewables, 2006 (0%, $n = 13$, boat-based obs)	
	Paton et al., 2010 (7.1%, $n = 1226$ land-based obs)	
	Paton et al., 2010 (28.3%, $n = 106$ ship-based obs)	
	Krijgsveld et al., 2011 (5%, no sample size, radar)	
	Cook et al., 2012 (2%; 95% ci <0.1–22%, $n = 9715$ ship-based obs)	
Black-throated diver	Garthe and Hüppop, 2004 (score of 2, median height 5–10 m);	5
	Cook et al., 2012 (0.1%; 95% ci <0.1–30%, $n = 126$ ship-based obs)	
	Score is a compromise between conflicting data in these two studies and a	
	view from reviewers that all divers should be same.	
Great northern diver	Sadoti et al., 2005a (4%, $n = 27$, boat-based obs)	5
	Paton et al., 2010 (22.8%, $n = 2762$ landb-ased obs)	
	Paton et al., 2010 (5.8%, $n = 292$ ship-based obs)	
	Cook et al., 2012 (0%, $n = 14$ ship-based obs);	
	Kerlinger, 1982 (migration can occur at 1000 to 3000 m heights)	
Great-crested grebe	Garthe and Hüppop, 2004 (score of 2, median height 5–10 m);	4
	Leopold et al., 2004 (0%, $n = 32$, ship-based obs)	
	Cook et al., 2012 (0%, $n = 82$ ship-based obs)	
Slavonian grebe	Paton et al., 2010 (23.5%, $n = 85$ land-based obs)	4
	Assumed similar to great-crested grebe	
Northern fulmar	Garthe and Hüppop, 2004 (score of 1, median height <5 m);	1
	Leopold et al., 2004 (0%, $n = 178$, ship-based obs)	
	Krijgsveld et al., 2005 (0%, $n = 10$, radar)	
	Cook et al., 2012 (0.2%; 95% ci <0.1–22%, $n = 29{,}168$ ship-based obs).	
Sooty shearwater	Paton et al., 2010 (0%, $n = 5$ land-based obs)	0
	Paton et al., 2010 (0%, $n = 16$ ship-based obs)	
	Cook et al., 2012 (0%, $n = 2$ ship-based obs)	
	Assumed similar to Manx shearwater	
Manx shearwater	Cook et al., 2012 (0.04%; 95% ci <0.01–10%, $n = 6957$ ship-based obs)	0
European storm-petrel	Cook et al., 2012 (2% (range 0–2.5%), $n = 52$ ship-based obs)	2
Leach's storm-petrel	Assumed similar to European storm-petrel	2

Table 1 (*continued*)

Species	Reference	Estimated % at blade height
Northern gannet	Garthe and Hüppop, 2004 (score of 3, median height 10–20 m but few above 50 m); Leopold et al., 2004 (13%, $n = 803$, ship-based obs) Krijgsveld et al., 2005 (44%, $n = 143$, radar, excludes birds following fishing boats) Sadoti et al., 2005a (20%, $n = 85$, boat-based obs) Npower renewables, 2006 (16%, $n = 50$, boat-based obs) Rothery et al., 2009 (13%, $n = 414$, land-based obs) Paton et al., 2010 (10%, $n = 8560$ land-based obs) Paton et al., 2010 (6.9%, $n = 1278$ ship-based obs) Krijgsveld et al., 2011 (30%, no sample size, radar) Cook et al., 2012 (estimated 9.6%; 95% ci <0.1–20%, $n = 44{,}851$); Garthe et al., 2012b (9%, no sample size, boat-based obs)	16
Great cormorant	Garthe and Hüppop, 2004 (score of 1, median height <5 m). Leopold et al., 2004 (7.5%, $n = 929$, ship-based obs) Rothery et al., 2009 (13%, $n = 352$, land-based obs) Npower renewables, 2006 (7%, $n = 113$, boat-based obs) Paton et al., 2010 (7.3%, $n = 2014$ land-based obs) Paton et al., 2010 (0%, $n = 15$ ship-based obs) Krijgsveld et al., 2011 (28%, no sample size, radar)	4
Shag	Npower renewables, 2006 (0%, $n = 5$, boat-based obs) Cook et al., 2012 (12.4% with model fit relatively poor; 95% ci 1.9–60%, $n = 233$ ship-based obs).	5
White-tailed eagle	Nygård et al., 2010 (24% of flights in study wind farm were at blade height (hub height = 70 m, blade radius = 38–41 m)	24
Arctic skua	Garthe and Hüppop, 2004 (score of 3, median height 10–20 m but few above 50 m); Npower renewables, 2006 (0%, $n = 2$, boat-based obs) Paton et al., 2010 (21.1%, $n = 19$ land-based obs) Paton et al., 2010 (0%, $n = 1$ ship-based obs) Cook et al., 2012 (3.8%; 95% ci <0.1–16%, $n = 331$ ship-based obs) Observations of Arctic skuas from seawatching and from birds foraging at sea in breeding areas suggest higher flying than Cook et al., 2012 model, as does G&H 2004 score. Our estimate follows Garthe and Hüppop, 2004 and suggestions from reviewers more closely than the data in Cook et al. (2012).	10
Great skua	Garthe and Hüppop, 2004 (score of 3, median height 10–20 m but few above 50 m); Cook et al., 2012 (4.3%; 95% ci 1.2–28%, $n = 1202$ ship-based obs) Observations of great skuas from seawatching, from birds foraging at sea in breeding areas, and from deployment of GPS data loggers by H. Wade, C. Thaxter and colleagues) suggest higher flying than Cook et al., 2012 model, as does G&H 2004 score. Our estimate follows Garthe and Hüppop, 2004, unpublished GPS logger data, and suggestions from reviewers more closely than data in Cook et al. (2012).	10
Black-headed gull	Bergh et al., 2002 (4%, $n = 82$, land-based obs at Slufterdam) Bergh et al., 2002 (78%, $n = 41$, land-based obs at Slag Dobbelsteen) Scored 5 by Garthe and Hüppop (2004) (median height 10–20 m with 10% above 100 m). Krijgsveld et al., 2005 (25%, $n = 334$, radar, excludes birds following fishing boats) Rothery et al., 2009 (4%, $n = 978$, land-based obs) Krijgsveld et al., 2011 (30%, no sample size, radar) Cook et al., 2012 (7.9%; 95% ci 0.4–50%, $n = 4490$ ship-based obs). Estimate also considers values for related gull species, and radar studies reporting a higher flight height than obtained from boat-based windfarm surveys (Cook et al., 2012)	18
Common gull	Bergh et al., 2002 (58%, $n = 120$, land-based obs at Slag Dobbelsteen) Garthe and Hüppop, 2004 (score of 3, median height 10–20 m but few above 50 m); Krijgsveld et al., 2005 (48%, $n = 1517$, radar, excludes birds following fishing boats) Npower renewables, 2006 (18%, $n = 102$, boat-based obs) Cook et al., 2012 (22.9%; 95% ci 8.5–47%, $n = 10{,}190$ ship-based obs); Garthe et al., 2012b (11%, no sample size, boat-based obs)	23
Lesser black-backed gull	Bergh et al., 2002 (34%, $n = 92$, land-based obs at Slufterdam) Bergh et al., 2002 (90%, $n = 1828$, land-based obs at Slag Dobbelsteen) Garthe and Hüppop, 2004 (score of 4, median height 10–20 m with 10% above 50 m); Krijgsveld et al., 2005 (55%, $n = 2470$, radar, excludes birds following fishing boats) Npower renewables, 2006 (35%, $n = 66$, boat-based obs) Krijgsveld et al., 2011 (60%, no sample size, radar) Cook et al., 2012 (estimated 25.2%; 95% ci 7.8–52%, $n = 35{,}142$ ship-based obs); Garthe et al., 2012b (29%, no sample size, boat-based obs)	30
Herring gull	Bergh et al., 2002 (33%, $n = 71$, land-based obs at Slufterdam) Bergh et al., 2002 (84%, $n = 7327$, land-based obs at Slag Dobbelsteen) Garthe and Hüppop, 2004 (score of 4, median height 10–20 m with 10% above 50 m); Krijgsveld et al., 2005 (50%, $n = 2223$, radar, excludes birds following fishing boats) Sadoti et al., 2005a (22%, $n = 63$, boat-based obs) Sadoti et al., 2005b (5%, $n = 63$, boat-based obs) Npower renewables, 2006 (37%, $n = 142$, boat-based obs) Rothery et al., 2009 (33%, $n = 1408$, land-based obs) Paton et al., 2010 (15.0%, $n = 51{,}036$ land-based obs) Paton et al., 2010 (13.8%, $n = 1652$ ship-based obs) Krijgsveld et al., 2011 (55%, no sample size, radar) Cook et al., 2012 (estimated 28.4%; 95% ci 16–48%, $n = 25{,}252$ ship-based obs); Garthe et al., 2012b (40%, no sample size, boat-based obs)	35

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Table 1 (*continued*)

Species	Reference	Estimated % at blade height
Great black-backed gull	Garthe and Hüppop, 2004 (Score 3, median height 10–20 m but few above 50 m)	35
	Krijgsveld et al., 2005 (57%, $n = 143$, radar, excludes birds following fishing boats)	
	Sadoti et al., 2005a (28%, $n = 163$, boat-based obs)	
	Sadoti et al., 2005b (6%, $n = 35$, boat-based obs)	
	Npower renewables, 2006 (23%, $n = 22$, boat-based obs)	
	Rothery et al., 2009 (44%, $n = 564$, land-based obs)	
	Paton et al., 2010 (8.5%, $n = 8610$ land-based obs)	
	Paton et al., 2010 (8.8%, $n = 1001$ ship-based obs)	
	Krijgsveld et al., 2011 (60%, no sample size, radar)	
	Cook et al., 2012 (33.1%; 95% ci 18–57%, $n = 8911$ ship-based obs)	
	Garthe et al., 2012b (56%, no sample size, boat-based obs)	
Black-legged kittiwake	Day et al., 2003 (10%, $n = 36$, land-based obs)	16
	Garthe and Hüppop, 2004 (Score 2, median 5–10 m)	
	Leopold et al., 2004 (9%, $n = 637$, ship-based obs)	
	Chamberlain et al., 2005 (4%, $n = 2036$, ship-based obs)	
	Krijgsveld et al., 2005 (38%, $n = 2459$, radar, excludes birds following fishing boats)	
	Npower renewables, 2006 (8%, $n = 79$, boat-based obs)	
	Rothery et al., 2009 (11%, 1350, land-based obs)	
	Paton et al., 2010 (0%, $n = 56$ land-based obs)	
	Paton et al., 2010 (10.9%, $n = 55$ ship-based obs)	
	Krijgsveld et al., 2011 (45%, no sample size, radar)	
	Cook et al., 2012 (15.7%; 95% ci 8–24%, $n = 62{,}975$ ship-based obs);	
	Garthe et al., 2012b (18%, no sample size, boat-based obs)	
Little tern	Everaert and Stienen, 2007 (23%, $n = 2124$, land-based obs at a coastal wind farm)	7
	Assumed similar to other terns	
Sandwich tern	Krüger and Garthe, 2001 (0%, $n = 959$, land-based obs)	7
	Garthe and Hüppop, 2004 (Score 3, median height 10–20 m but few above 50 m)	
	Leopold et al., 2004 (4.5%, $n = 419$, ship-based obs)	
	Krijgsveld et al., 2005 (56%, $n = 236$, radar, may include some other tern species)	
	Npower renewables, 2006 (5%, $n = 20$, boat-based obs)	
	Everaert and Stienen, 2007 (10%, $n = 27{,}571$, land-based obs at a coastal wind farm)	
	Rothery et al., 2009 (3%, $n = 2137$, land-based obs)	
	Perrow et al., 2011a (48%, no sample size, boat-based obs)	
	Krijgsveld et al., 2011 (50%, no sample size, radar)	
	Cook et al., 2012 (3.6%; 95% ci 0.7–35%, $n = 33{,}392$ ship-based obs);	
	Garthe et al., 2012b (8%, no sample size, boat-based obs)	
Common tern	Bergh et al., 2002 (8%, $n = 113$, land-based obs at Slufterdam)	7
	Krüger and Garthe, 2001 (0%, $n = 271$, land-based obs)	
	Garthe and Hüppop, 2004 (score of 2, median 5–10 m)	
	Leopold et al., 2004 (0.6%, $n = 1011$, ship-based obs, includes some Arctic terns)	
	Sadoti et al., 2005a (6.2%, $n = 130$, boat-based obs 2003; 5.7%, $n = 163$, boat-based obs 2004, may include some roseate terns)	
	Sadoti et al., 2005b (4%, $n = 29$, boat-based obs)	
	Npower renewables, 2006 (41%, $n = 79$, boat-based obs)	
	Everaert and Stienen, 2007 (13%, $n = 14{,}491$, land-based obs at a coastal wind farm)	
	Hatch and Brault, 2007 (6.2%, $n = 3154$ (including some roseate terns)	
	Paton et al., 2010 (3.8%, $n = 3644$ land-based obs)	
	Paton et al., 2010 (11.5%, $n = 61$ ship-based obs)	
	Cook et al., 2012 (12.7%; 95% ci 6–19%, $n = 19{,}332$ ship-based obs);	
	Garthe et al., 2012b (0%, no sample size, may include a few Arctic terns)	
Roseate tern	Paton et al., 2010 (0%, $n = 125$ land-based obs)	5
	Paton et al., 2010 (12.5%, $n = 8$ ship-based obs)	
	Assumed similar to other terns	
Arctic tern	Alerstam and Gudmundsson, 1999 (migrate mainly above collision risk height from 200 to 2000 m, radar)	5
	Gudmundsson et al., 2002 (migration mainly nocturnal, mean height 800 m, radar)	
	Garthe and Hüppop, 2004 (score of 1, median <5 m);	
	Cook et al., 2012 (2.8%; 95% ci <0.1–23%, $n = 2571$ ship-based obs)	
Common guillemot	Day et al., 2003 (0%, $n = 172$, land-based obs, may include a few Brunnich's guillemots)	1
	Garthe and Hüppop, 2004 (score of 1, median <5 m);	
	Leopold et al., 2004 (0.6%, $n = 316$, ship-based obs)	
	Krijgsveld et al., 2005 (3%, $n = 440$, radar, may include some razorbills and puffins)	
	Npower renewables, 2006 (5%, $n = 56$, boat-based obs)	
	Paton et al., 2010 (0%, $n = 131$ ship-based obs)	
	Cook et al., 2012 (0.01%; 95% ci <0.01–3.9%, $n = 36{,}260$ ship-based obs)	
	In breeding areas, often commutes to/from colony at heights relating to altitude of nest sites which can be up to 200 m above sea level	

Table 1 (*continued*)

Species	Reference	Estimated % at blade height
Razorbill	Garthe and Hüppop, 2004 (score of 1, median <5 m); Leopold et al., 2004 (0%, $n = 33$, ship-based obs) Sadoti et al., 2005a (0%, $n = 3$, boat-based obs) Npower renewables, 2006 (0%, $n = 17$, boat-based obs) Paton et al., 2010 (0%, $n = 135$ land-based obs) Paton et al., 2010 (0%, $n = 93$ ship-based obs) Cook et al., 2012 (0.4%; 95% ci <0.1–25%, $n = 13{,}171$ ship-based obs) In breeding areas, often commutes to/from colony at heights relating to altitude of nest sites which can be up to 200 m above sea level	1
Black guillemot	Paton et al., 2010 (0%, $n = 1$ land-based obs) Assume similar to closely related species	1
Little auk	Paton et al., 2010 (0%, $n = 125$ ship-based obs) Cook et al., 2012 (0.03%; 95% ci <0.01–15%, $n = 1287$ ship-based obs)	1
Atlantic puffin	Garthe and Hüppop, 2004 (score of 1, median <5 m); Paton et al., 2010 (0%, $n = 5$ ship-based obs) Cook et al., 2012 (0.1%; 95% ci <0.1–8%, $n = 5981$ ship-based obs)	1

2.2. Flight manoeuvrability

This factor takes into account the aerial agility of species and hence their potential to avoid collision with wind turbines at sea. Following Garthe and Hüppop (2004), we assume that, all other factors being equal, birds with low flight manoeuvrability are more likely to collide with wind turbines at offshore wind farms than birds with high flight manoeuvrability. Scores were taken from Garthe and Hüppop (2004), but adjusted where more recent data suggest appropriate. For additional species, scores were based on peer-reviewed literature and subjective judgement moderated by expert opinion. Species were classified from 'very high flight manoeuvrability' (score 1) to 'very low manoeuvrability' (score 5) (electronic Appendix 1). The value is considered to be a consequence of morphology rather than behaviour. It may vary seasonally (for example in relation to moult) but such detail is beyond the scope of this assessment.

2.3. Percentage of time flying

This factor was assumed to indicate risk of collision because marine birds that spend more time flying while at sea (whether while breeding, migrating, wintering, or as prebreeders) are more likely to be at risk of collision. Where available, scores were taken from Garthe and Hüppop (2004) and adjusted where more recent data suggest appropriate. For other species, scores were calculated from data on activity budgets following the procedure outlined by Garthe and Hüppop (2004). Species were scored 1 if 0–20% of time at sea was spent in flight, 2 if 21–40% was spent flying, 3 if 41–60% was spent flying, 4 if 61–80% was spent flying, and 5 if 81–100% was spent flying (electronic Appendix 2). This factor will probably vary seasonally, with the literature indicating more flight activity while rearing chicks than during the incubation period, and more flight while breeding than during winter. Peaks of flight activity occur in migrant species during the migration, while flight activity may be reduced during post-breeding moult. However, these refinements are not yet well enough documented to assess scores separately for different seasons, although that could be a useful development of the method.

2.4. Nocturnal flight activity

Although various observations exist, detailed data on nocturnal flight activity are not available for many species. Geolocation and GPS logger data are starting to change this situation, predominantly for large Southern Ocean seabirds (Mackley et al., 2010, 2011; Phalan et al., 2007), but with data now available for northern gannet (Garthe et al., 2012a) and black-legged kittiwake *Rissa tridactyla* (Kotzerka et al., 2010). Similar data should soon be available for some other North Atlantic seabirds including great skuas *Stercorarius skua* for which logger deployments have already provided data on migrations and wintering areas (Magnusdottir et al., 2012). We used scores published in Garthe and Hüppop (2004) for the species where these were available: Score 1 (limited flight activity at night) to score 5 (much flight activity at night) (electronic Appendix 3). We used published data where possible, and information (often qualitative rather than quantitative) from individual species studies or from handbooks (Cramp and Simmons, 1977, 1980; del Hoyo et al., 1992, 1996; Glutz von Blotzheim and Bauer, 1982). Classifications were also moderated by experts.

2.5. Disturbance by wind farm structures, ship and helicopter traffic

Marine bird species vary in their reactions to offshore wind farms and the ship and helicopter traffic that occurs during maintenance of the turbines. Where available, scores were taken from Garthe and Hüppop (2004) and adjusted where more recent data were available (e.g. Schwemmer et al., 2011). For additional species, information on disturbance sensitivity was taken from the peer-reviewed literature, and scores were moderated by experts. Scoring categories were: 1 (limited escape behaviour and a very short flight distance when approached), to 5 (strong escape behaviour, at a large response distance) (electronic Appendix 4).

2.6. Habitat specialisation

Marine birds vary in the range of habitats they use, for example relating to water masses and frontal systems and whether they use these as specialists or generalists. This score classifies species into categories from 1 (tend to forage over large marine areas with little known association with particular marine features) to 5 (tend to feed on very specific habitat features, such as shallow banks with bivalve communities, or kelp beds) (electronic Appendix 5). Where available, scores presented by Garthe and Hüppop (2004) were used. Scores for other species were based on foraging ecology described in single species studies in the literature, or from standard handbook descriptions.

2.7. Calculating species vulnerability scores for collision risk and displacement concern

Garthe and Hüppop (2004) calculated species vulnerability scores according to Equation (1), where *e*, *f*, *g*, *h*, *i* and *j* represent flight altitude, flight manoeuvrability, percentage of time flying, nocturnal flight activity, disturbance by wind farm structures, ship and helicopter traffic, and habitat specialisation respectively.

$$\text{Species vulnerability score} = (e+f+g+h)/4 \times (i+j)/2 \times \text{conservation score} \quad (1)$$

This recognised that the first four factors all relate to flight ability and flight behaviour, while the last two factors relate to habitat use and susceptibility to disturbance. Thus their index combined both collision risk and disturbance/habitat loss considerations into a single score.

We use an alternative approach and score separately for collision concern and for disturbance/habitat displacement concern. For collision risk, we give a high weighting to flight altitude (*e*), and lower weighting to manoeuvrability (*f*), percentage of time flying (*g*), and nocturnal flight activity (*h*) (Equation (2)).

$$\text{Collision risk score} = e \times (f+g+h)/3 \times \text{conservation importance score} \quad (2)$$

For disturbance/habitat displacement we calculated a vulnerability index according to Equation (3) where *i* and *j* represent disturbance by wind farm structures, ship and helicopter traffic, and habitat specialisation respectively. We divided the total by 10 (an arbitrary value) to recognise that the disturbance/displacement impact on populations is likely to be considerably less than a direct mortality impact such as from collisions and therefore the two scales should not be compared in a quantitative way but only in terms of the species ranking within one scale.

$$\text{Disturbance/displacement score} = ((\mathrm{i} \times \mathrm{j}) \times \text{conservation importance score})/10 \quad (3)$$

3. Results

Factors and species-scores were sent to a panel of seabird experts for comment. Most reviewers suggested no change to the factors used, and no change to most scores. Most reviewers felt that one or two out of the 228 scores should be adjusted, so agreed with more than 99% of the scores. The few scores that were consistently questioned by reviewers were altered to bring them in line with this consensus opinion.

The ranked species' vulnerability scores in the context of collision risk (and considering the conservation importance of the species) are summarised in Table 2. The percentage of a species' flight altitude at turbine blade height ranged from values of 0–35. The mean of the other factors ranged from 1.3 to 3.7 (within a theoretically possible range of 1–5). Details of the derivation of the component scores are given in Table 2 and the electronic Appendix. The highest scores indicate that the larger gull species (great black-backed gull *Larus marinus*, herring gull *Larus argentatus*, and lesser black-backed gull *Larus fuscus*), northern gannet and white-tailed eagle are the species likely to be most at risk of collision with offshore wind turbines.

The ranked species vulnerability scores in the context of disturbance or habitat displacement (and considering the conservation importance of the species) are summarised in Table 3. The highest scores indicate that all diver species and common scoter are most likely to be at risk of disturbance or displaced from habitats as a result of offshore wind turbines. Details of the derivation of the component scores are given in the electronic Appendix tables.

4. Discussion

The key results from our assessment are ranked species lists referring to species' vulnerability to offshore wind farms in the context of collision risk (Table 2) and vulnerability in the context of disturbance or habitat displacement (Table 3). For collision risk, the five top ranking species had scores that were greater than half of the maximum score achieved overall (by herring gull, with a collision risk score of 1306). These species were herring gull, great black-backed gull, lesser black-backed gull, white-tailed eagle and northern gannet. In the rankings for disturbance or habitat displacement concern, only three species had scores that were over half of the maximum score possible (maximum score possible was 50). In contrast to species indicated as having a high collision risk, the species identified as most vulnerable to disturbance or displacement were diver species and common scoter. Garthe and Hüppop (2004) identified divers and common scoter as the species with the highest sensitivity index scores in the southern North Sea, suggesting that their index weighted disturbance/displacement impacts more strongly than collision mortality risk. The differing results for the two potential risks shown by our assessment are a result of species-specific variability in behaviour. The species identified as of highest concern in the table rankings should be the focus for monitoring and for further research investigating the effects of offshore wind farms on marine birds in Scottish waters, and we suggest that this approach should be extended to consider marine bird communities in other countries.

4.1. Development/evolution of our approach

Following discussions and comments from reviewers, we developed the approach taken by Garthe and Hüppop (2004) to recognise that there is broad support for the view that collision concern should be considered separately from displacement concern, as the rankings of species in each context are very different. We also differ in our approach to Garthe and Hüppop (2004) by not ranking all scores for factors measuring the vulnerability of species to offshore wind turbines on a scale of 1–5. Instead, the factor assessing flight altitude was considered in greater detail by presenting the percentage of a species flight altitude spent at wind turbine blade height. In combination with an increased weighting given to this factor when calculating collision risk scores, our method incorporates the broad consensus of opinion among reviewers that flight height is considerably more important than the other factors considered here in assessing collision risk of marine birds with wind turbines. There were mixed views regarding the relative importance of the other three factors of manoeuvrability, percentage of time flying and the amount of nocturnal flight in affecting collision risk, and the down-weighting of these three factors recognised this. This change in approach is more appropriate now than it was when Garthe and Hüppop (2004) prepared their paper, because there are now considerably more detailed data on marine bird flight height from the work of Paton et al. (2010), Krijgsveld et al. (2011), Cook et al. (2012), Garthe et al. (2012b) and others.

4.2. Limitations of indices of sensitivity

The method of calculating indices and ranking species has been criticised for the uncertainty in how to incorporate a variety of different factors, often scored on different scales and sometimes with

Table 2
Ranked species concern in the context of collision impacts: percent flying at blade height × 1/3(manoeuverability score + % time flying score + nocturnal flight score) × conservation importance score (ranked by index value).

Species	Flight % at blade height	Flight agility	% of time flying	Night flight	Conservation importance score	Total risk score
Herring gull	35	2	2	3	16	1306
Great black-backed gull	35	2	2	3	15	1225
Lesser black-backed gull	30	1	2	3	16	960
White-tailed eagle	24	3	5	1	12	864
Northern gannet	16	3	3	2	17	725
Common gull	23	1	2	3	13	598
Black-legged kittiwake	16	1	3	3	14	523
Arctic skua	10	1	5	1	14	327
Great skua	10	1	4	1	16	320
Black-headed gull	18	1	1	2	12	288
Sandwich tern	7	1	5	1	15	245
Black-throated diver	5	5	3	1	16	240
Great northern diver	5	5	2	1	18	240
Common tern	7	1	5	1	14	229
Red-throated diver	5	5	2	1	16	213
Little tern	7	1	5	1	13	212
Arctic tern	5	1	5	1	17	198
Roseate tern	5	1	5	1	15	175
Shag	5	3	2	1	15	150
Slavonian grebe	4	4	2	2	13	139
Greater scaup	3	4	2	5	11	121
Common eider	3	4	2	3	13	117
Great cormorant	4	4	2	1	11	103
Common goldeneye	3	3	2	3	12	96
Common scoter	3	3	2	3	12	96
European storm-petrel	2	1	3	4	17	91
Velvet scoter	3	3	2	3	11	88
Leach's storm-petrel	2	1	3	4	16	85
Great-crested grebe	4	4	3	2	7	84
Long-tailed duck	3	3	2	3	8	64
Northern fulmar	1	3	2	4	16	48
Common guillemot	1	4	1	2	16	37
Razorbill	1	4	1	1	16	32
Black guillemot	1	4	1	1	13	30
Atlantic puffin	1	3	1	1	16	27
Little auk	1	3	1	1	9	15
Manx shearwater	0	3	3	3	17	0
Sooty shearwater	0	3	3	3	12	0

correlations among the factors being used (Desholm, 2009). Considering birds of all species and not just marine birds, Desholm (2009) argued that in order to prioritise bird species for assessment of the impact of collision mortality at offshore wind farms, it is possible to consider just two criteria; proportion of the biogeographic population at risk at a wind farm site, and population 'elasticity', which is mainly determined by adult survival rate. Birds with high adult survival rates (such as most marine birds) will be more severely impacted by wind farm mortality than birds with low natural survival rates (such as most terrestrial migrant passerines). That approach has the benefit of great simplicity, and works well when ranking bird species ranging from small passerines with very short life expectancy and high reproductive output through to marine birds with long life spans and low fecundity (Desholm, 2009). However, that method does not discriminate well between species with similar demographic parameters (most marine birds have similarly high adult survival rates), and it does not take into account the fact that some kinds of marine birds are more, or less, likely to collide with wind turbines as a consequence of their species-specific ecology or behaviour. For example, among marine birds with comparable demography, species that regularly fly at turbine blade height are apparently more at risk from collision mortality than species that only fly low over the water. Nor does that method take account of the possible effects of offshore wind farms on marine birds through effects such as deflection of flight paths (Masden et al., 2009; Speakman et al., 2009) or loss of foraging habitat (Fox et al., 2006a,b). In our analysis we are dealing mainly with species at the long-lived end of this spectrum of life histories and so the simple model focusing only on adult survival rate, but not on differences in ecology and behaviour, becomes less useful as there is relatively little variation in this among most marine bird species.

4.3. Limitations of data sources, implications for results, and future improvements

There are limitations inherent in the index resulting from data sources, which should be considered when interpreting our results. Flight altitude was considered to be the most important factor in calculating collision risk for marine bird species at offshore wind farms. In calculating the collision risk for species, Cook et al. (2012) provide the most detailed review of available data on flight altitude, however there are inherent limitations in these data, which should be considered. These include the fact that most data are from boat-based observations and as such, observations potentially include significant numbers of marine birds scared into flight by the boat (Camphuysen et al., 2004). Such birds tend to fly low over the water so will bias the distribution of flight heights. Birds of some species may also be attracted to boats. Additionally, most estimates of the height of flying marine birds used by Cook et al. (2012) are very crude, mostly being based on estimates by observers and not on measurements. It is worth noting that the data reported by Cook et al. (2012) can differ quite considerably to flight heights of

Table 3
Ranked species concern in the context of disturbance and/or displacement from habitat (Disturbance score × Habitat flexibility score × Conservation Importance score)/10.

Species	Disturbance by ship and helicopter traffic	Habitat use flexibility	Conservation importance score	Species concern index value
Black-throated diver	5	4	16	32
Red-throated diver	5	4	16	32
Great northern diver	5	3	18	27
Common scoter	5	4	12	24
Common goldeneye	4	4	12	19
Greater scaup	4	4	11	18
Velvet scoter	5	3	11	16
Common eider	3	4	13	16
Black guillemot	3	4	13	16
Slavonian grebe	3	4	13	16
Common guillemot	3	3	16	14
Razorbill	3	3	16	14
Shag	3	3	15	14
Great cormorant	4	3	11	13
Little tern	2	4	13	10
Arctic tern	2	3	17	10
Atlantic puffin	2	3	16	10
Long-tailed duck	3	4	8	10
Roseate tern	2	3	15	9
Sandwich tern	2	3	15	9
Common tern	2	3	14	8
Great-crested grebe	3	4	7	8
Great black-backed gull	2	2	15	6
Black-legged kittiwake	2	2	14	6
Common gull	2	2	13	5
Black-headed gull	2	2	12	5
Little auk	2	2	9	4
Northern gannet	2	1	17	3
Herring gull	2	1	16	3
Great skua	1	2	16	3
Lesser black-backed gull	2	1	16	3
Arctic skua	1	2	14	3
White-tailed eagle	1	2	12	2
Manx shearwater	1	1	17	2
European storm-petrel	1	1	17	2
Leach's storm-petrel	1	1	16	2
Northern fulmar	1	1	16	2
Sooty shearwater	1	1	12	1

marine birds that have been measured by radar (Krijgsveld et al., 2011; Cook et al., 2012). This discrepancy is unexplained, but it seems likely that the radar measurements of flight height are more reliable where confident of species attribution. Our index indicates that many marine bird species rarely fly at turbine blade height, and so appear to have negligible risk of population-level impacts from collision mortality. These include sea ducks, alcids, storm-petrels and shearwaters (though the possibility that such birds may occasionally fly higher than normal as a result of disturbance needs to be borne in mind). The low risk for these species is consistent with empirical data from long-established offshore wind farms (ICES, 2011). However, it would be desirable to have more data on flight heights to allow this inference to be converted into a confident conclusion that might permit species to be scoped out of assessments. In light of new, more detailed data it will be possible to revise the scorings presented here and establish with more confidence those species at risk. We do suggest however, that species presented here with high scores should be of particular concern in relation to offshore wind developments. Gulls, white-tailed eagles, northern gannets, skuas and divers are identified as being the groups whose populations are most at risk in a Scottish context.

The recent increase in the use of data loggers on seabirds is starting to provide more detailed information on the at-sea activity of seabirds. This should help achieve an increase in the amount of data available and improve the quality of flight altitude data and information required for some of the other factors considered in calculating collision risk (e.g. percentage of time spent in flight and nocturnal flight activity). It is also possible that, in the near future, collection of quantitative data on time spent in flight from geolocation data loggers (for example, based on salt-water switch recording time spent with the logger immersed in seawater) will allow scorings to be converted into a quantitative scale rather than the present qualitative one.

Previous studies indicate that some species of marine bird avoid wind farms and as such, collision levels are low (Fox et al., 2006b; ICES, 2011; Petterson, 2005; Lindeboom et al., 2011). Our collision risk index is likely to require modification in future as more data become available, and at present should be considered precautionary in that the relative avoidance responses of different species are not yet well known. In assessing the potential importance of displacement for different marine bird species, although there was strong consensus among reviewers for the scores used, this consensus may be more a result of uncertainty than confident agreement, and so the ranking of species needs to be treated with caution. However, we suggest that species with scores over 15 (divers, scoters, goldeneye *Bucephala clangula*, scaup *Aythya marila*, common eider, black guillemot *Cepphus grylle*, Slavonian grebe *Podiceps auritus*) should be considered as focal species for concern with regards to potential displacement effects. Species with scores below 8 (northern fulmar *Fulmarus glacialis*, storm-petrels, shearwaters, gulls, skuas, northern gannet, little auk *Alle alle*, and white-tailed eagle) seem very unlikely to be affected by displacement. It is worth noting that whilst it is clear that some marine birds do strongly avoid wind turbines at sea, recent work modelling the cumulative impact of disturbance by wind turbines suggests that the impact of these through increased travel distances and habitat loss is trivial, even for species that show especially strong avoidance behaviour and have a high displacement ranking, such as red-throated divers *Gavia stellata* (Topping and Petersen, 2011).

We are aware that our vulnerability index deals with a limited set of factors and that there are other potential impacts that are not necessarily covered. For example, Perrow et al. (2011b) presented evidence suggesting that little tern *Sternula albifrons* breeding success in a colony in Norfolk may have been reduced by a shortage of young herring *Clupea harengus* around Scroby Sands offshore wind farm caused by monopile installation affecting fish reproduction locally. To an extent, the high sensitivity of little tern would be indicated by our six factors because they are seabirds with a very short foraging range that utilise a very particular and restricted foraging habitat, so score as sensitive on the habitat flexibility factor. However, complex and indirect ecosystem effects such as alteration of fish and benthic invertebrate abundance by wind farms is something that is extremely difficult to predict, so caution is needed in interpretations and collecting post-construction monitoring data will be important. Once operational, offshore wind farms may possibly enhance food supplies for marine birds by acting as marine protected areas (e.g. closed to trawl fishing; Defew et al., 2012). Siting of wind farms can also be influential at specific sites. For example, turbines placed between a common tern *Sterna hirundo* colony and their feeding habitat have had a high impact on a particular colony (Everaert and Stienen, 2007; Stienen et al., 2008), which might not be the case where a wind farm is placed away from the obligatory flight line of birds from a specific breeding site. In addition, the perception of risk, which seems to vary among species, and possibly may be as important a factor as anatomical constraints, may also be relevant, but cannot readily be scored on a scale, though avoidance may possibly be a proxy for perception of risk. We considered these indirect and uncertain

effects to be beyond the scope of this review, but emphasise that they should not be assumed to be negligible.

An additional factor that we could not easily consider here but should not be ignored is the possibility that weather conditions may affect collision risk for marine birds. For example, weather conditions such as fog or heavy rain, may obscure turbines. Such effects might over-ride any species-specific differences in vulnerability. In reviewing the results of studies at demonstration offshore wind farms in Denmark, Fox et al. (2006b) stated "observations at Nysted that waterbirds tend not to fly in the area of the turbines at night, or under adverse weather conditions (as found elsewhere; Petterson, 2005) suggest that collision risk is not likely to be high even under conditions when the turbines are less visible". These observations suggest that catastrophic mortality incidents caused by adverse weather conditions are less likely at offshore wind farms than has been suggested by some, although breeding birds may not have the flexibility to respond in this way. In addition, presence of lights on wind farms may affect collision risk for marine birds, but the susceptibility of different species to attraction to lights at sea is not well known (Merkel and Johansen, 2011).

4.4. Applications and future testing

In scoping potential areas for offshore wind farm development in Scottish waters, Davies and Watret (2011) considered constraints implied by seabird SPAs, and the distribution at sea of seabirds as indicated by the European Seabirds at Sea database. These data were combined with the flight height data presented by Cook et al. (2012) to assess numbers of marine birds flying at collision height risk in different parts of the Scottish marine area. The development of sensitivity scoring and conservation importance scoring for individual species of marine birds may help to refine such assessment of environmental constraints by allowing a focus on the marine bird species of greatest concern. This would most usefully be combined with mapping (e.g. Garthe and Hüppop, 2004) of the distribution of seabird SPAs and the numbers of each species protected at these sites.

The scores presented in this paper, and similar scoring for other species, will help developers preparing Environmental Statements for new sites, by providing a clearer focus on the key species likely to be of concern in relation to collision risk and in relation to displacement impacts. Nevertheless, these scores should be seen as iterative, requiring to be updated as more data become available, and possibly to be made obsolete when macro- and micro-avoidance rates are measured for a range of species at a range of offshore wind farms. The ranking of marine bird species in our collision risk index accords well with the concerns of offshore wind farm developers in Scottish waters, gulls and gannets being a particular concern at many offshore wind farms as expressed in Environmental Statements and Environmental Impact Assessments. However, there is a need for post-construction monitoring data from offshore wind farms to assess more fully the accuracy of these predictive indices, and there is an urgent need to study both macro-avoidance and micro-avoidance behaviour of different seabird species.

5. Conclusions

This paper advances previous work on likely impacts of offshore wind farms on marine bird populations by separating risks of collision mortality from risks of disturbance/displacement. These two aspects affect different marine bird species. An offshore wind farm collision risk index identifies populations of gulls, white-tailed eagles, northern gannets and skuas as the marine birds in Scottish waters most vulnerable to collision mortality impacts. In contrast, populations of divers and common scoters appear most vulnerable to displacement impacts. The collision risk index is particularly affected by the proportion of birds flying at collision risk height, and there is a need to collect more data on marine bird flight heights. Populations identified as most vulnerable should be the focus of future monitoring studies, but there is a need to develop research into macro- and micro-avoidance rates and how these vary among species.

Acknowledgements

This work was funded by The Scottish Government. We particularly thank Dr Ian Davies (Marine Scotland Science) for very helpful advice throughout the project and for making available relevant 'grey' literature. We are extremely grateful to the following, who reviewed the scores and provided very helpful comments, and relevant literature to consider: Rob Barrett (University of Tromsø, Norway), Mark Bolton (RSPB), Nigel Buxton (SNH), Kees Camphuysen (NIOZ, Netherlands), Stefan Garthe (University of Kiel, Germany), David Gremillet (CNRS, France), Ross McGregor (SNH), Will Miles (Fair Isle Bird Observatory), Greg Mudge (SNH), Henrik Österblom (Stockholm Centre for Resilience Studies, Sweden), Jim Reid (JNCC) and Mark Tasker (JNCC). HMW and EAM are funded through the Marine Renewable Energy and the Environment (MaREE) project by Highlands and Islands Enterprise, the European Regional Development Fund and the Scottish Funding Council.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvman.2013.01.025.

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Marine Environmental Research 101 (2014) 69–80

Contents lists available at ScienceDirect

Marine Environmental Research

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Great skua (*Stercorarius skua*) movements at sea in relation to marine renewable energy developments

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ARTICLE INFO

Article history:
Received 12 June 2014
Received in revised form
14 September 2014
Accepted 15 September 2014
Available online 17 September 2014

Keywords:
Environmental impact
Remote sensing
Seabirds
GPS
Marine spatial planning
Wind power
Wave power
Bio-logging

ABSTRACT

Marine renewable energy developments (MREDs) are an increasing feature of the marine environment. Owing to the relatively small number of existing developments and the early stage of their associated environmental monitoring programmes, the effects of MREDs on seabirds are not fully known. Our ability to fully predict potential effects is limited by a lack of knowledge regarding movements of seabirds at sea. We used GPS tracking to improve our understanding of the movements at sea of a protected seabird species breeding in Scotland, the great skua (*Stercorarius skua*), to better predict how this species may be affected by MREDs. We found that the overlap of great skuas with leased and proposed MREDs was low; particularly with offshore wind sites, which are predicted to present a greater risk to great skuas than wave or tidal-stream developments. Failed breeders overlapped with larger areas of MREDs than breeding birds but the overall overlap with core areas used remained low. Overlap with wave energy development sites was greater than for offshore wind and tidal-stream sites. Comparison of 2011 data with historical data indicates that distances travelled by great skuas have likely increased over recent decades. This suggests that basing marine spatial planning decisions on short-term tracking data could be less informative than longer-term data.

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1. Introduction

Marine renewable energy developments (MREDs) are an increasing feature of the marine environment. The rapid development of these industries is being driven by the need to reduce carbon emissions and to increase energy security. In Scotland, The Crown Estate and the Scottish Government are working in partnership to realise the Scottish Government's target of generating 100% of electricity from renewable sources by 2020 (Scottish Government, 2011). An advanced marine plan has been developed for Scotland with sites leased to offshore wind, wave and tidal-stream energy developers, with additional sites proposed for future development (Scottish Government, 2013a).

Owing to the small number and limited extent of existing MREDs, the effects of offshore wind, wave and tidal-stream installations on seabirds are not fully known (Fox et al., 2006; McCluskie et al., 2013; Witt et al., 2012). This is despite several leased and proposed MRED sites in north-east Scotland being located in close proximity to special protection areas (SPAs) designated specifically to safeguard breeding populations of internationally important seabird species (JNCC, 2014a) (Fig. 1). SPA legislation (EC Birds Directive 2009/147/EC) requires that any development does not damage the integrity of protected seabird populations (Scottish Natural Heritage, 2014). Some of the potential negative effects of MREDs on seabirds include collision with devices (Everaert and Stienen, 2007; Langton et al., 2011); displacement from areas where devices are located (Larsen and Guillemette, 2007; McDonald et al., 2012); increased flight costs if devices act

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http://dx.doi.org/10.1016/j.marenvres.2014.09.003
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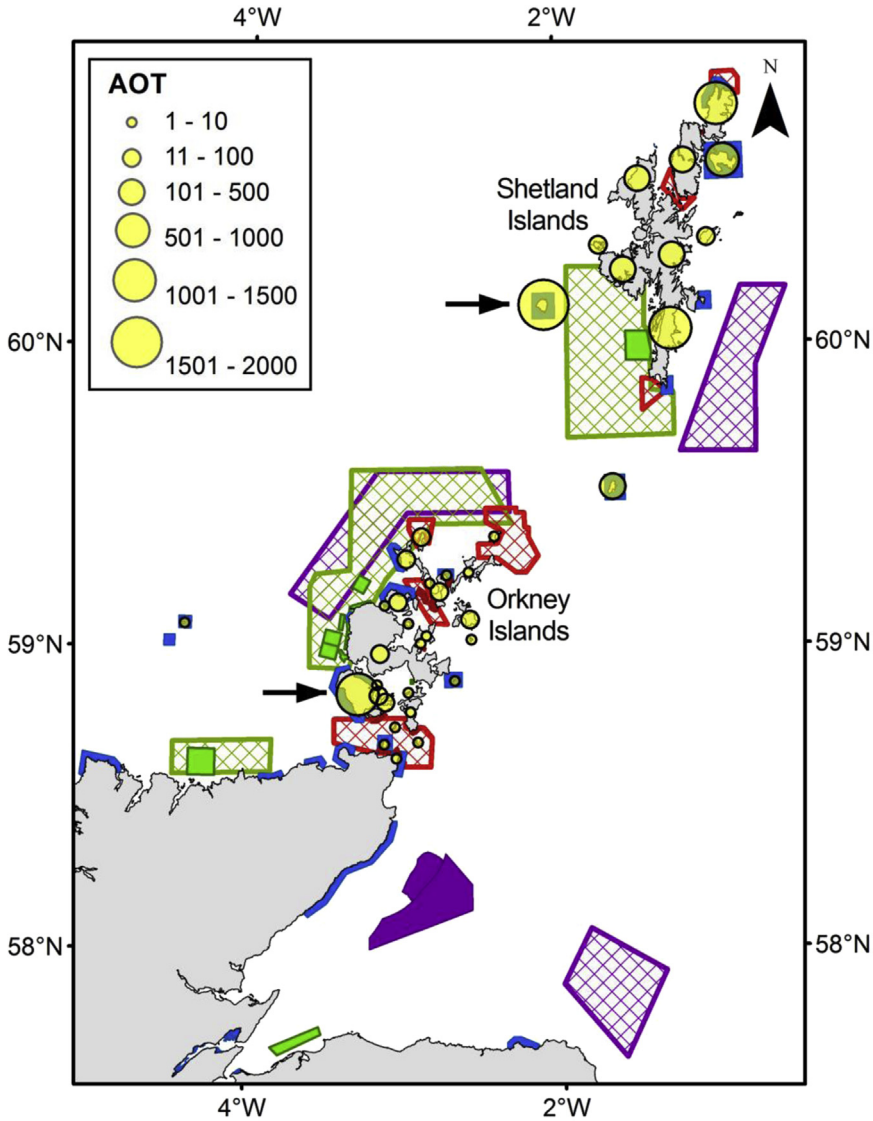

Fig. 1. Locations of breeding colonies (yellow circles; defined by number of Apparently Occupied Territories (AOT)) and special protection areas (blue polygons) designated for breeding seabird populations in north-east Scotland, and the proximity of leased and proposed marine renewable energy development sites (solid polygons = leased sites; cross-hatched = proposed sites; purple = offshore wind sites; green = wave sites; red = tidal-stream sites). Study colonies indicated with black arrows. See Supplementary Table 1 for colony sizes and year of AOT count. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

as barriers to movement (Desholm and Kahlert, 2005; Masden et al., 2009, 2010); alteration of foraging habitats if devices alter currents and sediment movements (Gill, 2005; Langhamer et al., 2010; McCluskie et al., 2013); and disturbance from increased vessel traffic (Bellefleur et al., 2009; Garthe and Hüppop, 2004; Ronconi and Clair, 2002; Schwemmer et al., 2010). Potential positive effects include the increased provision of loafing and roost structures at sea (Deurs et al., 2012; Grecian et al., 2010); and increased foraging opportunities if devices act as fish aggregation devices (FADs) (Langhamer and Wilhelmsson, 2009; Wilhelmsson et al., 2006), artificial reefs (Langhamer et al., 2009; Reubens et al., 2013a, 2013b) or *de facto* marine protected areas (*df*MPAs) (Grecian et al., 2010; Inger et al., 2009). Although knowledge of seabird ecology and behaviour allows moderately robust predictions of which species of seabirds are most likely to be at risk from impacts (Furness et al., 2013, 2012; Garthe and Hüppop, 2004), a lack of knowledge regarding seabird movements at sea limits our ability to identify and quantify potential effects of MREDs on populations (McCluskie et al., 2013).

The great skua (*Stercorarius skua*) is a protected species breeding in Scotland from April to late August/early September, with numerous breeding colonies located in close proximity to leased and proposed MREDs. Currently we know very little about the movements of this species at sea, which makes it difficult to predict how they may be affected by MREDs. Great skuas have a restricted global distribution with over 60% of the global population (*ca.*16,000 pairs) breeding in Scotland at the most southern part of their range and migrating to winter off Iberia and northwest Africa (Magnusdottir et al., 2012; Mitchell et al., 2004). This internationally important seabird population is experiencing low levels of breeding success (Orkney Bird Report Committee, 2013; Shetland Bird Club, 2013). In Orkney, the breeding population of great skuas has undergone a 23% reduction over the last decade (Meek et al., 2011) and numbers at the largest colonies in Shetland have

also decreased (JNCC, 2014b). These declines are likely to have been driven by a reduced sandeel (*Ammodytes marinus*) abundance (Caldow and Furness, 2000), decreases in fisheries discarding and possibly the impacts of climate change (Oswald et al., 2008, 2011), which have resulted in lower prey availability for great skuas (Hamer et al., 1991; Votier et al., 2004; Meek et al., 2011). This evidence suggests that great skuas are already under pressure from changing climate and human activities such as fishing, potentially in both their breeding and wintering foraging areas. As such, this species is at risk from additional impacts of MREDs that could compound existing declines in productivity and population sizes. Existing knowledge suggests that great skuas are most likely to be at risk from collision with offshore wind turbines, owing to their tendency to spend the majority of their time at sea in flight (except at night when they sit on the water or return to the colony), and because they fly at altitudes overlapping with turbine rotors (Furness et al., 2013; Garthe and Hüppop, 2004). The species is less likely to be negatively affected by wave and tidal-stream MREDs as they are surface feeders and so unlikely to collide with devices on or below the water surface (Furness et al., 2012). Birds struggling to find sufficient prey or birds from larger breeding colonies travel further to forage than those from smaller colonies or when food is abundant (Bertrand et al., 2012; Lewis et al., 2001; Wakefield et al., 2013). Therefore, great skuas that travel over greater distances and range more widely increase their probability of encountering and potentially being affected by a MRED. Conversely, if MREDs are located in close proximity to breeding colonies, breeding birds that are constrained to return to their nest and limited in the area over which they can forage (Shaffer et al., 2003), may repeatedly encounter and be affected by MREDs (Masden et al., 2010). Given this evidence, different subsections of great skua populations may be affected in different ways by MREDs.

To investigate how great skuas may be affected by MREDs we used tracking technology to improve our understanding of how this species uses the marine environment. We investigated the movements of breeding birds and failed breeders from two of the largest and best studied colonies in the UK (Fig. 1 and Supplementary Table 1) to assess differences according to colony location and breeding status of the bird (actively breeding or failed breeder). We also aimed to investigate the movements of birds according to their nest status (incubating, chick-rearing, following successful fledging of a chick and following a failed breeding attempt). The overlap of great skuas with leased and proposed MREDs was quantified for birds during different breeding stages and from two different colonies. In addition, we compared tracking data with historical data to assess whether great skua movements have changed over time, with the intention of exploring the implications of predicting the effects of MREDs on species based on short-term tracking data.

2. Materials and methods

2.1. Data collection

Solar-powered GPS data loggers were fitted to 20 breeding great skuas in the period from 03 June 2011 to 14 June 2011. The loggers were attached for the duration of the breeding season. Ten loggers were fitted to birds from Hoy, Orkney, UK (58°52′N, 3°24′W) and ten loggers fitted to birds from Foula, Shetland, UK (60°8′N, 2°5′W). The study sites represent the largest breeding colonies for great skuas in Orkney and Shetland and are located in close proximity to leased and proposed MRED sites (Fig. 1 and Supplementary Table 1). Birds were caught on the nest using a remote-controlled noose trap and logger attachment took a mean of 25 min (range = 19–32 min). UvA-BiTs GPS data loggers (Bouten et al., 2013) weighing *ca.* 25 g (<3% of adult mass) were attached dorsally using a Teflon® ribbon backpack harness, consisting of a neck loop and wing loops. Only birds with breeding partners in attendance on territory were targeted for trapping. Dummy eggs were temporarily placed in the nest during trapping and logger deployment to prevent damage through trampling or predation. GPS loggers provided location data at intervals ranging from 1 to 30 min. Loggers transmitted data remotely to a base-station via an antenna in each colony until birds left on migration. A control group of 20 breeding individuals (ten from each colony) were monitored to enable adverse effects of the logger and harness attachment to be identified. All 40 birds were marked with uniquely numbered Darvic colour rings for identification purposes and the location of their nest site was recorded by handheld GPS.

2.2. Trip characteristics

Movements >500 m beyond a pre-established colony perimeter and lasting >20 min were defined as 'trips'. Trips covering distances >2000 km ($n = 2$), which were an order of magnitude longer than the next longest trips, were excluded from analyses as they biased utilisation distribution (UD) calculations. When comparing movements of birds from Hoy and Foula, trips were categorised as occurring either during breeding or following a failed breeding attempt. When comparing movements of birds from Hoy only, trips were defined in more detail as occurring during incubation, chick-rearing, following successful fledging of a chick or following a failed breeding attempt. For each trip, we calculated the trip duration (time elapsed between departure and return to the colony), the foraging range (the maximum distance reached from the colony), and the total distance travelled (summation of distances between GPS points along the route). The number of trips per day were defined as the number of times a bird left the colony on a foraging trip. When birds did not leave the colony or were away from the colony on a foraging trip lasting longer than a day, a value of zero trips undertaken that day was recorded. Trip metrics were calculated using R package 'trip' ver.1.1-17 (Sumner, 2013) and Geospatial Modelling Environment ver.0.7.2.0 (Beyer, 2012).

2.3. Utilisation distributions

To estimate the core area used by birds in each category (outlined in Section 2.2) we used kernel density estimates (KDE) to calculate the 50% UD (Harris et al., 1990). Owing to variable intervals between GPS fixes and missing GPS locations, data were linearly interpolated to establish regular 10 min time intervals between GPS points. Interpolation of GPS locations eliminated overestimation and bias in habitat/area use as a result of clustering of short-interval GPS fixes in one area. KDEs were calculated using all location fixes within each category (outlined in Section 2.2) using a fixed smoothing parameter of 10,000 m and a grid size of 2,500 m. The smoothing parameter and grid size were identified as most appropriately representing the original data through visual assessment of a series of UDs calculated with a range of bandwidths. For birds in each category, we calculated the percentage overlap of the 50% UD with leased and proposed MRED sites (Scottish Government, 2013b). Analyses were performed using R packages 'adehabitatLT' ver.3.14 (Calenge, 2006), 'adehabitatHR' ver.4.10 (Calenge, 2006), and ArcGIS (ArcMap ver.10. ESRI, USA).

2.4. Variation in movements

Multiple trips were recorded for each bird. To account for non-independence of data as a result of repeated sampling, we used generalized linear mixed models (GLMM) with bird identity included as a random effect. In analyses comparing the movements

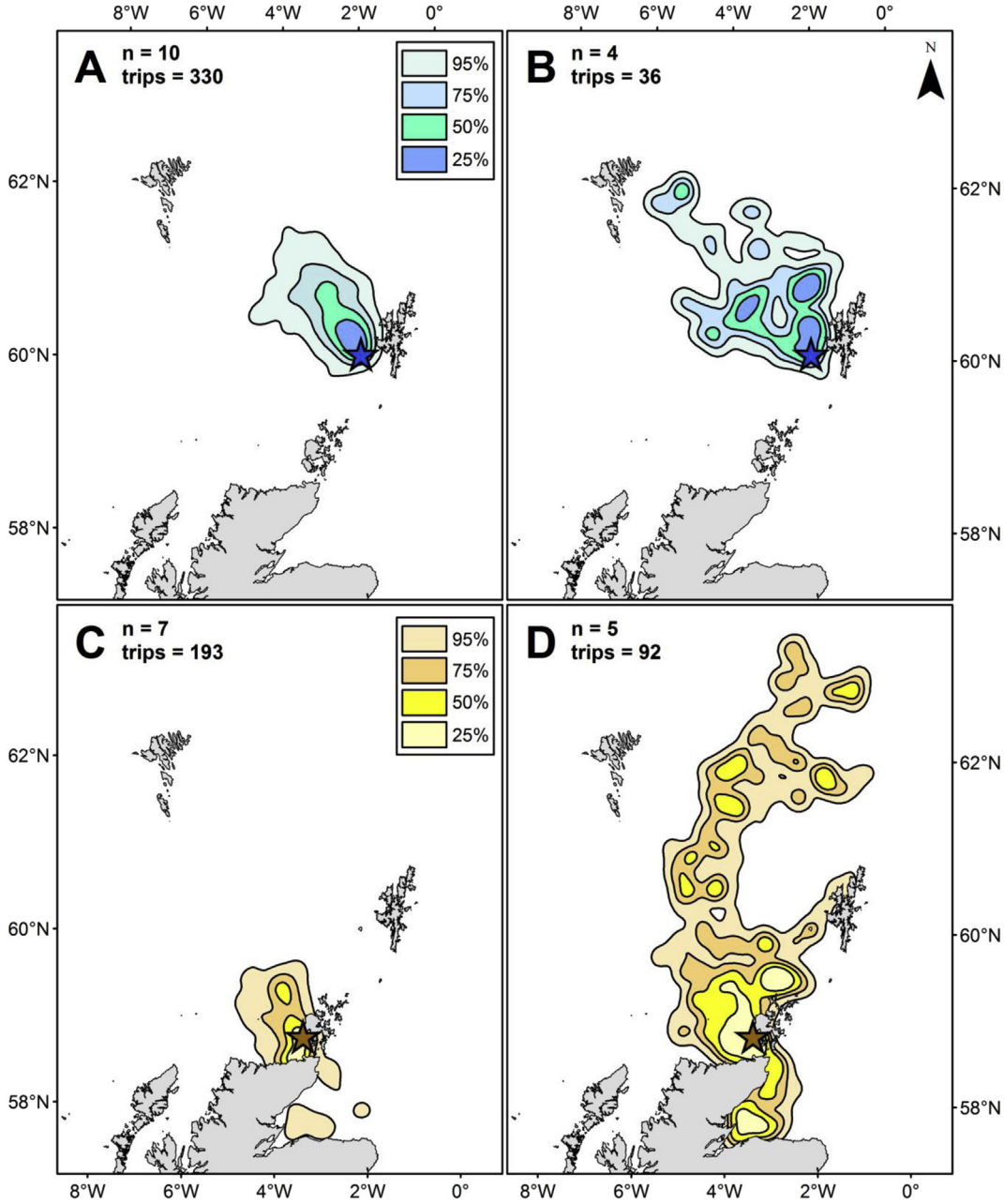

Fig. 2. Kernel density estimates of great skua movements from Foula (A–B) and Hoy (C–D) during breeding (A, C) and following breeding failure (B, D) until 16 July 2011.

of breeding and failed breeders from Hoy and Foula, models were constructed using a Gaussian distribution in R package 'lme4' ver.1.1-5 (Bates et al., 2012). In analyses comparing the movements of birds from Hoy during the breeding season (during incubation, chick-rearing, following successful fledging of chicks or following a failed breeding attempt), models were constructed using a Gaussian distribution in R package 'nlme' ver.3.1-109 (Pinheiro et al., 2013). This different model structure was required to compare the movements of birds from the Hoy colony alone as there was a greater variance in trips incorporating freshwater bathing or visits to probable club sites (sites where non-breeders congregate; Furness, 1987) than in trips that went directly out to sea, which was not evident when jointly analysing the Hoy and Foula movements. To account for this heterogeneity in variance that was attributed to trip type, a fixed variance structure was applied. In all models, trip length (km), duration (min) and maximal distance from the colony (km) were response variables and were log-transformed prior to statistical analysis to conform to assumptions of normality. Significance was assessed using likelihood ratio tests (LRTs) following backward step-wise model selection (Crawley, 2007).

All analyses were implemented in R ver.3.0.1 (R Core Team, 2013).

2.5. Historical data

Productivity on Foula, and data on nest attendance and trip characteristics during chick rearing, were extracted from published and unpublished literature from 1974 to 2011 (Catry, 1997; Furness, 1977; JNCC, 2014b; SOTEAG, 2012). Trip duration (time away from

nest), the number of trips undertaken per day and nest attendance were recorded during nest watches. Where available, productivity data for Hoy were also extracted (JNCC, 2014b).

3. Results

Of the 20 loggers deployed, data from 17 were analysed. The remaining three loggers (all from Hoy) did not charge their batteries sufficiently for GPS fixes to be taken regularly or the tagged bird left the colony very soon after logger attachment. The duration of data collection ranged from 23 to 106 days. A total of 2039 trips were recorded. Despite tagged birds not returning to their colony to breed the following year, monitoring of the control group of 20 breeding individuals (ten from each colony) found no adverse effects of the logger and harness attachment on territory attendance or breeding productivity during the 2011 breeding season (Thaxter et al., in review-a).

3.1. Comparison of movements of breeding birds and failed breeders from Hoy and Foula

A total of 651 foraging trips were recorded for 17 birds of known breeding status from Hoy and Foula between 03 June 2011 and 15 July 2011.

3.1.1. Trip characteristics

Failed breeders from each colony spent more time at sea, travelled over greater distances and undertook fewer trips per day than birds engaged in a breeding attempt. Birds from Foula tended to spend more time at sea and travelled further than birds from Hoy (Supplementary Table 2; Figs. 2 and 3A).

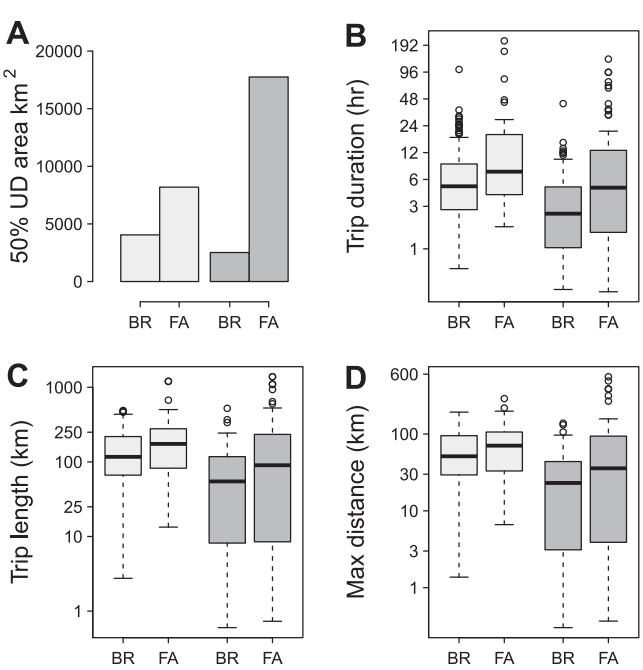

Fig. 3. Comparison of birds from Foula (light grey) and Hoy (dark grey) during breeding (BR) and following breeding failure (FA) in: A) the core area used, B) the duration of trips, C) the length of trips and D) the maximum distance travelled from the colony.

3.1.2. Variation in movements between colonies and according to breeding status

The duration of trips differed according to whether a bird was breeding or had failed in its breeding attempt (GLMM: $X^2 = 33.35$, d.f. = 1, $p < 0.001$); with breeding birds spending less time away from the colony than failed breeders (Fig. 3B).

The length of trips and the maximum distance reached from the colony during trips also differed according to breeding status (GLMM: $X^2 = 12.04$, d.f. = 1, $p < 0.001$; GLMM: $X^2 = 9.12$, d.f. = 1, $p < 0.01$ respectively) (Fig. 3C and D).

The duration of trips also differed according to colony (GLMM: $X^2 = 12.35$, d.f. = 1, $p < 0.001$); with birds from Foula undertaking trips that lasted longer than trips by birds from Hoy (Fig. 3B). The distance covered during trips and the maximum distance reached from the colony during trips differed according to colony (GLMM: $X^2 = 18.15$, d.f. = 1, $p < 0.001$; GLMM: $X^2 = 18.37$, d.f. = 1, $p < 0.001$ respectively) (Fig. 3C and D).

3.1.3. Overlap with marine renewables

There was no overlap of core areas with leased or proposed offshore wind sites for birds from Foula. A very low overlap with leased sites for offshore wind development was observed for failed breeders from Hoy (<1%) and a 9% overlap with proposed sites for future offshore wind development (Fig. 4 and Table 1).

Only birds from Hoy overlapped with leased sites for wave development (breeders: 3%, failed breeders: 1%). However, breeding birds and failed breeders from both colonies overlapped with proposed sites for wave developments; with a greater overlap observed for birds from Hoy (breeders: 11%, failed breeders: 12%) than for birds from Foula (breeders: 2%, failed breeders: 1%) (Fig. 4).

There was no overlap of Foula birds with leased or proposed tidal-stream development sites. Only failed breeders from Hoy overlapped with leased tidal-stream development sites (<1%), and there was a low overlap of breeding birds (5%) and failed breeders (3%) with proposed tidal-stream sites. The limited levels of overlap observed were expected given the small area covered by tidal-stream sites (Fig. 4).

3.2. Movements of birds from Hoy during the breeding season in relation to nest status

A total of 547 foraging trips were recorded for seven birds of known nest status from Hoy between 11 June 2011 and 15 August 2011.

3.2.1. Trip characteristics

Chick-rearing birds and birds that had successfully fledged chicks undertook more trips per day than incubating birds and birds that had failed in their breeding attempt. Failed breeders used a greater area at sea than incubating birds, chick-rearing birds or birds that had successfully fledged chicks (Supplementary Table 2; Figs. 5 and 6A).

3.2.2. Variation in movements according to nest status

The duration of trips differed according to the nest status of the bird (GLMM: $X^2 = 11.59$, d.f. = 3, $p < 0.01$). Chick-rearing and incubating individuals spent less time away from the nest than failed breeders and birds that had successfully fledged chicks. However, there was no significant difference according to nest status in the distance covered during trips or the maximum distance reached from the colony (Fig. 6B–D).

3.2.3. Overlap with marine renewables

Birds from Hoy only overlapped with leased wave and tidal-stream sites. There was no overlap with leased offshore wind

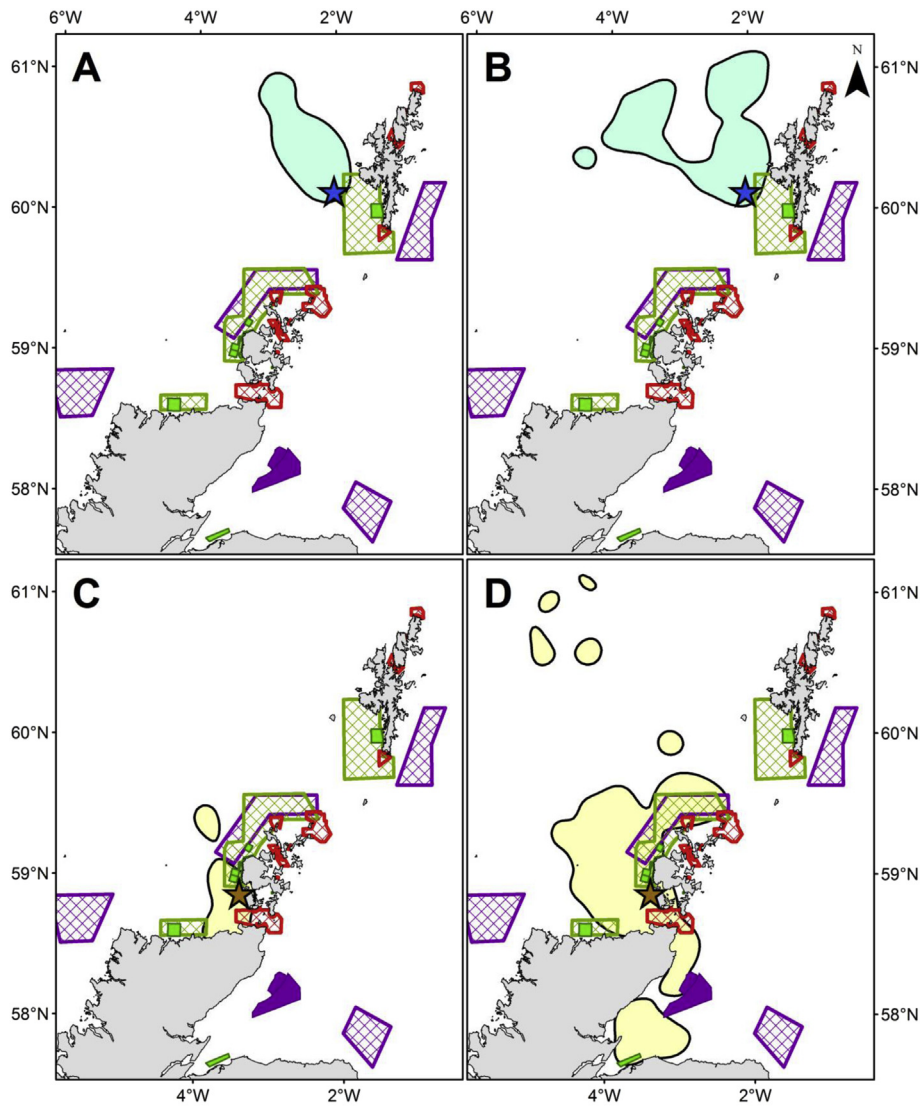

Fig. 4. Overlap of great skua core areas of use (50% utilisation distribution) with leased and proposed marine renewable energy development sites (solid polygons = leased sites; cross-hatched = proposed sites; purple = offshore wind sites; green = wave sites; red = tidal-stream sites) for birds from Foula (A–B) and Hoy (C–D), during breeding (A, C) and following breeding failure (B, D). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 1
Percentage of overlap of core areas used (50% utilisation distribution) with leased and proposed offshore wind, wave and tidal-stream renewable energy developments for birds from Foula and Hoy during breeding and once birds had failed in their breeding attempt.

Colony and breeding status	Offshore wind % overlap		Wave % overlap		Tidal % overlap	
	Leased	Proposed	Leased	Proposed	Leased	Proposed
Foula:						
Breeding	–	–	–	1.57	–	–
Failed	–	–	–	1.42	–	–
Hoy:						
Breeding	–	–	2.89	10.55	–	4.81
Failed	0.35	8.56	1.26	11.92	0.09	2.57

sites. The overlap of core areas with leased wave sites never exceeded 4% and the overlap with leased tidal-stream sites was never greater than 1% (Fig. 7 and Table 2).

Birds from Hoy overlapped to a greater extent with proposed sites for all energy types. Overlap with proposed offshore wind sites was only observed for birds that had successfully fledged chicks or had failed in their breeding attempts (<1 and 9% respectively). Birds of all breeding stages overlapped with proposed wave and tidal-stream sites. Incubating birds and successful breeders overlapped with proposed wave sites by 5%, whilst chick rearing birds and failed breeders overlapped by 10% and 13% respectively. For tidal-sites, incubating birds and failed breeders overlapped by less than 4%, chick-rearing birds by 7% and birds that had successfully fledged chicks by 13% (Fig. 7).

3.3. Comparison of tracking data with historical data

Historical records show an overall negative trend in great skua productivity over time on Foula (JNCC, 2014b) and Hoy (JNCC, 2014b; Meek et al., 2011). This overall decline at both colonies is evident in spite of the different regimes of breeding success in the Shetland Islands and Orkney Islands, which are driven by sandeel abundance (Cury et al., 2011; Ratcliffe et al., 1998).

In addition to the negative trend in productivity seen on Foula, there was a concurrent decline in nest attendance. For example, a

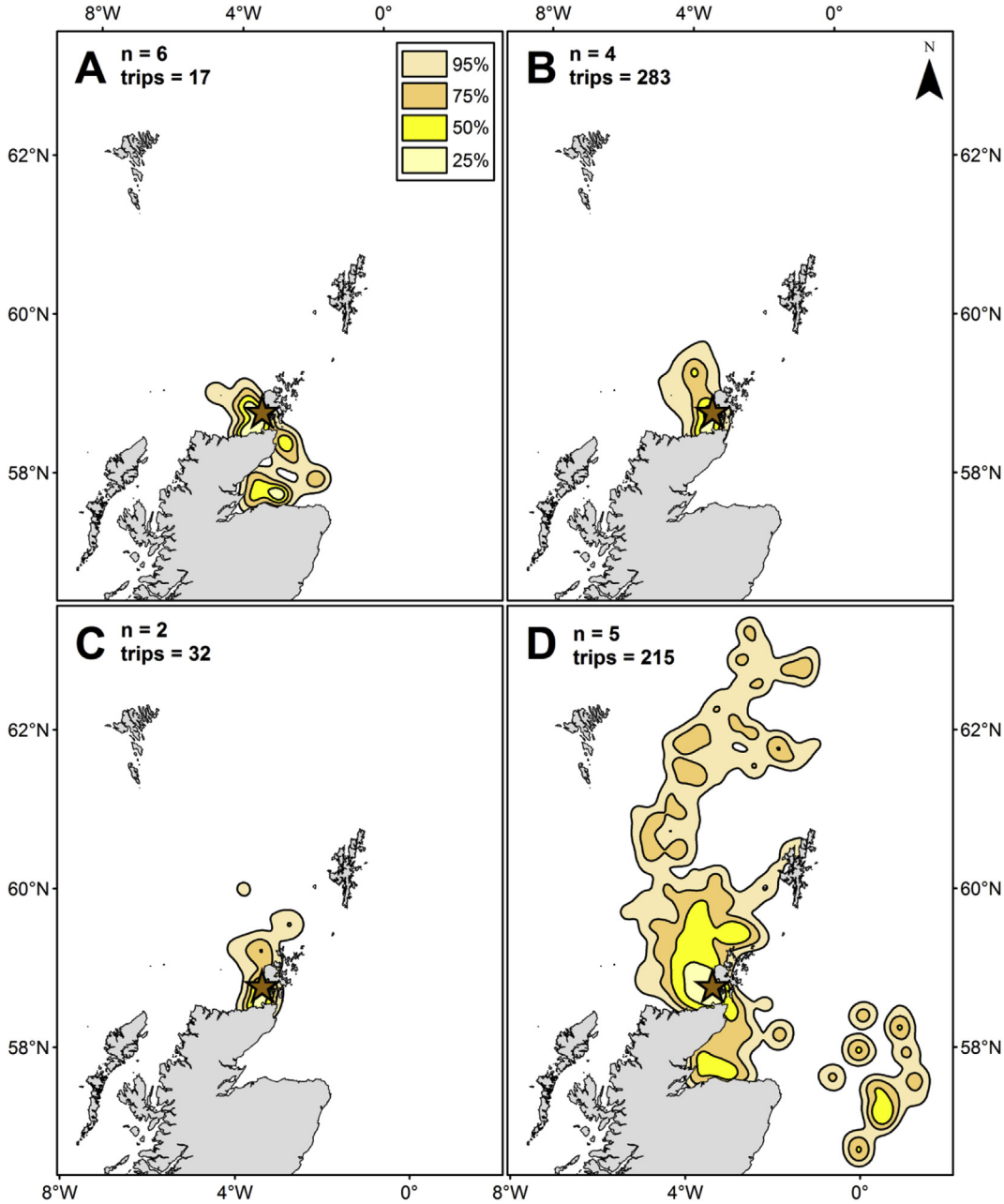

Fig. 5. Kernel density estimates of great skua movements from Hoy during incubation (A), chick-rearing (B), following successful fledging of chicks (C), and following breeding failure (D) until 16 August 2011.

mean of 1.5 birds were recorded in attendance on territory during spot checks on Foula in 1974, which had declined to 0.7 birds by 2011. There was also a large increase in the median trip duration since the 1970s; from trips lasting less than 40 min to trips lasting over 200 min in the 1990s and 2011. The mean number of trips per day has decreased over time, with 1 trip per day in 2011 compared to between 5 and 8 per day in the 1970s (Table 3).

4. Discussion

Our study is the first to track individual great skuas throughout the breeding season using GPS technology. We tracked great skuas from two colonies to improve our understanding of how this species uses the marine environment and to predict potential effects associated with MREDs. We also compared movement and productivity data during the tracking period with historical data to assess how great skua movements may have changed over time. We found that the movements of great skuas vary throughout the breeding season, with actively breeding birds remaining closer to the nest and spending less time away from the colony than birds that had failed in their breeding attempt. We also found evidence that movements differ according to breeding colony. Overall overlap with MREDs was low but historical data suggest that great skua movements have changed over time, which could have implications for predicting the long-term effects of MREDs on this species. These data also suggest that basing marine spatial planning decisions on short-term data could be less informative than longer-term data.

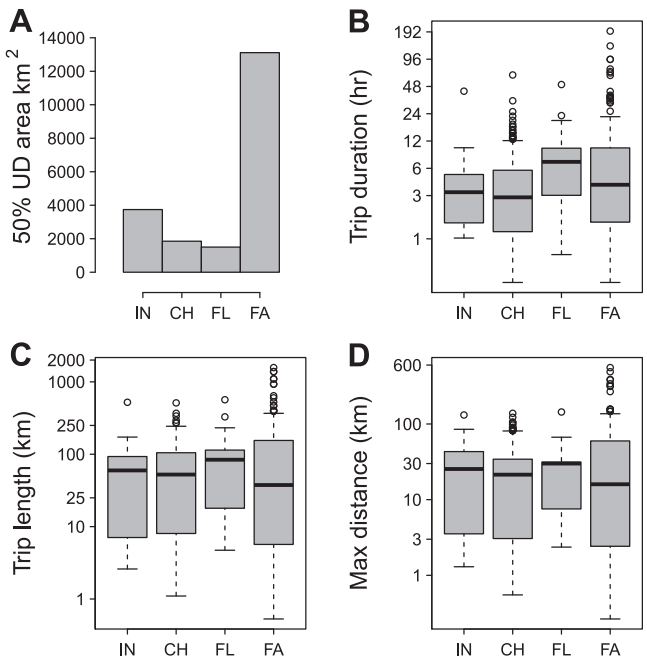

Fig. 6. Comparison of birds from Hoy during incubation (IN), chick-rearing (CH), following successful fledging of chicks (FL), and following breeding failure (FA) in: A) the core area used, B) the duration of trips, C) the length of trips and D) the maximum distance travelled from the colony.

We found that great skuas move differently through the marine environment when actively breeding compared to following a failed breeding attempt. UDs clearly show that failed breeders travelled further and ranged more widely from the colony than active breeders (Figs. 2 and 5). This is to be expected given that when breeding, birds are central place foragers constrained to regularly return to the nest to incubate eggs or provision chicks, which limits how far the bird is able to travel during foraging trips (Carlson and Moreno, 1985; Kacelnik and Cuthill, 1990). This behaviour has implications for how great skuas may be affected by MREDs. Failed breeders that range more widely may be at risk of encountering more MREDs than breeding birds and this could result in failed breeders being more affected by MREDs as a result of the cumulative effects of several development sites. Alternatively, if a MRED is located in close proximity to a colony, breeding birds restricted in their foraging range may be forced to encounter a development site more often than failed breeders, who are more flexible in their foraging movements (Everaert and Stienen, 2007; Fox et al., 2006). This could result in repeated negative effects on breeding birds, for example through displacement or collision with devices, that could affect adult survival and/or chick survival and colony productivity (Masden et al., 2010; McDonald et al., 2012). The effects of repeated encounters with specific MREDs may be compounded by the greater number of trips undertaken per day by breeding birds when compared to failed breeders; more specifically by chick-rearing individuals and birds that have successfully fledged a chick, than by incubating birds or failed breeders. By increasing the number of trips undertaken per day, adults are likely attempting to respond to additional nutritional requirements of their chicks. We found that birds that were categorised as having successfully fledged chicks appeared to be undertaking more trips than failed breeders. This may be because they were still provisioning an almost fully grown chick, fledged but still partly dependent on its parents. In addition, we suggest caution should be taken when interpreting the results of the more detailed analyses of nest status, as the categories of incubating birds and birds that successfully fledged chicks comprised small sample sizes.

Our analyses suggest that birds potentially differ in their movements according to the colony in which they are breeding. Statistical analyses suggest that when actively breeding, birds from Foula travel over greater distances, spend more time away from the nest and range more widely than actively breeding birds from Hoy (Fig. 3B–D). However, UDs indicate that the converse is true for failed breeders (Figs. 2 and 3A). Previous evidence has shown that birds from larger breeding colonies range further from the nest during foraging trips as a result of density-dependent prey depletion around the colony (Ashmole, 1963; Lewis et al., 2001; Wakefield et al., 2013). Therefore, it could be expected that movements by birds from the larger colony of Foula (Mitchell et al., 2004) would result in larger core areas of use than for birds from Hoy. Model outputs support this hypothesis (Fig. 3B–D). However, the UD calculations for failed breeders from Hoy do not follow this pattern, with failed breeders from Hoy making use of a larger core area than those from Foula. This lack of consistency between model outputs and UD calculations may be caused by the two colonies not differing in size enough for a clear effect to be seen or perhaps the proximity of prey to the colony differs according to colony location. It is also worth bearing in mind that there is greater variance in the maximal distance travelled from the nest by birds from Hoy; particularly failed breeders from Hoy (Fig. 3D). This greater variance may account for the greater UD areas calculated for failed breeders from Hoy, and is likely driven by five comparatively long trips to the north of the colony undertaken by three individuals.

We found evidence for a low overall overlap with MREDs (<13% maximum overlap with core areas used) (Tables 1 and 2), which suggests that great skuas are unlikely to be adversely affected by current development plans for MREDs if they use the marine environment in similar ways across years. UDs indicate a low overlap of core areas used with leased and proposed offshore wind development sites (<10%), which are predicted to represent a relatively high collision risk to this species, as birds tend to fly at altitudes overlapping with turbine swept areas (Figs. 4 and 7) (Furness et al., 2013). There was no overlap for birds from Foula owing to their consistent north-west direction of movement from the colony and only failed breeders from Hoy overlapped with leased and proposed sites. This could be expected given that failed breeders from Hoy ranged more widely than actively breeding birds thus increasing their chances of overlap. Despite this, the percentage of overlap with offshore wind sites was still low in comparison to the core area over which birds ranged. We found a low overlap of birds from Hoy with leased wave development sites (<4%) but a greater overlap with proposed sites; particularly for failed breeders. Birds from Foula only overlapped to a small degree with proposed wave development sites (<2%), again owing to their consistent movements to the north-west of the colony. A similar low overlap was observed with tidal-stream developments; with no overlap by birds from Foula and overall low levels of overlap with both leased and proposed sites by birds from Hoy (<13% maximum overlap). This low overlap with tidal-stream sites is not unexpected given the small area that these sites cover. In addition to the low levels of overlap with wave and tidal-stream development sites, these renewable energy types are predicted to present a low level of risk to great skuas (Furness et al., 2012). This is largely owing to the devices' limited profile above sea level, which as great skuas are surface feeders, lessens the risk of collision (Grecian et al., 2010). In addition, great skuas exhibit a low response to disturbance by structures and vessel traffic (Furness et al., 2012). Wave and tidal-stream developments are therefore unlikely to negatively affect great skuas. Conversely, it is possible that MREDs, particularly wave developments, could benefit breeding great skuas if they act

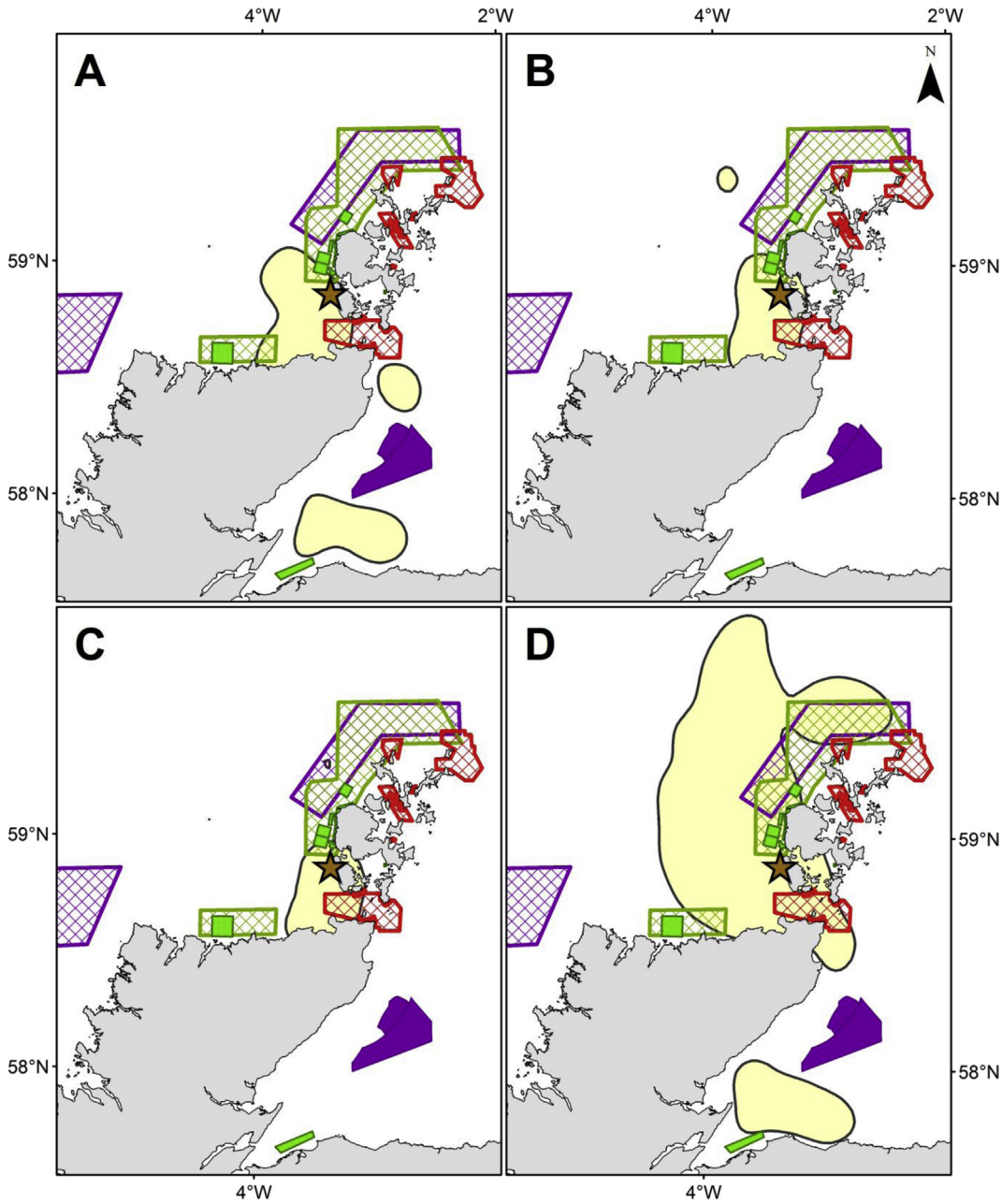

Fig. 7. Overlap of great skua core areas of use (50% utilisation distribution) with leased and proposed marine renewable energy development sites (solid polygons = leased sites; cross-hatched = proposed sites; purple = offshore wind sites; green = wave sites; red = tidal-stream sites) for birds from Hoy during incubation (A), chick-rearing (B), following successful fledging of chicks (C), and following breeding failure (D). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

as FADs, artificial reefs or *df*MPAs. For example, if fish abundance increases within the area of MREDs, as has been observed at some offshore wind farms (Reubens et al., 2013a), great skuas could benefit from accumulated prey in close proximity to the colony (Russell et al., 2014). This has the potential to reduce foraging effort, decrease the time spent away from the nest and subsequently increase productivity. In this study we investigate the movements of adult great skuas during the breeding season; however, there is extremely limited knowledge of the movements of juvenile birds, which therefore restricts predictions regarding how this part of the population may be affected by MREDs. In addition, interactions with MREDs as birds migrate should also be incorporated into predictions regarding the cumulative impacts of MREDs on great skuas throughout their annual cycle.

When comparing the tracking data with historical data there is evidence that great skua movements have changed over time. Chick rearing birds from Foula are spending more time away from the nest than in the 1970s, and are undertaking fewer trips per day. A concurrent decline in the levels of adult attendance at the nest and a reduction in annual productivity was also found (Table 3). Cairns (1992, 1988) and Montevecchi (1993) have concluded that changes in activity budgets are good indicators of changes to marine food supplies. In particular, previous research has shown that increased time spent away from the colony on foraging trips is linked to low prey availability and that a decline in the attendance of adults on

Table 2
Percentage of overlap of core areas used (50% utilisation distribution) with leased and proposed offshore wind, wave and tidal-stream renewable energy developments for birds from Hoy during incubating, chick rearing, once chicks had fledged and once birds had failed in their breeding attempt.

Breeding status (Hoy)	Offshore wind % overlap		Wave % overlap		Tidal % overlap	
	Leased	Proposed	Leased	Proposed	Leased	Proposed
Incubating	–	–	0.43	5.02	–	3.54
Chick-rearing	–	–	3.10	9.64	0.08	7.17
Fledged	–	0.47	1.14	4.99	0.73	12.43
Failed	–	8.77	1.30	12.99	0.14	3.23

2014; Hamer et al., 2007; Pettex et al., 2012; Thaxter et al., in review-b), which will influence how seabirds are affected by MREDs. Soanes et al. (2013) and Taylor et al. (2004) also provide evidence that sample sizes of tracked individuals and the number of trips incorporated in analyses can affect the extent of the area identified when calculating UDs, which should be borne in mind when interpreting this type of data. These issues highlight a challenge to, and a shortcoming of using short-term tracking studies in isolation to predict the effects of MREDs on seabird species and incorporating them as a basis for marine spatial planning. A potentially useful method to address some of these issues is species

Table 3
Comparison between 2011 tracking data and historical data where trip durations were available (all data from Foula unless specified). Annual productivity, attendance and the number of trips undertaken per day shown. All data are for chick-rearing individuals only. 25th and 75th refer to the 25th and 75th percentile of the interquartile range.

Year	Productivity (mean number of chicks per pair)			Attendance Foula (mean number of adults on territory)	Trip duration (m) Foula				Trips per day Foula			*n*
	Hoy (Stourdale)	Hoy (Grutfea)	Foula		25th	Median	Mean	75th	Median	Mean	Max	
1974			1.20	1.50	27.00	37.00	39.86	48.50	9.00	8.30	16.00	10
1976			1.30	1.47	18.50	28.00	64.55	76.00	5.50	5.50	8.00	2
1994		0.61	0.80	1.20		210.00						6
1995		0.93	1.10	1.30		210.00						6
2011	0.50		0.21	0.70	148.00	245.00	357.70	466.00	1.00	1.32	5.00	9

their territories reflects an increased difficulty in finding food (Boyd, 1999; Caldow and Furness, 2000; Furness and Hislop, 1981; Hamer et al., 1991). Other research suggests that trips per day can be used as a proxy for prey delivery rate (Wakefield et al., 2013). Based on this previous research we can conclude that chick rearing great skuas on Foula are currently experiencing lower prey availability than in the 1970s. As Chivers et al. (2012) found in black-legged kittiwakes (*Rissa tridactyla*), this decline in prey availability is forcing breeding birds to travel greater distances to locate sufficient food to sustain themselves and their chicks, affecting how many trips per day a bird can undertake, and thus reducing the rate of prey delivery to the chick. Increased foraging effort and the subsequent decrease in parental investment has been found to be directly linked to offspring mass and survival (Elliott et al., 2009; Kuhn et al., 2014), which could be reflected on Foula in the overall decline in annual productivity over time (Table 3). In addition to prey availability affecting chick survival, a reduction in nest attendance by adults has previously been proven to increase predation of eggs and chicks by conspecifics and other seabird species (Caldow and Furness, 2000; Hamer et al., 2007, 1991). This further compounds declines in annual productivity by further reducing chick survival and is likely occurring on Foula. We suggest that great skuas may have reached an energetic maximum in terms of effort available to invest in foraging, as it is clear that despite an increase in foraging effort over time, productivity has failed to improve and populations are continuing to decline (Meek et al., 2011; Mitchell et al., 2004). Based on this evidence great skuas appear to be a species under pressure and therefore vulnerable to additional threats, which makes them potentially sensitive to negative effects of MREDs.

Given the change that we found in the movements of great skuas during the breeding season since the 1970s, it is likely that areas identified as important in short-term tracking work may alter in the future. This is important to consider when using tracking data to inform marine spatial planning. In addition to long-term changes in movement behaviour, previous studies provide evidence for seasonal and annual variation in the movements of seabird populations and the core areas used (Bogdanova et al.,

distribution modelling (SDM). Arcos et al. (2012) provide evidence that SDMs can produce useful predictions of seabird hotspots for marine spatial planning even when incorporating only one year of data. However, the authors also acknowledge that SDM performance can differ according to species and that current rapid climatic change could limit the predictive power of such models. We advocate that although tracking data can be useful in marine spatial planning and Environmental Impact Assessments, where possible tracking studies should be conducted over several years, in conjunction with monitoring of populations and distributions at site specific and colony levels to fully assess effects.

Acknowledgements

This project was jointly funded by the Marine Renewable Energy and the Environment (MaREE) project (funded by Highlands and Islands Enterprise, the European Regional Development Fund, and the Scottish Funding Council) and the Department of Energy and Climate Change (DECC). We would particularly like to thank Eliza Leat, Chris Booth and Andy Knight, and the numerous volunteers who assisted with fieldwork and data collection. We are also very grateful to those who have contributed to discussions regarding analysis methods; particularly Paul Johnson and James Grecian, and two anonymous referees for constructive comments on a draft of this manuscript. We thank the RSPB and Scottish Natural Heritage for permission to carry out fieldwork.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.marenvres.2014.09.003.

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