A Framework for Studying the Effects of Offshore Wind Development on Marine Mammals and Turtles

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Executive Summary

The Massachusetts Clean Energy Center (MassCEC), the Massachusetts Executive Office of Energy and Environmental Affairs, the Bureau of Ocean Energy Management (BOEM), and the New England Aquarium (NEAq) convened a workshop on 30 and 31 May 2018 that included marine scientists, NGO representatives, regulators, public stakeholders, and offshore wind leaseholders to inform the development of a scientific research framework (the "Framework") to guide studies of potential impacts to endangered whales and sea turtles associated with offshore wind facility construction and operation in the U.S. Northeast.

Baleen whales and sea turtles are migratory species that rely on North Atlantic waters for all aspects of their life history. Recent surveys of wind energy areas offshore of Massachusetts and Rhode Island have documented their presence in the area at various times of the year. In order to assess the ecological impacts of offshore wind facility construction and operation on marine mammals and sea turtles in U.S. waters, a carefully designed research plan is needed. Because of multiple variables, changing oceanic conditions, and inter-annual variability, any such research to determine effects will require careful experimental design, appropriate statistical methods, and data collection methods designed to collect adequate sample sizes.

The workshop informed this marine mammal and wind research framework. The framework identifies options to assess potential population-level impacts to marine mammals and sea turtles associated with offshore wind facility construction and operation. This includes both the immediate effects of short-term construction activities at the project-specific scale, and the long term effects of and potential population-level impacts of windfarm placement and operations on distribution, abundance, behavior, or demography of endangered marine mammals and sea turtles. The framework was developed with a focus on assessing potential impacts to baleen whales and sea turtles associated with offshore wind facility construction and operation within the Massachusetts and Rhode/Island Massachusetts Wind Energy Areas (MA and RIMA WEAs). This is because adequate baseline marine mammal and sea turtle data exist, and wind facility construction is anticipated to begin in the next two years. However, the framework approach should be applicable in other offshore wind developments along the Atlantic coast.

The generic research question was "Do wind farms cause a change in some parameter of interest for species of concern?" To generate more specific questions, researchers will need to define the spatial and temporal scope and the parameters of interest. In terms of scope, one can measure temporal change (short-term or long-term, i.e., trend) over some defined area, or spatial change over some defined time, or both spatial and temporal change simultaneously. The potential parameters of interest include population size (stock abundance), relative population abundance (indices), occupancy, local spatial density/abundance, local spatial indices of abundance, movement (e.g., avoidance behavior of individuals), demographic parameters (e.g., birth, immigration, mortality), body condition/health, and/ or physiological/behavioral measures (e.g., stress hormones or changes in calling rates).

The hypotheses generated during the workshop fell into three categories. One, animals could be displaced from the wind energy area (by noise, construction, towers, etc), two, animal behavior and or physiological parameters could change (e.g. calling rates, feeding, breathing, movements, stress hormone levels), and three, wind farms could alter habitat in a way that disrupts prey species availability for relevant whales or sea turtles (Table 1). In all cases, it will be important to differentiate minor effects from those that will impact particular species at the population level.

Table 1. Hypotheses for testing the effects of wind farms on marine mammals and sea turtles.

Hypothesis	Importance	Testability	Assessment
Short-term effects of wind energy development			
1) Displacement. Construction activities result in displacement of whales or sea turtles away from activity locations.	High	High	There are multiple approaches to test this hypothesis, including aerial surveys and passive acoustic monitoring before, during and after construction activities, then evaluating the distribution of animals under each of the conditions.
2) Behavior disruption. Construction activities disrupt critical behaviors of whales or sea turtles, such as feeding, socializing or nursing.	High	Medium	This is worth further study using aerial and shipboard behavioral observations before, during, and after construction activities, both in the impacted and control sites. Could Cape Cod Bay be used as a control area?
3) Stress. Construction activities cause elevated stress hormone levels in whales or sea turtles.	High	High	Using concurrent acoustic measurements and blow sampling methods, it may be possible to characterize the relationship between stress-related hormones and underwater noise from pile-driving and other activities.
4) Prey. Construction activities cause zooplankton or fish prey to change their vertical distribution, density or patch structure.	Low/Med.	Medium	A possible study design would involve sampling down-current from an active pile-driving site and from a control site within the planned array, concurrent with appropriate prey sampling strategies.
Long-term effects of wind energy development			
1) Distribution change. Wind turbine presence either excludes or attracts whales and sea turtles.	High	High	This is an important hypothesis. It will be critical to compare changes in whale and turtle abundance, prey abundance, and oceanographic conditions in the MWEA to a nearby control area.
2) Prey. Wind turbine presence affects long-term feeding opportunities for whales and sea turtles.	Low	Low	Too little information on turbine tower wakes and the oceanographic consequences on biology exist. If that model existed, it would be possible to explore possible impacts.
3) Ecosystem enhancement. The development of artificial reefs on wind turbine foundations affects the regional ecosystem, potentially enhancing some characteristics of marine productivity.	Low	Medium	Unlikely to enhance copepods that right whales feed upon because patch formation is not due to food-chain processes. Unlikely that tower reef effects would support the small pelagic planktivorous fishes that comprise the prey of other baleen whales. Sea turtles could find enhanced feeding opportunities due to the reef effect.

Recent efforts to develop tools for detecting and measuring the population-level consequences of disturbance (PCoD) include a set of mathematical methods to quantitatively assess the magnitude of these effects. Incorporating the concept of animal "health" (often quantified in terms of energy stores), was a way to link short-term effects of disturbance with long-term demographic outcomes on individuals. A number of case studies have been created, and work is ongoing to transition the methods to an operational context. In 2017, a National Academies report reviewed the wider context of the cumulative impact of multiple stressors. An expanded conceptual framework was developed, but implementing it in practice will be very difficult due to lack of knowledge on cumulative effects. This body of work is relevant to the marine renewable energy situation because installation and operation of wind farms may cause behavioral disturbance, potentially leading to population-level effects. Research studies should, therefore, be designed in such a way that they can help parameterize a PCoD model.

There are several potential data-collection methods available for testing hypotheses. These include aerial surveys, remote sensors (e.g., infrared, radar, LIDAR), passive acoustic monitoring including both archival and real-time acoustic methods, tagging, drones, hormones in scat and blow, and habitat monitoring/oceanographic sampling. It is likely that the well-known data collection methods (aerial surveys, passive acoustics, DTAGs) are best suited for answering most questions. However, the chosen monitoring program should be flexible, and be able to incorporate new technologies that may enhance data collection. Additional data considerations for every method includes species identification capacity, acoustic and behavioral characteristics of the species of interest, cost, data turnaround time, data-processing time, technology development stage, geographic scale, detection range, limitations due to ocean and weather conditions, ease of implementation, suitability for short-term or long-term studies, durability, and reliability of detections.

Workshop participants made several recommendations with regard to the links between data collection and managing wind farm development. One, data must be collected in a manner that can inform regulatory and management decisions on individual project review and long-term cumulative impacts. Two, the framework should be adaptable to new lease areas as they come online and other stressors emerge (e.g., fishing, climate change), so that each wind project can be informed by the data collected from previous projects. Three, the framework should be designed to provide usable information about cumulative effects in order to respond to managers and regulators. Finally, the data collected following the research framework should help regulators and developers determine the best timing for construction.

Finally, there is still much to learn about whale and sea turtle behavior and physiology; these gaps in knowledge will be a challenge when designing a long-term study. There are outstanding questions about how whales find food, how they navigate, migration routes, and the scope of their sensory capabilities. Regulators and industry should proceed with caution because these unknowns may be important for designing monitoring and research programs to determine the effects of wind energy facilities, and could have implications for the timing and magnitude of both construction and operations.

1 Introduction

The Massachusetts Clean Energy Center (MassCEC), the Massachusetts Executive Office of Energy and Environmental Affairs, the Bureau of Ocean Energy Management (BOEM), and the New England Aquarium (NEAq) convened a workshop on 30 and 31 May 2018 that included marine scientists, NGO representatives, regulators, public stakeholders, and offshore wind leaseholders to inform the development of a scientific research framework (the “Framework”) to guide studies of potential impacts to endangered whales and sea turtles associated with offshore wind facility construction and operation in the Atlantic waters of the U.S. This report lays out a framework for determining the effects of wind energy installation and operation on marine mammals and turtles off the US Atlantic coast.

This framework was authored by S.D Kraus, R.D. Kenney, and L. Thomas with input from a panel of subject matter experts. The workshop was attended by numerous stakeholders who participated in working groups that also informed the framework. Subsequent editorial suggestions have been provided by BOEM and other stakeholders. A list of all participants is provided in Appendix 1. In addition, a proceedings document from the work shop was produced (Field and Gilbert, 2019).

The intention of the Framework is to make it applicable to address other offshore wind development along the Atlantic coast (Figure 1.1). However, for discussion purposes, the Framework was developed with a focus on assessing potential impacts to baleen whales and sea turtles associated with offshore wind facility construction and operation within the Massachusetts and Rhode/Island Massachusetts Wind Energy Areas (MA and RIMA WEAs). In these areas, sufficient current biological data exist, and offshore wind facility construction is anticipated to begin in the foreseeable future (Figure 1.2).

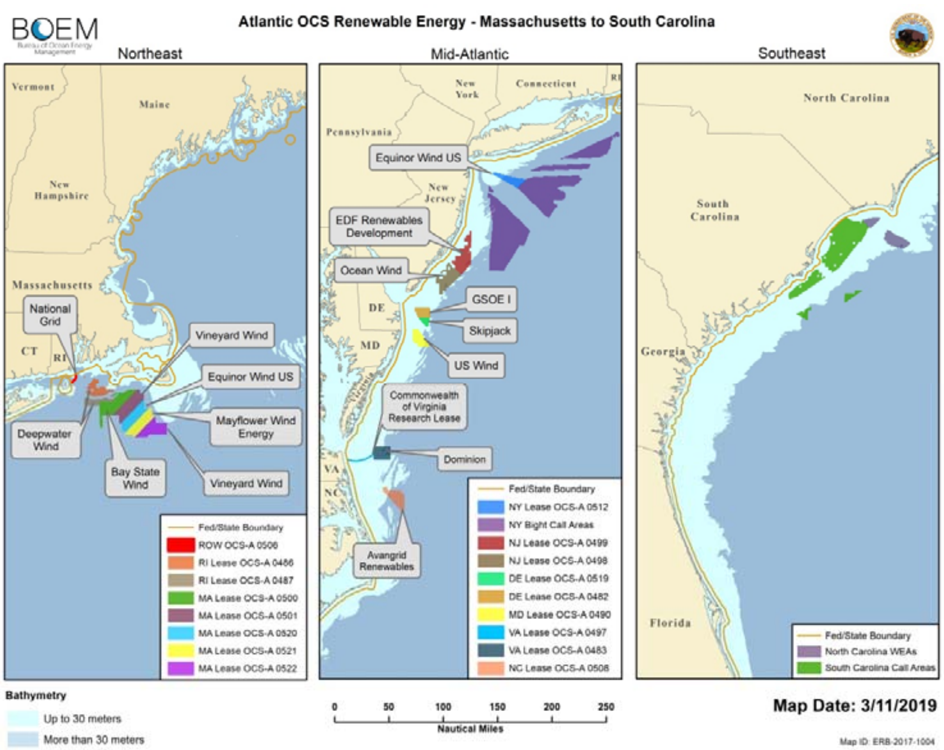

Figure 1.1. Planned wind energy development areas in the US Atlantic (source BOEM).

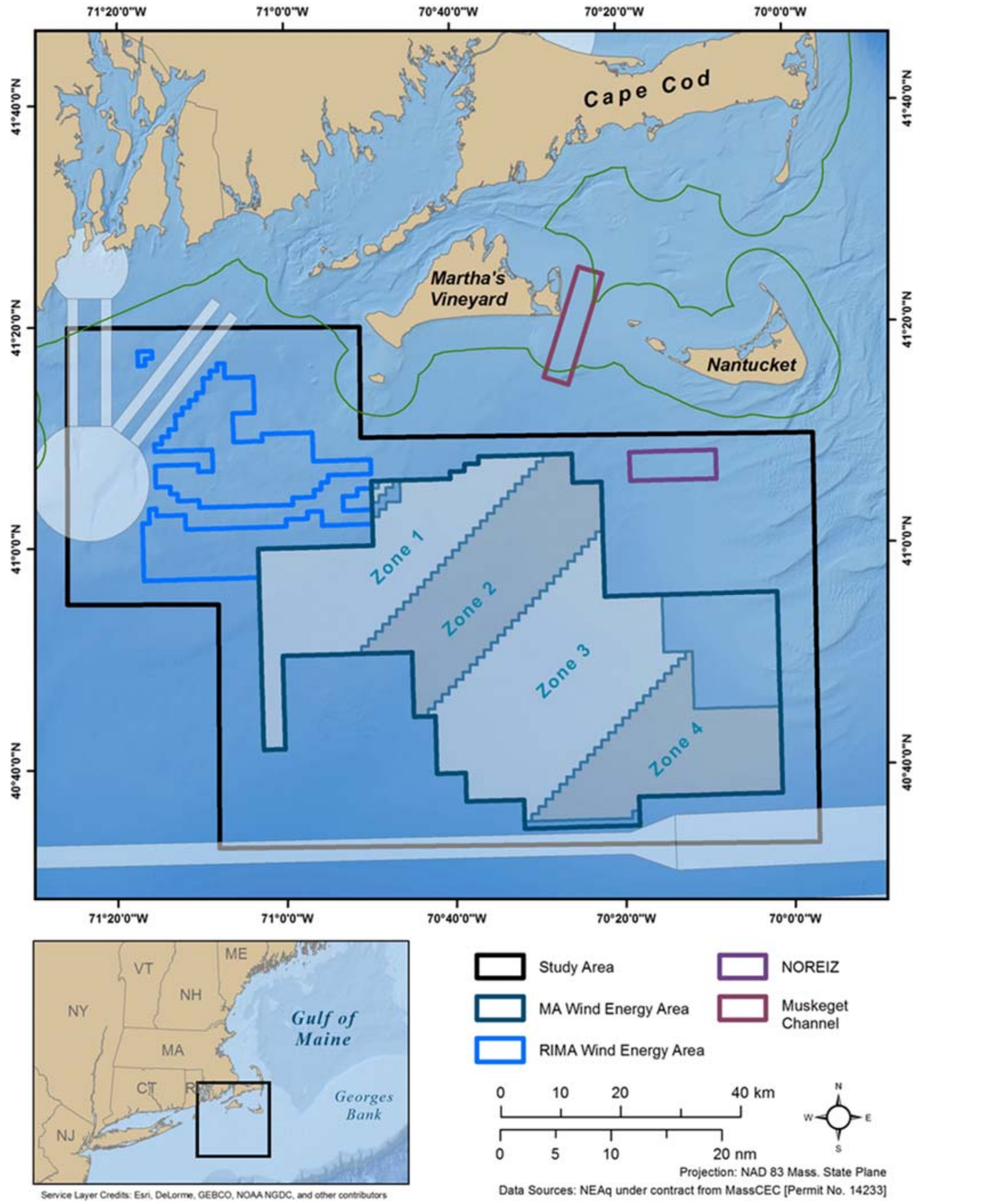

Figure 1.2. Wind energy areas (WEAs) offshore of Massachusetts (MA WEA, outlined in dark blue) and Rhode Island (RIMA, outlined in bright blue), and the study area (SA) outlined in black. Current lease areas are depicted as Zones 1 – 4).

We divided possible effects into two scales of exposure. *Short-term* exposures due to installation may be limited to days, weeks, or months at a given location, whereas *long-term* exposure may be on the order of years of operations and regular maintenance. With these definitions, we considered (1) short-term exposure and effects of installation activities (pile driving, boat traffic, cable laying, etc.); (2) long-term (multi-year) effects of operations and maintenance. We did not consider research related to the effectiveness of real-time mitigation strategies (e.g., Lee et al., 2012; Wochner et al., 2013), nor the possible effects of decommissioning wind farms.

To generate more specific questions about exposure and possible effects, researchers need to define the spatial and temporal scope and the parameters of interest. In terms of scope, one can measure temporal change (short-term or long-term, i.e., trend) over some defined area, or spatial change over some defined time, or both spatial and temporal change simultaneously. The potential parameters of interest include population size (stock abundance), relative population abundance (indices), occupancy, local spatial density/abundance, local spatial indices of abundance, movement (e.g., avoidance behavior of individuals), demographic parameters (e.g., birth, immigration, mortality), body condition/health, and/ or physiological/behavioral measures (e.g., stress hormones or changes in calling rates).

This report is divided into four chapters. In the remainder of this section we briefly review the potential stressors associated with installation and operation of marine wind energy and the regulatory framework within which these developments occur; we summarize existing knowledge on the effects of relevant anthropogenic activities; we then discuss methods for determining effects and lay out a set of hypotheses developed at this workshop. We then consider potential research projects for assessing effects of short-term (Chapter 2), and long-term (Chapter 3) exposures. The final chapter (Chapter 4) contains a set of conclusions and recommendations. Because this is an inter-disciplinary and cross-taxonomic report, and because words have different meanings among different professional groups, we have provided a glossary of terminology and abbreviations in Appendix 2.

1.1 Potential stressors associated with offshore wind energy

There are several activities associated with wind energy development and operation that may affect marine mammals and turtles. While we recognize that there is the possibility of hearing damage if animals are too close to impulsive sounds (permanent or temporary threshold shifts), we assume that mitigation measures will address these risks, and they are not addressed further here. Instead, this document is concerned with other consequences that may not be immediately obvious. First, animals could be displaced from the wind energy area, either by noise from construction or operations, or by the presence of wind towers, or by the boats and noise associated with regular maintenance activity in the wind farm area. Second, animal behavior (e.g. calling rates, feeding, breathing, movements) may change in ways that will reduce food or mate finding abilities (reducing fecundity and or animal health), or it may increase chronic stress levels leading to reduced animal health. Third, wind farms may alter the physical habitat in a way that disrupts the occurrence and or aggregation of prey species for relevant whales or sea turtles. In addition, there may be indirect effects. For example, animals could be displaced away from a wind installation into nearby shipping lanes, or possibly into areas with more fishing gear. It is also important to note that there may be positive impacts, including the potential for reef effects by the towers, possibly increasing productivity and feeding opportunities for several marine mammal and turtle species. Further, wind installations may serve as de facto marine protected areas if shipping and commercial fishing are restricted within their boundaries.

1.2 Regulatory context

The Bureau of Ocean and Energy Management (BOEM) designated two wind energy areas (WEAs) in New England: one offshore of Massachusetts (MA WEA) and the other offshore from both Rhode Island and Massachusetts (RIMA WEA) (Figure 1.2). Under the National Environmental Policy Act of

1969 (42 U.S.C. 4371 et seq.), BOEM and other relevant federal agencies are required to conduct environmental assessments of offshore development and construction plans.

Under the Marine Mammal Protection Act of 1972 (MMPA) and the Endangered Species Act of 1973 (ESA), many species that occur in wind energy lease areas are afforded legal protections. Many WEAs are inhabited frequently and seasonally by five species of large whale and three species of sea turtle that are listed as Endangered or Threatened under the ESA (Lazell, 1980; CETAP, 1982; Kenney and Winn, 1986; Waring et al., 2015; LaBrecque et al. 2015; Hodge et al., 2015). The whales found in the area include the fin whale (*Balaenoptera physalus*), sei whale (*B. borealis*), North Atlantic right whale (*Eubalaena glacialis*), humpback whale (*Megaptera novaeangliae*), minke whale (*B. acutorostrata*), and occasional sperm whales (*Physeter macrocephalus*). Of these, all but the minke and humpback whale are listed as Endangered under the ESA (NMFS OPR, 2016a). Sea turtles recently observed in southern New England waters include the loggerhead (*Caretta caretta*), leatherback (*Dermochelys coriacea*), and Kemp's ridley (*Lepidochelys kempii*). All of these sea turtles are listed as Endangered or Threatened under the ESA (NMFS OPR, 2016b).

1.3 Existing knowledge on marine mammals and turtles in the MA/RIMA WEAs

For discussion purposes, this workshop used information collected from two wind energy areas (WEAs) offshore from Massachusetts and Rhode Island, which were surveyed for marine mammals and sea turtles between October 2011 and June 2015. (Additional survey data has been collected between late 2015 and 2019.) A total of 969 cetacean sightings of over 10,000 animals were documented during systematic line-transect aerial surveys (67,525 km flown). Twelve cetacean species were documented, including seven odontocete and five mysticete species. Cetaceans were recorded in all seasons demonstrating inter-annual, seasonal, and spatial use of the WEAs. Peak presence occurred in the spring and summer for most species, with the exception of North Atlantic right whales, which occurred primarily in the winter and spring. Seasonal estimates of sighting rates and abundance were calculated for seven species, including five baleen whales (fin, minke, humpback, right, and sei) and two dolphins (common and bottlenose) (Figure 1.3, from Kraus et al, 2016).

North Atlantic right whales were a primary target species of the study, and the aircraft deviated from transects so observers could obtain photographs of the animal(s) for individual identification (Kraus et al., 1986). Observers collected oblique photographs of the entire rostral callosity pattern of each right whale sighted, and any other scars or markings that were obvious, and attempts were made to document each individual within a given aggregation.

That study also collected passive acoustic data to complement aerial efforts, and to characterize patterns of baleen whale occurrence, and the ambient noise environment in the vicinity of MA WEA and RIMA (Kraus et al. 2016). The acoustic analyses focused on five whale species: North Atlantic right, humpback, fin, blue, and minke whales. Acoustic data were collected using marine autonomous recording units (MARUs) (Calupca et al. 2000). Between November 2011 and October 2012, an array of 6 MARUs was deployed at 6 sites in or near the MA WEA. Due to a broadening of the scope of the project at the request of BOEM, between February 2013 and March 2015, three additional MARUs were deployed at 3 sites in the RIMA WEA (Table 1), in addition to the existing array of six MARUs in the MA WEA. The two arrays are hereafter referred to as the MA array and RIMA array (Figure 2). The locations of the MARUs are shown in Figure 1.3.

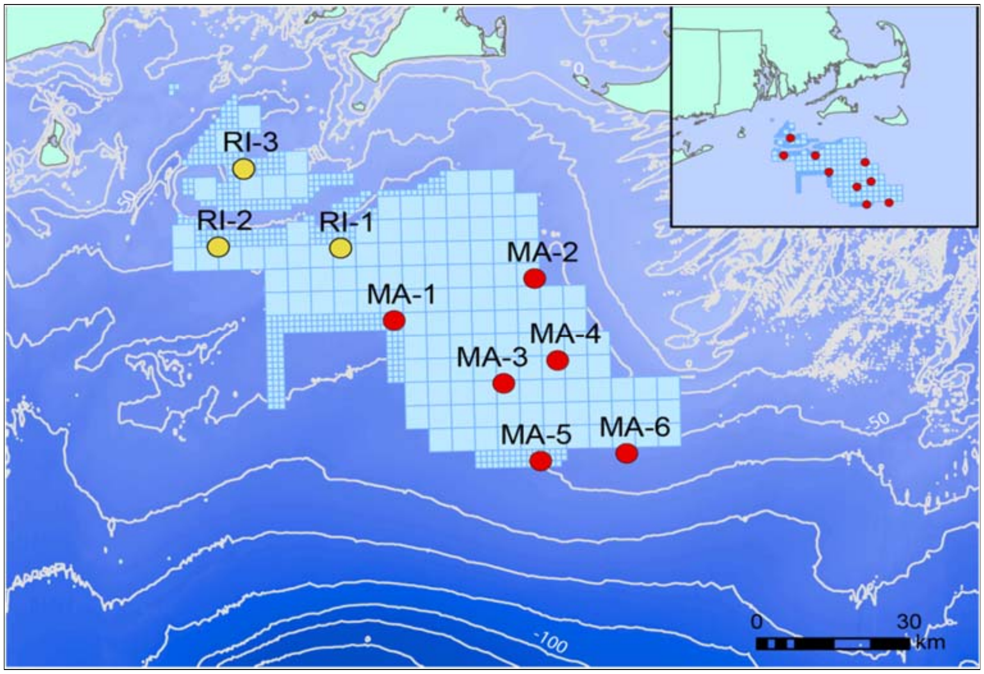

Figure 1.3.1 Map of the MA array of MARUs within the MA WEA (red circles) and the RIMA array of MARUs within the RIMA WEA (yellow circles). White lines represent isobaths in 10-m intervals. The blue squares represent lease sub-blocks within the wind energy area.

1.3.1 Spatial and temporal patterns of presence and abundance

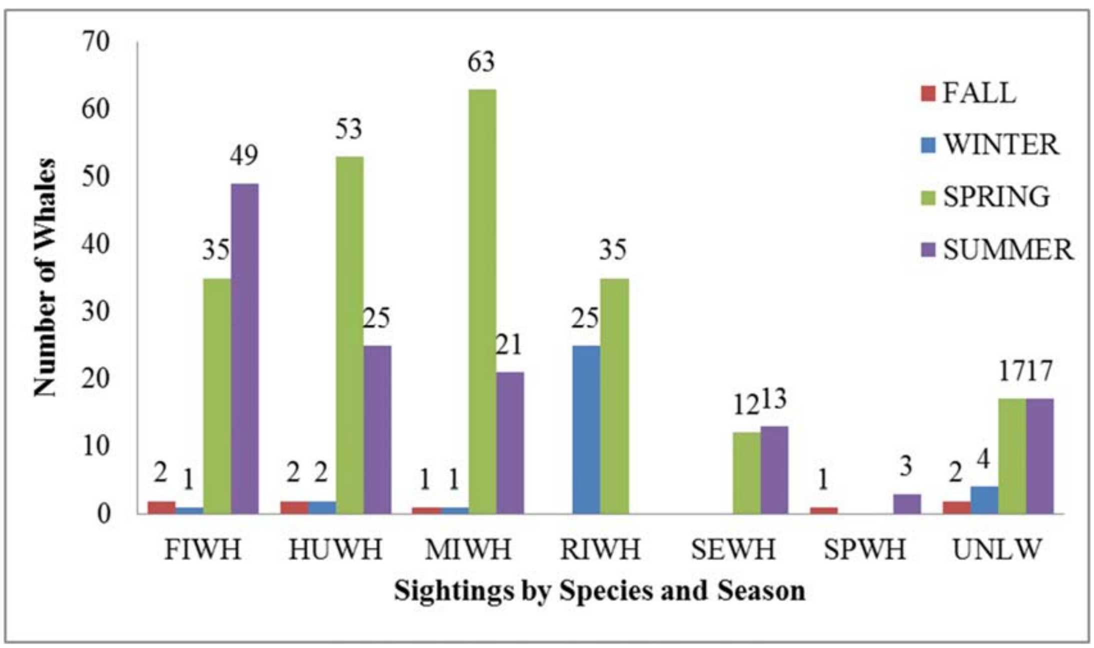

Figure 1.3.2 Numbers of whale sightings in the SA by season across all years (FIWH = fin whale, HUWH = humpback whale, MIWH = minke whale, RIWH = North Atlantic right whale, SEWH = sei whale, SPWH = sperm whale, UNLW = any whale sightings not identified to species).

Table 1.3.3 Mean sighting rates (number per 1,000 km of aerial survey) by month for five large whale species and all large whales combined (including unidentified sightings) (from Kraus et al., 2016).

	Mean SR					
Month	**RIWH**	**HUWH**	**FIWH**	**SEWH**	**MIWH**	**All**
January	4.557	0.194	0.227	0.000	0.000	5.852
February	6.684	0.000	0.000	0.000	0.000	6.684
March	8.425	0.416	0.000	0.137	0.409	9.763
April	2.543	5.926	2.816	0.793	3.092	12.730
May	0.000	4.162	5.617	2.373	6.406	13.080
June	0.000	9.431	5.468	1.726	1.939	17.960
July	0.000	1.447	5.102	0.000	1.627	7.393
August	0.000	0.000	3.372	0.000	0.472	5.326
September	0.000	0.000	0.230	0.000	0.191	0.460
October	0.000	0.503	0.328	0.000	0.000	1.159
November	0.000	0.000	0.000	0.000	0.000	0.203
December	1.647	0.183	0.000	0.000	0.000	2.013

North Atlantic Right Whales

Based on aerial observations and acoustic data, North Atlantic right whales appear to have a distinct seasonal occurrence in the SA during winter and spring between December and May. There was no significant variation from year to year, indicating a fairly consistent annual presence. During spring right whales were widely distributed throughout the SA and were detected in each of the lease areas (RIMA and MA WEA Zones 1 – 4), however during winter, distribution was shifted to the north and outside of most of the lease areas (to the waters just south of Martha's Vineyard and Nantucket) and to the east (Kraus et al., 2016). Abundance tended to be higher in spring (Table 1.2).

Table 1.3.3. Density and abundance of North Atlantic right whales (Eubalaena glacialis) by season-year. Density and variance are the means of the transect estimates, weighted by transect lengths. Multiple surveys are included in each season, so estimates (N) and 95% C.I.'s are based upon multiple surveys/season. T = number of transects used in the analysis; G, I = number of groups and individuals (based upon photo-identification data, not transect data) sighted; D = density in animals/km2 for each season; V = variance of the density; N = estimated abundance in the SA by season; CI95 = 95% confidence interval, with the lower limit changed to zero if it was negative.

Season-Year	**T**	**G, I**	**D**	**V**	**N**	**CI95**
Autumn-2011	32	0, 0	0	–	0	–
Winter-2012	30	0, 0	0	–	0	–
Spring-2012	56	8, 13	0.0035	0.0027	24	0–118
Summer-2012	48	0, 0	0	–	0	–
Autumn-2012	24	0, 0	0	–	0	–
Winter-2013	16	3, 5	0.0045	0.004	35	0–296
Spring-2013	39	1, 1	0.0005	0.0003	4	0–43

Summer-2013	46	0, 0	0	–	0	–
Autumn-2013	36	0, 0	0	–	0	–
Winter-2014	26	1, 3	0.0008	0.0006	7	0–83
Spring-2014	41	4, 11	0.0019	0.0016	15	0–109
Summer-2014	60	0, 0	0	–	0	–
Autumn-2014	39	0, 0	0	–	0	–
Winter-2015	28	4, 15	0.0027	0.002	21	0–155
Spring-2015	65	10, 44	0.0029	0.0021	23	0–111

The estimates for right whale abundance in the SA ranged from 7 to 35 in the winter and spring months of each year, with 95% CI's from 0 to 296 (Table 1.3.4). When right whales were sighted, the numbers of individuals identified within each season ranged from 1 to 44, and a yearly average of 15.4 individuals were sighted in the SA across all survey years. Acoustic detections found that right whales are present within or near the WEAs during all months of the year, implying that aerial surveys missed individual animals or small groups outside of the window of greatest seasonal presence. When normalized, the spatial patterns of right whale acoustic detections in the SA were consistent with aerial detections (Figure 1.3.3.; Kraus et al. 2016).

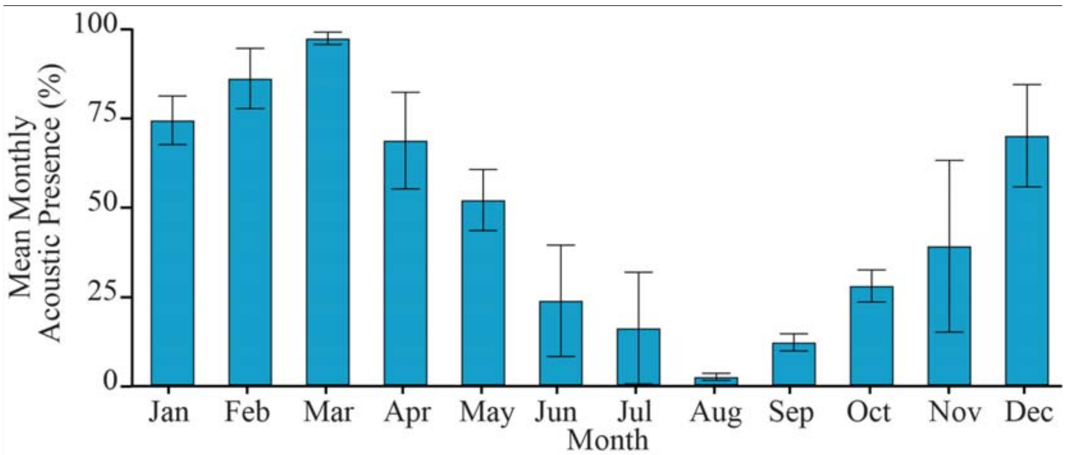

Figure 1.3.4. Right whale mean monthly acoustic presence ± standard error for all years combined.

Fin Whales

Seasonal abundance estimates of fin whales ranged from 0 to 59 animals with upper 95% confidence limits ranging up to 267 (Table 13). These estimates tended to be highest in spring and summer. Abundance was estimated at zero during the winter months.

Table 1.3.4. Density and abundance of fin whales (Balaenoptera physalus) by season-year. Density and variance are the means of the transect estimates, weighted by transect lengths. T = number of transects flown; G, I = number of groups and individuals sighted; D = density in animals/km2; V = variance of the density; N = estimated abundance in the SA; CI95=95% confidence interval, with the lower limit changed to zero if it was negative.

Season-Year	T	G, I	D	V	N	CI95
Autumn-2011	32	0, 0	0	–	0	–
Winter-2012	30	0, 0	0	–	0	–
Spring-2012	56	4, 4	0.0013	0.0005	9	0–48
Summer-2012	48	4, 5	0.0012	0.0007	8	0–60

Autumn-2012	24	1, 1	0.0006	0.0001	4	0–38
Winter-2013	16	0, 0	0	–	0	–
Spring-2013	39	6, 12	0.0022	0.0005	17	0–70
Summer-2013	46	3, 15	0.0009	0.0002	7	0–41
Autumn-2013	36	1, 1	0.0004	0.0001	3	0–25
Winter-2014	26	0, 0	0	–	0	–
Spring-2014	41	7, 10	0.0025	0.0011	19	0–98
Summer-2014	60	18, 34	0.0042	0.0018	32	0–116
Autumn-2014	39	0, 0	0	–	0	–
Winter-2015	28	0, 0	0	–	0	–
Spring-2015	65	7, 11	0.0015	0.0005	12	0–56
Summer-2015	17	8, 9	0.0076	0.0027	59	0–267

During the summer months, when estimated abundances were highest, fin whales were more likely to be observed in the RIMA and to a lesser extent, in the MA WEA, Zones 1–4. Fin whales were acoustically detected throughout the year; however, due to estimated detection ranges in excess of 200 km, the detections do not confirm that fin whales were vocalizing within the WEAs (Kraus et al. 2016).

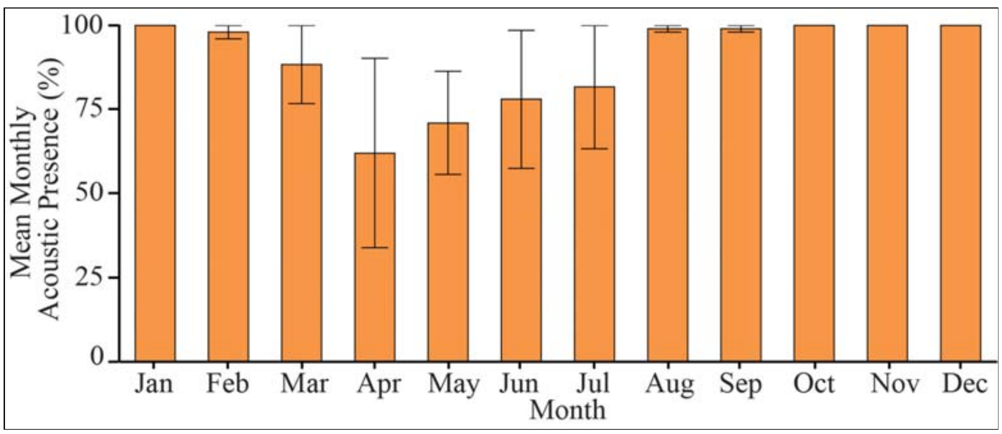

Figure 1.3.5. Fin whale mean monthly acoustic presence (± SE) between November 2011 and March 2015.

Humpback Whales

Humpback whales were sighted in the SA during all seasons, however they were primarily sighted in the spring and summer seasons (Table 5). The greatest number of sightings of humpback whales occurred during the month of April (n = 33), and their presence in the area seemed to start in March and end in July (Figure 1.3.6).

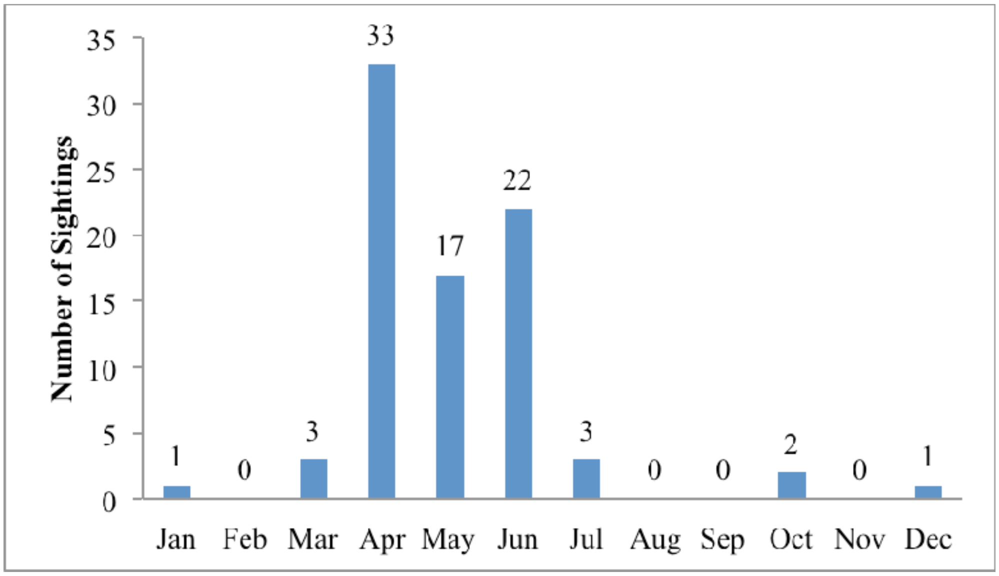

Figure 1.3.6. Humpback whale sighting totals by month, combined across all survey years (October 2011–June 2015).

Seasonal abundance estimates of humpback whales ranged from 0 to 41, with 95% upper confidence intervals of up to 168 (Table 1.3.5). These estimates tended to be highest in spring and summer with some exceptions.

Table 1.3.5. Density and abundance of humpback whales (Megaptera novaeangliae) by season-year. Density and variance are the means of the transect estimates, weighted by transect lengths. T = number of transects flown; G, I = number of groups and individuals sighted; D = density in animals/km2; V = variance of the density; N = estimated abundance in the SA; CI95=95% confidence interval, with the lower limit changed to zero if it was negative (Kraus et al. 2016).

Season-Year	T	G, I	D	V	N	CI95
Autumn-2011	32	2, 2	0.001	0.0006	7	0–66
Winter-2012	30	0, 0	0	–	0	–
Spring-2012	56	3, 4	0.0011	0.0003	7	0–39
Summer-2012	48	0, 0	0	–	0	–
Autumn-2012	24	0, 0	0	–	0	–
Winter-2013	16	0, 0	0	–	0	–
Spring-2013	39	14, 21	0.0052	0.0024	41	0–160
Summer-2013	46	13, 17	0.0034	0.0021	26	0–128
Autumn-2013	36	0, 0	0	–	0	–
Winter-2014	26	0, 0	0	–	0	–
Spring-2014	41	13, 17	0.005	0.0029	39	0–168
Summer-2014	60	7, 29	0.0018	0.0011	14	0–79
Autumn-2014	39	0, 0	0	–	0	–
Winter-2015	28	2, 2	0.0011	0.0003	9	0–63
Spring-2015	65	6, 13	0.0014	0.0005	11	0–55
Summer-2015	17	0, 0	0	–	0	–

The distribution of humpbacks tended to be farther offshore in spring, although detections occurred in both the RIMA and Zones 1–4 of the MA WEA. Aerial detections in the RIMA occurred only in summer. Abundance estimates tended to be highest in spring and summer. Acoustic detections of humpback whales occurred over a longer seasonal period (more total months), and were similar for both the RIMA Array and the MA WEA Array. Acoustic presence data suggest more humpback whales occur in the winter (December through February) than aerial detections indicated (Figure 1.3.7). There was little variation in humpback acoustic presence among years (Kraus et al. 2016).

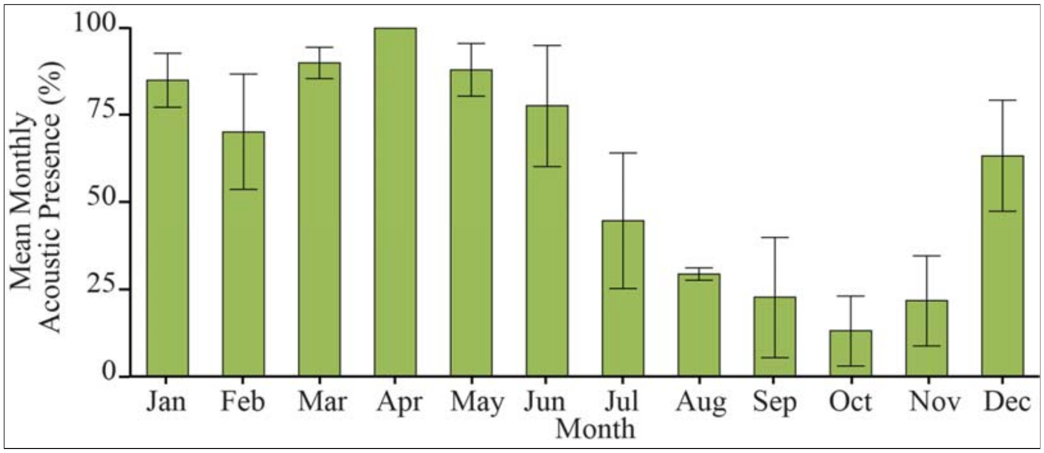

Figure1.3.7. Mean monthly acoustic presence (± SE) for humpback whales between November 2011 and March 2015.

Sei Whales

Seasonal abundance estimates of sei whales ranged from 0 to 27 animals with upper 95% confidence limits ranging up to 202 (Table 18). These estimates were only calculated during spring and summer when animals were seen, and there were no sightings available in spring and summer of 2012.

Table 1.3.6. Density and abundance of sei whales (Balaenoptera borealis) by season-year. Density and variance are the means of the transect estimates, weighted by transect lengths. T = number of transects flown; G, I = number of groups and individuals sighted; D = density in animals/km2; V = variance of the density; N = estimated abundance in the SA; CI95=95% confidence interval, with the lower limit changed to zero if it was negative.

Season-Year	T	G, I	D	V	N	CI95
Autumn-2011	32	0, 0	0	–	0	–
Winter-2012	30	0, 0	0	–	0	–
Spring-2012	56	0, 0	0	–	0	–
Summer-2012	48	0, 0	0	–	0	–
Autumn-2012	24	0, 0	0	–	0	–
Winter-2013	16	0, 0	0	–	0	–
Spring-2013	39	4, 6	0.0013	0.0007	10	0–75
Summer-2013	46	0	0	–	0	–
Autumn-2013	36	0	0	–	0	–
Winter-2014	26	0	0	–	0	–
Spring-2014	41	1, 2	0.0003	0.0004	3	0–48
Summer-2014	60	6, 12	0.0013	0.0013	10	0–80
Autumn-2014	39	0	0	–	0	–
Winter-2015	28	0	0	–	0	–
Spring-2015	65	3, 7	0.0006	0.0005	5	0–47
Summer-2015	17	4, 4	0.0035	0.0019	27	0–202

Sei whales appear to only frequent the SA in spring and early summer. The distribution of sei whales was throughout the SA, and abundance estimates of sei whales tended to be higher in the summer than in the spring. Due to the uncertainty associated with sei whale vocalization, they were not included as one of the focal species for systematic acoustic surveys.

Minke Whales

Minke whales are the smallest of the baleen whales observed in the SA, and were seen primarily during the spring and summer seasons (Figure 1.3.8.). No abundance estimates were generated for this species in the WEA's (Kraus et al. 2016).

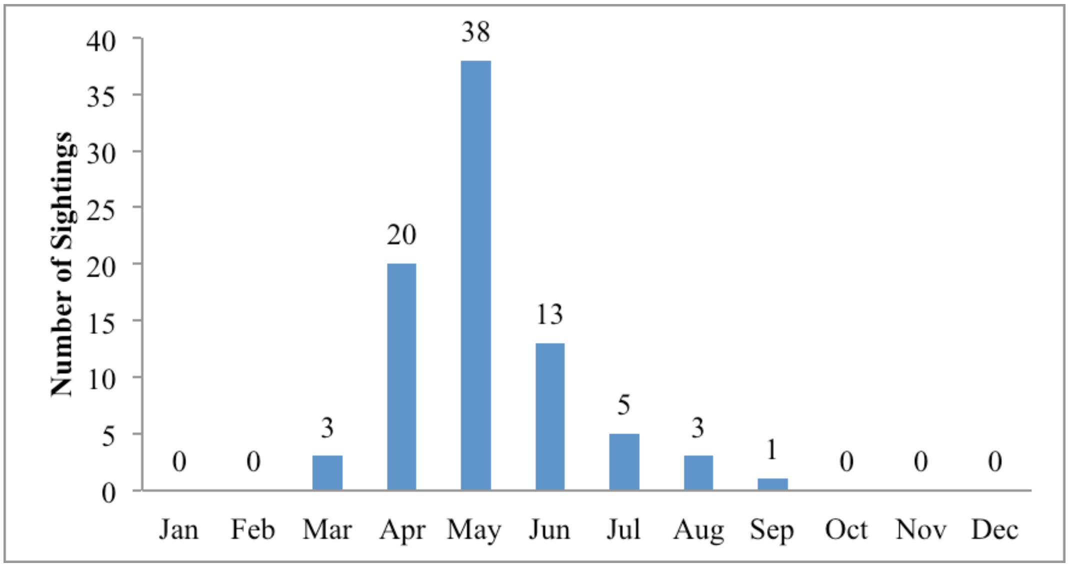

Figure 1. Minke whale sighting totals by month, combined across all survey years (October 2011–June 2015).

Minke whales occurred in the SA between March and September, with a peak in May. Distribution appears to be slightly more concentrated in the southern portion of the SA in spring, although sightings were reported in the RIMA and all four Zones of the MA WEA during that season. Acoustic detections occurred in more months of the year than visual observations. The mean monthly acoustic presence shows an overall trend of a gradual increase in presence starting in February, peaking in April, and then gradually decreasing through the summer. There is another slight increase in September and October.

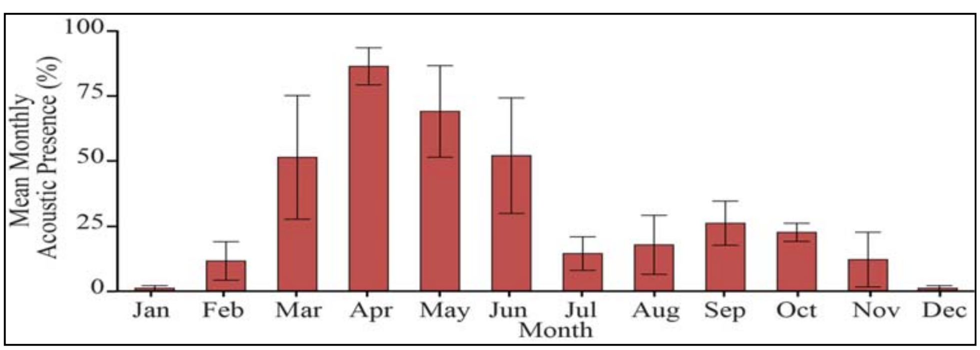

Figure 1.3.9. Minke whale mean monthly acoustic presence (± SE) between November 2011 and March 2015.

Sea Turtles

There were three species of sea turtles sighted in the SA; leatherback turtle, loggerhead turtle, and Kemp's Ridley turtle. Leatherback and loggerheads were sighted primarily during summer and autumn (Table 1.3.7 and Figure 1.3.10). There were no sightings of any species of sea turtle during the winter season.

Table1.3.7. Effort-weighted average sighting rates (SR, the number of animals per 1000 km), numbers of sightings (S), and numbers of animals observed (A) for three sea turtle species (only definite and probable identifications) and all sea turtles combined, by season. Total effort (km) is shown below each season name.

Species	Autumn (13,298.08 km)			Winter (11,846.17 km)			Spring (23,348.20 km)			Summer (18,683.15 km)		
	SR	S	A	SR	S	A	SR	S	A	SR	S	A
Leatherback	4.59	59	62	0	0	0	0.08	2	2	4.65	92	95
Loggerhead	3.97	45	45	0	0	0	0.07	2	2	1.52	31	31
Kemp's Ridley	NA	4	4	NA	0	0	NA	0	0	NA	0	0
All turtles	10.46	133	140	0	0	0	0.19	5	5	8.66	146	165

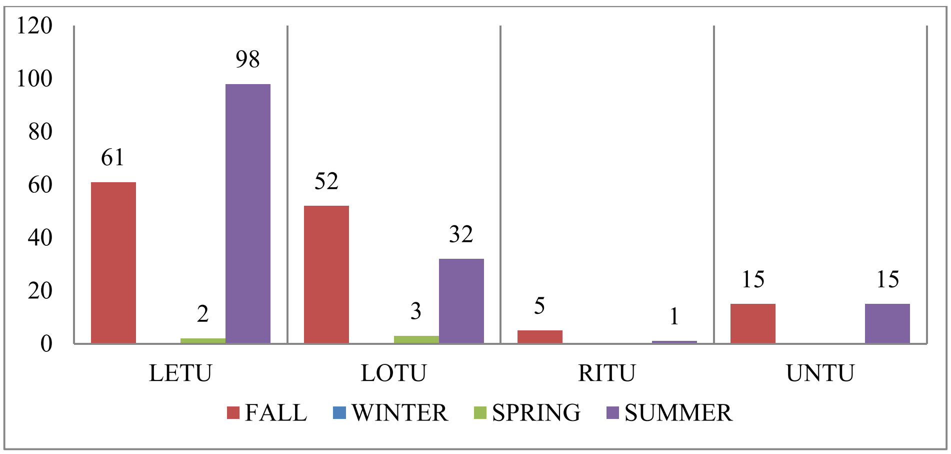

Figure 1.3.10. Sea turtle sightings in the SA by season across all years (LETU = leatherback turtle, LOTU = loggerhead turtle, RITU = Kemp's ridley turtle, UNTU = any sea turtle sightings not identified to species).

Only Leatherbacks had adequate number to estimate abundance. Seasonal abundance estimates of leatherbacks ranged from 0 to 99 animals, with upper 95% confidence limits ranging up to 616 (Table 1.3.8). Abundance estimates were highest in 2012 and 2014.

Table 1.3.8. Density and abundance of leatherback sea turtles (Dermochelys coriacea) by season-year. Density and variance are the means of the transect estimates, weighted by transect lengths. T = number of transects flown; G, I = number of groups and individuals sighted; D = density in animals/km2; V = variance of the density; N = estimated abundance in the SA; CI95=95% confidence interval, with the lower limit changed to zero if it was negative.

Season-Year	T	G, I	D	V	N	CI95
Autumn-2011	32	9, 12	0.0082	0.0220	57	0–412
Winter-2012	30	0, 0	0	–	0	–
Spring-2012	56	0, 0	0	–	0	–
Summer-2012	48	24, 25	0.0131	0.0579	90	0–560
Autumn-2012	24	12, 12	0.0133	0.0322	92	0–616

Winter-2013	16	0, 0	0	–	0	–
Spring-2013	39	0, 0	0	–	0	–
Summer-2013	46	2, 2	0.0012	0.0009	9	0–79
Autumn-2013	36	1, 1	0.0007	0.0005	6	0–61
Winter-2014	26	0, 0	0	–	0	–
Spring-2014	41	0, 0	0	–	0	–
Summer-2014	60	16, 16	0.0072	0.0087	56	0–239
Autumn-2014	39	16, 16	0.0127	0.0643	99	0–719
Winter-2015	28	0, 0	0	–	0	–
Spring-2015	65	0, 0	0	–	0	–
Summer-2015	17	2, 2	0.0037	0.0034	29	0–263

Leatherback turtles occurred in the WEA's between May and November, peaking in late summer. During this seasonal occurrence in the SA, leatherback turtles were most highly concentrated south of Nantucket, although there were sightings throughout the area. Loggerhead turtles primarily occurred in the SA in August and September. Distribution of turtles was widely dispersed throughout the SA.

Finding: *Updated supplementary spatial density models are needed for wind energy areas to manage potential conflicts with marine mammals and sea turtles.* For example, in the Massachusetts WEA case, the MassCEC and BOEM studies on distribution and abundance of wildlife in the SA presented abundance estimates and spatial maps based upon sighting per unit of effort (Kraus et al., 2016; Leiter et al., 2017; Stone et al., 2017). Roberts et al. (2016, 2017) have created more sophisticated models of seasonal and spatial occurrence of multiple species based upon environmental variables and sightings data from systematic surveys. However, the Roberts et al. model does not include the aerial survey data from the New England WEAs. In addition to including these aerial survey data, it may also be possible to incorporate additional sources of information, such as from the passive acoustic surveys.

1.4 Existing knowledge of habitat and oceanography

Adequate understanding of the potential impacts of any stressor on a marine population, especially the indirect impacts, presumes some basic information about their habitat requirements and prey resources. For the continental shelf south of Massachusetts and Rhode Island, the physical oceanography is relatively well understood. The general pattern of currents in the Gulf of Maine region were described by Bigelow (1927) (see also Beardsley and Flagg 1976, Limeburner and Beardsley 1982, Beardsley et al. 1985, Butman et al. 1987, Ramp et al. 1988, Brooks 1996, Townsend et al. 2006, Wilkins 2006). The interior Gulf of Maine is dominated by a cyclonic gyre with relatively discrete south-flowing coastal currents along the western margin (Fig. 1.4.1). When the coastal circulation reaches the northern part of the Great South Channel (east of Cape Cod), the flow splits, with part turning toward the northeast and becoming an anti-cyclonic eddy around Georges Bank and part turning toward the west over Nantucket Shoals. The mean, non-tidal circulation over the shelf south of Cape Cod and Rhode Island is westward, with about the same average volume transports in summer and winter. The sources are mainly from the flow around Cape Cod and secondarily from the southern part of Georges Bank, but there is very little exchange with the slope region because of a persistent shelf-slope front. Short-term variability in the mean circulation in the area of the WEAs is most prevalent during the winter, when stronger storms are more frequent.

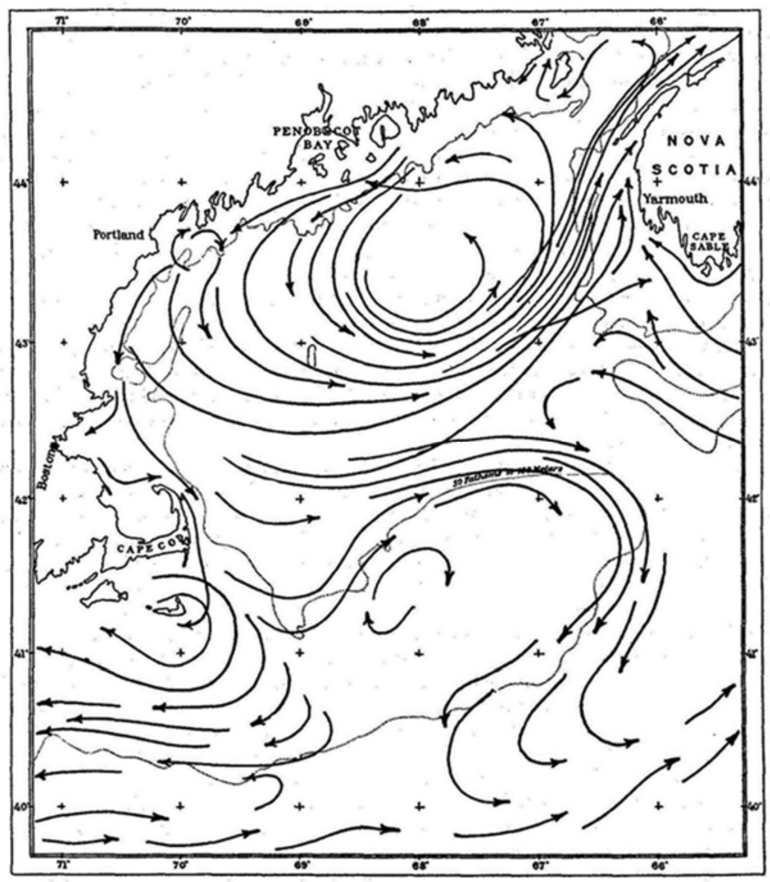

Figure 1.4.1. General ocean circulation in the Gulf of Maine, Georges Bank, Nantucket Shoals, and surrounding regions of the Northwest Atlantic (from Bigelow, 1927, Fig. 207).

Potential short- or long-term indirect effects of offshore industrial development on whales and sea turtles essentially boil down to effects on prey species. For the whale and turtle species of concern in the MA and RIMA WEAs, there are four main classes of prey resources to be considered (see Kenney and Vigness-Raposa 2010 for a detailed literature review concerning prey of all marine mammal and sea turtle species known to occur off southern New England)—zooplankton, pelagic fish, jellies, and benthic mollusks. North Atlantic right whales are obligate zooplanktivores, mainly feeding on larger calanoid copepods. Most of the other baleen whales are piscivores, feeding on small, schooling fishes such as sand lance and herring. Sometimes they can feed on krill, which are functionally more similar to pelagic fishes than to zooplankton. Sei whales may feed on either copepods or krill. Leatherback sea turtles feed primarily on large gelatinous animals such as jellyfish, ctenophores, and salps. Loggerhead sea turtles are benthic feeders, consuming mollusks such as gastropods (e.g., whelks) and bivalves (e.g., clams).

1.5 Existing knowledge on response to WEA construction and operation

1.5.1 Behavioral and physiological response to stressors

Assessing the effects of wind farm development on marine mammals and sea turtles will be a challenge, due to inter-annual variability, seasonal patterns, high animal mobility, and site characteristics (Scheidat et al. 2011). Most studies to date have been in the relatively well developed wind utility regions of Europe, and most have focused on harbor porpoises and pinnipeds (Schuster et al. 2015; Brandt et al., 2011; Bailey et al., 2010; Russell et al., 2015; Tougaard et al., 2009), on projects that did not include large scale mitigation measures, e.g. bubble curtains. In these taxa, studies have shown displacement during construction and variable levels of recovery in the wind energy areas after construction ceases. In harbor and gray seals, tagging data around wind energy sites showed behavior consistent with foraging after construction was completed (Russell et al., 2014; 2016). In harbor porpoise, displacement from pile driving areas was immediate (Brandt et al, 2011), and in some cases, long lasting (Teilmann and Carstensen, 2012). However, at a wind farm in the northern Irish Sea, there was a significant reduction in relative harbor porpoise abundance both within and surrounding the offshore wind farm during construction, but no significant difference was detected between the preconstruction and operational phases (Vallejo et al., 2017). One study in the Dutch North Sea found that there was an overall increase in harbor porpoise acoustic activity from baseline to operation (Scheidat et al., 2011). Brandt et al. (2016) analyzed the effects of the construction of eight wind farms within the German North Sea (2009-2013) on harbor porpoise. In this study they found clear short-term displacement (1-2 days duration) that was also likely affected by general construction vessel activity, but concluded that even without properly functioning mitigation techniques (bubble curtains), there was no indication that harbor porpoise within the German Bight were/are negatively affected by wind farm construction at the population level. Thus these two different taxonomic groups behaved very differently after construction and during wind farm operations. Pinnipeds appear to either habituate quickly, or may take advantage of the windfarm physical structures as a foraging opportunity, whereas harbor porpoise show high variability in displacement and recovery response to wind farm construction and operations.

Studies on large whales and sea turtles have not been done because few members of these species occur in the wind energy areas in European waters with any frequency. However, pile-driving noise has the potential to be a major potential stressor for right and other large whales. Pile-driving sounds are impulsive, low frequency and broadband, traveling across large distances of ocean and overlapping with acoustic frequencies that baleen whales depend on for communication. In feeding areas, displacement could lead to reduced foraging time, with reduced body condition and health as a consequence. Reduced foraging time could also be caused by direct disturbance of foraging events, or reduced ability to find prey (or direct others to the prey) or because of indirect effects such as changes in prey behavior or abundance due to prey responses to acoustic disturbance (see also Forney et al., 2017).

All species are subject to potential acoustic masking, as reverberation, multi-path transmission, and oceanographic factors will cause impulsive sounds to raise ambient noise levels (Clark et al. 2009; Estabrook et al., 2016). Because humpback and right whale mother-calf pairs communicate very quietly (probably as an anti-predator strategy) they are likely susceptible to masking (Videsen et al. 2017). Right whales have demonstrated a stress response to broadband noise from shipping (Rolland et al. 2012), and bowhead whales respond to seismic exploration signals at ~100 dB (Blackwell et al. 2015). There is enough evidence to suggest that pile-driving around wind farm construction has the potential to harm endangered whale species. In addition, since right whales currently experience chronic stress from continuous exposure to shipping noise, there are concerns that any additional anthropogenic noise sources, even intermittent ones like pile-driving noise, could increase the chronic physiological stress levels in right whales. Chronic stress in all mammals (including humans) reduces immune and endocrine function, negatively affecting health and reproductive fitness and leaving them vulnerable to disease (Schick et al., 2013; Rolland et al., 2017). The acoustic characteristics of pile-driving activities occurring during offshore windfarm construction have been detected out to 80 kilometers, beyond which they were no longer detectable above background noise (Bailey et al. 2010). These sound levels were related to noise exposure criteria for marine mammals to assess possible effects. For bottlenose dolphins, auditory injury would only have occurred within 100 m of the pile-driving and behavioral disturbance, defined as modifications in behavior, could have occurred up to 50 km away (Bailey et al. 2010).

Findings:

1) There is a need for a comprehensive review of effects such as behavioral responses and energetic consequences of impulsive sound sources on marine mammals.

2) There is an urgent need for empirical data collection on the effects (i.e responses) of impulsive sounds on marine mammal species to help validate the existing modeling efforts (that are currently mostly hypothetical).

Both the review and additional empirical data on responses, would inform both mitigation and management measures as well as population models (PCoD, PCoMs) to assess population consequences of such disturbance. In addition, empirical data collection before, during and after windfarm construction is needed.

1.5.2 Demographic and population-level response to stressors

Measuring short-term behavioral and physiological responses is relatively tractable. Demonstrating long-term impact on individual demography and then population level response is much harder – chronic effects over long time scales are hard to attribute to specific causes. To identify population level responses to disturbance, the effect size usually needs to be large. Recent efforts to develop tools for detecting and measuring the population-level consequences of disturbance (PCoD) include a set of mathematical frameworks that can be used quantitatively to assess the magnitude of these effects (Pirotta et al. 2018). A key step was to include the concept of "health" (often quantified in terms of energy stores) as a way to link short-term effects of disturbance with long-term demographic outcomes on individuals. A number of case studies have been created (Pirotta et al. 2018), and work is ongoing to transition the methods to an operational context. An interim approach (iPCoD) exists, for use in data-poor situations (King et al. 2015). This body of work is relevant to the marine renewable energy situation because installation and operation of wind farms may cause behavioral disturbance, potentially leading to population-level effects.

1.5.3 Cumulative effects

One complicating feature of assessing wind farm effects is that most marine mammal and sea turtle populations are exposed to many other kinds of stressors during a year. While some stressors may be minor by themselves, it is possible that the cumulative effect of the different stressors may create a biologically significant population level response. As an example, short-term exclusion of right whales from a feeding area may have little effect on survival or reproduction in a healthy population, but may have a strong effect in a population where many individuals are already in poor body condition due to entanglement. The concept of cumulative effects of multiple stressors is clearly closely linked to that of PCoD, since the latter considers the aggregate effect of multiple instances of a single stressor that causes behavioral disturbance. Cumulative effects generalize this by considering multiple stressors and going beyond just disturbance as a response. One reason that cumulative effects are harder to study and predict is that stressors may interact to produce large, synergistic effects. In 2017, a National Academies report was commissioned to review the wider context of the cumulative impact of multiple stressors on marine mammals. An expanded conceptual framework was developed (PCoMS – Population Consequences of Multiple Stressors), but it was recognized that this would be hard to implement in practice due to lack of knowledge about stressor interactions (National Academies (2017)).

Finding: *Research studies should be designed in such a way that they can contribute to a PCoD or PCoMS model.* This means that specific parameters for these models should be considered with each investigation.

1.6 Approaches to determining anthropogenic effects

Scientific investigations are classically divided into two categories: manipulative experiments and observational studies. In the former, multiple replicate experimental units are created and an experimental manipulation (a "treatment") is applied to a random set of these units, with the remaining units being left as controls. A measured difference in average response between the manipulated and control units can then be inferred to be due to the treatment. Observational studies can be similar in all respects except that there is no random allocation of units to treatment or control because the researcher is not in control of the manipulation. In this case, it is generally agreed to be impossible to definitively assign any observed effect to the treatment because there may be some other underlying factor, correlated with the treatment, that is the cause of the observed effect (although see, e.g., Sugihara et al. 2012 for an opposing view). In studies of wind energy development, it will rarely, if ever, be possible to adhere to the strict experimental paradigm. However, it may be possible to come close, and thereby reduce the chance of mis-diagnosing the cause of any observed change. This involves ensuring that studies have adequate replication to have a good chance of detecting an observed change of biologically significant magnitude (see below, on power analysis), and having adequate control sites or a gradient of locations around the treatment site.

Fleischman et al. (2016) suggest that four components are needed in programs designed to monitor the physiological, behavioral, and population level effects of human activities on marine mammals. First, they recommend the development of a set of plausible mechanistic hypotheses how a given activity might have one or more measurable effects on individuals and populations, as well as the magnitude, timing, and duration of those effects. Second, Fleischman et al (2016) recommend defining biologically meaningful effects, to determine if the proposed monitoring program can detect those magnitudes of effect with the desired levels of precision. Third, identifying appropriate response variables will be critical for assessing whether changes in the status of individuals or populations are attributable to a given activity. Visual surveys, passive acoustics, tagging, and direct physiological measurements all can provide data for quantitative hypothesis testing. Finally, they recommend the development of a temporal sequence for disturbance monitoring, i.e., before, during, and after a disturbance.

A classic design for detecting impact is the Before-After-Control-Impact (BACI, see, e.g., review by Smokorowski and Randall 2017). Here, in addition to the treatment ("impact") site, a control site is chosen, and both are monitored before, during and after the treatment takes place. If the parameters being monitored change in a different way in the control site vs the treatment site, this is attributed to the treatment. Note that multiple impact and control sites may be necessary, depending on whether the inference required is to be specific just to that one site, or more general. Note also that selection of appropriate control sites is often problematic, because they should identical, or as similar as possible, to the treatment site.

An alternative design is a spatial gradient, where an area around the treatment site, large enough that no effect is expected at the margins, is monitored before, during and after the treatment. Here, a change in the spatial surface for the parameter(s) of interest, during or after the treatment, is inferred to be an effect of the treatment if that change is centered on the treatment site (see Mackenzie et al. 2013 for a detailed description of analysis methods for spatial gradient designs).

Critical to the success of any monitoring program is the high probability of detecting the expected effect changes in the variables identified above. Here "power analyses" are needed to understand the data collection requirements with regard to sample size. We put power analysis in quotes because the term strictly relates to a study of the probability of correctly rejecting a statistical null hypothesis in a classical hypothesis test, while here we acknowledge that a wider range of statistical techniques may be used to detect an effect, including Bayesian approaches and estimation of effect sizes and corresponding confidence intervals, rather than statistical hypothesis tests. Nevertheless, the concept is the same: before a study, it is important to optimize the sampling so that there is a high probability of detecting a biologically meaningful effect should one exist. One conceptually straightforward way to achieve this is through a simulation study, where multiple realistic scenarios with a biologically meaningful effect are simulated on a computer, analyzed as if they were real data, and the proportion where the effect is detected using the proposed analysis method is taken as an estimate of the statistical power of the approach. An example of a simulation package for spatial gradient designs is MRSeaPower (Mackenzie et al. 2017).

Determining effects of marine renewables is likely to be a long term undertaking. Hence, studies designed in the light of knowledge at the time they were commenced may need to be updated, as new information becomes available. This has strong parallels with the concept of adaptive monitoring (Lindenmayer and Likens, 2009), where the monitoring program should change what is measured as the scientific hypotheses under consideration are updated.

1.6.1 Data Collection Methods

Generally established methods are preferable in hypothesis testing, as the strengths and weaknesses of those methods are well understood. However, there is value in having a parallel research stream for supporting emerging technologies that may be more effective.

There are several potential data-collection methods available for testing hypotheses about disturbance, both short and long term. To test whether sightings or occurrence of animals has changed, approaches could include aerial surveys, remote sensors (e.g., infrared, radar, LIDAR), passive acoustic monitoring including 1) archival methods (e.g. bottom-moored recorders) and 2) real-time acoustic monitoring (moored buoy, Slocum glider, wave glider). To assess short term behavioral and physiological responses, methods could include tagging (implantable and/or suction cups), drone monitoring, and assessments of hormones in scat and blow. To control for other variables that could influence animal responses, it will be necessary to implement some habitat monitoring/oceanographic sampling. The chosen monitoring program will need to be flexible, and should be able to incorporate new technologies that may come online. Finally, it will be important to determine whether the existing location is adequate as its own control, or whether alternative sites will be necessary to assess responses of animals to wind utility activities.

When choosing data-collection methods, additional considerations include prioritizing species of interest (e.g., acoustic and visual detection characteristics), cost, data turnaround time (real-time or archival?), technology development stage (deployable, proven methods or R&D approaches?), geographic scale, detection range, limitations due to ocean and weather conditions, local capacity, ease of implementation, suitability for short-term or long-term studies, durability, and reliability of detections.

1.6.2 Candidate Species

Baleen Whales: Right whales, because of their highly endangered status, are of major interest to all parties. Because of their winter and spring occurrence, and well-known behavioral and acoustic detection characteristics, this species is amenable to survey and acoustic approaches to testing the disturbance hypotheses. However, their highly endangered status, and some behaviors (for example, physical contact with each other and the sea floor makes tagging problematic), indicate that other baleen whales may be more suitable for collecting certain types of information on disturbance. For example humpback whales are readily detectable acoustically and visually, feeding behavior is often visible at the surface, and density can be moderately high in the MA WEA. Other species have less desirable traits. For example, finback whales are readily detectable, and occur in moderately high density, however, acoustic detections occur over long distances (ca 100-200km), so it would be difficult to determine if they were near the pile-driving of wind farm operations area. Minke whales also occur in relatively high density, and their low acoustic detection range is suitable for near field effect determination, but they are less visible and their behavior is difficult to discern. Given this variability, not all data collection approaches will be appropriate for all species.

Toothed Whales and Dolphins: Both Common dolphins and white sided dolphins occur frequently in the MA WEA, and could be good candidates for studying the effects of disturbance. Survey and acoustic detections are well-characterized, and numbers may be adequate for robust statistical treatments of disturbance effects. However, these species tend to be highly mobile in the area, potentially making interpretation of results challenging.

Sea Turtles: Because sea turtles are acoustically non-detectable, disturbance studies of the effects on endangered sea turtles are likely to rely entirely upon changes in distribution and abundance, making visual surveys critical to assessments. Tagging studies may be possible as well, requiring captures and releases in the area.

1.7 Summary of scientific hypotheses generated during workshop

Following the first step advocated by Fleischman et al. (2016), a set of core scientific (and as far as possible mechanistic) hypotheses were generated by those attending the workshop, relating to the hypothesized short-term effects of wind energy development and long-term effects of operation. These were then partitioned into a set of core hypotheses that were identified as being testable using existing field and statistical methods, and the others, which were judged to be less tractable. The criteria for selection included two factors. First, testing can be done using proven technology, although we could envisage parallel development of emerging technology for future studies. Second, it appears to be feasible to undertake both a power analysis (or equivalent quantitative assessment of the likelihood of detecting a biologically meaningful effect), and cost-benefit analysis in advance of testing the hypothesis.

The core hypotheses are listed here without comment, but they are expanded upon in the following two chapters, together with a consideration of their relative scientific importance, testability and potential study designs for addressing them. These chapters also contain details on the other hypotheses generated.

Note that each hypothesis is written here in a generic way – without specifying a particular activity, species, spatial/temporal scale (beyond short-term and long-term) or, in some cases, effect. However, before they are used to design any study, they will need to be made specific. For example, the first hypothesis under short-term effects is “Construction activities result in displacement of whales or sea turtles away from activity locations.” One specific version of this hypothesis could be “Pile driving in the MA WEA results in displacement of North Atlantic right whales an average of 20km away from pile driving sites, with return to baseline conditions 2 days after activities stop.”

Note also that the wording of the hypotheses has been edited slightly from the form of words used in the workshop, with the aim of clarifying them while retaining their meaning.

Other considerations for testing hypotheses include the ability to infer causation, whether a monitoring program should be adaptive (i.e., could it contribute to adaptive management?), and the features of successful, long-term monitoring programs (Thomas et al. 2004). The tools for conducting such assessments are openly available and straightforward (e.g., Schick et al. 2013, 2015; New et al., 2013, Pirotta et al., 2018). Furthermore, these models include the ability to test the importance of various parameters (e.g., behavioral changes, New et al. 2013, 2014) for population level significance.

Hypotheses relating to short-term effects of wind energy development

Hypothesis 1: Construction activities result in displacement of whales or sea turtles away from activity locations.

Hypothesis 2: Construction activities disrupt critical behaviors of whales or sea turtles, such as feeding, socializing, or nursing.

Hypothesis 3: Construction activities cause elevated stress hormone levels in whales or sea turtles.

Hypothesis 4: Construction activities cause zooplankton or fish prey to change their vertical distribution, density or patch structure.

Hypotheses relating to long-term effects of wind energy operation

Hypothesis 1: Wind turbine presence either excludes or attracts whales and sea turtles.

Hypothesis 2: Wind turbine presence affects long-term feeding opportunities for whales and sea turtles.

Hypothesis 3: The development of artificial reefs on wind turbine foundations affects the regional ecosystem, potentially enhancing some characteristics of marine productivity.

2 Short-term effects of wind energy development

In this report, “short-term” is defined as the time period during which construction-related activities, principally pile driving, may take place. We focus on pile driving because it is by far the noisiest element of construction; however, we note that related activities such as increased ship traffic may also potentially have an effect. As an example of timing, in the MA WEA, developers estimate it may take from 6 to 9 months to install the first phase of wind turbine foundations, with 90-120 minutes of pile driving taking place every day, during daylight hours. Pile driving during construction will have large acoustic impacts with sound pressure levels from 200 dB re 1µPa @ 1m to 270 dB re 1µPa @ 1m at the source (Degraer et al., 2010). Variables such as pile diameter, subsurface soil composition, and frequency of pile driving impacts will affect the sound source levels, and submarine topography and oceanographic conditions can affect underwater acoustic transmission. Nevertheless, these sounds are likely to be audible to marine mammals and fish at distances of tens of kilometers or more (Tougaard et al., 2008; Bailey et al., 2010).

In the study designs identified within each hypothesis below, there are some overarching considerations. In addition to animal safety and welfare concerns, each study will need to take into account project safety and construction or lease requirements (e.g. exclusion zones) and study designs will require close co-ordination between researchers, industry, and appropriate State and Federal agencies.

2.1 Displacement hypothesis

Hypothesis 1: *Construction activities result in displacement of whales or sea turtles away from activity locations.*

Evidence

No direct evidence on large whales is available. Aerial surveys have been used in studies of harbor porpoise responses to pile driving in Germany (Dähne et al. 2013). Passive acoustic studies have also shown significant displacement (mean of 17.8km) of harbor porpoise from pile driving in the Danish North Sea, and the effect lasted as long as pile driving was underway (5 months) (Brandt et al 2011), although other studies have shown high variability in porpoise responses in other areas (Teilmann and Carstensen, 2012; Scheidat et al, 2011; Dähne et al., 2013)

Aerial survey studies on acoustic disturbance displacement of large whales have been done to evaluate short-term bowhead responses to seismic activity (Richardson et al., 1999) and humpback responses to low frequency broad-band transmissions by the North Pacific Research laboratory (Mobley et al., 2005). Based upon the above studies, there is reason to believe that displacement of large whales away from the pile driving source sounds is likely. Sea turtle responses to loud impulsive sounds are unknown.

Importance

This was classified as highly important, since numerous endangered species occur in the area, presumably attracted by feeding opportunities. Displacement from feeding could lead to energetic losses that may have repercussions for reproduction and health.

Testability

High

Study design

There are multiple approaches to test this hypothesis. Frequently used methods include aerial surveys and passive acoustic monitoring before, during and after construction activities, then evaluating the distribution of animals (either visually or acoustically) under each of the conditions.

Visual aerial surveys can provide population-level data on abundance and distribution for most species, and occasionally individual-level responses for photo-identified right whales. The advantages of this approach include information on density and local displacement responses of multiple animals, and it will work for all species, and to some extent, all life history stages. Aerial surveys can also observe behavior (e.g., feeding) and may be able to detect changes. Visual survey methods have proven effective in studies of harbor porpoise responses to pile driving in Germany (Dähne et al., 2013). Aerial survey studies to assess displacement of large whales from acoustic disturbance have been done to evaluate bowhead responses to seismic activity (Richardson et al., 1999) and humpback responses to low frequency broad-band transmissions (Mobley et al., 2005). The potential downside is that this approach may need many flights to get adequate sample sizes, and power analysis in advance will be required. For certain species and times of year there may be inadequate numbers of animals to provide enough statistical power to determine if displacement has occurred (see McDonald et al., 2012)

Passive acoustic studies can provide population-level (as opposed to individual-level) inference. Significant displacement of harbor porpoise was detected using porpoise click detectors around pile driving in the Danish North Sea (Brandt et al., 2011). The success of this approach is dependent upon the volubility of the target species, and will be most effective in obligate echolocators (like harbor porpoise). For acoustic signals from large whales, using a "Gradient" design, strategically placed autonomous underwater recording devices can provide another measure of distribution and abundance shifts. Depending upon the species of interest, researchers will have to determine whether to use broadband recorders vs lower frequency recorders (for baleen whale) or higher frequency loggers for delphinids. Difficulties with this approach are that it cannot tell the difference between animals leaving the area and animals changing call behavior (e.g. going quiet for the duration of the pile driving). Also, passive acoustics will be less effective for some life history stages (e.g., mother-calves), and it will be ineffective for sea turtles. In one study on Bowheads, McDonald et al (2012) used fixed bottom hydrophones to localize calling whales, and used those locations to evaluate the spatial response of calling whales relative to the low frequency acoustic disturbance from a drilling operation. McDonald et al's (2012) approach may be applicable to passive acoustic methods in the wind energy areas. The success of this approach will depend on the species, vocalization rates, location, and time of year, but may have applications in wind areas.

Another possibility includes the application of satellite or radio tagging individuals before, during, and after construction activities. However it may be difficult to tag enough animals to determine if the displacement is a population level event, rather than individual variability in movements and responses, and these methods are likely more suitable for studies on behavioral disruption (Hypothesis 2.2), where it is discussed further.

Pseudo-experimental exposure (PEE) studies have been done with studies of Navy sonar (AUTEC, SOCAL, Atlantic BRS, 3S) and seismic (3S, BRAHSS) activities (Southall et al. 2012). Generally these are one event studies, looking at specific individual animal responses to a specific stimulus, so small sample sizes are the norm. However, there may be the potential to do a controlled exposure study by adjusting industrial-scale pile driving deployment schedules.

2.2 Behavior disruption hypothesis

Hypothesis 2: *Construction activities disrupt critical behaviors of whales or sea turtles, such as feeding, socializing or nursing.*

Evidence

Marine mammals are acoustically sensitive, and there is evidence of industrial sound affecting the behavior of gray and bowhead whales (Cooke et al., 2015; Blackwell et al., 2015; Richardson et al., 1999). Harbor porpoise respond specifically to pile driving by moving away from the sound source (Dähne et al., 2013). Many studies have examined behavioral responses to other acoustic stimuli, including sonar, seismic airguns, shipping, conspecific-call playbacks, and novel sounds, in many cetacean species by evaluating diving, movements, and acoustic behaviors (Cerchio et al., 2014; Novacek et al., 2002; Castellote et al., 2012; Nieukirk et al., 2012; Risch et al., 2012; McCarthy et al., 2011). Disruptions in foraging behavior were observed in harbor porpoise exposed to high rates of vessel noise (Wisniewska et al., 2018). Likewise, Naval sonar appeared to disrupt foraging behavior in humpbacks (Sivle et al., 2016) and sperm whales (Isojunno et al. 2016). In contrast, studies of short-term airgun effects on sperm whales did not indicate a behavioral response between exposure levels (Miller et al., 2009), although repeated exposure could have potential population level effects (Farmer et al. 2018). In a recent review of sonar studies, Southall et al (2016) state "*Responses were highly variable and may not be fully predictable with simple acoustic exposure metrics (e.g. received sound level). Rather, differences among species and individuals along with contextual aspects of exposure (e.g. behavioral state) appear to affect response probability*." It is likely that comparable variability in responses will be found in response to pile driving and other wind farm installation activities. This suggests that large sample sizes will be needed to determine the important species-specific responses vs individual variability, regardless of whether the data are acoustic, observations, or tagging tracks.

Importance

High

Testability

Medium

Study design

Passive acoustics has been used to assess changes in acoustic behavior for humpbacks (Risch et al 2012), bowheads (Blackwell et al., 2015), harbor porpoise (Brandt et al, 2011) and could be used to test the behavioral alteration hypothesis here. Risch et al (2012) showed changes in the acoustic behavior of humpbacks in response to a "sonar" type signal over 200 km away. Even though the sound characteristic of various sound sources can be dramatically different, and made more so depending on the associated environmental factors that affect sound propagation because most marine mammals are dependent upon vocalizations for communication and frequently for foraging, changes in acoustic behavior may be easily detected. Passive acoustic monitoring offers a relatively high return of data for multiple species, it is effective in most weather conditions, and can be configured to produce near-real time information if desired. Recordings of the species of interest before, during, and after construction activities, would allow the development of baseline acoustic characteristics (calling rates, frequencies, intensity) for the study area and season, against which subsequent changes can be compared.

Movement responses to pile driving could be assessed by appropriately designed short-term tagging studies (Tyack, 2009). It may be feasible to tag several species of baleen whales with DTAGs before pile driving commences to collect data on whale behavioral responses. Right, humpback, sei, and finback whales are all potential targets for these studies. Other archival and implantable tag types may be considered for longer duration studies, as this technology is rapidly improving (Szesciorka, et al., 2016). However, in some cases, notably right whales, implantable tagging should be considered with extreme caution, given the health status of the population, the low tag survival rates on this species, and the experimental nature of contemporary implantable tags. During the summer months, sea turtles are also appropriate targets for tagging studies, as this taxonomic group is fairly amenable to long-term attachments (Hays and Hawkes, 2018).

Finally, if there are species for which there are easily visible behavioral characteristics, it may be possible to conduct aerial surveys in control area and within the MWEA to assess whale and turtle behavior before, during and after construction. However, comparing changes in behavior between sites before, during, and after construction would be useful only if those changes in behavior were very obvious.

2.3 Stress hypothesis

Hypothesis 3: *Construction activities cause elevated stress hormone levels in whales or sea turtles.*

Evidence

Acoustic disturbance from shipping has been shown to elevate stress levels in right whales (Rolland et al. 2012). Physical stressors of whales (e.g. entanglement events) have been shown to increase stress hormone levels as long as the stressor is present (Rolland et al., 2017; Lysiak et al., 2018). Although the physiological responses of marine mammals to pile driving or construction activities have not been measured, it is likely that noise that is much louder than shipping will evoke a stress response in marine animals that depend upon sound for food-finding and communication. Methodologies to characterize those effects on physiology through the use of hormone panels have recently become available (Burgess et al., 2018; 2017; Rolland et al., 2017; Hunt et al., 2013). The effects of various stressors can cause changes in behavior or physiology that affect fitness by reducing an individuals' health and vital rates (survival, reproductive success, and growth rates). The population-level consequences of changes in individual fitness depend on what proportion of the population is affected and the magnitude of those changes across the affected portion of the population (Pirotta et al., 2018). Measurable physiological biomarkers could afford managers insights into sub-lethal effects on health and reproduction.

Importance

High

Testability

High

Study design

The approach would be to characterize the relationship between stress-related hormones measured in blow and variable exposure to ambient underwater noise occurring from pile-driving and other construction activities in the MA WEA's. This would require collecting ambient underwater acoustic data using a dedicated hydrophone system prior to collecting blow samples from whales in calm sea conditions (Beaufort <3). Recorded acoustic data should be matched to the relative time that blow sampling occurred for each whale, with the goal to record for 1 h prior to blow collection using a drifting autonomous recorder deployed in close proximity to the subject in order to best estimate received levels at the whale's location. In addition, remote bottom mounted recorders should be used in the general vicinity of feeding aggregations of whales to assess medium-term noise levels over several days. Acoustic data following NOAA standardized technical guidelines (National Marine Fisheries Service, 2018) would be collected before, during and after the acoustic stimulus. Blow hormone profiles of whales exposed to sound disturbances need to be interpreted relative to life-history data of each individual whale. Multiple relevant variables could be statistically modelled (using generalized additive mixed models) to determine their potential influences on individual whale physiological state – and ultimately, to assess whether acoustic disturbances are associated with an elevated stress response. These data may help inform the PCOD approaches to population effects.

2.4 Prey hypothesis

Hypothesis 4: *Construction activities cause zooplankton or fish prey to change their vertical distribution, density or patch structure.*

Evidence

Weilgart (2018) reviewed a substantial number of studies of noise effects on marine invertebrates and fishes. McCauley et al. (2017) conducted a field study exposing zooplankton to a seismic air gun. The air gun used had a source level roughly equivalent to pile-driving noise (Illingworth & Rodkin, 2007; Deepwater Wind 2012; Amaral et al., 2018), but with significantly different sound propagation characteristics. They showed significant mortality of zooplankton, especially juvenile stages, out to at least 1.2 km. The spatial scale of the mortality, should it occur, would likely be quite small and confined to the area immediately adjacent to pile driving, and the duration of any mortality event should be short-term. It is unlikely that pile-driving would take place on more than one foundation at a time, further limiting the spatial extent of possible impacts. Zooplankton in the area of the WEAs is advected into the region from upstream on the westward currents, mostly from the Great South Channel, and therefore would be continually replenished. Any mortality caused by pile-driving therefore is unlikely to affect regional zooplankton abundance. A recent agreement signed by Vineyard Wind and three NGOs (NRDC et al. 2019) commits the developer to mitigation that would endeavor to reduce the source level from pile driving by 12 dB, further reducing the extent of any impact, although this does not yet extend to any other developer.

Aggregation of zooplankton into patches suitable for right whale feeding is controlled by physical oceanographic factors (e.g., stratification, horizontal and vertical currents, bathymetry, and frontal boundaries) augmented by the behavior of the zooplankton themselves (e.g., depth-keeping, diel vertical migration) (Kenney and Wishner 1995, Beardsley et al. 1996, Epstein and Beardsley 2001). Pile-driving is not likely to have any effect on physical oceanography sufficient to cause any change in zooplankton aggregation or patch structure. The southern New England shelf does not have the consistent physical features that cause predictable copepod aggregations as those which occur in Cape Cod Bay (Jiang et al. 2007), the Great South Channel (Kenney and Wishner 1995; Beardsley et al. 1996), or off Nova Scotia (Baumgartner and Mate 2003), and the features that do occur are more likely to be ephemeral and short-term and even less likely to be disrupted by a localized effect. It may be feasible that behavioral disturbance in the vicinity of pile-driving noise might disrupt a behavior such as diel vertical migration, but that would be small-scale and short duration.

The prey of sea turtles—jellyfish and benthic mollusks—are similar to zooplankton in having a reduced capability of moving away from sources of impact. There have been few studies of noise impacts on these species (see Weilgart 2018). There are suggestions of effects on mussels and scallops from air-gun noise, but at higher intensities, longer durations, and/or closer ranges than the zooplankton study described above. Solé et al. (2016) exposed two species of jellyfish in a laboratory experiment to low-frequency sound (1-second sinusoidal sweeps of 50-400 Hz, received levels of 157 ± 5 dB re 1µPa and peak of 175 db) for 2 hours and found microscopic evidence of injuries. Such injuries are typically caused by particle motion (i.e., the near-field displacement component of a sound rather than the far-field pressure-variation component) and would not occur except in the immediate vicinity of the source. The pre-installation modeling and monitoring data from the Block Island Wind Farm (Deepwater Wind 2012, Amaral et al. 2018) would address the ranges of potential impacts, both without and with 12 dB mitigation. It is likely that any impacts would be small-scale and short-duration, as with the zooplankton. Effects on the behavior of jellyfish or benthic mollusks that would affect their availability for sea turtle feeding are difficult to conceptualize.

The pelagic fish prey of baleen whales are highly mobile and capable of quickly moving away from pile-driving or any other source of potentially harmful noise. Avoidance behavior would be enhanced by the practice of "ramping up" during pile driving—beginning each pile-driving event with lower-energy strikes to alert sensitive species in the vicinity so they could move away (Bailey et al., 2010). Therefore, one should expect that mortality of pelagic fishes would be insignificant.

Short of actual mortality, sub-lethal impacts on pelagic fishes might be possible, e.g., from stress (see Weilgart 2018). Some studies, however, have failed to show any significant behavioral effects. Wardle et al. (2001) exposed coral reef communities to repeated passes of a seismic air-gun array and could detect no observable changes in behavior in either the fishes or the invertebrates. Hassel et al. (2004) compared the behavior of lesser sand eels (enclosed in cages on the bottom) exposed to seismic air-gun sounds to unexposed controls and found no apparent differences. The most logical prediction might be that pelagic fish exposed to pile-driving noise that they found disturbing would move farther away from the source, potentially reducing the availability of prey to any whales that did not also move away from the noise.

Importance

Low to medium

Testability

Studies conducted to date have been largely inconclusive (Weilgart 2018). Laboratory studies are often not applicable because any observed mortality is more likely due to near-field particle-motion effects. Natural spatio-temporal variability of occurrence of these prey types also complicates testing.

Study design

A possible study design would involve sampling down-current from an active pile-driving site and from a control site within the planned array (or two controls—with and without an installed foundation). It could include repeated net tows or other sampling (e.g., video plankton recorder tow-yos), perpendicular to the prevailing current and possibly at multiple distances, during a day of pile-driving and comparing zooplankton densities and frequencies of dead or damaged individuals between the active and control sites. VPR could distinguish between live and dead zooplankters. If the sampling also included larger-scale video, some measure of fish presence might also be feasible. Dead fish might be catchable in a zooplankton net, but not live fish. Simultaneous hydroacoustic sampling might also quantify fish density, however might not be capable of distinguishing living from dead individuals. None of these methods would address benthic mollusks. Diver surveys and collections at active and control sites (after the pile-driving for diver safety) could quantify mortality. A study setting out individuals (clams, mussels, whelks, etc.) in cages at various ranges from active and control locations would quantify any mortality and likely be more amenable to statistical analyses.

2.5 Other hypotheses

Other questions were raised during the workshop. These included topics such as the potential for co-occurring ecological shifts (e.g., plankton and oceanography studies) as well as the potential for indirect effects, for example, changes in vessel patterns or fishing effort that might create de-facto marine protected areas. These were not considered of primary importance and were not developed further.

3 Long-term effects of wind energy operation

Unlike the short term effects, which are likely to be observed over a period of 6 to 9 months, the long term effects question is concerned with the life-time of wind energy facilities (~30 years), and the potential effects that might last for years or decades. There are several scenarios possible, including physical alteration of the habitat and local oceanography by the towers, potential low level acoustic or electromagnetic energy emissions, and the consistent presence of maintenance vessels in and around the installation. In addition, there may be secondary effects, for example, the potential for the pilings to become artificial reefs, or the exclusion of commercial fishing and large vessel traffic from the area. Here the main hypotheses of concern are addressed in detail, although many unknowns about baseline information and climatological trends will challenge studies to determine long-term changes that are specifically due to wind installations.

3.1 Distribution change hypothesis

Hypothesis 1: *Wind turbine presence either excludes or attracts whales and sea turtles.*

This is an important hypothesis to address. We think it is critical to compare changes in whale abundance (aerial surveys), prey abundance, and oceanographic conditions in the MWEA to a nearby control area. Observations in the control area can help explain changes in the MWEA that are related to changes in larger scale oceanographic conditions (e.g., over the entire northeast continental shelf). For example, in the case of right whales, it might be possible to use Cape Cod Bay as a control area for the MWEA, where whale distribution, zooplankton species composition and phenology between the two areas could be compared.

Evidence

Like the short term studies on displacement, there are multiple approaches to test this hypothesis. These will include aerial surveys and passive acoustic monitoring before, during and after construction activities, then evaluating the distribution of animals (visually and/or acoustically) under each of the conditions. However, the main difference is that these studies will need to be continued for several years.

Visual aerial surveys can provide long-term population-level data on abundance and distribution for most species of interest. This approach can provide information on density and local abundance for all species, and to some extent, all life history stages. This approach may need multiple years of survey effort, and a power analysis in advance will be required to help determine the number of flights and sightings needed to detect changes. For certain species and times of year there may be inadequate numbers of animals to provide enough statistical power to determine if long-term displacement has occurred (see McDonald et al., 2012)

Passive acoustic studies can provide another measure of long-term distribution and abundance shifts. This method may not be able to tell the difference between animals no longer in the area and animals that are quiet in the vicinity of the wind installation. However, the presence or absence of various species detected on recorders can be validated by concurrent visual surveys. Passive acoustic studies on long-term effects will still be less effective for some life history stages (e.g., mother-calves), and it will be ineffective for sea turtles. The McDonald et al (2012) method could be used to assess long term changes in habitat use patterns by several acoustically active species, and if employed long-term, could also detect any habituation to wind energy installations.

Importance

High

Testability

High

Study design

The long term commitment required for these studies dictates that consistent methods need to be used for extended periods. Well understood monitoring methods include aerial surveys and passive acoustic methods. The concerns over uncontrolled variables (climate change) suggest that it would be wise to design a monitoring program that incorporates comparable studies in both the MA WEA and a control area to assess whale abundance and behavior before, during and after construction. To control for oceanographic and climate variable, it will be necessary to conduct prey and oceanographic studies (see hypothesis 3.2 below). Comparisons of abundance and trends in abundance between sites before and after installation of wind farm will be feasible only for very large changes in abundance.

3.2 Long-term prey hypothesis

Hypothesis 2: *Wind turbine presence affects long-term feeding opportunities for whales and sea turtles*

Evidence

It is possible that the foundations could cause wakes that might disrupt prey aggregations. This possible effect is only relevant to North Atlantic right whales, since their zooplankton prey are the only mammal or turtle prey in the region whose aggregations have a substantial physical causative factor. Turbulent wakes should have no effect on schooling of pelagic fishes or on benthic mollusks, therefore no impacts would be predicted to piscivorous whales or to loggerhead turtles. Drifting jellyfish could be influenced by the wake downstream of a turbine foundation, however since leatherback turtles do not depend on aggregations of jellyfish, no significant effect on their feeding should expected.

There have been modeling efforts that appear to show that offshore wind farms can cause wakes that will result in detectable changes in vertical motion and/or structure in the water column (e.g., Paskyabi, 2015; Segtnan and Christakos, 2015). There was also a remote-sensing study that detected wakes downstream from a wind farm by increased turbidity (Vanhellemont and Ruddick, 2014), showing that wakes certainly do occur. However, we currently know very little about the physical oceanographic phenomena (e.g., mechanisms, forcing factors, spatial scale, persistence) responsible for right whale feeding habitat in the southern New England region, so it is not possible to predict whether turbine wakes could impact right whale feeding, or to what degree. If there is some effect, it is likely to be small and restricted to a short distance down-current from the foundations.

A possible effect on loggerhead feeding might be sediment scour around the foundation bases, which theoretically could change availability of benthic mollusks. This is likely to be very localized and small-scale, and might be counter-balanced by increased prey resources in the communities that develop attached to the foundations and bases. Monitoring at the Block Island Wind Farm showed no detectable effects on benthic communities (Bartley et al. 2018).

Importance

Low

Testability

This does not appear to be testable given our current understanding.

Study design

The first step in addressing the hypothesis that turbine wakes affect the development of the dense copepod patches that right whales require would be better knowledge of how those patches form in this region. If that model existed, then it would be possible to add turbine wakes to it and explore the possible impacts. Since copepod aggregations suitable for right whales are not really predictable or detectable except by finding feeding whales, designing a field study to examine the effect of turbine wakes on copepod patch formation does not seem possible.

3.3 Ecosystem enhancement hypothesis

Hypothesis 3: *The development of artificial reefs on wind turbine foundations affects the regional ecosystem, potentially enhancing some characteristics of marine productivity.*

Evidence & Assessment

Artificial reefs formed by oil platforms (active and decommissioned) are a well-known "disturbance" of regional ecosystems. A review of the potential reef effects from offshore wind turbines by Linley et al. (2007) suggested largely positive or neutral effects on a variety of species, including mussels and kelp directly attached to the foundations and scour pads; crabs, lobsters, and oysters nearby, and even finfish in the vicinity. Slavik et al. (2018) estimated that up to four tons of mussels could grow on a single turbine foundation. The attached community might not be exactly the same as similar communities in the same vicinity; Wilhelmsson and Malm (2008) found that the fouling communities on wind farm foundation structures had lower species diversity than those on nearby boulders. At the Block Island Wind Farm, there have been many recent news reports, including underwater video, about dense mussel populations growing attached to the foundations and increased fish abundance (e.g., black sea bass) in the vicinity. There have been reports that commercial gillnet fishermen are setting their gear right up to the foundations to target the increasingly abundant fish. Wilber et al. (2018) examined the abundance, size, and condition of seven species of flatfish at impact and control sites in the area of the Block Island Wind Farm before construction (baseline), during construction, and during operation. They found no reef effect on flatfish, and no impacts, either positive or negative, from construction activity or wind farm operation.

Slavik et al. (2018) found that the substantial mussel populations attached to wind farm foundations could affect primary productivity in the vicinity. This is not likely to have any effect on the copepod patches that right whales could feed upon because (1) patch formation is not due to food-chain processes (Kenney and Wishner 1995) and (2) zooplankton in southern New England shelf waters are more likely to be advected in from source regions to the east rather than "home grown" (Ji et al., 2017).

It seems unlikely that reef effects from wind turbine foundations would have a significant effect on the small pelagic fishes that comprise the prey of other baleen whales. Those species are planktivorous, unlike fish species that might be enhanced near a turbine because they do feed on mussels or other attached biota (e.g., black sea bass, tautog, scup).

Both of the common species of sea turtles in the region could find enhanced feeding opportunities due to the reef effect. Loggerheads can very likely feed on the mussels, and leatherbacks might also eat mussels, or tunicates (sea squirts) that are another common component of local biofouling communities. Turtle researchers have often speculated that one of the factors causing sea turtle entanglement in fixed fishing gear is turtles feeding on biofouling organisms attached to the gear (e.g., to the vertical lines on pot gear) (Schwartz, 2009). It is feasible that turtles attracted to the wind farm to feed on biota attached to the foundations could be at increased risk of entanglement if the presence of fixed-fishing gear is also increased near the turbines.

Importance

Low / unknown

Testability

Testable, although since the only potential impacts on ESA-listed species would be on sea turtles, and those are more likely to be positive than negative, the priority for testing is very low.

Study design

Studies of the biofouling communities on the Block Island Wind Farm foundations are already underway, as are studies of fish stocks in the vicinity. A potential added study addressing sea turtles feeding on biofouling communities could be done relatively simply and inexpensively using video cameras attached to the foundations and monitoring the waters around the legs.

4 Conclusions and recommendations

A carefully designed research plan is needed in order to assess the ecological impacts of offshore wind facility construction and operation on marine mammals and sea turtles in U.S. waters. The May 2018 workshop provided expert input to develop a framework for studying the effects of offshore wind development on marine mammals and sea turtles. The authors have compiled all of the recommendations into this document for reference. At the outset, it was clear that with multiple variables, changing oceanic conditions, and inter-annual variability, research to determine wind installation effects will require careful experimental design, appropriate statistical methods, and data collection methods designed to collect adequate sample sizes.

Among the hypotheses that were considered, the concerns about displacement, behavioral disturbance, and physiological stressors rose to the top priorities. Recommended studies included both short and long-term displacement research, using both aerial surveys and passive acoustic methods. Behavioral studies were recommended for short term displacement and disturbance, using both passive acoustics and DTAGs. Physiological disturbance, or stress studies, were also recommended in conjunction with passive acoustics work. The hypotheses for lost or enhanced feeding opportunities by wind turbine foundations affecting the prey base were not rated by workshop participants as highly important to the wind farm installations as the other studies. However, workshop participants emphasized that the lack of understanding on prey species abundance in the WEA's will inhibit the ability of managers and researchers to distinguish between displacements effects, and the fact that animals may have simply left the area to look for better foraging elsewhere. For that reason, plankton and prey fish studies may be important for determining cause and effect of any construction or post construction on marine mammal and sea turtle observations.

Understanding the effects of wind installation development on several whale and turtles populations is the ultimate goal of this work. One recommendation is to apply PCoD modelling to the hypotheses, to determine the effects upon a given population by looking at the worst case scenario of development. For example, suppose a hypothetical group of animals are unable to migrate through or use the areas because they are somehow blocked by the installation – a PCoD approach can evaluate what percentage of feeding time is lost, or how much longer migration might take, and apply energy budget models to assess the consequences on those animals.

There were also several recommendations for pre-installation work that are summarized below.

- Updated supplementary spatial density models are needed for wind energy areas to manage potential conflicts with marine mammals and sea turtles. For example, in the Massachusetts WEA case, the MassCEC and BOEM studies on distribution and abundance of wildlife in the study area presented abundance estimates and spatial maps based upon sighting per unit of effort (Kraus et al., 2016; Leiter et al., 2017; Stone et al., 2017). Roberts et al. (2016, 2017) have created more sophisticated models of seasonal and spatial occurrence of multiple species based upon environmental variables and sightings data from systematic surveys. However, the Roberts et al. model does not include the aerial survey data from the New England WEAs. In addition to including these aerial survey data, it may also be possible to incorporate additional sources of information, such as the data from the passive acoustic surveys.
- There is a need for a comprehensive review of behavioral, physiological and population effects of impulsive sound sources on marine mammals.

- There is an urgent need for empirical data collection on the behavioral and physiological effects of impulsive sounds on marine mammal species to help validate the existing population consequences modeling efforts (that are currently mostly hypothetical).
- Research studies on the impact of wind energy installations should be designed in such a way that they can contribute to a PCoD or PCoMS model. This means that specific parameters for these models should be considered with each investigation.
- A comprehensive review of tagging data on the species of interest would be valuable for assessing the efficacy of tagging as a potential method for detecting behavioral changes in marine mammals and sea turtles in response to construction activities.
- A review of all passive acoustic work to date on species of interest is needed to help inform passive acoustic study design. For example, animals with low call rates will be less suitable candidates than those that are more voluble. A review will help select the candidate species, and can refine the power analyses to detecting changes over time.
- Zooplankton modelling was recommended. While we don’t understand the processes that generate patches that whales can feed on, biological oceanography modelling would be useful to provide some bounds on the unknowns, essentially serving as a gap analysis for future work.

Several workshop participants made recommendations with regard to the links between data collection and managing wind farm development. One, data should be collected in a manner that can inform regulatory and management decisions on individual project review and long-term cumulative impacts. Two, the framework should be adaptable to new lease areas as they come online and other stressors emerge (e.g., fishing, climate change), so that each wind project can be informed by the data collected from previous projects. Three, the framework should be designed to provide usable information about cumulative effects in order to respond to managers and regulators who may be required to assess them. Finally, the data collected following the research framework should help regulators and developers determine the best timing and methods for construction.

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Appendix 1. List of Participants

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Debra Palka	National Marine Fisheries Service
Julie Crocker	National Marine Fisheries Service
Peter Corkeron	National Marine Fisheries Service
Catherine Bowes	National Wildlife Federation
Francine Kershaw	Natural Resources Defense Council
Dan Pendleton	New England Aquarium
Ester Quintana	New England Aquarium
Scott Kraus	New England Aquarium
Meghan Rickard	New York State Department of Environmental Conservation
Laura Morse	Ørsted
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Dom Tollit	SMRU Consulting
Chris McGuire	The Nature Conservancy
Kelly Macleod	UK Joint Nature Conservation Committee
Bob Kenney	University of Rhode Island
Len Thomas	University of St. Andrews
Sue Moberg	VHB
Matt Robertson	Vineyard Wind
Rachel Pachter	Vineyard Wind
Elizabeth James Perry	Wampanoag Tribe of Gay Head Aquinnah
Howard Rosenbaum	Wildlife Conservation Society
Mark Baumgartner	Woods Hole Oceanographic Institution

Appendix 2. Glossary

abundance	The number of animals in a biological population
BACI	Before-After-Control-Impact. An experimental design for studying the effects of a stressor. In this design, one or more control sites are paired with one or more impact sites (i.e., sites where the stressor will operate). These are monitored both before and after the start of the stressor. Using this paired design, any possible change due to the stressor (which affect the impact site alone) can be distinguished from background changes (which affect both control and impact sites).
BOEM	Bureau of Ocean Energy Management
DTAG	Digital Acoustic Recording Tag (see Johnson et al. 2009)
ESA	Endangered Species Act
iPCoD	Interim Population Consequences of Disturbance (PCoD – see above). An interim approach to quantifying the population consequence of disturbance in data-poor situations, based on the use of expert elicitation. See King et al. 2015. MARU Marine Autonomous Recording Units (Section 1.3)
MA WEA	Massachusetts Wind Energy Area (see Figure 1.2)
MMPA	Marine Mammal Protection Act
PCoD	Population Consequences of Disturbance. Conceptual framework for studying the population-level effects (i.e., changes in population trend) caused by repeated behavioral disturbance from anthropogenic noise and other chronic stressors. A set of quantitative approaches have been created based on this framework for particular case studies. See Pirotta et al. (2018).PCoMS . . .
Power analysis	Power analysis allows researchers to determine the sample size required to detect an effect of a given size with a given degree of confidence.
RIMA WEA	Rhode Island and Massachusetts Wind Energy Area (see Figure 1.2)
SA	Study Area (see Figure 1.2)
VPR	. . . (Section 2.4)
WEA	Wind Energy Area