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Passive acoustic monitoring of baleen whales using autonomous gliders in relation to offshore wind energy areas in the New York Bight

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ABSTRACT: The rapid expansion of offshore wind development in the northeast USA has amplified the need to understand baseline patterns of habitat use for protected species relative to offshore wind lease areas. Here, we used 5 years of intermittent passive acoustic monitoring via autonomous glider deployments to monitor baleen whales in the New York Bight, a region of the US east coast where offshore wind construction is imminent. We assessed the daily presence of humpback *Megaptera novaeangliae*, sei *Balaenoptera borealis*, North Atlantic right *Eubalaena glacialis*, and fin *B. physalus* whale vocalizations, and examined spatial (10s—100s of km) and temporal (seasonal) habitat use patterns in the wind energy areas and the broader New York Bight region. All species were observed year-round in the New York Bight, with humpback and sei whales detected most frequently in spring, and right whales detected more frequently in winter. Humpback and North Atlantic right whales were detected closer to the coast in winter than in other seasons. The overlap between whale habitat and wind energy areas was greatest in winter for all species except for sei whales, which showed similar overlap in summer and winter. This new understanding of the overlap of baleen whale habitat use relative to offshore wind energy areas illustrates the utility of routine seasonal glider surveys as passive acoustic monitoring platforms. Furthermore, this information will be helpful for developing appropriate mitigation and management strategies to minimize impacts of wind energy development on baleen whales.

KEY WORDS: New York Bight · New Jersey · Wind energy areas · North Atlantic right whale · Humpback whale · Sei whale · Fin whale

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1. INTRODUCTION

A growing human population and the increasing impacts of climate change have amplified the demand for renewable energy resources (Bilgili et al. 2011, Sun et al. 2012, Li et al. 2020). Offshore wind energy development has become an increasingly popular renewable energy source (Sadorsky 2021), with global capacity nearly doubling between 2020 and 2024 (McCoy et al. 2024). Considerable offshore wind development is currently underway on the continental shelf along the east coast of the USA, with 62 959 MW of energy-generating capacity planned, under construction, or in operation. A total of 50 areas along the coast of the eastern US have been leased to developers, with an additional 19 planning areas on and off the continental shelf from South Carolina to Maine (McCoy et al. 2024). As of May 2024, 174 MW (<0.01%) were operational in 3 wind energy areas (WEAs) along the US east coast (McCoy et al. 2024). Areas representing 5831 MW (9.3%) were under construction, 1644 MW (2.6%) were approved, and 55 310 (87.8%) MW were in permitting or surveying stages (McCoy et al. 2024).

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Given the rapid expansion of offshore wind in US waters, there is a growing need to understand its impacts on marine organisms (Masden et al. 2010, Garel et al. 2014, Lindeboom et al. 2015, Gill et al. 2020, Mooney et al. 2020, Perry & Heyman 2020, van Berkel et al. 2020). Understanding patterns of baleen whale habitat use in and around WEAs has been of particular interest as they are protected under the US Marine Mammal Protection Act and, for several species, the Endangered Species Act (ESA). Baleen whale species are highly mobile, with some individuals migrating long distances between low-latitude breeding grounds and mid- to high-latitude foraging grounds, while others remain on the foraging grounds year-round (Lockyer & Brown 1981, Stevick et al. 2003, Geijer et al. 2016, Kowarski et al. 2018). They may therefore be impacted by multiple WEAs during annual migrations, as well as in their foraging and breeding grounds. Furthermore, multiple species of baleen whale inhabit the coastal zones identified for wind farm development in US waters for all or a portion of their life cycles (Van Parijs et al. 2021). Understanding potential overlap between wind development and baleen whale habitat use is a priority for management and mitigation.

The assessment of WEAs and the construction of wind turbines produce sounds that could impact baleen whales (Van Parijs et al. 2023). Seismic surveys during assessment and pile driving during construction may result in habitat avoidance by marine mammals (Tougaard et al. 2003, Madsen et al. 2006). Additionally, vessel traffic and noise associated with operational wind turbines could increase the risk of masked acoustic communication (Bailey et al. 2014, Van Parijs et al. 2023) and vessel strikes, a major cause of mortality in baleen whales (Vanderlaan & Taggart 2007, Schoeman et al. 2020, Barkaszi et al. 2021). Operational turbines may also influence the distribution of zooplankton prey by altering local current dynamics (Floeter et al. 2017, van Berkel et al. 2020, National Academies of Sciences, Engineering, and Medicine 2024). Collecting baseline data on the occurrence and distribution of baleen whales prior to the construction of WEAs is critical to understanding, quantifying, and assessing potential impacts of offshore wind development (Estabrook et al. 2022, 2025, Van Parijs et al. 2023).

The abundance and distribution of baleen whales has traditionally been estimated using standardized visual line transect surveys from ships or aircraft (Buckland et al. 2001, Hammond et al. 2021). These surveys count individuals along set transect lines and can be used to develop density and population size estimates for baleen whales (Buckland & Turnock 1992, Buckland et al. 2001, Buckland & York 2018). Ship-based or aerial surveys rely on the visual detection of animals at or near the surface of the water, which is dependent on factors such as sea state and animal behavior (Eberhardt et al. 1979, Beavers & Ramsey 1998, Kinzey et al. 2000, Barlow 2015).

Passive acoustic monitoring (PAM) can also be used to detect baleen whale acoustic occurrence and, when applied across a sufficient spatial extent, distribution (Zimmer 2011, Browning et al. 2017, Davis et al. 2017, 2020, Macrander et al. 2022). PAM uses hydrophones to record whale vocalizations, which can be identified through manual or automatic annotation methods (Browning et al. 2017, Fleishman et al. 2023). While marine mammal presence can be estimated from PAM, assessing the true absence of animals can be a challenge due to animal behavior and environmental conditions. Individuals must be vocalizing within a certain range of the hydrophone, and their calls must be sufficiently loud, in comparison to background noise, to be detected (Zimmer 2011, Parks et al. 2011, Browning et al. 2017, Estabrook et al. 2022, Fleishman et al. 2023).

Monitoring marine mammals has traditionally required a trade-off in spatial versus temporal extent. Visual surveys can cover wide areas, but the significant amount of person-hours and vessel- or plane-hours required can limit the temporal frequency and duration of surveys (Aragones et al. 1997, Thomas et al. 2007, Hammond et al. 2021, Laake et al. 2021). PAM recorders have typically been either deployed at fixed stations or towed from vessels. Fixed stations allow data to be collected continuously for months or years at a time at specific locations with minimal ship time, but their spatial extent is limited to the detection ranges around the instruments (Laurinolli et al. 2003, Baumgartner & Fratantoni 2008, Clark et al. 2010, Cholewiak et al. 2018, Davis et al. 2020). Towed PAM deployments from vessels provide a snapshot of marine mammal occurrence in time across a wider spatial extent (Van Parijs et al. 2009), but require and are limited by ship time, similar to line transect surveys. Regardless of the platform, PAM monitoring can, however, facilitate observations across a wider range of sea states than visual surveys (Zimmer 2011, Browning et al. 2017, Macrander et al. 2022).

Gliders have emerged in recent decades as PAM platforms providing a balance of spatial and temporal coverage (Moore et al. 2007, Baumgartner & Fratantoni 2008, Klinck et al. 2012, Johnson et al. 2022). Gliders are a class of autonomous underwater vehicle (AUV) that move through the water column by changing their buoyancy. Buoyancy-driven gliders consume little power, meaning that they can be deployed for weeks or months at a time, depending on the sensors being used. They move slowly (~15 km per day on average) along a pre-determined track (Schofield et al. 2007, Rudnick 2016). This allows them to provide near-continuous monitoring across hundreds of kilometers in a single mission (Schofield et al. 2007, Rudnick 2016). These platforms require minimal vessel time for deployment and recovery, as well as fewer resources (i.e. person-hours) during operation than traditional scientific vessels (Schofield et al. 2007, Rudnick 2016). They also survey continuously, regardless of sea state. For these reasons, gliders provide advantages for monitoring efforts within and around WEAs (Baumgartner et al. 2013, 2020, Johnson et al. 2022, Indeck et al. 2025).

Here, we assessed baleen whale detections from gliders deployed in and around WEAs in the New York Bight (NYB) on the east coast of the USA from 2020 to 2024. The NYB is bounded by the coastline between Cape May, New Jersey, and Montauk, New York, out to the continental shelf break (Ketchum et al. 1951; see Fig. 1). Eleven WEAs are present within the western NYB, and construction in these areas had not yet begun as of summer 2025. Together, these areas have the potential to deliver 19 703 MW of power, representing 31% of the total estimated energy from offshore wind projects currently planned on the US east coast (McCoy et al. 2024).

The NYB serves as migratory and foraging habitat for several baleen whale species (Davis et al. 2017, 2020, Muirhead et al. 2018, King et al. 2021, Zoidis et al. 2021, Chou et al. 2022, Murray et al. 2022, Estabrook et al. 2025). We focused on the following species that occur regularly in the NYB: humpback *Megaptera novaeangliae*, sei *Balaenoptera borealis*, North Atlantic right *Eubalaena glacialis* (NARW), and fin *B. physalus* whales. Of these, sei and fin whales are endangered and NARWs are critically endangered (Nieukirk et al. 2012, Edwards et al. 2015, National Oceanic and Atmospheric Administration 2022, 2023). While humpbacks are not endangered under the ESA, the West Indies distinct population segment that occupies the NYB has been experiencing an unusual mortality event since 2017 (National Oceanic and Atmospheric Administration 2020). Humpback and fin whales occupy the region year-round, with humpbacks occupying waters across the continental shelf (<30–200 m depth, within 20 km of shore) and fin whales occupying waters closer to the continental shelf break (200 m isobath; Morano et al. 2012, Muirhead et al. 2018, Davis et al. 2020, Zoidis et al. 2021). Sei and NARWs are less common in the NYB; previous acoustic and visual surveys suggested that sei whales occupy mid-shelf and offshore waters, whereas NARWs are found closer to shore from spring to late fall (Davis et al. 2020, Zoidis et al. 2021).

We describe spatial (10s–100s of km) and temporal (seasonal) patterns in the occurrence of these species in the NYB and examine the utility of gliders as monitoring platforms in WEAs. Our specific objectives were to quantify: (Objective 1) spatial and (Objective 2) seasonal patterns in baleen whale acoustic detections; and (Objective 3) the spatiotemporal overlap of baleen whale detections and WEAs. We hypothesized that (1) all species would be detected furthest from shore in the summer months, (2) season would have a significant impact on the number of detection days for each species, following previous visual and PAM surveys (Davis et al. 2017, 2020, Muirhead et al. 2018, Zoidis et al. 2021), and (3) baleen whales would be detected at the same frequency within WEAs within the study region, indicating no preference for WEAs.

2. MATERIALS AND METHODS

2.1. Glider deployments and survey effort in WEAs

We conducted a total of 29 missions with Slocum gliders (Teledyne Webb Research) in and around WEAs in the NYB between July 2020 and August 2024 (Fig. 1, Table 1; Table S1 in the Supplement at www.int-res.com/articles/suppl/n058p257_supp.pdf). Across the 29 missions, gliders were deployed for a total of 748 d and covered 14 338 km of trackline (Fig. 1, Table 1; Table S1). All gliders were equipped with integrated digital acoustic monitoring (DMON; Johnson & Hurst 2007, Baumgartner et al. 2013) instruments that recorded audio continuously at 2 kHz. The glider deployments used here were from a variety of projects, some of which were focused on broad surveys in the NYB, and some of which specifically targeted WEAs. Of the missions analyzed here, 17 targeted WEAs directly, and the remaining 12 surveyed the region broadly. The missions that targeted WEAs were designed to maximize time in the WEAs.

Forward propulsion in Slocum gliders is provided by the buoyancy engine and the horizontal wings, and steering is provided by the rudder on the tail (Schofield et al. 2007, Rudnick 2016). The movement of the buoyancy pump adds noise to the audio. Noise is also introduced via surfacing events that occur every 2–3 h. During these events, the glider remains at the surface to send or receive data to/from shore, during

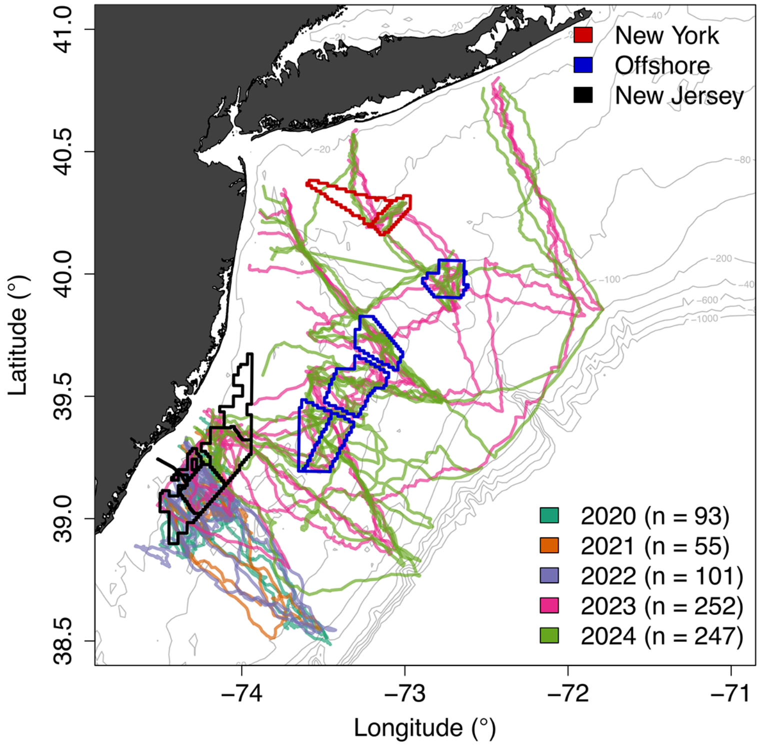

Fig. 1. New York Bight study region and glider tracks for the 29 glider missions used to analyse baleen whale occurrence. Glider tracks are colored by year of deployment, with n representing the number of deployment days per year. Wind energy areas (WEAs) are outlined according to subregion; depth contours are in 20 m increments

which time waves breaking near and over the glider as well as electromagnetic interference from the satellite data transmissions can induce noise. Most noise is short in duration, lasting for only a few seconds, apart from surfacing events which can last as long as 20 min, but only occur every 2–3 h. This represents approximately 10% of the deployment time, which aligns with previous studies on other glider models (Cauchy et al. 2023). Therefore, PAM only occurs when the instrument is gliding and silent, which makes up a majority (~90%) of the deployment time (Cauchy et al. 2023).

WEA shapefiles were taken from the Bureau of Ocean Energy Management (BOEM) website (https://www.boem.gov/renewable-energy/mapping-and-data/renewable-energy-gis-data, accessed November 2023; Fig. 1). WEAs were grouped by geographic location into 3 subgroups: New York (NY), Offshore, and New Jersey (NJ) (Fig. 1). Total survey effort across time (in days) and space (in km) within each of these subgroups is shown by season in Table 1. Seasons were defined as winter (December through February), spring (March through May), summer (June through August), and fall (September through November).

2.2. Acoustic data processing

DMON hydrophones were encapsulated in a urethane housing on the top of the glider science bay (Baumgartner et al. 2020). One of these hydrophones measured at low frequencies (8–7500 Hz bandwidth, 36 dB re µPAa/√Hz noise floor at 2 kHz, and −169 dB re V/µPa total system sensitivity) (Baumgartner et al. 2013). The DMON's integrated programmable digital signal processor and flash memory continuously recorded audio at 2 kHz sample rate, and processed those audio files using the Low Frequency Detection and Classification System (LFDCS; Baumgartner & Mussoline 2011, Baumgartner et al. 2020) to identify and report to the glider detections of whale calls in real

Table 1. Glider survey effort in time and distance throughout the New York Bight study region and the 3 wind energy area (WEA) subregions defined in Fig. 1

Region	Season	Survey effort (d)	Survey effort (km)	Survey years	No. of missions
New York Bight	Winter	175.2	3417.4	2020–2024	29
	Spring	262.6	5075.1		
	Summer	182.9	3497.9		
	Fall	127.2	2347.2		
	Total	747.8	14337.7		
New Jersey WEAs	Winter	31.6	558.1	2020–2024	23
	Spring	26.8	453.1		
	Summer	28.4	527.5		
	Fall	29.9	538.4		
	Total	116.7	2077.1		
Offshore WEAs	Winter	5.8	114.4	2023–2024	18
	Spring	14.3	274.8		
	Summer	24.4	490.6		
	Fall	8.2	179.9		
	Total	52.9	1059.8		
New York WEAs	Winter	0.8	14.1	2023–2024	6
	Spring	3.4	64.6		
	Summer	3.1	46.3		
	Fall	1.8	40.6		
	Total	9.1	165.7		

time. These real-time detections were not used in this study, but rather a desktop computer implementation of the LFDCS was used to process the recorded audio after the glider was recovered. Using archival audio has been shown to improve both false and missed detections rates since an analyst can use the audio (both aurally and visually via spectrograms) to improve the overall detection process (Baumgartner et al. 2020).

The LFDCS creates spectrograms of acoustic data using short-time Fourier transforms (Baumgartner & Mussoline 2011, Baumgartner et al. 2020). Contour lines are then automatically drawn through tonal sounds to characterize variation in pitch and amplitude over time. Multivariate discriminant function analysis is used to assign each contour line, or 'pitch track', to a call type based on a call library (Baumgartner & Mussoline 2011). We used the desktop computer LFDCS implementation described above to generate pitch tracks using 2 sets of LFDCS parameters and call libraries: one optimized to detect humpback, sei, and NARWs, and a second optimized to detect fin whales. Our LFDCS parameters and call libraries are the same as those used by Davis et al. (2017, 2020). The call libraries are described in Text S1. All calls were manually confirmed by a trained acoustic analyst for all species.

We used confirmed calls to determine daily acoustic presence and absence of baleen whale vocalizations. We determined if humpbacks, sei whales, and NARWs were present or absent on a given day using previously published thresholds (number of confirmed detections within a given time frame) and generated a new threshold for fin whales based on previously published protocols and thresholds (Davis et al. 2017, 2020). These daily presence thresholds are described in Text S2.

Timestamps of confirmed calls were matched with the glider clock to estimate the location of the glider at the times of detections. Glider position was linearly interpolated between surfacing events at which position was measured with the glider's GPS receiver. Confirmed calls were only matched to the glider data if baleen whales were considered present on that day following the definitions described in Text S2.

The LFDCS, like any detector, will have a missed and false detection rate. Therefore, we validated the performance of the LFDCS by performing a manual analysis on a subset of the data following previously established protocols (Baumgartner et al. 2019, 2020, Davis et al. 2020). Every 10th day was manually screened for daily presence of the target species in each glider deployment longer than 10 d to ensure days were screened across all seasons and geographic regions. A total of 57 days were screened. Missed and false detection rates, as well as detector accuracy, were calculated on the resulting confusion matrix of daily acoustic presence and acoustic absence of each species following methods from Baumgartner et al. (2019).

2.3. Objective 1: Spatial distribution of baleen whale occurrence

To quantify spatial patterns of baleen whale habitat use within the NYB, we considered (1) spatial autocorrelation of whale daily acoustic presence and absence and (2) sound transmission in the water. Detecting a species in one location along a glider track makes it more likely that the species is detected at nearby locations. Therefore, we must account for spatial autocorrelation in our analysis. We tested for spatial autocorrelation using the 'ncf' package in R v. 4.3.1. We used the position of the glider at noon each day and daily acoustic presence/absence (following methods of Silva et al. 2019). We calculated the distance at which spatial autocorrelation became zero (*x*-intercept), which we refer to as the spatial decorrelation scale, for each glider mission. We then determined the 95th percentile of the spatial decorrelation scale across all glider missions (Fig. S1).

The spatial autocorrelation of daily acoustic presence and absence theoretically accounts for the detection ranges of the whale calls, which vary from 10 to 40 km (Newhall et al. 2012, Cholewiak et al. 2018, Baumgartner et al. 2019, Davis et al. 2020, Johnson et al. 2022). For example, a whale with a smaller detection range (such as a NARW) will have a lower spatial decorrelation scale since it will likely not be detected multiple times by a glider over the distance the glider travels in a day (~15 km). To test this assumption, we performed a simple detection range calculation for each species under best case oceanic and background noise scenarios. This analysis is described in Text S3. Ultimately, the spatial decorrelation scales were used to determine the spatial resolution (i.e. grid size) of our analysis for each species as they were greater or equal to our detection range estimates, and were more similar to previously published detection range estimates (Newhall et al. 2012, Cholewiak et al. 2018, Baumgartner et al. 2019, Davis et al. 2020, Johnson et al. 2022).

This analysis resulted in grid sizes of 23 × 23 (529), 16 × 16 (256), 12 × 12 (144), and 38 × 38 (1444) km × km (km^2) for humpback, sei, NARW, and fin whales, respectively (Fig. S1). Since effort is not evenly distributed across grid cells, we summed the distance surveyed within a grid cell seasonally across all missions (Figs. S2–S5). We then normalized the number of days baleen whales were detected by the cumulative effort within each species-specific grid.

To quantify seasonal shifts in baleen whale distributions relative to the coastline, we calculated the distance between grid cell centroids and the coast using the function 'dist2Line' in the R package 'geosphere' (Hijmans 2022). The interquartile range (25th–75th quartile) of the distances to shore were 39–112, 41–106, 41–102, and 38–112 km for the humpback, sei, NARW, and fin whale specific grids, respectively. This illustrates that our sampling design sufficiently covers the continental shelf in this region, but undersamples inshore (<40 km from shore) and offshore (>100 km from shore) waters.

We performed an ANOVA to compare how far grid cells with baleen whale detections were from shore across seasons. We weighted the distance to shore by the number of days baleen whales were detected normalized by survey effort described above to account for both survey effort and the number of baleen whale detection days within each grid cell. We also calculated a weighted mean distance to shore. Assumptions were tested visually using residual and Q–Q plots, and Tukey post hoc tests were performed as needed.

2.4. Objective 2: Quantifying seasonal patterns in baleen whale distributions

To examine temporal variability in baleen whale detections, we examined trends normalized by effort in both time (distance surveyed) and space (days surveyed). First, we used an ANOVA to test if the number of detection days in a model grid cell differed across seasons. We normalized the number of detection days by the distance surveyed within each grid cell per month to account for differences in effort spatially. Distance surveyed was log-transformed to meet normality and variance assumptions. We performed this test seasonally, rather than monthly, due to high month-to-month variability and to prevent zero-inflating the data set. Assumptions were tested visually using residual and Q–Q plots, and Tukey post hoc tests were performed as needed.

Second, we examined temporal trends, normalized in time by the number of survey days. We calculated the proportion of glider deployment days per month in which whales were detected for each mission and target species. We averaged and calculated standard deviations of these proportions monthly across glider missions. We did not perform a statistical test on these data due to high month-to-month variability.

2.5. Objective 3: Quantifying the spatial overlap and temporal patterns of whales in WEAs

To quantify the spatial overlap between whale detections and WEAs, we calculated the percentage of surveyed grid cells which had whale detections, both across the entire survey area and within WEAs exclusively. We calculated these percentages both for the entire year and by season. All grid cells that overlapped with WEAs were considered, regardless of the amount of overlap. We calculated the number of grid cells with detections for a given species within WEAs divided by the number of grid cells with detections across the entire survey area. We called this metric 'relative habitat overlap'. We performed this analysis both with all WEAs pooled and for each WEA subregion.

To test potential habitat preference for WEAs, we used chi-squared goodness of fit tests to compare the observed number of grid cells within WEAs with and without baleen whale detections to the expected proportions. These expected proportions were based on the proportion of grid cells with detections across the NYB and the number of surveyed grid cells within WEAs. WEAs were pooled across subregions for chi-squared tests since the number of grid cells differed significantly across geographical subregions.

3. RESULTS

3.1. LFDCS performance

Daily baleen whale presence and acoustic absence from the LFDCS and manual analysis generally agreed (Table 2). Accuracy was greatest for NARWs (98%), followed by sei (91%), fin (88%), and humpback (86%) whales (Table 2). Humpback whales had the highest missed detection rates (20%), followed by fin whales (17%), sei whales (13%), and NARWs (0%) (Table 2). False detection rates were highest for sei whales (13%) followed by NARWs (10%), humpback whales (3%), and fin whales (0%) (Table 2).

3.2. Spatial distribution of baleen whales in the NYB

Humpback whales were detected on the continental shelf over all seasons, but their locations and rates

Table 2. Comparison of Low-Frequency Detection and Classification System (LFDCS) performance to manual analysis on a subset of the data by a trained analyst. Analysis was conducted on daily acoustic presence/absence data collected using underwater gliders to monitor baleen whale occurrence. Missed and false detection rates and accuracy are calculated following the protocols of Baumgartner et al. (2019)

Species	LFDCS analysis	Manual analysis Detected	 Not detected	Missed detection rate (%)	False detection rate (%)	Accuracy (%)
Humpback	Detected	28	1	20	3	86
	Not detected	7	21			
Sei	Detected	13	2	13	13	93
	Not detected	2	40			
North Atlantic right whale	Detected	9	1	0	10	98
	Not detected	0	47			
Fin	Detected	34	0	17	0	88
	Not detected	7	16			

of detection varied seasonally. Detections were significantly closer to shore in winter (Fig. 2; Table S3; ANOVA, $F_{3,166} = 6.9$, $p < 0.01$). Humpbacks had the widest spatial distribution in the spring, ranging from near-shore to the continental shelf break and occurred in 83% of surveyed grid cells (Fig. 2, Table 3).

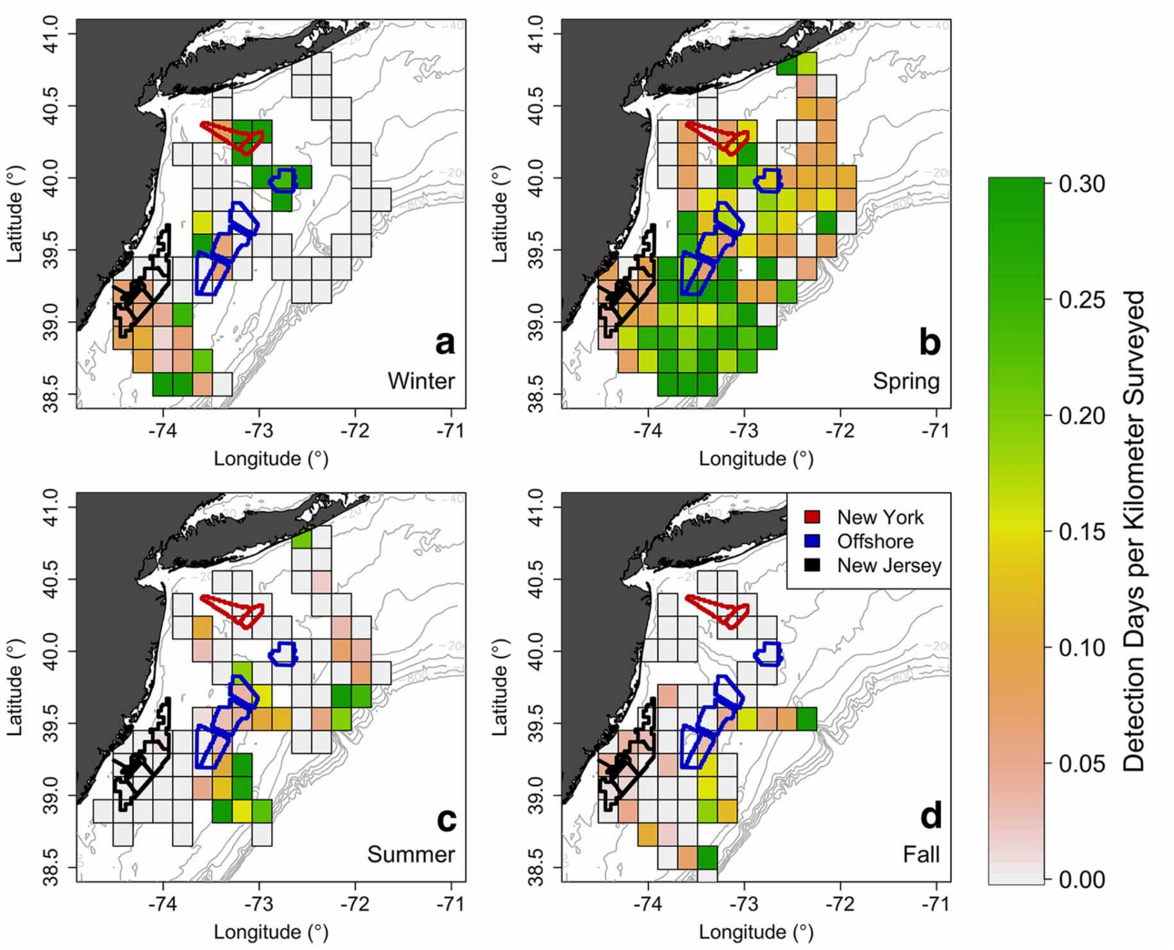

Fig. 2. Number of days humpback whales were detected per kilometer surveyed across seasons: (a) winter; (b) spring; (c) summer; (d) fall. Wind energy areas are colored by subregion following Fig. 1. White grid cells reflect areas that were surveyed where no humpback whales were detected. Grid cells are 529 km^2 (23 km × 23 km), as determined by spatial autocorrelation analyses

Table 3. Percent of surveyed grid cells with baleen whale detections across the study area and within all wind energy areas (WEAs) combined for all baleen whale species across seasons. Superscripts indicate periods when the number of grid cells within WEAs with detections was significantly ($p \leqslant 0.05$) greater (G) or less (L) than expected as determined by a chi-squared goodness of fit test. Relative habitat overlap is calculated as the percentage of grid cells with baleen whale detections within WEAs. All grid cells that overlapped with WEAs were considered, regardless of the amount of overlap

Species	Season	No. of grid cells			No. of grid cells within WEAs			Relative habitat overlap (%)
		Surveyed	With detections	Percent	Surveyed	With detections	Percent	
Humpback	All	123	95	77	34	33	97^{G}	35
	Winter	76	29	38	30	16	53	55
	Spring	106	88	83	33	30	91	34
	Summer	82	30	37	28	10	36	33
	Fall	63	23	37	25	10	40	43
Sei	All	220	83	38	59	25	42	30
	Winter	127	8	6	51	4	8	50
	Spring	181	80	44	54	23	43	29
	Summer	138	10	7	53	5	9	50
	Fall	103	3	3	45	2	4	67
North Atlantic right whale	All	349	94	27	80	31	39^{G}	33
	Winter	184	31	17	68	20	29^{G}	65
	Spring	276	42	15	75	14	19	33
	Summer	201	29	14	68	3	4^{L}	10
	Fall	157	19	12	62	8	13	42
Fin	All	55	48	87	18	18	100	38
	Winter	41	29	70	17	15	88	52
	Spring	47	39	83	18	16	89	41
	Summer	42	24	57	17	8	47	33
	Fall	35	23	65	17	11	65	48

They had the narrowest distribution in fall, when humpbacks occurred primarily in offshore waters (Fig. 2, Table 3).

Sei whales were generally detected in mid- to outer regions of the continental shelf (Fig. 3) and were the most widespread in the spring, occurring in 44% of surveyed grid cells (Table 3, Fig. 3). Distance to shore varied significantly with season (Table S3; ANOVA, $F_{3,97} = 2.7$, $p = 0.05$), but only differed between winter (89 km) and spring (104 km; $p < 0.05$, Table S3).

NARWs were detected in the largest proportion of surveyed grid cells in the winter, with detections in 17% of grid cells, though spring and summer showed similar proportions (Table 3). Detections were also significantly closer to shore in winter (Table S3; ANOVA, $F_{3,117} = 19.5$, $p < 0.01$). NARWS were detected across the continental shelf in spring (Fig. 4). Summertime NARW detections were only offshore, while fall detections occurred sporadically across the continental shelf, from coastal waters to Hudson Canyon (Fig. 4, Table 3).

Fin whales were detected across the continental shelf year-round (Fig. 5), with more detections near the continental shelf break. Fin whales were detected in the greatest proportion of surveyed grid cells in spring, with detections in 83% of grid cells (Table 3). The distance from shore did not differ significantly across seasons (Table S3; ANOVA, $F_{3,111} = 1.9$, $p = 0.14$).

3.3. Temporal distribution of whale detections in the NYB

When normalized spatially (by kilometers surveyed per grid cell), humpback whales were detected on significantly more days in spring than in all other seasons (Table S4; ANOVA, $F_{3,323} = 25.5$, $p < 0.01$). When normalized by temporal effort (number of survey days), humpback whales were also detected in the highest proportion of survey days (>50%) from March to May (Fig. 6). The proportion of detection days was ~20% from July through November and increased to 40% in January and February (Fig. 6).

Similarly, sei whales were also detected on significantly more days in spring than other seasons, when normalized by spatial effort (Table S4; ANOVA, $F_{3,545} = 33$, $p < 0.01$). They were detected on 20% of survey days from March through May, when normalized by temporal effort (Fig. 6). During the rest of the year, the proportion of days sei whales were detected was less than 10% (Fig. 6).

NARWs were detected on the greatest number of days, when normalized for spatial effort, in the winter (Table S4; ANOVA, $F_{3,814} = 8.9$, $p < 0.01$). When normalized for temporal effort, NARWs were detected on 20–30% of survey days from December to March (Fig. 6). The proportion of NARW detection days was less than 10% the rest of the year, apart from August and September, where they increased to ~25% (Fig. 6).

The number of fin whale detection days did not vary significantly across seasons when normalized for spatial effort (Table S4; ANOVA, $F_{3,161} = 2.4$, $p = 0.07$). When normalized for temporal effort, fin whales were detected on over 40% of survey days from July through March (Fig. 6).

The number of detection days in June was low for all species (Fig. 6) due to low sampling effort in June across all years (Fig. S6). This lack of June effort did not affect the overall temporal (survey days) or spatial (km surveyed) effort on the seasonal scales at which statistical tests were performed, as summer had the second highest amount of effort across all seasons in the NYB (Table 1).

3.4. Distribution of whale detections within WEAs

Humpback whales were detected within 97% of grid cells in WEAs over the course of all deployments, and this percentage was significantly greater than the percentage of surveyed grid cells with detections across the entire study area (Table 3, $\chi^2_1 = 8.0$, $p < 0.01$). Within each of the WEA subgroups, this percentage ranged from 89 to 100% (Tables S5–S7).

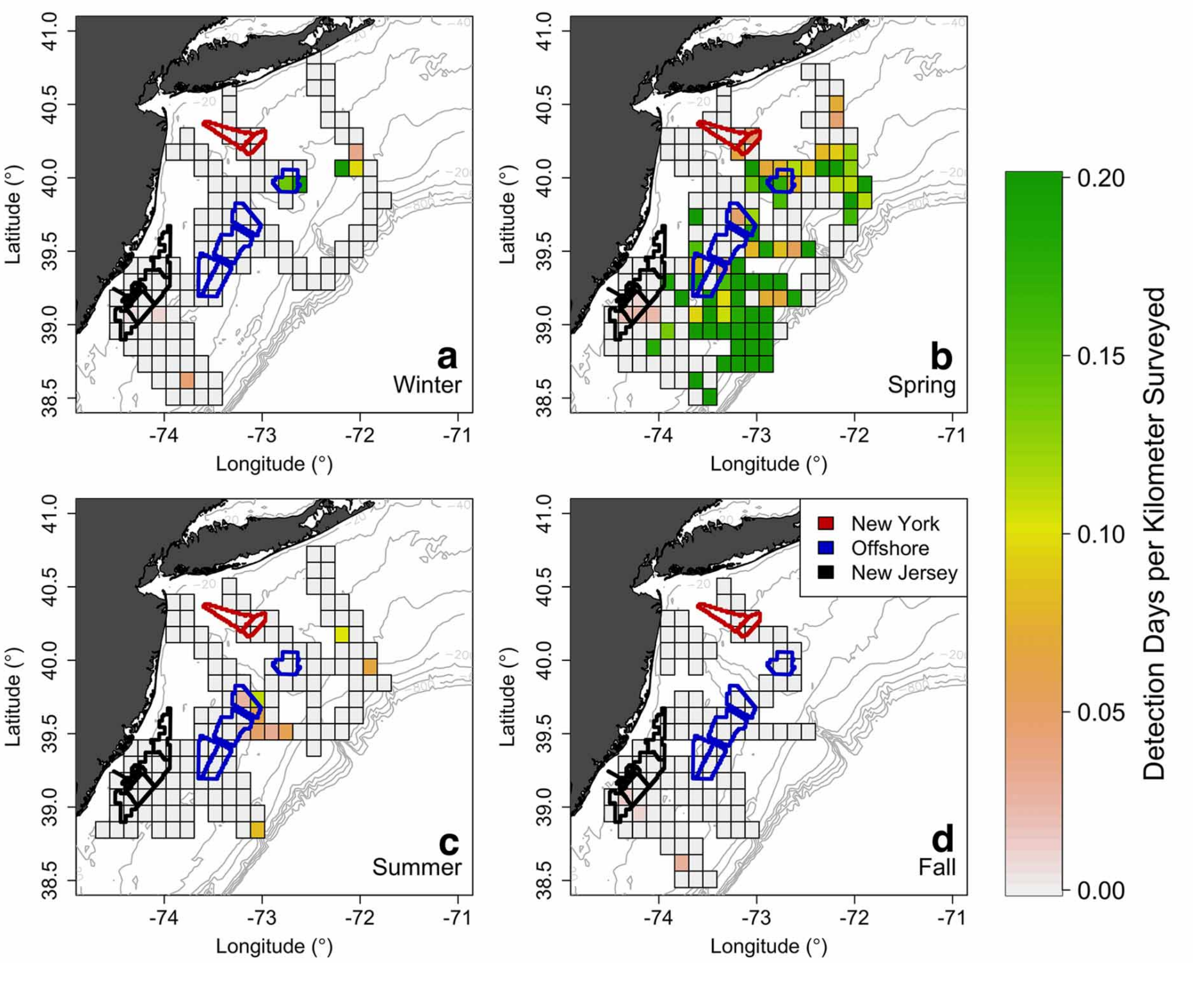

Fig. 3. Number of days sei whales were detected per kilometer surveyed across seasons. Details as in Fig. 2, except that grid cells are 256 km^2 (16 km × 16 km), as determined by spatial autocorrelation analyses

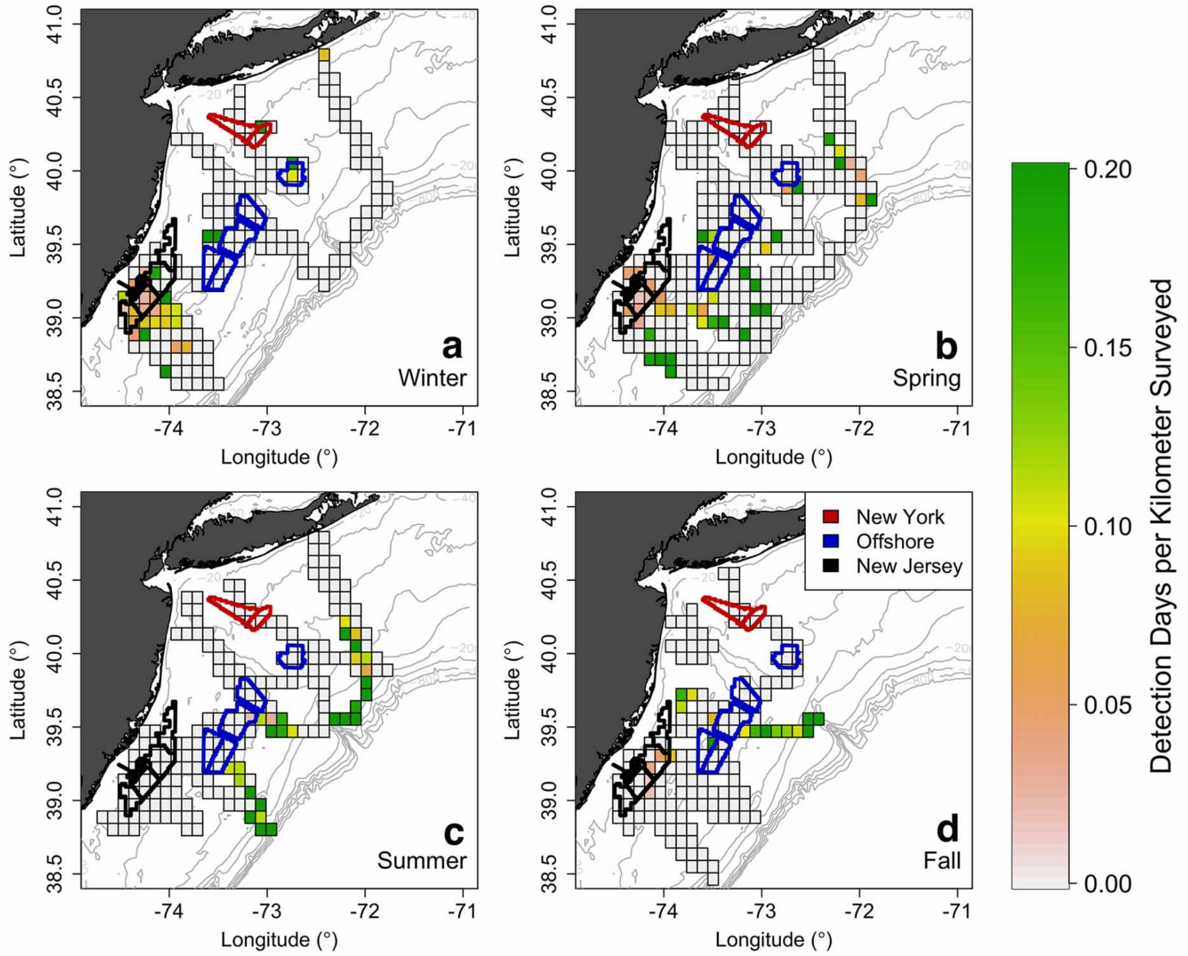

Fig. 4. Number of days North Atlantic right whales (NARW) were detected per kilometer surveyed across seasons. As in Fig. 2, except that grid cells are 144 km^2 (12 km × 12 km), as determined by spatial autocorrelation analyses

Humpback whales were detected in the greatest percentage of grid cells throughout the study area (91%) and within all 3 WEA subregions (83–95%) in spring (Tables 3; Tables S5–S7). The percent of grid cells across the study area and within WEAs with detections did not differ across seasons (χ^{2}_{1} = 0–3.6, p = 0.06–1). Relative habitat overlap across all deployments was 35% (Table 3). The greatest relative habitat overlap across WEAs occurred in winter (55%, Table 3). Humpback relative habitat overlap was greatest in winter within New York (14%), fall in the New Jersey WEAs (26%), and summer in the Offshore WEAs (30%; Tables S5–S7).

Sei whales were detected in 42% of WEA grid cells across all deployments and seasons (Table 3). This ranged between 22 and 61% within WEA subgroups (Tables S5–S7). Sei whales were detected in the greatest percentage of grid cells across the NYB and within WEAs (43% across all WEAs; 25, 59, and 28% for NY, Offshore, and NJ WEAs, respectively) in the spring (Tables 3; Tables S5–S7). These percentages did not differ across seasons (χ^{2}_{1} = 0–1.2, p = 0.27–1). The relative habitat overlap for sei whales was 30% across the calendar year (Table 3). Sei whale relative habitat overlap was greatest in the fall at 67%. Patterns differed within the different WEA subregions, where sei whale relative habitat overlap was greatest in spring (3%) in New York, summer (50%) in Offshore, and fall (66%) in New Jersey WEAs (Tables S5–S7).

NARWs were detected in 39% of grid cells within WEAs across all deployments, which was significantly greater than expected (Table 3, χ^{2}_{1} = 8.2, $p < 0.01$). Within WEA subgroups, this ranged from 8 to 77% (Tables S5–S7). NARWs were detected in the greatest percentage of grid cells across all WEAs in winter (29%) and this percentage was greater than expected (χ^{2}_{1} = 8.9, $p < 0.01$). NARWs were also

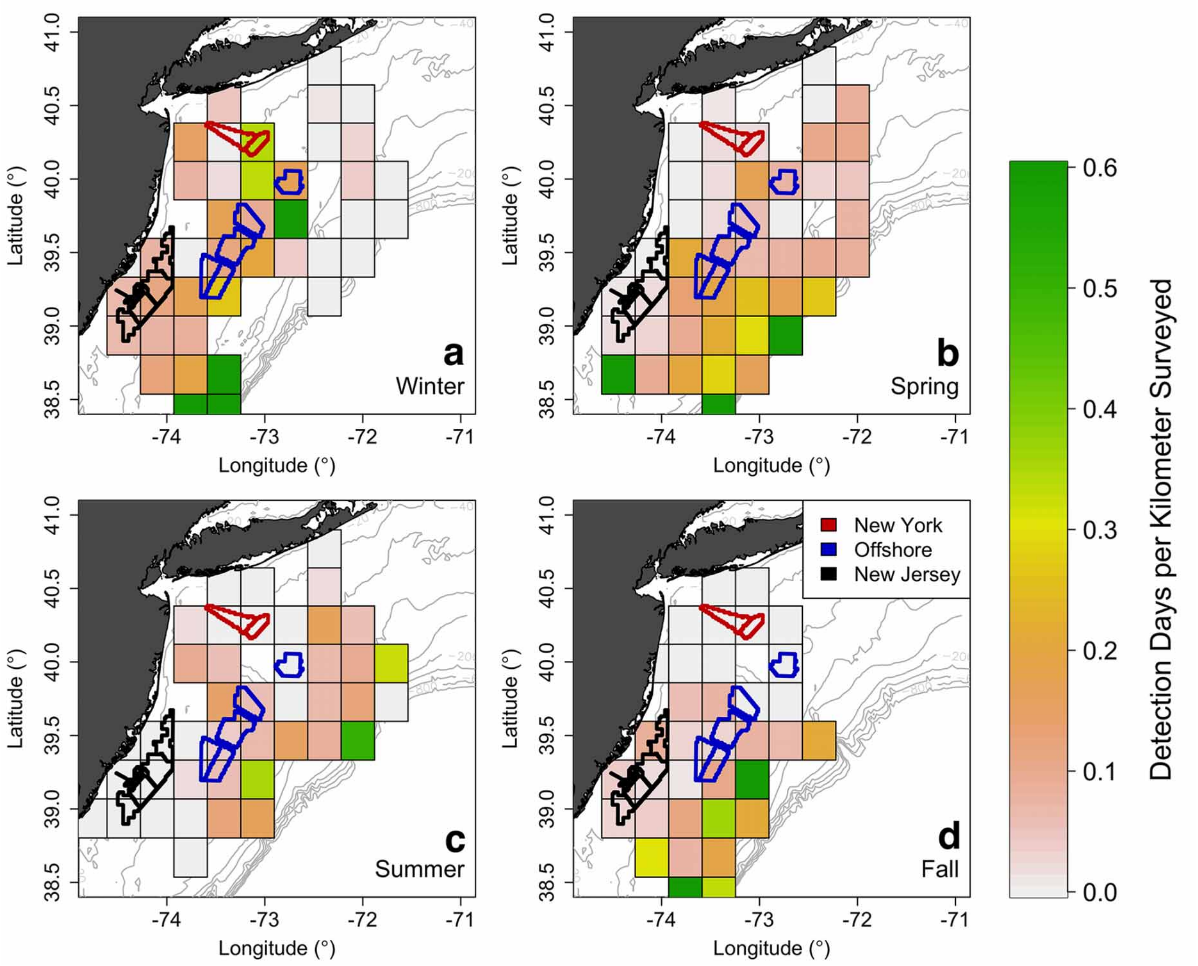

Fig. 5. Number of days fin whales were detected per kilometer surveyed across seasons. As in Fig. 2, except that grid cells are 1444 km^2 (38 km × 38 km), as determined by spatial autocorrelation analyses

detected the most within NY and New Jersey WEAs (3 and 52% for New York and New Jersey WEAs, respectively) in winter (Table 3; Tables S5 &S7). In offshore WEAs, this percentage was greatest in spring (11%; Table S6). NARWs were detected in the fewest WEA grid cells in summer (3%), which was significantly less than expected (Table 3; $\chi^2_1 = 4.6$, $p = 0.03$). The percentages in spring and fall did not differ ($\chi^2_1 =$ 0.2–1.0, p = 0.33–0.69). Relative habitat overlap with WEAs was 33% across all seasons. Like the other species, overlap was highest in the winter (65%) (Table 3). Within subregions, relative habitat overlap was greatest in winter for the New York (3) and New Jersey (52%) WEAs, but was greatest in fall in the Offshore WEAs (16%; Tables S5–S7).

Fin whales were detected in 100% of surveyed grid cells within WEAs (Table 3). Within subgroups, this ranged from 27 to 100% (Tables S5–S7). This percentage was greatest (89%) in spring (Table 3). These percentages did not differ from expected (χ^2_1 = 0–2.6, p = 0.11–1). Within the Offshore WEAs, it was greatest in spring (100%; Table S6) and within the New Jersey and New York WEAs, the percentage was greatest in winter (100 and 75%, respectively; Table S7). Fin whales had a 38% relative habitat overlap with WEAs across all seasons (Table 3). This overlap was greatest in the winter at 52% (Table 3). Within WEA subregions, relative habitat overlap was greatest in winter in the New York WEAs (10%, Table S5) and fall within the Offshore and New Jersey WEAs (26 and 22%, respectively; Tables S6 & S7).

4. DISCUSSION

With offshore wind development actively occurring along the US east coast and increasing worldwide (McCoy et al. 2024), there is an urgent need to improve our understanding of baleen whale habitat use relative to WEAs prior to construction. This is especially true in the northeast USA where construction has already occurred in WEAs to the east of the NYB and is expected to begin in the NYB by the end of 2025.

4.1. Spatial and temporal distribution of baleen whales within the NYB

Baleen whale habitat use in the NYB was highly seasonal. Humpback and sei whales were detected on the greatest proportion of survey days and surveyed grid cells in the spring. Humpback whales were detected closer to shore in the winter, whereas sei whales were detected closest to shore in the fall. While fin whales were the most widespread across the NYB in spring months, the proportion of detection days and distance to shore did not change significantly with season. These patterns are similar to those observed in previous PAM and visual surveys (Davis et al. 2017, Muirhead et al. 2018, Zoidis et al. 2021).

NARWs had the greatest proportion of detection days, and were detected significantly closer to shore, in winter months. Detections were the most widespread in spring, similar to other species, which aligns well with previous studies (Muirhead et al. 2018, Davis et al. 2020, Zoidis et al. 2021). However, we detected NARWs offshore in the summer months during 2 glider missions in summer 2024. Visual surveys detected nearly 100 NARWs near the continental shelf break and Hudson Canyon in summer 2024 (New England Aquarium 2024). Anomalous oceanographic conditions in the northeast USA during this time may have altered prey availability for NARWs (Record et al. 2024), causing them to forage farther south than is typical during summer months.

4.2. Spatial and temporal patterns within WEAs

Detections of baleen whales were more common in the coastal WEAs (New Jersey and New York) during the fall, winter, and spring, whereas detection rates in the offshore WEAs were greatest in the summer. Although seasonal trends in detections varied between species, our relative habitat overlap metrics suggest that, on broad scales, WEAs comprised a larger proportion of the observed winter habitat for all species. We found that all species were detected in the greatest proportions of the survey areas in the spring, but humpback and NARWs were detected significantly closer to the coast in winter months, which drove the higher overlap with WEAs.

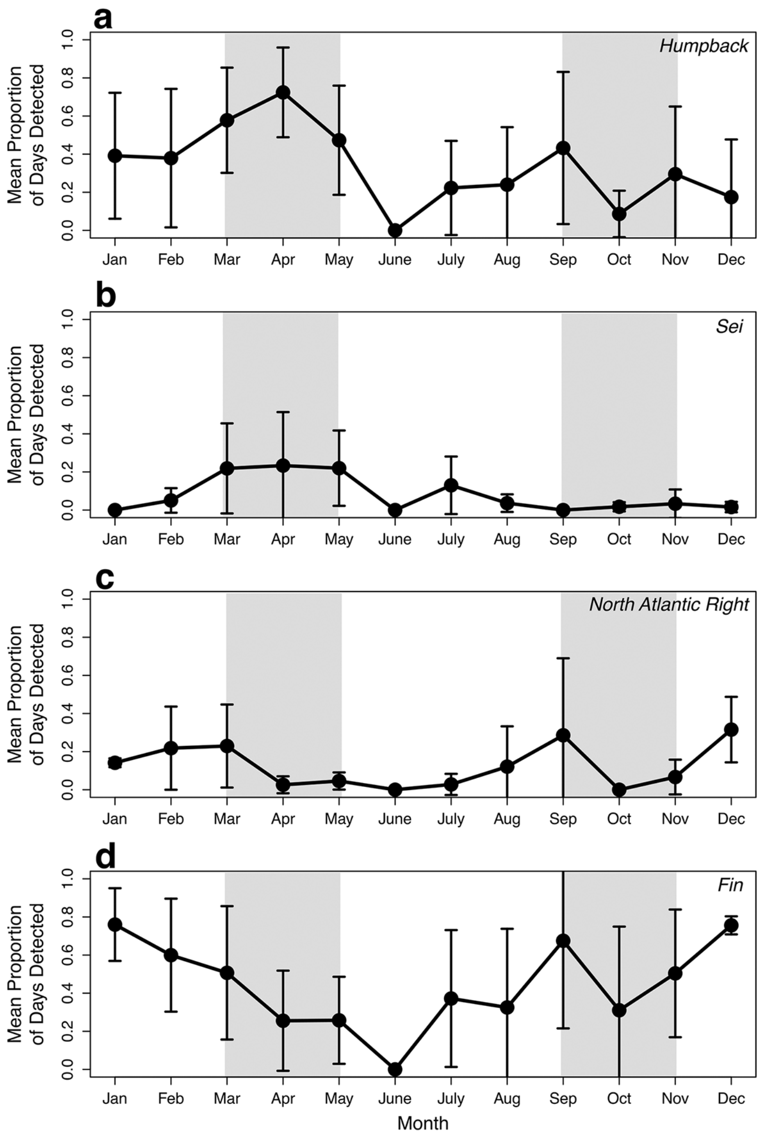

Fig. 6. Mean (±SD) proportion of days in which baleen whales were detected in each month within the New York Bight for (a) humpback, (b) sei, (c) North Atlantic right, and (d) fin whales. Alternating gray and white bands illustrate the different seasons: winter (December–February), spring (March–May), summer (June–August), and fall (September–November)

Current mandates prevent wind development companies from pile driving from January through April in the NYB (National Oceanic and Atmospheric Administration 2024). While this mandate is in place primarily to protect NARWs, our results indicate that it may also provide protection to humpback, sei, and fin whales since their relative habitat overlap is also greatest with WEAs, especially the inshore New York and New Jersey WEAs, in the winter months. However, our results also suggest that this mandate should be extended to include fall and early winter (September—December), as relative habitat overlap for all species was also high in these periods.

4.3. LFDCS performance and limitations

We used the LFDCS to automatically detect baleen whale vocalizations within our acoustic data collected on glider missions. Missed detection rates were similar to previous studies using gliders reporting in near-real time (Baumgartner et al. 2020, Johnson et al. 2022) and stationary moored platforms (Davis et al. 2017, 2020, Baumgartner et al. 2019, Johnson et al. 2022). This suggests that the LFDCS analysis on the archival audio performed similarly to previous methods.

False detection rates were higher in this study (0—13%) than in previous studies using stationary moorings and gliders (0%; Davis et al. 2017, 2020, Baumgartner et al. 2019, 2020). This is likely due to the few (1—2) days where the manual annotation did not detect a species, where the LFDCS did. Manual annotations such as the ones performed here are labor intensive and can be subject to human error (Knight et al. 2017, Leroy et al. 2018, Baumgartner et al. 2020). This therefore highlights one of the strengths of the LFDCS and other automatic annotation efforts that can identify species calls where manual annotation could not.

However, our study had higher accuracies (both the LFDCS and manual analysis agreed) than previous glider-based PAM studies using the LFDCS (Baumgartner et al. 2020). While these are lower than the accuracies reported by previous moored platforms (96—98%, Davis et al. 2020), the relatively high accuracy in comparison to previous glider studies illustrates the strength of analyzing the entire archival data set, regardless of the amount of data that are transmitted back in the near-real time analysis, for assessing baleen whale acoustic occurrence using glider data.

Whale vocalization behavior, how these calls propagate through the environment, and how well a receiver can detect these calls may vary for a variety of reasons, all of which can impact the detectability of a species (Browning et al. 2017). Whale calling behavior, including the frequency and amplitude of calls, can be altered by inter- or intra-species interactions or individual behavior (foraging, transiting, etc.) (Parks et al. 2011, Dunlop et al. 2013). Call propagation will be impacted by water depth, whale depth, and sound speed (Tennessen & Parks 2016, Etter 2018). These factors, in addition to background noise levels and baleen whale call source levels, will also impact how well a receiver can detect these calls (Zimmer 2011, Parks et al. 2011, Browning et al. 2017, Estabrook et al. 2022, 2025, Johnson et al. 2022, Fleishman et al. 2023). All of these factors can change seasonally, geographically, or daily due to animal behavior, oceanographic conditions, and changes to the background noisescape.

In addition, gliders as PAM platforms introduce additional variability as they move vertically through the water column. Sound transmission can change throughout the water column, due to seasonal stratification and changes with sound propagation with depth. Sounds can be reflected off the surface or bottom, increasing transmission loss. Furthermore, sound can be absorbed by bottom sediments. Previous glider studies have illustrated that transmission loss is greatest near the surface, and that the depth of the glider changes detection ranges and detection probabilities in a uniform modeled environment (Johnson et al. 2022). The detectability of baleen whale calls likely changes not only seasonally, but will change, for better or worse, over the course of the glider profile. Understanding how glider behavior, baleen whale behavior, and environmental changes impact the detectability of baleen whale calls on mobile PAM platforms, both in the NYB and broadly, will help us better understand whale acoustic presence and absence.

4.4. Gliders as monitoring platforms

In the present study, the sampling scale of gliders allowed us to illustrate trends in baleen whale habitat use across longer temporal scales than previous visual surveys and across larger spatial scales than previous PAM studies. The gliders in this study accumulated over 20 times more temporal survey effort than previous monthly aerial surveys (~30 survey days in Zoidis et al. 2021 versus more than 700 deployment days in our study). However, our glider-based PAM deployments did not provide the same temporal coverage as previous PAM studies using stationary moorings. For example, Davis et al. (2017, 2020) examined over 35000 days of acoustic data across their study area. Of those, approximately 1869 days occurred within our study region (Davis et al. 2020), equating to 2.5 times more temporal effort than our glider surveys.

While the temporal coverage of our glider surveys was shorter than previous PAM surveys, we had greater spatial coverage due to the mobility of the glider. Previously published PAM studies used ~15 stationary moorings within our study area (Davis et al. 2017, 2020). If the detection ranges of each of these platforms ranged from 12 to 38 km (same as our species-specific grid sizes and spatial autocorrelation analysis and similar to previously published detection ranges), then each mooring would have a survey area of ~450–4500 km^2 (assuming the detection range is a circle around the mooring). This would give a 15-unit moored array a maximum survey area of ~6800–68000 km^2 (not accounting for overlap between adjacent moorings). The grid cells used in our study had areas ranging from 144 to 1444 km^2. We surveyed 55–349 grid cells, for a total survey area of ~50000–79000 km^2, 7 and 1.2 times greater than previous moored PAM arrays, respectively. In comparison, previous aerial visual surveys covered an area of 43449 km^2 and over 140000 km of transect lines (Zoidis et al. 2021). While the overall area covered by our gliders is 1.2–1.8 times greater, the total distance covered by previous aerial surveys is 10 times greater than our glider surveys (~14000 km).

We can compare the spatiotemporal sampling scale of different survey methods, using the differences in area and number of days surveyed across methods. We find that the spatiotemporal sampling scale (difference in area sampled multiplied by difference in days sampled) of gliders is 24–36 times greater than aerial surveys (covering 1.2–1.8 times more area multiplied by 20 times longer), but only 0.48–2.80 (1.2–7 times more area multiplied by 40% of the time) times greater than moored PAM arrays. This suggests that gliders and moorings cover similar spatiotemporal scales, with moorings capable of surveying nearby areas for long periods, and gliders capable of surveying over longer distances, for shorter periods of time. However, it is important to note that both of these platforms are bounded by the limitations of PAM. It is challenging to estimate baleen whale density or distinguish between individuals to determine residency from PAM surveys, while visual vessel-based or aerial surveys allow individuals to be counted and identified to determine residency. Visual surveys, as well as glider-based surveys and some moored platforms, allow for immediate (near-real time) results, in comparison to moored archival PAM arrays that rely on the analysis of recorded audio. Furthermore, while the amount of noise that gliders introduce to archival PAM recordings is minimal, they do introduce more noise than their moored counterparts, which can provide more robust insights into the entire soundscape over longer time periods.

Gliders should be considered a cost-effective tool to be deployed in conjunction with traditional stationary mooring platforms and synoptic visual surveys. For example, a long-term PAM array of 1–2 moorings may be able to detect baleen whales within a WEA, in conjunction with periodic glider deployments in and around the WEA to put mooring data in context of the broader ecosystem. Periodic visual surveys would supplement these PAM methods to identify individuals and estimate density. However, understanding baleen whale habitat use both within target areas such as WEAs and across the entire ecosystem requires sustained long-term monitoring programs, such as the state-funded initiatives that funded this work. The need for long-term monitoring is especially great in and around WEAs where extended monitoring can be used to understand habitat use prior to construction, as we have done here. This monitoring should continue through construction and operation phases to understand the impacts of WEAs on baleen whale habitat use.

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