Technical Report

Assessment of Impacts to Marine Mammals, Sea Turtles, and ESA-Listed Fish Species Revolution Wind Offshore Wind Farm

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Acronyms

AEP	auditory evoked potential
AMAPPS	Atlantic Marine Assessment Program for Protected Species
ASMFC	Atlantic States Marine Fisheries Commission
ASSRT	Atlantic Sturgeon Status Review Team
BIWF	Block Island Wind Farm
BOEM	Bureau of Ocean Energy Management
CETAP	Cetacean and Turtle Assessment Program
CFR	Code of Federal Regulation
COP	Construction and Operations Plan
CPA	closest point of approach
dB	decibel
DP	dynamic positioning
DPS	distinct population segment
EA	environmental assessment
EEZ	Exclusive Economic Zone
EIS	Environmental Impact Statement
$ER_{95\%}$	95^{th} percentile exposure-based ranges
ESA	Endangered Species Act
EZ	exclusion zone
FHWG	Fisheries Hydroacoustic Working Group
HF	high-frequency
HRG	high-resolution geophysical
IAC	inter-array cable
IPF	impact producing factor
ISO	International Organization for Standardization
IUCN	International Union for Conservation of Nature
JASCO	JASCO Applied Sciences
Lease Area	Lease Area OCS-A 0486
LF	low-frequency
μPa	micropascal
MEC	munitions and explosives of concern
MF	mid-frequency
MMPA	Marine Mammal Protection Act
MMS	Minerals Management Service
MZ	monitoring zone
NEPA	National Environmental Policy Act
NMFS	National Marine Fisheries Service
NMS	noise mitigation system

NOAA	National Oceanic and Atmospheric Administration
NRC	National Research Council
NYSDEC	New York State Department of Environmental Conservation
NYSERDA	New York State Energy Research and Development Authority
O&M	operation and maintenance
OnSS	onshore substation
OCS	Outer Continental Shelf
Orsted NA	Orsted North America Inc.
OSRP	Oil Spill Response Plan
OSS	offshore substation
OW	otariid pinnipeds in water
PAM	passive acoustic monitoring
PBR	potential biological removal
PCW	phocid carnivores in water
PK	zero-to-peak sound pressure level
PPW	phocid pinnipeds in water
Project	Revolution Wind Farm Project
PSMMP	Protected Species Mitigation and Monitoring Plan
PSO	protected species observer
PTS	permanent threshold shift
re	referenced to
Revolution Wind	Revolution Wind, LLC
RI-MA WEA	Rhode Island-Massachusetts Wind Energy Area
RWEC	Revolution Wind Farm Export Cable
RWEC - OCS	Revolution Wind Farm Export Cable on the Outer Continental Shelf
RWEC - RI	Revolution Wind Farm Export Cable in Rhode Island state waters
RWF	Revolution Wind Farm
SAR	stock assessment report
SEL	sound exposure level
SEL_{24h}	sound exposure level over 24 hours
SL	source level
SMA	Seasonal Management Area
SPL	root-mean-square sound pressure level
TEWG	Turtle Expert Working Group
TTS	temporary threshold shift
UME	Unusual Mortality Event
U.S.	United States
USFWS	U.S. Fish and Wildlife Service
UXO	unexploded ordnance
VHF	very high-frequency
VLF	very low-frequency
WTG	wind turbine generator

1.0 INTRODUCTION

Revolution Wind, LLC (Revolution Wind) (formerly DWW REV I, LLC), a 50/50 joint venture between Orsted North America Inc. (Orsted NA) and Eversource Investment LLC, proposes to construct, own, and operate the Revolution Wind Farm Project (hereinafter referred to as the Project). The wind farm portion of the Project will be located in federal waters on the Outer Continental Shelf (OCS) in the designated Bureau of Ocean Energy Management (BOEM) Renewable Energy Lease Area OCS-A 0486 (Lease Area). The Lease Area is approximately 30 km south of the coast of Rhode Island (**Figure 1.1-1** in **Section 1.1** of the Project's Construction and Operations Plan [COP]). Other components of the Project will be located in state waters of Rhode Island and onshore in North Kingstown, Rhode Island. The proposed interconnection location for the Project is the existing Davisville Substation, which is owned and operated by National Grid and located in North Kingstown, Rhode Island.

The Project will specifically include the following offshore and onshore components:

- Up to 100 wind turbine generators (WTGs) connected by a network of Inter-Array Cables (IACs);
- Up to two offshore substations (OSSs) connected by an OSS-Link Cable;
- Up to two submarine export cables (referred to as the Revolution Wind Export Cable [RWEC]), generally co-located within a single corridor;
- A landfall location located at Quonset Point in North Kingstown, Rhode Island (referred to as the Landfall Work Area);
- Up to two underground transmission circuits (referred to as the Onshore Transmission Cable), co-located within a single corridor; and
- A new Onshore Substation (OnSS) located adjacent to the existing Davisville Substation with up to two interconnection circuits (overhead or underground) connecting the OnSS to the existing substation.

The Project's components are further grouped into four general categories: the Revolution Wind Farm (RWF), inclusive of the WTGs, OSSs, IAC, and OSS-Link Cable; the RWEC – OCS inclusive of up to 40 km of the RWEC in federal waters; the RWEC – RI, inclusive of up to 37 km of the RWEC in state waters; and Onshore Facilities, inclusive of an up to 100-m segment of the RWEC, Landfall Work Area, Onshore Transmission Cable, and OnSS (including interconnection circuits). These categories collectively are referred to in this report as the Project Area.

1.1 Contents of Technical Report

This Technical Report is intended to provide the reader with a substantial overview of the baseline conditions in the Project Area as they pertain to marine mammals, sea turtles, and fish species listed under the Endangered Species Act (ESA). The Technical Report is designed to provide supplemental information for the Project-related impact producing factors (IPFs) discussed in **Sections 4.3.3.2**, **4.3.4.2**, and **4.3.5.2** of the Project's COP that have the potential to result in greater than negligible impacts on marine mammals, sea turtles, or Atlantic sturgeon. For the purposes of this report, negligible impacts are defined as those that, if perceptible, would not result in measurable impacts on the potentially affected resources. IPFs which may result in greater than negligible impacts were determined to be habitat alteration, underwater noise, and vessel traffic. (see **Table 1.2-1**). The underwater noise IPF is treated in more detail in this report because the affected resources are known to be vulnerable to potential impacts from underwater noise. The assessment of underwater noise impacts resulting from the construction for the Project are largely based on the underwater acoustic analysis conducted by JASCO Applied Sciences (JASCO) (Denes et al., 2020). Impact assessments for underwater noise produced during operations and maintenance (O&M), and decommissioning are based on literature and assessment of similar activities. A summary of the proposed environmental protection measures, which will be implemented during Project activities to reduce the potential for impacts, is also provided in **Section 5.5**.

1.2 Regulatory Context and Resource Definition

The Project's COP provides the basis for assessed environmental and socioeconomic effects resulting from the Proposed Activities (**Section 3.0** of the COP) during construction, O&M, and decommissioning of the Project. It is prepared in accordance with 30 Code of Federal Regulation (CFR) Part 585, BOEM's Guidelines for Information Requirements for a Renewable Energy Construction and Operations Plan (BOEM, 2016), and other BOEM policy, guidance, and regulations (**Section 1.1** of the COP). The underwater acoustic propagation and animal exposure modeling results presented in the Underwater Acoustic Analysis report (Denes et al., 2020), in combination with the assessment provided in this Technical Report, are intended to provide BOEM with the necessary information to evaluate their permitted actions under the National Environmental Policy Act (NEPA) and Marine Mammal Protection Act (MMPA). As discussed in **Section 1.4** of the Project's COP, NEPA requires that Federal actions undertake an environmental assessment (EA) to produce an Environmental Impact Statement (EIS) to determine impacts to resources.

The resources of interest in this Technical Report include marine mammals, sea turtles, and ESA-listed fish species. All marine mammals are protected under the MMPA; some species are also listed as Endangered under the ESA (**Section 2.2.1**). Sea turtle and fish species included in this assessment are listed as either Endangered or Threatened under the ESA (**Section 2.2.2** and **Section 2.2.3**). The National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (USFWS) share regulatory responsibility for these species under the MMPA and ESA. The MMPA requires any Project Activities that may produce noise be assessed for the potential "take" of marine mammals, as defined in the MMPA, and provided to NMFS for approval. ESA species will also be assessed under Section 7 inter-agency consultations between BOEM and NMFS for all activities that have the potential to affect listed species. The information presented in both the Project's COP and this Technical Report will provide the basis for these MMPA and ESA consultations.

1.3 Significance Threshold

Resources may be vulnerable to one or more IPF. Each IPF that has the potential to impact marine mammals, sea turtles, or Atlantic sturgeon were assessed in **Sections 4.3.3.2**, **4.3.4.2**, and **4.3.5.2** of the Project's COP. In the analysis for the Technical Report, IPFs associated with each resource were first categorized as: 1) having greater than negligible impacts (i.e., measurable, either negative or beneficial) and require analysis; 2) having negligible impacts to a resource (i.e., an impact that if perceptible, is not measurable); or 3) no expected impacts on the resource (i.e., no perceptible impact to a resource is evident). Those IPFs assessed in the COP which had the potential to result in greater than negligible impacts to the resources are further discussed in this Technical Report (**Table 1.2-1**). Supplementary information regarding the affected resources and potential impacts is provided to further support the impact assessment provided in the COP. There are multiple sources of noise during all phases of RWF development; however, not all sources have equivalent impact potential on a given resource. Therefore, each source is discussed separately in the Technical Report to allow the reader an understanding of the underwater noise components that contribute to the overall impact determination for the underwater noise IPF.

Table 1.2-1. Summary of impact producing factors (IPFs) included in the Technical Report for marine mammals, sea turtles, and fish during construction, operation, or decommissioning of the Revolution Wind Farm and Revolution Wind Farm Export Cable.

Resource	IPF							
	Habitat Alteration	Underwater Noise						Vessel Traffic
		DP Vessel Noise	Impact Pile Driving Noise	Vibratory Pile Driving Noise	Geophysical Survey Noise	WTG Operational Noise	Aircraft Noise	
Marine Mammals	+/++	+	+	-	-	+	-	+
Sea Turtles	+/++	-	+	-	-	-	-	+
Atlantic Sturgeon	-	-	+	-	-	-	-	+

+ indicates a greater than negligible impact; ++ indicates a potential beneficial impact; - indicates negligible or no impact expected; DP = dynamic positioning; WTG = wind turbine generator.

Broad significance criteria were developed for the three resources addressed in this Technical Report. In order to assess the potential impacts, the IPFs were characterized as either direct or indirect, and short-term or long-term (as defined in **Section 4.0** of the Project's COP) primarily using the following four parameters:

- Detectability (i.e., measurable or detectable impact);
- Duration (i.e., short-term, long-term);
- Spatial extent (i.e., localized, extensive); and
- Severity (i.e., severe, less than severe).

Elements such as distribution, range, life history, sensitivity to the IPF, and potential outcomes of the impact were considered for each resource. The significance evaluations in **Sections 4.3.3.2**, **4.3.4.2**, and **4.3.5.2** of the Project's COP considered the potentially affected environment and the degree of the impact following NEPA regulations (40 CFR § 1501.3). The potentially affected area for a particular IPF considers the extent (i.e., national, regional, or local) of the effect and any special circumstances affecting resources within this area (e.g., ESA-listings or designated habitat). The degree of an impact considers the severity of the effect based on whether impacts are short-term or long-term, beneficial or adverse. The evaluation process also assessed the risk or likelihood (i.e., likely, not likely) of an effect to occur based on species' expected presence and perception of an IPF by the resource.

During the preparation of the impact assessment, each impact determination was accompanied by a statement or statements explaining how the impact determination was reached. The determinations were based on the best available information. Data or information from referenced journals used to support each determination were cited, as applicable, and professional judgement by experienced subject matter experts and impact analysts was considered in each evaluation. The impact assessment in **Section 5.0** of this Technical Report provides additional information intended to justify the assessment in **Sections 4.3.3.2**, **4.3.4.2**, and **4.3.5.2** of the Project's COP, with a focus on the duration of impacts (i.e., short-term, long-term) and identifying if impacts were direct or indirect, as defined in **Section 4.0** of the Project's COP. The impact determination process was designed to assess impacts at a population-scale rather than an individual-scale. Potential impacts to species listed as Endangered or Threatened under the ESA and marine mammal stocks listed as strategic by NMFS were given greater "weight" than impacts to non-listed species or non-strategic marine mammal stocks.

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2.0 UNDERWATER NOISE AS AN IPF

This document follows International Organization for Standardization (ISO) 18405:2017 (ISO, 2017) for all acoustic terminology. Acoustic terminology used in this report are provided in **Table 2.0-1**.

Table 2.0-1. Acoustic terminology used in this report based on International Organization for Standardization 18405 (ISO, 2017).

Metric Name	Abbreviation	Units
Root-mean-square sound pressure level	SPL	dB re 1 µPa
Zero-to-peak sound pressure level	PK	dB re 1 µPa
Sound exposure level	SEL	dB re 1 µPa^2 s
Sound exposure level over 24 hours	SEL_{24h}	dB re 1 µPa^2 s
Source level	SL	dB re 1 µPa m

dB = decibel; µPa = micropascal; re = referenced to.

Underwater noise generated by construction, operations, and decommissioning of an offshore wind farm can be assessed in the framework of impacts that may have physical or behavioral consequences for the animal exposed to the noise; or impacts that result in changes to the acoustic habitats (**Section 2.2**) from the introduction of man-made noise sources into the marine environment. Noise generated by human activities may be introduced into the environment for a specific purpose (e.g., navigational sonar, seismic exploration), or as an indirect by-product of activities such as shipping, pile driving, and other industrial activities. The propagation characteristics of these various noise sources are determined by the local physical and environmental conditions, while the perception of the noise by an animal "receiver" will be largely dependent upon individual hearing sensitivities. Outside of physiological effects, impacts on marine species from man-made noise are largely influenced by the context within which the noise is perceived by the animal.

2.1 Sources of Noise in the Project Area

Noise contributing to the acoustic habitat of the Rhode Island-Massachusetts Wind Energy Area (RI-MA WEA) is produced by both natural processes and offshore human activities within this region. Ambient noise sources can typically be divided into three general categories: physical, biological, and anthropogenic.

Physical Noise

The dominant cause of naturally occurring noise in the ocean resulting from physical processes occurs at or near the ocean surface in the form of wind and wave activity. As shown in **Figure 2.1-1**, noise produced by wind and waves are generally correlated with one another and fall within the 100 Hz to 100 kHz frequency band. Ambient noise levels tend to increase with increasing wind speed and wave height (Urick, 1962; Wenz, 1964; Erbe, 2011). In the frequency band between 3 and 30 MHz, "thermal noise" caused by the random motion of water molecules is the primary source contributing to ambient noise levels (Urick, 1962; Wenz, 1964; Hildebrand, 2009). Natural noise sources, especially noise from wave and tidal action, contribute to higher ambient noise levels typically found in shallower coastal environments.

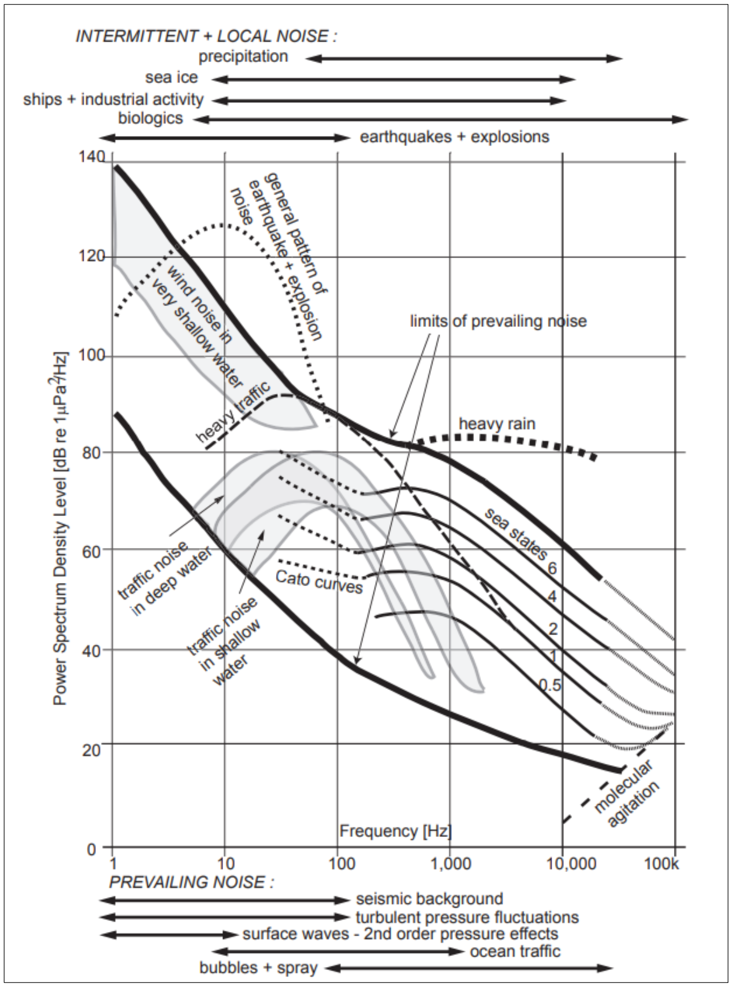

Figure 2.1-1. Wenz curve showing frequency and amplitude range of common sources of noise in the ocean. Figure from Erbe (2011) based on work from Wenz (1964).

Precipitation falling on the ocean's surface also contributes to natural noise in ocean environments. In general, noise from rain or hail is an important component of total noise at frequencies >500 Hz during periods of precipitation (**Figure 2.1-1**). Rain can increase natural ambient noise levels by up to 35 decibels (dB) across a broad range of frequencies from several hundred Hz to more than 20 kHz (National Research Council [NRC], 2003; Richardson et al., 1995). Heavy precipitation associated with large storms can generate noise at frequencies as low as 100 Hz and can significantly affect ambient noise levels at considerable distances from the storm's center (NRC, 2003). Movement of sediment by ocean currents across the ocean bottom can also be a significant source of ambient noise at frequencies from 1 kHz to over 200 kHz (NRC, 2003).

Biological Noise

Biological noise is created by marine animals and can contribute significantly to ambient noise levels in certain areas of the ocean. Marine mammals are major contributors, but noise produced by some crustaceans (e.g., snapping shrimp [Alpheidae]) and vocalizing fish can also be significant (NRC, 2003; Richardson et al., 1995).

Surveys conducted in the RI-MA WEA indicate that delphinids are the most commonly observed species in this region. Vocalizations from these mid- to high- frequency species can influence the local ambient noise conditions for short periods of time (Varga et al., 2017). Reported mid-frequency species include common bottlenose dolphins (*Tursiops truncatus*), common dolphins (*Delphinus delphis*), Risso's dolphins (*Grampus griseus*), Atlantic white-sided dolphins (*Lagenorhynchus acutus*), Atlantic spotted dolphins (*Stenella frontalis*), and long-finned pilot whales (*Globicephala melas*) (BOEM, 2013; Kraus et al., 2016). These species were observed during all seasons, with the highest number of recorded sightings in summer and fall. Harbor porpoises (*Phocoena phocoena*), the only high frequency species likely to occur in the Project Area, were also observed in this region, primarily in winter and spring (Kraus et al., 2016).

Acoustic detections of large whale species indicated that fin whales (*Balaenoptera physalus*) were the most commonly detected cetacean species in the RI-MA WEA, but humpback (*Megaptera novaeangliae*), minke (*Balaenoptera acutorostrata*), blue (*Balaenoptera musculus*), and North Atlantic right whale (*Eubalaena glacialis*) calls were also detected (BOEM, 2013; Kraus et al., 2016). Large whale vocalizations were primarily detected in the winter and spring, but fin and humpback whales were detected in all seasons, and minke whales showed a peak acoustic presence in May (BOEM, 2013; Kraus et al., 2016). Although there were no confirmed acoustic detections during the recording period, visual surveys indicated that sei whales (*Balaenoptera borealis*) were also present in the spring and summer, and sperm whales (*Physeter macrocephalus*) in the summer and autumn (Kraus et al., 2016). Baleen whale vocalizations have a marked effect on long term spectral average data with increases of up to 15 dB above ambient noise levels attributed to seasonal congregations of whales (Haver et al., 2018).

Fish vocalizations were also a substantial source of biological noise observed in this region. Series of buzzes, grunts, and thumps from unidentified fish species were heard primarily between December and February (Martin et al., 2014). The only identifiable fish call was detected between June and August, described as a jack-hammer sound, that was thought to correspond to striped cusk eel (*Ophidion marginatum*) vocalizations (Martin et al., 2014).

<u>Anthropogenic noise</u>

Vessels are a primary source of anthropogenic noise and contribute to ambient ocean noise, predominantly in low-frequency (LF) bands under 500 Hz (Hildebrand, 2009; NRC, 2003). A large portion of the noise from vessels comes from engine noise and propeller cavitation (Richardson et al., 1995). In the open water, vessel noise can influence ambient noise levels at distances of thousands of kilometers; however, the effects of vessel noise in shallower shelf and coastal waters are more variable due to physical and geological properties of the seabed, sea surface, and water column which influence reflection, refraction, and absorption and thus propagation, of noise in the water.

Underwater noise sources associated with Project Activities include impact and vibratory pile driving, geophysical surveys, and Project-related aircraft operations during the construction phase; vessels with and without dynamic positioning (DP) thrusters used during all Project phases; and WTG operations during the O&M phase. The potential for impacts on marine species from noise produced by these activities is highly dependent on the equipment scenarios and the context in which species perceive or are exposed to each noise source or activity.

The following sections provide further information about Project-related noise sources, and the corresponding acoustic characteristics and measurements based on previous assessments and published literature for all noise-producing Project Activities, and the results presented in the underwater acoustic analysis report (Denes et al., 2020) for impact pile driving activities.

2.1.1 Vessel Noise

Vessel noise is characterized as low frequency, typically <1,000 Hz with peak frequencies between 10 and 50 Hz, non-impulsive rather than impulsive like impact pile driving, and continuous, meaning there are no substantial pauses in the noise that vessels produce. The acoustic signature produced by a vessel varies based on the type of vessel (e.g., tanker, bulk carrier, tug, container ship) and vessel characteristics (e.g., engine specifications, propeller dimensions and number, length, draft, hull shape, gross tonnage, speed). Large shipping vessels and tankers produce lower frequency noise with a primary energy near 40 Hz and underwater source levels (SLs) for these commercial vessels can range from 177 to 188 dB referenced to (re) 1 micropascal (μPa) m (McKenna et al., 2012). Smaller vessels typically produce higher frequency noise (1,000 to 5,000 Hz) at SLs between 150 and 180 dB re 1 μPa m (Kipple and Gabriele, 2003, 2004). Vessels using DP thrusters are known to generate substantial underwater noise with SLs ranging from 150 to 180 dB re 1 μPa m depending on operations and thruster use (BOEM, 2013; McPherson et al., 2016). While vessel noise was not modeled for this Project, qualitative information about vessel noise which may be produced during Project activities is provided in the underwater acoustic analysis report (Denes et al., 2020).

2.1.2 Aircraft Noise

As discussed in **Section 4.1.4.1** of the Project's COP, helicopters will be used during construction and O&M activities to support crew transfers. Noise produced in air can be transmitted into the water column. Noise from a Bell 212 helicopter measured from a hydrophone deployed at 18 m depth showed frequencies ranged up to 340 Hz with received root-mean-square sound pressure levels (SPL) in the 10 to 500 Hz frequency band of approximately 106 dB re 1 μPa (Patenaude et al., 2002). Received SPL were generally higher at 3 m depth than 18 m depth by an average of 2.5 dB and decreased further as the altitude of the helicopter increased and speed decreased (Patenaude et al., 2002).

2.1.3 Impact Pile Driving Noise

Impact pile driving produces high intensity sound pulses at levels capable of producing injury to marine animals (Halvorsen et al., 2012a,b; NMFS, 2018; Popper et al., 2014). Subsequent effects from impact pile driving noise are dependent upon the physical characteristics of the environment, which influence noise propagation, receiver species, and the implementation and effectiveness of environmental protection measures (**Section 5.5**) such as noise attenuation systems. Impact pile driving noise produced from foundation installation is expected to fall predominately within LF bandwidths (below 1,000 Hz); however, Bailey et al. (2010) measured broadband noise within 1 km of impact pile driving in the Moray Firth off the coast of Ireland.

Noise produced during impact pile driving is a primary concern with respect to underwater noise impacts from RWF construction. Revolution Wind will use hydraulic (impact) hammers to install monopile foundations for the WTGs and either jacket or monopile foundations for the OSSs.

Environmental and seabed conditions, hammer type, and the size and type of pile will affect noise propagation and the estimated ranges to regulatory criteria. Due to the complexity of noise propagation generated from impact pile driving activities, modeled distances to acoustic thresholds often differ from field-measured distances and highlight the site-specific nature of noise propagation and impact radii during pile installation. While models and measurements from one project are not fully applicable across other similar projects, they do provide general information useful for predicting potential impacts during similar activities.

Modeled and *in situ* underwater noise measurements for jacket pile installation of the Block Island Wind Farm showed variability by distance and sample methods (Amaral et al., 2018). Similarly, Patricio et al. (2014) measured noise produced during impact pile driving for the Westernmost Rough Wind farm and compared modeled results to field measurements. The study found that modeled distances to injury criteria thresholds ranged from 15 to 300 m from the pile, while distances based on field measurements ranged from 200 to 1,500 m from the pile for cetaceans. Field measurements of offshore wind pile driving in Europe were summarized by Bellmann et al. (2020) and provide some of the most relevant information regarding sound levels expected during impact pile driving at RWF. Results from the Bellmann et al. (2020) measurements showed that piles without a noise mitigation system (NMS)(e.g., bubble curtain) produced noises with frequencies predominately within 32 Hz to 2 kHz and produced measured cumulative 24-h sound exposure levels (SEL_{24h}) up to 175 dB re 1 μPa^2 s at 750 m from the pile. When a single or combined NMS was applied to monopile installation, noise reductions ranging from 3 dB to 17 dB were achieved depending on the NMS combination, with some frequency-dependent reductions of >20 dB (Bellmann et al., 2020).

To help identify the potential for impacts to marine species, site-specific acoustic propagation modeling was conducted for impact pile driving for the Project, as described by Denes et al. (2020), and results of this modeling effort, as they are applied to impact assessment in this Technical Report are summarized in **Section 4.2**.

2.1.4 Vibratory Pile Driving Noise

Vibratory pile driving produces a non-impulsive, intermittent noise with maximum sound levels lower than those generated by impact pile driving (Popper et al., 2014). Measurements from vibratory pile driving of sheet piles during construction activities for bridges and piers indicate that SPL produced by this activity can range from 130 to 170 dB re 1 μPa depending on the measured distance from the source and physical properties of the location (Buehler et al., 2015; Illingworth & Rodkin, 2017). At approximately 10 m from the source, the average SPL was approximately 155 dB re 1 μPa, while measurements taken 200 m away were closer to 140 dB re 1 μPa (Illingworth & Rodkin, 2017). SEL over 1 s measured at 10 m from the source were approximately 162 dB re 1 μPa^2 s (Buehler et al., 2015).

2.1.5 Geophysical Survey Noise

Prior to construction of the RWF and RWEC, geophysical surveys will be conducted to identify any seabed debris or munitions and explosives of concern and unexploded ordnances (MEC/UXOs) (**Section 3.3** of the Project's COP). Equipment used to conduct MEC/UXO surveys may include multi-beam echosounders, side-scan sonars, shallow penetration sub-bottom profilers, medium penetration sub-bottom profilers, and marine magnetometers or gradiometers. Equipment will be comparable to those used during high-resolution geophysical (HRG) site investigation surveys conducted in the region (CSA Ocean Sciences Inc., 2018, 2020; Feehan and Daniels, 2018). Estimated distances to SPL of 160 dB re 1 μPa resulting from HRG equipment ranged from a maximum of 141 m to less than 5 m depending on the source (CSA Ocean Sciences Inc., 2018, 2020).

As discussed in **Section 3.3.3.2** of the Project's COP, avoidance is the preferred approach for MEC/UXOs, and in any situation in which avoidance is not possible, the confirmed MEC/UXO may be removed through *in situ* disposal or physical relocation. The removal method used will depend on the location, size, and condition of the MEC/UXO, and will be made in consultation with specialists and the appropriate agencies. *In situ* disposal will be done using methods such as deflagration or cutting of the MEC/UXO and relocation will be accomplished through a "Lift and Shift" operation, both of which are expected to be low-noise methods (**Section 3.3.3.2** of the Project's COP). The risk mitigation measures in place will be used to avoid munitions and prevent the potential for underwater explosions if removal of any MEC/UXOs is warranted; therefore, only noise associated with the HRG surveys is evaluated for impact assessment.

2.1.6 Wind Turbine Generator Operational Noise

WTGs primarily produce two types of noise: aerodynamic turbine blade noise and mechanical noise (Minerals Management Service [MMS], 2007). Mechanical noise may be transmitted underwater through the turbine towers and foundations producing underwater SPL noise levels between 80 and 150 dB re 1 μPa and can increase noise in frequencies below 100 Hz by 3 to 10 dB (Bergström et al., 2014; HDR, 2019). A study by Miller and Potty (2017) measured an SPL of 100 dB re 1 μPa 50 m from a set of five GE Haliade 150-6 MW wind turbines with a peak signal frequency 72 Hz. Other studies estimated SLs of operational noise from WTGs ranging from 125 to 130 dB re 1 μPa m across all octave bands (Lindeboom et al., 2011; Tougaard et al., 2009). Maximum SPL occurred in the 25 Hz one-third octave band for a 450-kW turbine during normal operations (Lindeboom et al., 2011; Tougaard et al., 2009).

In a compilation of case studies published by BOEM in 2017 (English et al., 2017), similar noise levels were identified:

- The one-third octave SPL were measured between 90 to 115 dB re 1 μPa 110 m from a 1.5-MW turbine in Sweden (Thomsen et al., 2006). The frequency range was 20 to 1,000 Hz with peak energy levels occurring at 50, 160, and 200 Hz.
- Pangerc et al. (2016) found the main signal associated with 3.6 MW turbine operations had a mean-square power spectral density level that peaked at 126 dB re 1 μPa^2 Hz^{-1} at the 162 Hz one-third octave band, and a broadband SPL of 128 dB re 1 μPa 50 m from the source at wind speeds of 10 m/s.
- Collett and Mason (2014) found that noise from operating 6 MW turbines dropped to ambient levels at approximately 100 m from the turbine.
- Noise associated with the 6 MW turbines at the Block Island Wind Farm were below SPL of 120 dB re 1 μPa measured 50 m from the turbines, except at wind speeds exceeding 13 m/s (HDR, 2019).

While underwater noise from turbines has been measured within the hearing frequency of marine animals, impacts at the anticipated noise levels would be limited to audibility, and perhaps some degree of behavioral response or auditory masking (MMS, 2007). Behavioral responses include changes in foraging, socialization, or movement, while auditory masking could impact foraging and predator avoidance. Due to the long-expected duration of this source and the low likelihood of impacts to marine animals, turbine noise was not included in the acoustic model presented by Denes et al. (2020). However, potential impacts from this noise source using published literature are discussed in **Section 3.3**.

2.2 Acoustic Habitat within the Project Area

The term acoustic habitat is defined here as the environment within which an animal perceives and transmits acoustic cues important for foraging, reproduction, socialization, and predator avoidance. Various natural and anthropogenic activities contribute noise to the ocean, creating a complex acoustic habitat. An animal's acoustic habitat is made up of concomitant noises generated biologically (biophony), physically (geophony), or anthropogenically (anthrophony) that create regional ambient noise conditions through which discrete signals must be sent and gathered by animals adapted to living in acoustically-dominated habitats. Changes in the acoustic habitat can therefore change an animal's ability to function within its environment. Acoustic habitats are not stagnant and will vary both temporally and spatially on large and small scales. Variations in the ambient noise level as a function of frequency can change by as much as 10 to 20 dB from day-to-day based on variations in the noise sources (Richardson et al., 1995; Kraus et al., 2016). Large- and small-scale temporal fluctuations (e.g., daily, seasonal) in the acoustic habitat and species vocalization patterns may influence or directly affect temporal patterns in animal communication systems and detections of other acoustic cues.

Marine animals can perceive underwater noise over a broad range of frequencies from about 10 Hz to more than 200 kHz. Where there is an overlap in the frequencies produced by anthropogenic noise sources and core frequencies used or produced by marine life, there is the potential for noise to interfere with their biological functions. The primary acoustic habitat for any species will fall within the bounds of that species' specific vocal and hearing ranges, and it is those primary acoustic habitats that were assessed when characterizing potential impacts. While many species hearing sensitivities overlap, there is evidence that acoustic habitats may be partitioned by species to maximize access to the necessary acoustic habitat (Gottesman et al., 2020). Resource partitioning may be viewed on a frequency-band or temporal basis as well as an energy basis (Ruppé et al., 2015; Gottesman et al., 2020). Ruppé et al. (2015) documented apparent resource partitioning in the acoustic communication behavior of a community of nocturnal marine fishes, in which 17 distinctive sounds that differed in peak frequency and pulsing characteristics were recorded. Furthermore, the sounds produced by soniferous species during the day did not overlap with those produced by nocturnal species and were far less diverse, indicating that the acoustic habitat use was maximized when visual resource use was less important (Hastings and Širović, 2015).

Acoustic habitats can be represented by plotting the ratios of sound energy within selected frequency bandwidths for the habitat of interest. The acoustic habitat and changes within that habitat are demonstrated by shifts in the dominant frequency range and by increases or decreases in sound energy within selected bandwidths. Modeled soundscapes and sound maps, such as those provided in National Oceanographic and Atmospheric Administration's (NOAA's) sound data mapping products (NOAA, 2019), are generated by incorporating environmental (e.g., bathymetric, oceanographic), biological, and anthropogenic noise data then modeling the noise propagation over space and time. These models represent the basis for assessing acoustic habitats and are the baseline for a potential impact analysis to species due to the introduction of acoustic sources, such as those expected during offshore wind farm construction and operations, within that environment.

The ambient noise analysis for the RI-MA WEA was provided by Kraus et al. (2016) through the deployment of passive acoustic recorders from 2011 through 2015, and with dedicated recorders deployed specifically within the RI-MA WEA between 2013 and 2015. The acoustic data were analyzed for both ambient noise levels and biological signals. In the analyses, Kraus et al. (2016) built power spectral densities, which provided the received SPL within selected frequency bands, and the cumulative distribution, which provided the percentage of time that noise within a selected frequency band reached specific SPL. The cumulative distribution enables analysis of the acoustic habitat available within a species' specific vocal range. Kraus et al. (2016) used a frequency band of 20 to 447 Hz to capture the acoustic habitat of LF cetaceans. By correlating the ambient SPL within this band with the average SPL of the LF cetacean calls, some predictions can be made regarding acoustic habitat availability and potential masking.

As shown in **Figure 2.2-1**, Kraus et al. (2016) found that the power spectrum levels above 200 Hz did not differ greatly among the nine recording sites; however, sites that were closest to shipping lanes showed an increase in power spectrum levels for spectral content below 100 Hz. The site labeled RI-3, centrally located within the Project Lease Area, had one of the lowest overall ambient noise levels with an increase around the 20 Hz frequency band, which was attributed to persistent fin whale vocal pulses. For frequencies between 70.8 and 224 Hz, the RI-3 site recorded SPL of 95 dB re 1 μPa or less for 40% of the recoding time, and SPL of 104 dB re 1 μPa or greater for only 10% of the recording time.

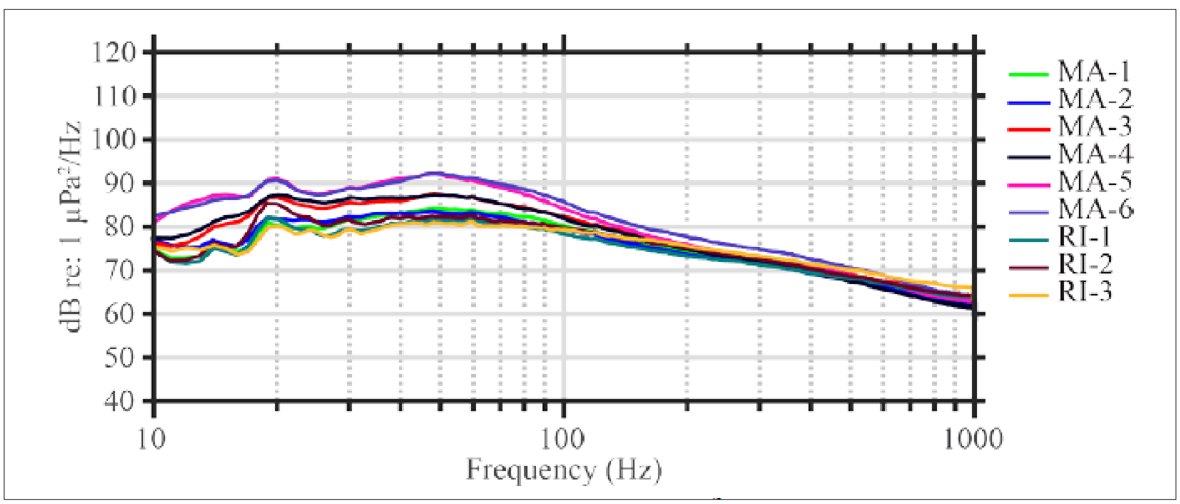

Figure 2.2-1. Power spectral density plot showing the 50th percentile power spectrum levels for each recording site within the Rhode Island-Massachusetts Wind Energy Area between November 2011 and March 2015. The yellow line labeled RI-3 represents the hydrophone located centrally within the Project Lease Area. From: Kraus et al. (2016).

Amaral et al. (2018) collected ambient noise measurements during quiet periods of impact pile driving activities for the Block Island Wind Farm (BIWF) offshore Rhode Island. Results show SPL range from 107.4 dB re 1 μPa 30 km east of the BIWF site to 118.7 dB re 1 μPa within 1 km of the site (Amaral et al., 2018). Power spectral density plots (**Figure 2.2-2**) showed higher noise levels in frequencies between 30 and 300 Hz attributed to vessel and equipment noise from BIWF construction activities (Amaral et al., 2018).

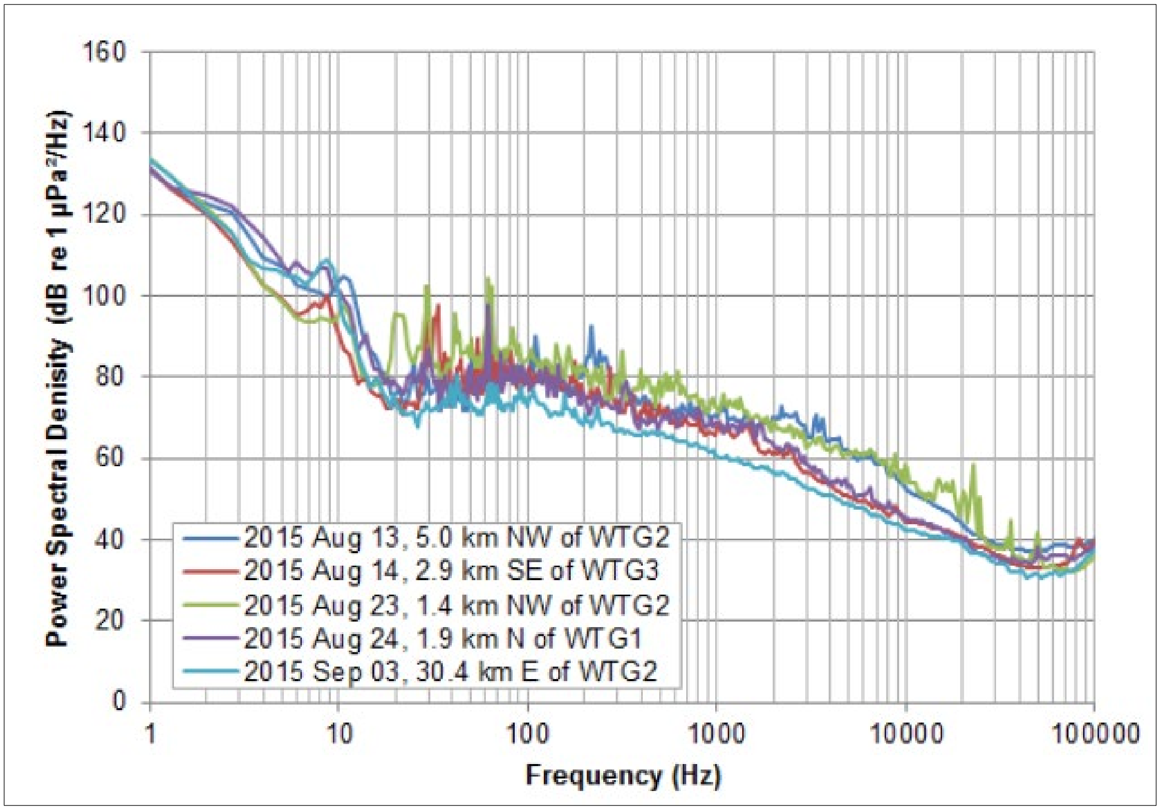

Figure 2.2-2. Power spectral density plot of ambient noise measurements collected within the vicinity of the Block Island Wind Farm. From Amaral et al. (2018).

2.3 Potential Impacts from Underwater Noise

Two primary components of underwater noise important for impact assessment include pressure and particle motion. Pressure can be characterized as the compression and rarefaction of the water as the noise wave propagates through it. Particle motion is the displacement, or back and forth motion, of the water molecules that creates the compression and rarefaction. Both factors contribute to the potential for impacts to affected resources from underwater noise. However, marine mammal and sea turtle hearing is based on the detection of sound pressure, and there is no evidence to suggest either group is able to detect particle motion for the purposes of hearing and noise detection (Bartol and Bartol, 2012; Nedelec et al., 2016). All discussions of particle motion are therefore focused on fish and invertebrate species.

All fishes can detect and use particle motion (Popper and Hawkins, 2019). The organ located in the inner ear of fishes contains a dense structure called the otolith (i.e., ear stone), which lies near the auditory sensory macula (i.e., layer of sensory hair cells). The otolith organ acts as an accelerometer and enables detection of particle motion. Particularly fish with primitive swim bladders that are not involved in hearing, like Atlantic sturgeon, particle motion is thought to play a key role in detection of underwater noise (Hawkins and Chapman, 2020). However, measurements of sensitivity to particle motion and pressure were rarely performed simultaneously, leaving a data gap in the understanding of particle motion sensitivity in fish

(Popper and Hawkins, 2018). Additionally, particle motion levels associated with a high intensity noise sources are often difficult to measure and isolate from sound pressure levels (Popper and Hawkins, 2018). There is currently very limited understanding of the potential effects of particle motion on fish and invertebrates, and it is expected that particle motion associated with impulsive noise sources, such as impact pile driving, will have similar effects to pressure waves in fish species.

Currently, there are no accepted thresholds for particle motion for any noise-producing Project Activities from which the potential for impact may be assessed. Therefore, information available on particle motion detection in fish and invertebrate species is provided in the following subsections for reference, but the impact assessment in **Section 5.0** of this report focuses on the pressure component of underwater noise.

Underwater noise is the primary IPF expected to result from construction of the RWF and RWEC. Acoustic impacts can be generalized for marine mammals, sea turtles, and Atlantic sturgeon based on the type of source (i.e., impulsive versus non-impulsive). The general impacts of hearing threshold shifts, acoustic injury (i.e., barotrauma), auditory masking, stress and behavioral responses, and reduction in prey availability are discussed in the sections below. While most available references focus on impacts on marine mammal species, the general impact categories also apply to sea turtles and Atlantic sturgeon.

2.3.1 Hearing Threshold Shifts

The minimum sound level an animal can hear at a specific frequency is called a hearing threshold. Sound levels above a hearing threshold are accommodated until a certain level of noise intensity or duration is reached, after which the ear's hearing sensitivity decreases (i.e., the hearing threshold increases) (Southall et al., 2007). This process is referred to as a threshold shift, meaning that only noises louder than a certain level will be heard within a given frequency range following the shift. Threshold shifts can be temporary (TTS) or permanent (PTS) and are defined as follows (Au and Hastings, 2008; NMFS, 2018; Southall et al., 2007):

- TTS – also known as auditory fatigue, is the milder form of hearing impairment, or threshold shift, that is non-permanent and reversible. It results from exposure to high intensity noises for short durations or lower intensity noises for longer durations. Both conditions are species-specific, and lead to an elevation in the hearing threshold, meaning it is more difficult for an animal to hear noises. TTS can last for minutes, hours, or days; the magnitude of the TTS depends on the level (frequency and intensity), energy distribution, and duration of the noise exposure, among other considerations.
- PTS – is a permanent elevation in hearing threshold (i.e., permanent loss of hearing), which is considered an auditory injury. PTS is attributed to exposure to very high peak sound pressure levels (PK) and rapid increases in intensity, or very prolonged or repeated exposures to noise strong enough to elicit TTS. Permanent damage to the inner ear such as irreparable damage to sensory hair cells in the cochlea is associated with noise-induced PTS. Because few direct data are currently available regarding noise levels that might induce PTS in marine mammals, sea turtles, and fish, PTS onset thresholds are inferred from TTS onset data (NMFS, 2018; Popper et al., 2014). For impulsive sources, dual metric criteria, PK and cumulative 24-h sound exposure level (SEL_{24h}), are often used to define PTS onsets, as well as the incorporation of applicable frequency weighting functions (e.g., M-weighting for marine mammals) to account for the differential hearing abilities in the different functional hearing groups or species (NMFS, 2018; Popper et al., 2014).

Auditory impairment, either temporary or permanent, is a possibility when animals are exposed to underwater noise. The minimum PK or SEL_{24h} necessary to reach the onset of PTS is higher than the level that indicates onset of TTS, although data are insufficient to determine the precise difference. Data indicate that TTS onset in animals is more closely correlated with the received SEL_{24h} than with the PK and that received sound energy over time, not just the single strongest pulse, should be considered a primary measure of potential impact (NMFS, 2018; Southall et al., 2007).

2.3.2 Barotrauma

Acoustic injury can occur in marine mammals, sea turtles, and fish exposed to rapid pressure changes that can theoretically be realized within close proximity to an impulsive noise source such as impact pile driving. However, barotrauma is typically only associated with explosives when considering impacts to marine mammals and sea turtles; therefore, it is only discussed within the context of impacts on fish for this Technical Report as they are the only species that could potentially be within the proximity of impact pile driving to receive the pressure changes necessary to induce barotrauma during Project construction.

Acoustic injury to fish from exposure to impulsive noise would likely be associated with barotrauma (Carlson, 2012; Halvorsen et al., 2012a, b). Barotrauma results from rapid and instantaneous changes in the ambient pressure level in the water as well as within the fluids and tissue of the animal, causing physical injury to soft tissue and organs. Barotrauma injuries in fish involve the swim bladder or dissolved gases in the blood and tissues. It can cause ruptured capillaries and internal hemorrhaging to the organs, fins, or eyes, hematoma, and a deflated or ruptured swim bladder. Depending on the affected tissues or organs, the resulting injuries may be mild (e.g., external fin hematoma; deflated, but not ruptured swim bladder), moderate (e.g., renal, intestinal, muscular hematoma), or lethal (e.g., pericardial or cerebral hemorrhage, gill embolism, ruptured swim bladder) (Brown et al., 2012; Christian, 1973; Gaspin, 1975; Goertner, 1978; Rummer and Bennett, 2005; Yelverton et al., 1975).

Some fishes, such as sturgeon and salmonids, can voluntarily release the gas from their swim bladder. The ability to rapidly vent swim bladder gas means that when the swim bladder is under pressure during an acoustic event, these fishes can decrease the volume of swim bladder gas, thereby partially protecting themselves from barotrauma injuries (Brown et al., 2016).

A controlled exposure laboratory study by Halvorsen et al. (2012a) exposed several fish species to an underwater SEL_{24h} ranging from 204 to 216 dB re 1 μPa^2 s. At SEL_{24h} >210 dB re 1 μPa^2 s, lake sturgeon (*Acipenser fulvescens*), whose swim bladder is not involved in hearing like Atlantic sturgeon, experienced recoverable barotrauma injuries characterized by hematomas on the swim bladder, kidney, and intestine, and a partially deflated swim bladder, but showed no external or mortal injuries. Conversely, Nile tilapia (*Oreochromis niloticus*) have a swim bladder that is involved in hearing, and they were shown to be more vulnerable to barotrauma at a relatively lower SEL_{24h}. They exhibited recoverable injuries including gonadal and swim bladder hematoma at 207 to 210 dB re 1 μPa^2 s, and lethal injuries such as a ruptured swim bladder and renal hemorrhage at 213 to 216 dB re 1 μPa^2 s. By contrast, no internal or external barotrauma injuries were observed at any of the SEL_{24h} for hogchoker (*Trinectes maculatus*), a flatfish that lacks a swim bladder (Halvorsen et al., 2012a). Although this study was conducted in a controlled laboratory setting, it replicated acoustic conditions in the field.

Barotrauma injuries may be more extensive in fish exposed to fewer hammer blows at higher energy versus a greater number of hammer blows at lower energy, even when the SEL_{24h} are equivalent. In a study by Halvorsen et al. (2012b), juvenile Chinook salmon (*Oncorhynchus tshawytscha*) were exposed to underwater SEL_{24h} ranging from 204 to 220 dB re 1 μPa^2 s and PK ranging from 199 to 213 dB re 1 μPa. The fish exposed to SEL_{24h} between 213 and 220 dB re 1 μPa^2 s and PK between 210 and 213 dB re 1 μPa exhibited a greater number of barotrauma injuries, specifically those that were classified as moderate or having the potential to cause lethal effects.

Overall, it is more likely that fish will experience sub-lethal impacts that increase the possibility for delayed mortality (Hawkins et al., 2014). Because the majority of Project construction sources produce LF noise that is within the sensitive hearing range of most fish, and most of the sources are non-impulsive, the potential for fish to experience TTS, masking, and behavioral impacts is higher than acoustic injury or mortality.

2.3.3 Auditory Masking

In addition to affecting hearing through physical injury, noise can partially or completely reduce an individual's ability to effectively transmit and receive acoustic signals important for detecting predator, prey, conspecific signals, and environmental features associated with spatial orientation (Clark et al., 2009). This phenomenon is defined as auditory masking, where a reduction in the detectability of a sound signal of interest (e.g., communication calls, echolocation) occurs due to the presence of another sound, which is usually part of ambient noise in the environment, that often occurs for sounds with similar frequency ranges. Under normal circumstances, in the absence of high ambient noise levels, an animal would hear a sound signal if it is above its absolute hearing threshold. Auditory masking prevents part or all of a sound signal from being heard and decreases the distances over which sounds can be detected by an animal (i.e., reduction in communication space). These effects could cause a long-term decrease in an animal's efficiency at foraging, navigating, or communicating (International Council for the Exploration of the Sea [ICES], 2005). For some marine mammal species, specifically common bottlenose dolphins, beluga whales (*Delphinapterus leucas*), and killer whales (*Orcinus orca*), empirical evidence confirms that the degree of masking depends strongly on the relative directions at which noise arrives and the characteristics of the masking noise (Bain et al., 1993; Bain and Dahlheim, 1994; Dubrovskiy, 1990; Penner et al., 1986).

Ambient noise from natural and anthropogenic sources can result in masking for marine animals, effectively interfering with the ability of an animal to detect a sound signal that it otherwise would hear. Spectral, temporal, and spatial overlap between the masking sound and the signal of interest determines the extent of interference, the greater the spectral and temporal overlap, the greater the potential for masking. As discussed in **Section 2.1**, naturally occurring ambient noise is produced by various sources, including environmental noise from wind, waves, and precipitation; thermal noise resulting from molecular agitation (at frequencies above 30 kHz); and biological noise produced by animals (Richardson et al., 1995). Biological sounds are commonly produced by fish, for example, which create LF sounds (50 to 2,000 Hz, most often from 100 to 500 Hz) that can be a significant component of local acoustic habitats (Martin et al., 2014; Zelick et al., 1999). Anthropogenic sources known to contribute to ambient noise levels can include vessels, sonar (military and commercial), geophysical surveys, acoustic deterrent devices, construction noise, and scientific research sensors. Ambient noise is highly variable in the shallower waters over continental shelves where many anthropogenic activities occur, effectively enabling anthropogenic noise to cover a wide range of sound levels and frequencies in these habitats (Desharnais and Hazen, 1999).

In coastal waters, noise from boats and ships, particularly commercial vessels, is the predominant source of anthropogenic noise (Parks et al., 2011). Over the past 50 years, commercial shipping, the largest contributor of anthropogenic noise (McDonald et al., 2008), has increased the ambient noise levels in the deep ocean at LFs by 10 to 15 dB re 1 µPa (Hatch and Wright, 2007). This increase in LF ambient noise coincides with a significant increase in the number and size of vessels making up the world's commercial shipping fleet (Hildebrand, 2009). Tournadre (2014) estimated from satellite altimetry data that, globally, vessel traffic grew by approximately 60% from 1992 to 2002 at a nearly constant rate of approximately 6% per year; however, after 2002, the rate of increase in vessel traffic rose steadily to more than 10% by 2011, except in 2008 and 2009 when traffic remained steady. The highest estimated rate of growth in vessel traffic was in the Indian and western North Pacific Oceans, especially in the continental seas along China; the rate of growth in shipping in the Atlantic Ocean and Mediterranean Sea, however, decreased after 2008.

2.3.4 Stress and Behavioral Responses

Stress and behavioral changes are the result of marine animals responding to extreme or excessive disturbances in their environment, either of natural or anthropogenic origin. Stress responses can be manifested as a physiological reaction such as changes in an animal's blood chemistry while behavioral responses involve changes in an animal's normal actions.

Marine mammals have been shown to respond to environmental stress by releasing hormones into their bloodstream and measuring changes in an animal's blood chemistry can determine whether there is a stress response. Stress responses in marine mammals are immediate, acute, and characterized by the release of neurohormones such as norepinephrine, epinephrine, and dopamine (Office of Naval Research, 2009). The NRC (2003) examined acoustically induced stress in marine mammals and determined that a one-time exposure to noise was less likely to have detrimental population-level effects than repeated exposure over extended periods of time. Various researchers have summarized the available evidence regarding stress induced events in marine mammals (e.g., Cowan and Curry, 2008; Eskesen et al., 2009; Mashburn and Atkinson, 2008; Romano et al., 2004).

Romano et al. (2004) examined the levels of three stress-related blood hormones (norepinephrine, epinephrine, and dopamine) in a beluga whale after exposure to varying PK signals produced by a seismic water gun between 198 and 226 dB re 1 µPa. Hormone levels were measured after a control, low-level sound, and a high-level sound exposure. No significant differences in the hormone blood concentrations were found between the control and low-level sound exposure, but elevated levels of all three hormones were measured in response to the high-level sound exposure. Furthermore, a regression analysis demonstrated a linear trend between increased hormone levels in the blood and sound levels. They also noted that no quantitative approach to estimating changes in mortality or fecundity due to stress has been identified, but qualitative effects may include increased susceptibility to disease and early termination of pregnancy.

Following the terrorist attacks of September 11, 2001, shipping traffic dramatically decreased in the Bay of Fundy, Canada, resulting in a 6-dB decrease in ambient underwater noise levels, including a significant reduction in frequencies below 150 Hz associated with vessel traffic. Decreased baseline levels of stress-related hormone metabolites in North Atlantic right whales were also observed during this period, which was thought to be the result of reduced noise levels (Rolland et al., 2012). This reduction in ambient noise levels associated with shipping was the first evidence that exposure to LF noise from shipping may be associated with chronic stress in whales, particularly North Atlantic right whales (Rolland et al., 2012).

Anthropogenic noise in aquatic environments has also been demonstrated to elicit a stress response in fish. This response has been measured in terms of short-term (i.e., <1 h) indicators such as a startle response, increased gill ventilation, increased heart rate and blood pressure, increased plasma cortisol and glucose levels, and increased oxygen intake, as well as long-term (i.e., days to months) indicators including reduced foraging, growth and reproductive fitness, diminished immune response, and increased vulnerability to predation (Bruintjes et al., 2016a,b; Sierra-Flores et al., 2015; Simpson et al., 2016; Smith et al., 2004). Increased levels of cortisol have been reported in giant kelpfish (*Heterostichus rostratus*) in response to vessel noise, and cod (*Gadus* spp.) exposed to linear frequency sweeps of sufficient amplitude (Slabbekoorn et al., 2019). Temporary stressors such as impact pile driving and vessel noise may cause a short-term stress response in fish, but the potential for these activities to cause longer term growth and fitness consequences has not been demonstrated in a field setting. In general, fish may acclimate to long-term exposure to acoustic stressors (Schreck, 2000). Goldfish (*Carassius auratus*) exposed to long-term, continuous noise sources, such as the hum or vibration of vessel traffic at SPL of 160 to 170 dB re 1 µPa, exhibited a short-term stress response characterized by increased cortisol and glucose levels, but they did not exhibit a long-term stress response (Smith et al., 2004). Additionally, Neo et al. (2014) indicated that the temporal nature of the noise may influence the rate of recovery following behavioral disturbance. Both intermittent (e.g., pile driving) and continuous (e.g., vessel traffic, drilling) noises elicited behavioral changes in fish, but the time it took to return to normal baseline behavior was longer in response to intermittent noises compared to continuous noises (Neo et al., 2014).

Disturbances can also cause subtle to extreme changes in normal behavior, with some behavioral responses resulting in biologically significant consequences. Behavioral responses including startle, avoidance (i.e., changes in swim speed and direction), displacement, diving, and vocalization alterations have been observed in marine animals. In some cases, these have occurred at ranges of tens to hundreds of kilometers from the noise source (Gordon et al., 2004; Miller et al., 2014; Tyack, 2008). However, behavioral observations are variable, some findings are contradictory, and the biological significance of the effects are not fully quantified (Gordon et al., 2004). Behavioral reactions of animals to noise are difficult to predict because reactions depend on numerous factors, including the species being evaluated; the animal's state of maturity, prior experience with or exposure to anthropogenic noises, current activity patterns, and reproductive state; time of day; and weather state (Wartzok et al., 2004). There is also the potential for differences in observed responses among individuals of the same species (Castellote et al., 2014). If a marine mammal reacts to underwater noise by changing its behavior or moving to avoid the noise, the impacts of that change may not be important to the individual, the stock, or the population as a whole. However, if a noise source displaces animals from an important feeding or breeding area, impacts on individuals and the population could be significant.

For marine mammals, assessing the severity of behavioral effects associated with anthropogenic noise exposure presents unique challenges due to the inherent complexity of behavioral responses and the contextual factors affecting them, both within and between individuals and species. Severity of responses can vary depending on characteristics of the noise source including whether it is moving or stationary, the number and spatial distribution of noise source(s), its similarity to predator sounds, and other relevant factors (Barber et al., 2010; Bejder et al., 2009; Ellison et al., 2012; NRC, 2005; Richardson et al., 1995; Southall et al., 2007).

Many examples have been reported of individuals of the same species exposed to the same noise reacting differently (Nowacek et al., 2004), as well as different species reacting differently to the same noises (Bain and Williams, 2006). Odontocetes appear to exhibit a greater variety of reactions to anthropogenic noise than mysticetes. Odontocete reactions can vary from approaching vessels (e.g., bow riding) to strong avoidance. Richardson et al. (1995) noted that most small and medium-sized odontocetes exposed to prolonged or repeated underwater noises are unlikely to be displaced unless the overall received SPL is at least 140 dB re 1 µPa.

Limited data exist on sound levels that may induce stress or behavioral changes in sea turtles, and no data exist on population impacts from acoustic disturbance in sea turtles (Nelms et al., 2016). Lavender et al. (2011) collected behavior audiograms from sea turtles and found that loggerheads (*Caretta caretta*) may be more sensitive to behavioral disturbance from underwater noise than electrophysiological studies suggest. Avoidance responses by sea turtles to seismic signals have been observed at received SPL between 166 and 179 dB re 1 µPa (McCauley et al., 2000); however, these studies were done in a caged environment, so the extent of avoidance could not be fully monitored. During experiments using airguns to repel sea turtles from dredging operations, Moein et al. (1995) observed a habituation effect to seismic noises; the animals stopped responding to the signal after three presentations, although it was not clear whether this was a result of behavioral habituation or physical effects from TTS or PTS. The potential effects of impulsive noise on sea turtles are likely to be varied and sometimes cryptic (Nelms et al., 2016). The frequency and duration of exposure are not discussed in the available literature; however, this topic is important when determining the level of risk to sea turtles.

2.3.5 Reduction of Prey Availability

There are limited data on hearing mechanisms and potential effects of noise on prey species of marine mammals and sea turtles (i.e., crustaceans, cephalopods, fish). These species have been increasingly researched as concern has grown related to noise impacts on the food web. Invertebrates appear to be able to detect both sound pressure and particle motion (André et al., 2016; Budelmann, 1992; Solé et al., 2016, 2017) and are most sensitive to LF noises (Budelmann and Williamson, 1994; Lovell et al., 2005a,b; Mooney et al., 2010; Packard et al., 1990). Reduction of prey fish availability could affect marine mammals and sea turtles if rising sound levels affect fish populations and alter prey abundance, behavior, and distribution (McCauley et al., 2000; Popper and Hastings, 2009; Slabbekoorn et al., 2010).

Cephalopods (i.e., octopus, squid) and decapods (i.e., lobsters, shrimps, crabs) are capable of sensing both particle motion and sound pressure at lower frequencies. Packard et al. (1990) showed that three species of cephalopod (common cuttlefish [*Sepia officinalis*], common octopus [*Octopus vulgaris*], and European squid [*Loligo vulgaris*]) were sensitive to particle motion rather than sound pressure, with the highest sensitivity to particle motion reported at 1 to 2 Hz. In longfin squid (*Loligo pealeii*), Mooney et al.

(2010) also observed responses to particle motion at lower frequencies between 100 and 300 Hz and also observed responses to sound pressure at 200 Hz. These data indicate that some prey species may be responding to both the particle motion and pressure component of LF noises, but thresholds for physiological or behavioral responses to particle motion in invertebrates are not currently available.

Potential onset thresholds for both physiological and behavioral respones to the pressure component of underwater noise are available in published literature. Solé et al. (2017) showed that SPL ranging from 139 to 142 dB re 1 µPa at one-third octave bands centered at 315 Hz and 400 Hz may be suitable threshold values for trauma onset from sound pressure in cephalopods. Hearing thresholds for sound pressure at higher frequencies have been reported, such as 134 and 139 dB re 1 µPa at 1,000 Hz for the oval squid (*Sepioteuthis lessoniana*) and the common octopus, respectively (Hu et al., 2009). Cephalopods have also exhibited behavioral responses to low frequency noises (<1,000 Hz) including inking, locomotor responses, body pattern changes, and changes in respiratory rates (Kaifu et al., 2008; Hu et al., 2009). McCauley et al. (2000) reported that of caged squid exposed to seismic airguns showed behavioral responses such as inking. Wilson et al. (2007) exposed two groups of longfin squid in a tank to killer whale echolocation clicks at SPL from 199 to 226 dB re 1 µPa, which resulted in no apparent behavioral effects or any acoustic debilitation. However, both the McCauley et al. (2000) and Wilson et al. (2007) experiments used caged squid, so it is unclear how unconfined animals would react. André et al. (2011) exposed four cephalopod species (European squid, common cuttlefish, common octopus, and Southern shortfin squid [*Ilex coindetii*]) to 2 h of continuous noise from 50 to 400 Hz at received SPL of 157 dB re 1 µPa, and reported lesions occurring on the sensory hair cells of the statocyst that increased in severity with time, suggesting that cephalopods are particularly sensitive to LF noise. Similarly, Solé et al. (2013) conducted an LF (50 to 400 Hz) controlled exposure experiment on two deep-diving squid species (Southern shortfin squid and European squid), which resulted in lesions on the statocyst epithelia. Solé et al. (2013) described their findings as “morphological and ultrastructural evidence of a massive acoustic trauma induced by…low-frequency sound exposure.” In experiments conducted by Samson et al. (2014), common cuttlefish exhibited escape responses (i.e., inking, jetting) when exposed to frequencies between 80 and 300 Hz with SPL above 140 dB re 1 µPa, and they habituated to repeated 200 Hz noises. The intensity of the cuttlefish response with the amplitude and frequency of the noise stimulus suggest that cuttlefish possess loudness perception with a maximum sensitivity of approximately 150 Hz (Samson et al., 2014). Jones et al. (2020) exposed longfin inshore squid (*Doryteuthis pealeii*) to playbacks of impact pile driving recorded at the Block Island Wind Farm ranging from approximately 190 to 194 dB re 1 µPa, which were meant to match sound levels recorded 500 m from the piles. Most of the squid tested showed alarm behavior (e.g., inking, jetting, body pattern change), but the proportion of the trial in which squid exhibited these behaviors decreased substantially following the first 30 impulses of the playback, indicating the squid may become habituated to the noise (Jones et al., 2020).

Several species of aquatic decapod crustaceans are also known to produce sounds. Popper et al. (2001) reviewed behavioral, physiological, anatomical, and ecological aspects of noise and vibration detection by decapod crustaceans and noted that many decapods also have an array of hair-like receptors within and upon the body surface that potentially respond to water- or substrate-borne displacements as well as proprioceptive organs that could serve secondarily to perceive vibrations. They concluded that many are able to detect substratum vibrations at sensitivities sufficient to tell the proximity of mates, competitors, or predators (Popper et al., 2001). However, the acoustic sensory system of decapod crustaceans remains poorly studied (Popper et al., 2001). Lovell et al. (2005a,b, 2006) reported potential auditory-evoked responses from prawns (*Palaemon serratus*) that showed auditory sensitivity of noises from 100 to 3,000 Hz. Filiciotto et al. (2016) also reported behavioral responses to vessel noise within this frequency range. Lovell et al. (2005b) found that the greatest sensitivity for prawns was an SPL of 106 dB re 1 µPa at 100 Hz, noting that this was the lowest frequency at which they tested and that prawns might be more sensitive at frequencies below this.

Marine fish are typically sensitive to the 100 to 500 Hz range, which is within the range of noise produced by impact pile driving, and several studies have demonstrated that seismic airguns and impulsive sources might affect the behavior of at least some species of fish. For example, field studies by Engås et al. (1996) and Løkkeborg et al. (2012) showed that the catch rate of haddock (*Melanogrammus aeglefinus*) and Atlantic cod (*Gadus morhua*) significantly declined over 5 days immediately following seismic surveys, after which the catch rate returned to normal. Other studies found only minor responses by fish to noise created during or following seismic surveys, such as a small decline in lesser sand eel (*Ammodytes marinus*) abundance that quickly returned to pre-seismic levels (Hassel et al., 2004) or no permanent changes in the behavior of marine reef fishes (Wardle et al., 2001). However, both Hassel et al. (2004) and Wardle et al. (2001) noted that when fish sensed the airgun firing, they performed a startle response and sometimes fled.

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3.0 DESCRIPTION OF AFFECTED RESOURCES

The expected occurrence of each species in the Project Area is based on information provided in EAs conducted by BOEM offshore Rhode Island and Massachusetts (BOEM, 2013, 2014); regional surveys such as the Northeast Large Pelagic Survey, the Atlantic Marine Assessment Program for Protected Species (AMAPPS), or the Cetacean and Turtle Assessment Program (CETAP) (CETAP, 1982; Kraus et al., 2016; Palka et al., 2017); stock information from NMFS and USFWS available for the region; density and other available information from published literature. Vulnerability of each species to potential impacts is determined based on the status of the stock (i.e., ESA- or MMPA-listing) and relevant publications indicating responses from previous exposures to similar activities. Available information was applicable to both the RWF and RWEC (including both the RWEC – OCS and RWEC – RI), so assessment methods did not differ between to the two Project Components. As discussed in the Project's COP (**Sections 4.3.3.1**, **4.3.4.1**, and **4.3.5.1**), impacts associated with the Onshore Facilities are not expected to occur to affected resources, and this Project Component will not be discussed further.

3.1 Marine Mammals

There are 36 marine mammal species in the Western North Atlantic OCS Region whose ranges include the Northeastern U.S. region where the Project will be located (BOEM, 2013, 2014). The marine mammal assemblage comprises cetaceans (whales, dolphins, and porpoises), pinnipeds (seals), and sirenians (manatee).

There are 31 cetacean species, including 25 members of the suborder Odontoceti (toothed whales, dolphins, and porpoises) and 6 of the suborder Mysticeti (baleen whales) within the region.

Along with cetaceans, there are also four phocid species (true seals) that are known to occur in the region, including harbor seals (*Phoca vitulina*), gray seals (*Halichoerus grypus*), harp seals (*Pagophilus groenlandica*), and hooded seals (*Cystophora cristata*) (Hayes et al., 2020). Finally, one species of sirenian, the Florida manatee (*Trichechus manatus latirostris*), is an occasional visitor to the region during the summer months (USFWS, 2019).

The protection status, stock identification, and abundance estimates of each marine mammal species with geographic ranges that include the Northeastern U.S. region are provided in **Table 3.1-1**. Density data are also available from Roberts et al. (2018) and Roberts (2020) for this region, but are not provided at this time because these data may be updated between now and final submission of the COP used by BOEM to prepare the EIS. Density estimates for the Project Area will be provided prior to this final COP submission. **Table 3.1-1** evaluates the potential occurrence of marine mammals in the Project Area based on five categories defined as follows:

- **Common** – Occurring consistently in moderate to large numbers;
- **Regular** – Occurring in low to moderate numbers on a regular basis or seasonally;
- **Uncommon** – Occurring in low numbers or on an irregular basis;
- **Rare** – Records for some years but limited; and
- **Not expected** – Range includes the Project Area, but due to habitat preferences and distribution information, species are not expected to occur in the Project Area although records may exist for adjacent waters.

Table 3.1-1. Marine mammals with geographic ranges that include the Northeastern U.S. region and their relative occurrence in the Project Area (Bureau of Ocean Energy Management, 2013, 2014; U.S. Fish and Wildlife Service [USFWS], 2019; National Marine Fisheries Service [NMFS], 2020a).

Common Name	Scientific Name	Stock	Current Population Status	Relative Occurrence in the RWF	Relative Occurrence in the RWEC – OCS	Relative Occurrence in the RWEC – RI	Best Abundance Estimate[1]
Order Cetacea							
Suborder Mysticeti (baleen whales)							
Fin whale	*Balaenoptera physalus*	Western North Atlantic	ESA Endangered MMPA Depleted and Strategic RI State Endangered	Common	Common	Common	6,802
Sei whale	*Balaenoptera borealis*	Nova Scotia	ESA Endangered MMPA Depleted and Strategic	Regular	Uncommon	Uncommon	6,292
Blue whale	*Balaenoptera musculus*	Western North Atlantic	ESA Endangered MMPA Depleted and Strategic	Rare	Not Expected	Not Expected	402
North Atlantic right whale	*Eubalaena glacialis*	Western North Atlantic	ESA Endangered MMPA Depleted and Strategic RI State Endangered	Common	Common	Common	412
Minke whale	*Balaenoptera acutorostrata*	Canadian East Coast	MMPA Non-strategic	Common	Common	Common	21,968
Humpback whale[2]	*Megaptera novaeangliae*	Gulf of Maine	MMPA Non-strategic[2] RI State Endangered	Common	Common	Common	1,393
Suborder Odontoceti (toothed whales, dolphins, and porpoises)							
Sperm whale	*Physeter macrocephalus*	North Atlantic	ESA Endangered MMPA Depleted and Strategic	Common	Common	Regular	4,349
Pygmy sperm whale	*Kogia breviceps*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	7,750
Dwarf sperm whale	*Kogia sima*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	7,750
Northern bottlenose whale	*Hyperoodon ampullatus*	Western North Atlantic	MMPA Non-strategic	Not Expected	Not Expected	Not Expected	Unknown
Cuvier's beaked whale	*Ziphius cavirostris*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	5,744
Mesoplodont beaked whales	*Mesoplodon spp.*	Western North Atlantic	MMPA Depleted	Rare	Rare	Rare	10,107
Killer whale	*Orcinus orca*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	Unknown

Common Name	Scientific Name	Stock	Current Population Status	Relative Occurrence in the RWF	Relative Occurrence in the RWEC – OCS	Relative Occurrence in the RWEC – RI	Best Abundance Estimate[1]
False killer whale	*Pseudorca crassidens*	Western North Atlantic	MMPA Strategic	Rare	Rare	Rare	1,791
Pygmy killer whale	*Feresa attenuata*	Western North Atlantic	MMPA Non-strategic	Not Expected	Not Expected	Not Expected	Unknown
Short-finned pilot whale	*Globicephala macrorhynchus*	Western North Atlantic	MMPA Strategic	Rare	Rare	Rare	28,924
Long-finned pilot whale	*Globicephala melas*	Western North Atlantic	MMPA Strategic	Common	Uncommon	Uncommon	39,215
Melon-headed whale	*Peponocephala electra*	Western North Atlantic	MMPA Non-strategic	Not Expected	Not Expected	Not Expected	Unknown
Risso's Dolphin	*Grampus griseus*	Western North Atlantic	MMPA Non-strategic	Common	Uncommon	Uncommon	35,493
Common dolphin	*Delphinus delphis*	Western North Atlantic	MMPA Non-strategic	Common	Common	Common	172,974
Fraser's dolphin	*Lagenodelphis hosei*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	Unknown
Atlantic white-sided dolphin	*Lagenorhynchus acutus*	Western North Atlantic	MMPA Non-strategic	Common	Common	Common	93,233
White-beaked dolphin	*Lagenorhynchus albirostris*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	536,016
Pantropical spotted dolphin	*Stenella attenuata*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	6,593
Clymene dolphin	*Stenella clymene*	Western North Atlantic	MMPA Non-strategic	Not Expected	Not Expected	Not Expected	4,237
Striped dolphin	*Stenella coeruleoalba*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	67,036
Atlantic spotted dolphin	*Stenella frontalis*	Western North Atlantic	MMPA Non-strategic	Uncommon	Uncommon	Uncommon	39,921
Spinner dolphin	*Stenella longirostris*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	4,102
Rough toothed dolphin	*Steno bredanensis*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	136
Common bottlenose dolphin	*Tursiops truncatus*	Western North Atlantic, offshore	MMPA Non-strategic	Common	Common	Common	62,851
		Western North Atlantic, northern migratory coastal	MMPA Depleted and strategic	Rare	Rare	Rare	6,639
Harbor Porpoise	*Phocoena*	Gulf of Maine/Bay of Fundy	MMPA Non-strategic RI State SGCN	Common	Common	Common	95,543

Common Name	Scientific Name	Stock	Current Population Status	Relative Occurrence in the RWF	Relative Occurrence in the RWEC – OCS	Relative Occurrence in the RWEC – RI	Best Abundance Estimate[1]
Order Carnivora							
Suborder Pinnipedia							
Harbor seal	*Phoca vitulina*	Western North Atlantic	MMPA Non-strategic RI State SGCN	Regular	Regular	Regular	75,834
Gray seal	*Halichoerus grypus*	Western North Atlantic	MMPA Non-strategic	Regular	Regular	Regular	27,131
Harp seal	*Pagophilus groenlandica*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	Unknown
Hooded seal	*Cystophora cristata*	Western North Atlantic	MMPA Non-strategic	Rare	Rare	Rare	Unknown
Order Sirenia							
Florida manatee[3]	*Trichechus manatus latirostris*	-	ESA Threatened MMPA Depleted and Strategic	Rare	Rare	Rare	13,000[4]

- = not applicable; ESA = Endangered Species Act; MMPA = Marine Mammal Protection Act; Project Area = includes the Revolution Wind Farm (RWF), Revolution Wind Export Cable (RWEC) – Outer Continental Shelf (OCS) and RWEC – Rhode Island (RI) state waters, and Onshore Facilities; SGCN = Species of Greatest Conservation Need.
[1]Best abundance estimate from the Draft 2020 Marine Mammal Stock Assessment Report, published by NMFS (NMFS, 2020a).
[2]Globally there are 14 Distinct Population Segments of humpback whale, four of which are listed as Endangered under the ESA. The Gulf of Maine population which is expected to occur in the Project Area is not listed under the ESA.
[3]Under management jurisdiction of USFWS rather than NMFS and therefore not included in Draft 2020 Stock Assessment Report.
[4]Current range-wide estimate from USFWS (2019).

Of the 36 marine mammal species with geographic ranges that include the Northeastern U.S. region, 15 species can be reasonably expected to reside, traverse, or routinely visit the Project Area in densities that could result in impacts from Proposed Activities, and therefore, be considered *potentially affected species*. Species not expected or rare are not carried forward in this Technical Report. The following affected species are those that have a common, uncommon, or regular relative occurrence in the Project Area, or have a very wide distribution with limited distribution or abundance details.

- Fin whale;
- Sei whale;
- North Atlantic right whale;
- Minke whale;
- Humpback whale;
- Sperm whale;
- Long-finned pilot whale
- Atlantic spotted dolphin;
- Atlantic white-sided dolphin;
- Common dolphin;
- Risso's dolphin;
- Common bottlenose dolphin;
- Harbor porpoise;
- Harbor seal; and
- Grey seal.

The following subsections summarize data on the status and trends, distribution and habitat preferences, behavior and life history, and auditory capabilities of ESA-listed and non-listed marine mammals expected to occur in the Project Area as available in published literature and reports, including NMFS marine mammal stock assessment reports (SARs). Expected occurrence for each species within the RWF area and RWEC corridor, including both the RWEC – OCS and RWEC – RI areas, was assessed separately.

3.1.1 ESA-listed Species

Six species known to occur in the Western North Atlantic are listed under the ESA; these include the fin whale (Endangered), sei whale (Endangered), blue whale (Endangered), North Atlantic right whale (Endangered), sperm whale (Endangered), and Florida manatee (Threatened). Of these six species, only the fin whale, sei whale, North Atlantic right whale, and sperm whale are expected to occur in the Project Area and are considered potentially affected species. These species are highly migratory and do not spend extended periods of time in a localized area. The following sections provide further information regarding species behavior and expected occurrence in the RWF and two RWEC areas (RWEC – OCS and RWEC – RI).

Fin Whale

Fin whales have a wide distribution and can be found in the Atlantic and Pacific Oceans in both the Northern and Southern Hemisphere (NMFS, 2020a). The population is divided by ocean basins; however, these boundaries are arbitrary as they are based on historical whaling patterns rather than biological evidence (NMFS, 2020a). In the Northeastern U.S., fin whales are the most commonly sighted species and account for 47% of the large whale sightings in the region (CETAP, 1982). They have been observed in all four seasons, and their distribution ranges from the Mid-Atlantic coast to Nova Scotia in Western North Atlantic OCS waters (Kenney and Vigness-Raposa, 2010).

Fin whales are often confused with other balaenopterid whales (e.g., blue whale, sei whale) during field surveys, but can be distinguished by the white, v-shaped patterns on their back behind the head (Jefferson et al., 1993). Fin whales also produce characteristic vocalizations that can be distinguished during passive acoustic monitoring (PAM) surveys (BOEM, 2013; Erbe et al., 2017). The most commonly observed calls are the "20-Hz signals," a short downsweep falling from 30 to 15 Hz over a 1-sec period. Fin whales can also produce higher frequency sounds up to 310 Hz, and SLs as high as 195 dB re 1 µPa m have been reported, making it one of the most powerful biological sounds in the ocean (Erbe et al., 2017). Anatomical modeling based on fin whale ear morphology suggests their greatest hearing sensitivity is between 20 Hz and 20 kHz (Cranford and Krysl, 2015; Southall et al., 2019).

Fin whales are listed as Endangered under the ESA and by the state of Rhode Island, and are listed as Vulnerable by the International Union for Conservation of Nature (IUCN) Red List (NMFS, 2020a; Rhode Island Department of Environmental Management [RI DEM], 2020; IUCN, 2021). The best abundance estimate available for the Western North Atlantic stock is 6,802 based on data from 2016 NOAA shipboard and aerial surveys and the 2016 Canadian Northwest Atlantic International Sightings Survey (NAISS) that extended from Newfoundland to Florida (NMFS, 2020a). A population trend analysis does not currently exist for this species because of insufficient data; however, based on photographic identification, the gross annual reproduction rate is 8% with a mean calving interval of 2.7 years (Agler et al., 1993; NMFS, 2020a). This stock is listed as strategic and depleted under the MMPA due to its Endangered status (NMFS, 2020a). Potential biological removal (PBR) for this stock is 11, and annual human-caused mortality and serious injury for the period between 2014 and 2018 was estimated to be 2.35 per year. This estimate includes incidental fishery interactions (i.e., bycatch/entanglement) and vessel collisions, but other threats to fin whales include contaminants in their habitat and potential climate-related shifts in distribution of prey species (NMFS, 2020a). There is no designated critical habitat for this species in or near the Project Area.

RWF

Two well-known feeding grounds for fin whales are present near the RWF. These include the Great South Channel and Jeffrey's Ledge and waters directly east of Montauk, New York (Kenney and Vigness-Raposa, 2010; NMFS, 2020a). The highest occurrences of fin whales in this region are identified south of Montauk Point, New York to south of Nantucket, Massachusetts (Kenney and Vigness-Raposa, 2010). **Figure 3.1-1** shows visual detections by month in the RI-MA WEA (Kraus et al., 2016), and **Figure 3.1-2** shows the number of detections of fin whales Southern New England based on 10 years of passive acoustic data (Davis et al., 2020). Results of data collected in region 7 (Southern New England where the Project Area is located) indicate the greatest number of detections from August through April with a decrease in fin whale presence in the summer (Davis et al., 2020), whereas visual detections are greatest in the summer (Kraus et al., 2020). Because of these high occurrences within the OCS waters and offshore near the OCS break where surveys occurred, it is likely that fin whales will be present within the RWF area, potentially occurring during all seasons.

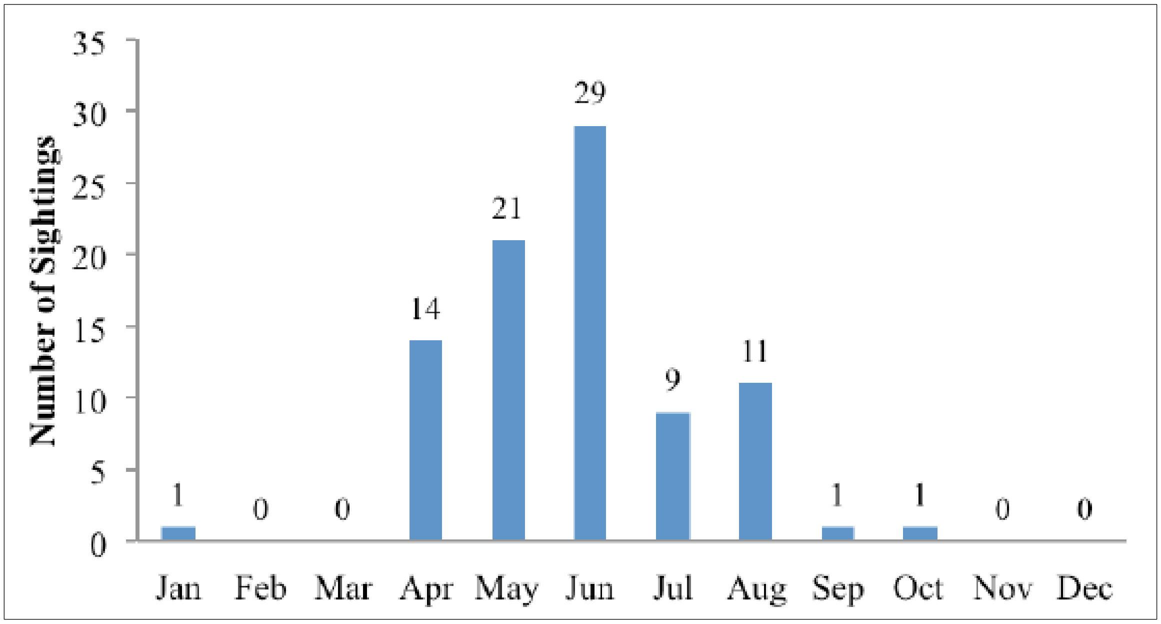

Figure 3.1-1. **Visual detections of fin whales by month for all survey years between October 2011 and June 2015. From: Kraus et al. (2016).**

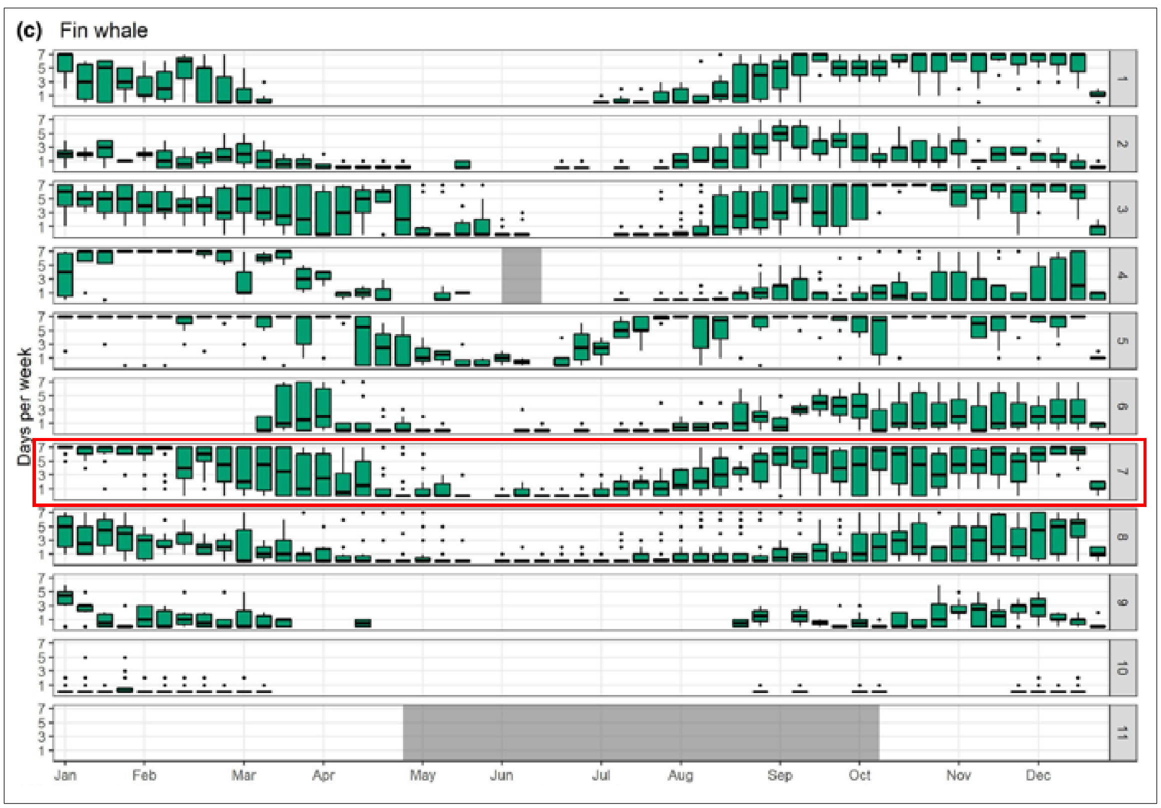

Figure 3.1-2. **Acoustic detections of fin whales from 10 years of passive acoustic data collected along the U.S. East Coast. Region 7 (red box) is Southern New England which contains the Project Area. Gray blocks indicate weeks where no data were collected. Adapted from: Davis et al. (2020).**

RWEC

Fin whales are common in Rhode Island state waters and adjacent OCS waters in this area, and aggregations of fin whales are often reported between Block Island, Rhode Island, and Montauk Point, New York (Kenney and Vigness-Raposa, 2010). They are typically centered along the 100-m isobath off the U.S. East Coast, but sightings have occurred in both shallower and deeper waters and they have been observed in Rhode Island state waters (Kenney and Vigness-Raposa, 2010; RI DEM, 2020). Because of their regular occurrence in this area, a large number of whale watching boats also frequent this area (Kenney and Vigness-Raposa, 2010). Fin whale sightings are greatest in the spring and summer, but they are known to occur in all four seasons in inner shelf waters (Kenney and Vigness-Raposa, 2010). Therefore, it is highly likely that fin whales will be encountered within the RWEC – OCS and RWEC – RI.

Sei Whale

Sei whales occur in all the world's oceans and migrate between feeding grounds in temperate and sub-polar regions to winter grounds in lower latitudes (Kenney and Vigness-Raposa, 2010; NMFS, 2020a). In the Western North Atlantic, most of the population is concentrated in northerly waters along the Scotian Shelf. Sei whales are observed in the spring and summer, utilizing the northern portions of the U.S. Atlantic Exclusive Economic Zone (EEZ) as feeding grounds, including the Gulf of Maine and Georges Bank. The highest concentration is observed during the spring along the eastern margin of Georges Bank and in the Northeast Channel area along the southwestern edge of Georges Bank. The winter habitat for this population remains unknown, but recent PAM data detected sei whale vocalizations from late fall through winter in Southern George's Bank region, with sporadic detections in the Southeast U.S. around Cape Hatteras and Blake Plateau (NMFS, 2020a). In general, sei whales are observed offshore with periodic incursions into more shallow waters for foraging (NMFS, 2020a).

Sei whales can often be confused with fin whales during field surveys; however, they do not have the characteristic v-shaped patterns on their backs that are present on fin whales, and their skin is often mottled with scars thought to be caused by lamprey bites (Jefferson et al., 1993). Although uncertainties still exist with distinguishing sei whale vocalizations during PAM surveys, they are known to produce short duration (0.7 to 2.2 sec) upsweeps and downsweeps between 20 and 600 Hz. SLs for these calls can range from 147 to 183 dB re 1 µPa m (Erbe et al., 2017). No auditory sensitivity data are available for this species (Southall et al., 2019).

Sei whales are listed as Endangered under the ESA and by the IUCN Red List (NMFS, 2020a; IUCN, 2021). Prior to 1999, sei whales in the Western North Atlantic were considered a single stock, but following the suggestion of the Scientific Committee of the International Whaling Commission (IWC), two separate stocks were identified for this species; a Nova Scotia stock and a Labrador Sea stock. Only the Nova Scotia stock can be found in U.S. waters, and the current abundance estimate for this population is 6,292 derived from recent surveys conducted between Halifax, Nova Scotia and Florida (NMFS, 2020a). Population trends are not available for this stock because of insufficient data (NMFS, 2020a). This stock is listed as strategic and depleted under the MMPA due to its Endangered status (NMFS, 2020a). The PBR for this stock is 6.2, and annual human-caused mortality and serious injury from 2014 to 2018 was estimated to be 1.20 per year (NMFS, 2020a). Like fin whales, major threats to sei whales include fishery interactions, vessel collisions, contaminants, and climate-related shifts in prey species (NMFS, 2020a). There is no designated critical habitat for this species in or near the Project Area.

RWF

CETAP surveys observed sei whales along the OCS edge only during the spring (237 sightings) and summer (101 sightings) (CETAP, 1982). This agrees with the Kraus et al. (2016) study, where sei whales were also only observed in the RI-MA WEA during the spring and summer (**Figure 3.1-3**). No sightings were reported during the fall and winter. A small cluster of five individuals was reported south of Montauk Point, New York, and Block Island, Rhode Island, in July 1981, August 1982, and May 2003 (Kenney and Vigness-Raposa, 2010). Davis et al. (2020) found detections of sei whales nearly year-round in Southern

New England, but the greatest number of detections were observed between March and July (**Figure 3.1-4**). Therefore, sei whales may be present seasonally in the RWF, primarily in the spring and summer.

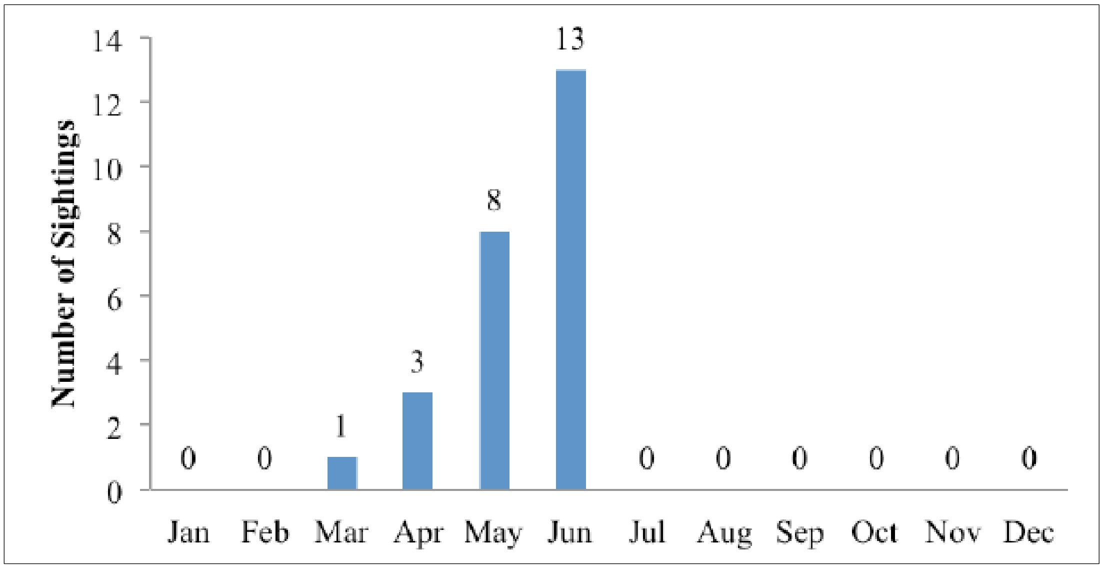

Figure 3.1-3. Visual detections of sei whales by month for all survey years between October 2011 and June 2015. From: Kraus et al. (2016).

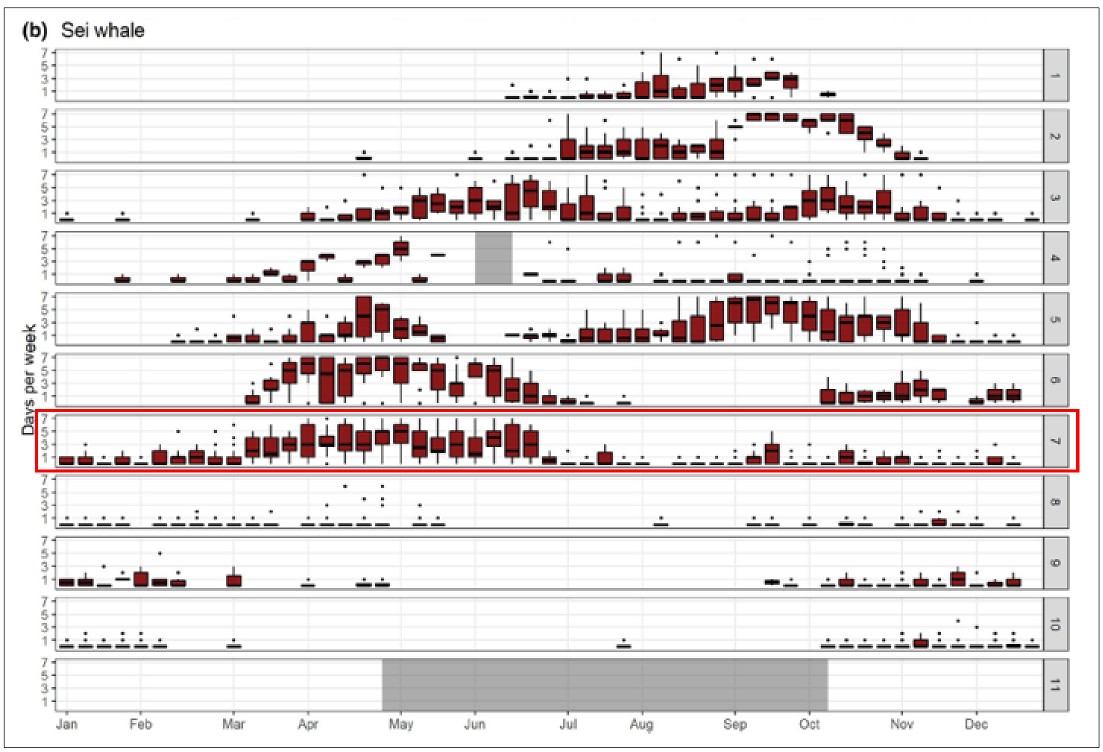

Figure 3.1-4. Acoustic detections of sei whales from 10 years of passive acoustic data collected along the U.S. East Coast. Region 7 (red box) is Southern New England which contains the Project Area. Gray blocks indicate weeks where no data were collected. Adapted from: Davis et al. (2020).

RWEC

Sei whales are associated with the deeper waters along the continental shelf edge and are observed in shallower waters when foraging. In the spring and summer, sei whales are seen in feeding habitats in Nova Scotia and Cape Cod north of the RWEC corridor (NMFS, 2020a). Sei whales are therefore not likely to enter shallower waters off Rhode Island and are not expected to occur in the RWEC - OCS or RWEC – RI.

North Atlantic Right Whale

The North Atlantic right whale occurs in all the world's oceans from temperate to subpolar latitudes. The primary habitat for this species is coastal or continental shelf waters ranging from calving grounds in the Southeastern U.S. to feeding grounds in the Northeastern U.S. (NMFS, 2020a). Acoustic surveys have also demonstrated their presence year-round in the Gulf of Maine, off New Jersey, and off Virginia (NMFS, 2020a). Important feeding habitats include coastal waters off Massachusetts, Georges Bank, the Great South Channel, Gulf of Maine, Bay of Fundy, and the Scotian Shelf. All waters within the Gulf of Maine are designated as a Foraging Area Critical Habitat (NMFS, 2020a).

One of the most distinguishing features of the right whale is the whitish callosities, or areas of roughened skin, covering their head, which can be up to one-third of their body length and their prominently curved jawline (Jefferson et al., 1993). Right whale vocalizations most frequently observed during PAM studies include upsweeps rising from 30 to 450 Hz, often referred to as "upcalls," and broadband (30 to 8,400 Hz) pulses, or "gunshots," with SLs between 172 and 187 dB re 1 μPa m (Erbe et al., 2017). However, recent studies have shown that mother-calf pairs reduce the amplitude of their calls in the calving grounds, possibly to avoid detection by predators (Parks et al. 2019). Modeling conducted using right whale ear morphology suggest that the best hearing sensitivity for this species is between 16 Hz and 25 kHz (Southall et al., 2019; Ketten et al., 2014).

The North Atlantic right whale is listed as Endangered under the ESA and by the state of Rhode Island, and are listed as Critically Endangered by the IUCN Red List (NMFS, 2020a; RI DEM, 2020; IUCN, 2021). Right whales are considered to be the most critically Endangered large whales in the world (NMFS, 2020a). The Western North Atlantic population size was estimated to be 412 individuals in the most recent draft 2020 SAR, which used data from the photo-identification database maintained by the New England Aquarium that were available in October 2019 (NMFS, 2020a). A population trend analysis conducted on the abundance estimates from 1990 to 2011 suggest an increase at about 2.8% per year from an initial abundance estimate of 270 individuals in 1998 (NMFS, 2020a). However, modeling conducted by Pace et al. (2017) showed a decline in annual abundance after 2011, further evidenced by the decrease in the abundance estimate from 451 in 2018 (NMFS, 2020a) to the current 2020 estimate of 412 (NMFS, 2020a). Highly variable data exists regarding the productivity of this stock. Over time, there have been periodic swings of per capita birth rates (NMFS, 2020a). Net productivity rates do not exist as the Western North Atlantic stock lacks any definitive population trend (NMFS, 2020a). The average annual human-related mortality/injury rate exceeds that of the calculated PBR of 0.8, and due to its listing as Endangered under the ESA this population is classified as strategic and depleted under the MMPA (NMFS, 2020a). Estimated human-caused mortality and serious injury between 2014 and 2018 was 8.15 whales per year (NMFS, 2020a). The predominant threats to North Atlantic right whales are entanglement and vessel collisions. Available data from 2000 to 2017 suggest an increase in the percent of injuries and mortalities (per capita) caused by entanglement, and while there no discernible trend in vessel strikes over the years, the annual rate of mortality and serious injury from 2014 to 2018 due to vessel strikes was 1.3 whales per year (NMFS, 2020a). There have been elevated numbers of mortalities reported since 2017 and continuing to through 2020 totaling 34 dead North Atlantic right whales which prompted NMFS to designate an Unusual Mortality Event (UME) for North Atlantic right whales (NMFS, 2021a). Although the majority (62%) of the mortalities occurred in Canadian waters, the U.S. population is not separated from those in Canada, and therefore the effects of mortality affect the population considered in the assessment process. Of these documented mortalities, 41% were of undetermined cause; however, of the remainder of the mortalities (59%) were determined to be the result of human interaction with ten mortalities resulting from vessel strikes and eight resulting from gear entanglement (NMFS, 2021a).

RWF

Kraus et al. (2016) only observed North Atlantic right whales in the RI-MA WEA during the winter and spring (**Figure 3.1-5**). Davis et al. (2017) analyzed 10 years of passive acoustic data and found a similar trend in the data collected in Southern New England where North Atlantic right whale detections began to increase in the winter through early summer (**Figure 3.1-6**).However, the North Atlantic right whale has the potential to occur within the waters off Rhode Island and Massachusetts any time of the year. Typically, right whale sightings begin in December and continue through April. A total of 77 individuals were sighted in the WEA from October 2011 to June 2015. The greatest numbers are seen in March. The Muskeget Channel and south of Nantucket, both located within the RI-MA WEA, were also identified as right whale hotspots during the spring (Kraus et al., 2016).

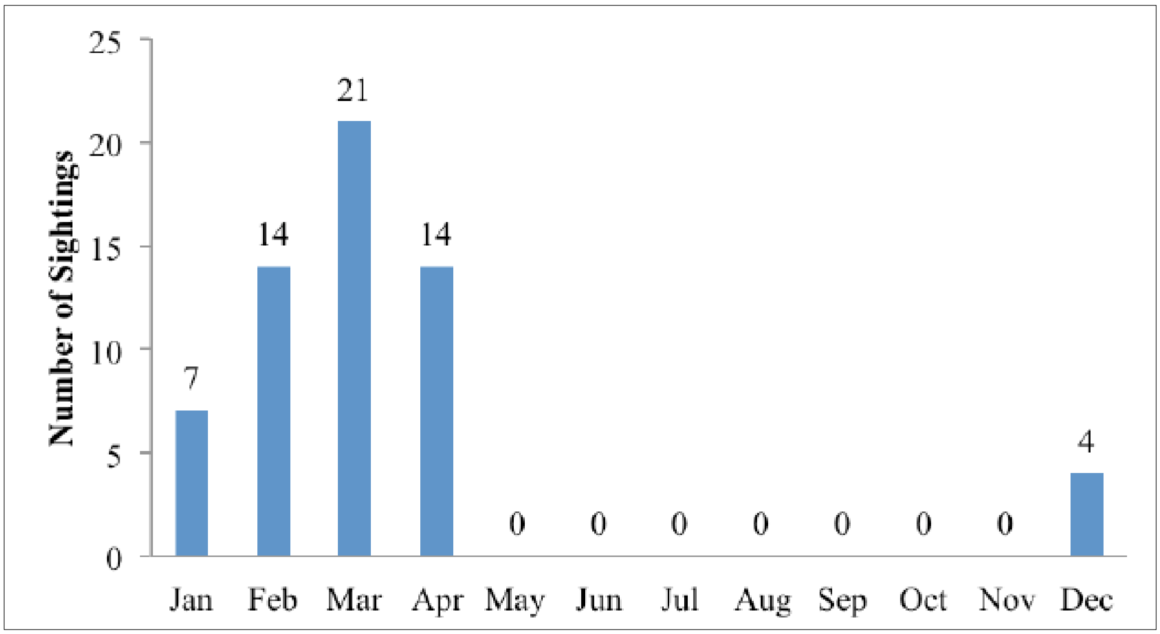

Figure 3.1-5. Visual detections of North Atlantic right whales by month for all survey years between October 2011 and June 2015. From: Kraus et al. (2016).

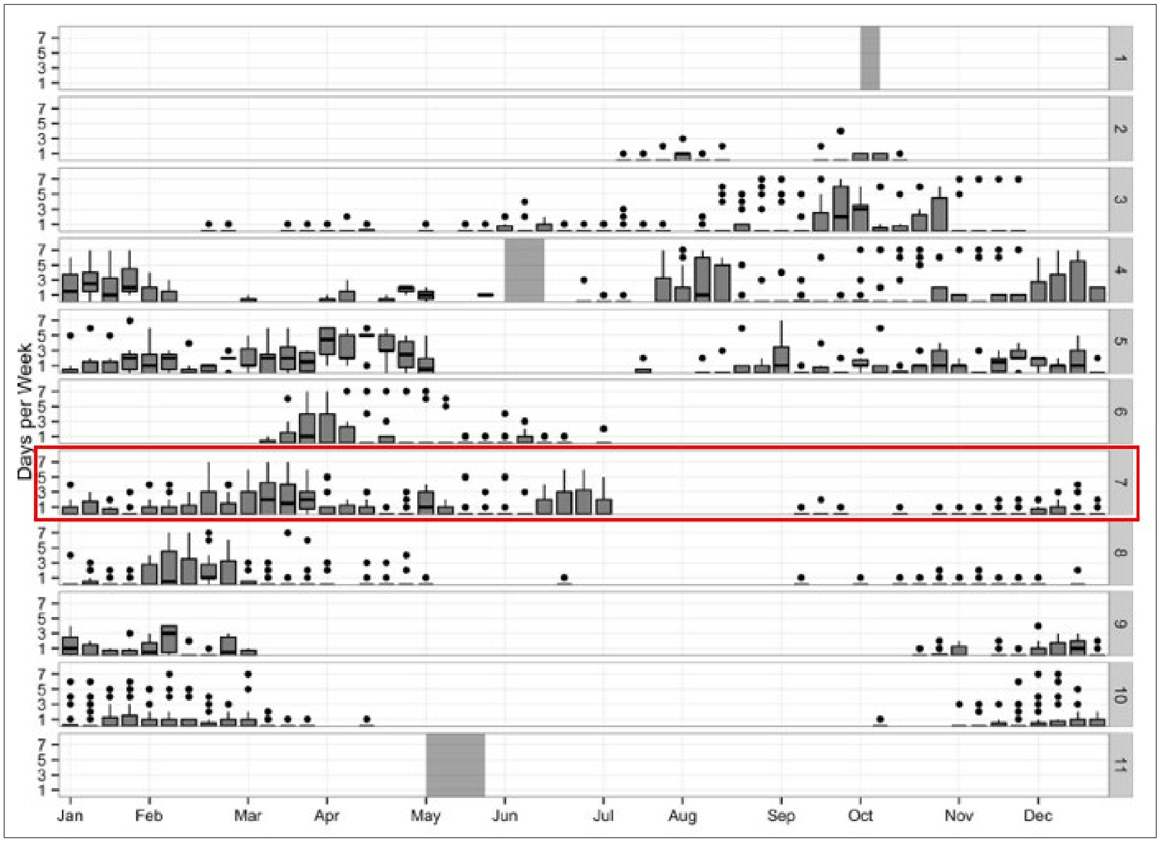

Figure 3.1-6. Acoustic detections of North Atlantic right whales from 10 years of passive acoustic data collected along the U.S. East Coast. Region 7 (red box) is Southern New England which contains the Project Area. Gray blocks indicate weeks where no data were collected. Adapted from: Davis et al. (2017).

Kraus (2018) provided recent right whale survey information for crew training prior to the 2017 South Fork Wind Farm site characterization surveys. North Atlantic right whale sighting results from 2011 to 2015 are presented in **Figure 3.1-7**. Kraus (2018) also presented the sighting locations from 2017 that reported skim (surface) feeding activity by right whales (**Figure 3.1-8**). Skim feeding is an important activity identified in impact assessments because first, it demonstrates a critical behavior (feeding) that could be disrupted by introduced noise; and second, it represents a vulnerable time for right whales to be exposed to ship strikes because they are active at or near the surface.

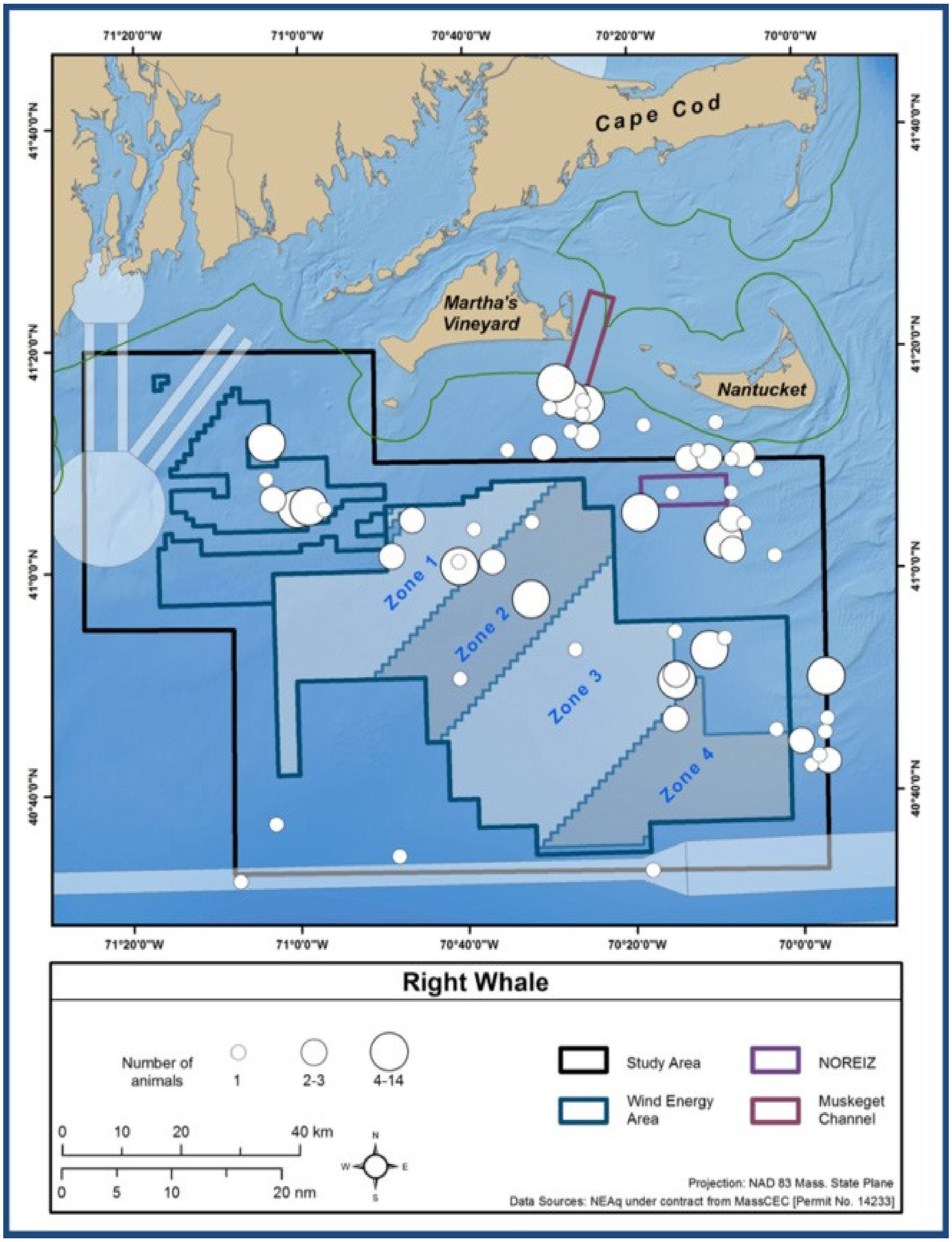

Figure 3.1-7. **North Atlantic right whale sighting data from 2011 to 2015. Figure and data from Kraus (2018). NOREIZ = Northeast Offshore Renewable Energy Innovation Zone.**

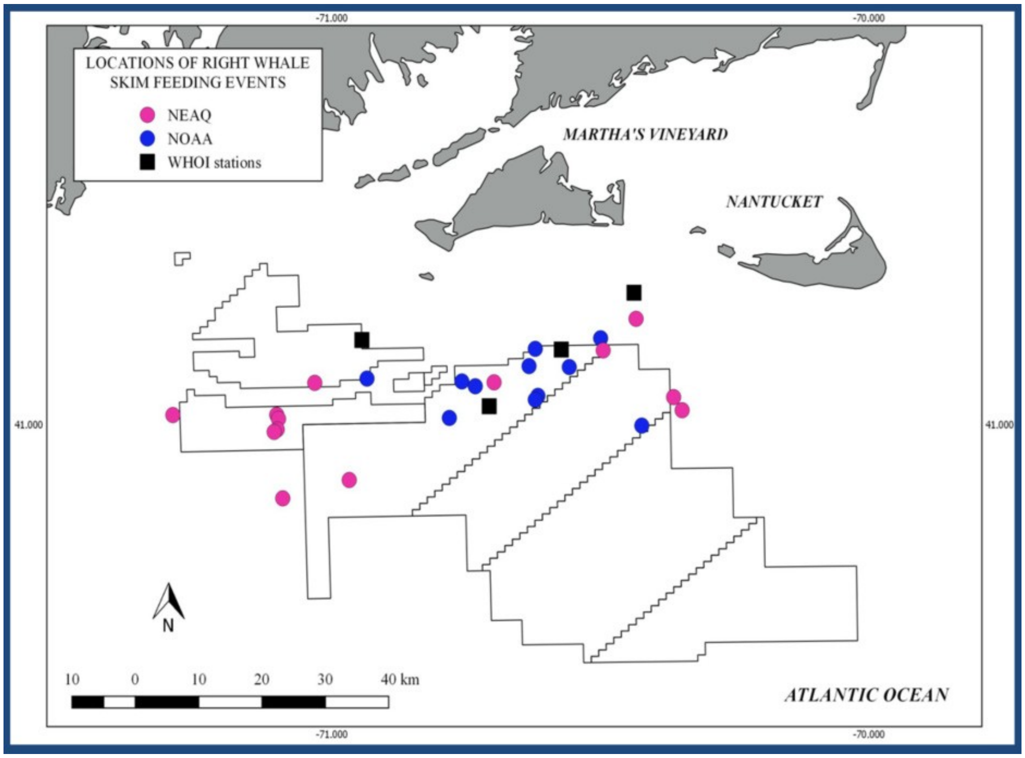

Figure 3.1-8. The 2017 North Atlantic right whale sightings that reported skim (surface) feeding activity. Figure from Kraus (2018). NEAQ = New England Aquarium; NOAA = National Oceanic and Atmospheric Administration.

Seasonal management areas (SMAs) also exist within the vicinity of the RWF, including the Great South Channel SMA (April 1–July 31), Cape Cod Bay SMA (January 1–May 15), Off Race Point SMA (March 1-April 30), and Block Island SMA (November 1–April 30) (NMFS, 2021b); therefore, right whales are likely to occur within the RWF.

RWEC

North Atlantic right whales are known to occur within both Rhode Island state and adjacent OCS waters year-round. The Gulf of Maine has been designated as a critical habitat area; therefore, they may migrate through the RWEC corridor as they travel to this feeding habitat. Kraus et al. (2016) reported a seasonal cluster of right whales south of Martha's Vineyard, Massachusetts, and east of Nantucket, Massachusetts, during the winter. This area is also designated as the Block Island SMA from November 1 through April 20, which contains the RWEC corridor. Therefore, it is likely right whales would occur within both the RWEC – OCS and RWEC – RI.

Sperm Whale

Sperm whales can be found throughout the world's oceans. They can be found near the edge of the ice pack in both hemispheres and are also common along the equator. The North Atlantic stock is distributed mainly along the continental shelf-edge, over the continental slope, and mid-ocean regions, where they prefer water depths of 600 m or more and are less common in waters <300 m deep (Waring et al., 2015; Hayes et al., 2020). In the winter, sperm whales are observed east and northeast of Cape Hatteras. In the spring, sperm whales are more widely distributed throughout the Mid-Atlantic Bight and southern portions of George's Bank (Hayes et al., 2020). In the summer, sperm whale distribution is similar to the spring, but they are more widespread in Georges Bank and the Northeast Channel region and are also observed inshore of the 100-m isobath south of New England (Hayes et al., 2020). Sperm whale occurrence on the continental shelf in areas south of New England is at its highest in the fall (Hayes et al., 2020).

Sperm whales can easily be distinguished in visual surveys by their large, blunt head, narrow underslung jaw, and characteristic blow shape resulting from the S-shaped blowhole set at the front-left of the head (Jefferson et al., 1993). Unlike mysticete whales that produce various types of calls used solely for communication, sperm whales produce clicks that are used for echolocation and foraging as well as communication (Erbe et al., 2017). Sperm whale clicks have been grouped into five classes based on the click rate, or number of clicks per second; these include “squeals,” “creaks,” “usual clicks,” “slow clicks,” and “codas.” In general, these clicks are broadband sounds ranging from 100 Hz to 30 kHz with peak energy centered around 15 kHz. Depending on the class, SLs for sperm whale calls range between approximately 166 and 236 dB re 1 µPa m (Erbe et al., 2017). Hearing sensitivity data for this species are currently unavailable (Southall et al., 2019).

The Western North Atlantic stock is considered strategic under the MMPA due to its listing as Endangered under the ESA, and the global population is listed as Vulnerable on the IUCN Red List (Hayes et al., 2020; IUCN, 2021). The best and most recent abundance estimate based on 2016 surveys conducted between the lower Bay of Fundy and Florida is 4,349 (Hayes et al., 2020). No population trend analysis is available for this stock. Thousands of sperm whales were killed during the early 18th Century. A moratorium on sperm whale hunting was adopted in 1986 and currently no hunting is allowed for any purposes in the North Atlantic. Occasionally, sperm whales will become entangled in fishing gear or be struck by ships off the east coast of the U.S. However, this rate of mortality is not believed to have biologically significant impacts. The current PBR for this stock is 6.9, and because the total estimated human-caused mortality and serious injury is <10% of this calculated PBR, it is considered insignificant (Hayes et al., 2020). Between 2013 and 2017, 12 sperm whale strandings were documented along the U.S. East Coast, but none of the strandings showed evidence of human interactions (Hayes et al., 2020). Other threats to sperm whales include contaminants, climate-related changes in prey distribution, and anthropogenic noise, although the severity of these threats on sperm whales is currently unknown (Hayes et al., 2020). There is no designated critical habitat for this population in the Project Area.

RWF

Sperm whales were the fifth most commonly sighted large whale in the CETAP study area and were observed in all four seasons. The study sighted 341 individuals, which accounted for only 8% of the total large whale sightings during their survey period (CETAP, 1982). Kraus et al. (2016) reported sightings of sperm whales in the RI-MA WEA during the summer and fall months; five individuals in August 2012, one in September 2012, and three in June 2015. There have also been occasional strandings in Massachusetts and Long Island (Kenney and Vigness-Raposa, 2010). Although accounts of sperm whales in the area are low, their occurrence within the RWF and surrounding waters is possible.

RWEC

CETAP reported that the distribution of sperm whales primarily centers at about the 1,000-m depth contour. However, their distribution can also extend shoreward, inshore of the 100-m contour, particularly in the summer and fall (CETAP, 1982; Hayes et al., 2020). Although relatively infrequent, sightings have been reported in waters as shallow as 60 m. Southern New England is one of the few locations in the world in which sperm whales frequent inshore areas (Kenney and Vigness-Raposa, 2010). Many reported sightings take place in a narrow band just south of Block Island, Rhode Island, Martha’s Vineyard, Massachusetts, and Nantucket, Massachusetts, from May through November, in which the RWEC corridor would intersect. This high occurrence of sperm whales is believed to be related to the presence of spawning squid (CETAP, 1982). Therefore, given their preference for deeper waters sperm whales are likely to occur in the RWEC – OCS, but may also occur seasonally within the RWEC – RI in the summer and fall when they enter shallower state waters in search of food.

3.1.2 Non-ESA listed Species

Of the 30 non-listed species whose ranges include the Northeastern U.S., 11 are expected to be present in the Project Area and are considered potentially affected species. The following sections provide further information regarding species behavior and expected occurrence in the RWF and two RWEC areas (RWEC – OCS and RWEC – RI).

Minke Whale

Minke whales prefer the colder waters in northern and southern latitudes, but they can be found in every ocean in the world. Available data suggest that minke whales are distributed in shallower waters along the continental shelf between the spring and fall and are located in deeper oceanic waters between the winter and spring (NMFS, 2020a). They are most abundant in New England waters in the spring, summer, and early fall (NMFS, 2020a).

A prominent morphological feature of the minke whale is the large, pointed median ridge on top of the rostrum. The body is dark gray to black with a pale belly, and frequently shows pale areas on the sides that may extend up onto the back. The flippers are smooth and taper to a point, and the middle third of each flipper has a conspicuous bright white band that can be distinguished during visual surveys (Kenney and Vigness-Raposa, 2010). In the North Atlantic, minke whales commonly produce pulse trains lasting 10 to 70 sec with a frequency range between 10 and 800 Hz. SLs for this call type have been reported between 159 and 176 dB re 1 μPa m (Erbe et al., 2017). Some minke whales also produce a unique "boing" sound which is a train of rapid pulses often described as an initial pulse followed by an undulating tonal (Erbe et al., 2017; Rankin and Barlow, 2005). The "boing" ranges from 1 to 5 kHz with an SLs of approximately 150 dB re 1 μPa m (Erbe et al., 2017). Auditory sensitivity for this species based on anatomical modeling of minke whale ear morphology is best between 10 Hz and 34 kHz (Southall et al., 2019; Ketten et al., 2014).

Minke whales are not listed under the ESA or classified as strategic under the MMPA and are list as Least Concern on the IUCN Red List (NMFS, 2020a; IUCN, 2021). The best available current global abundance estimates for the common minke whale, compiled by the IUCN Red List, is around 200,000 (Cooke, 2018). The most recent population estimate for the Canadian East Coast stock which occurs in the Project Area is 24,202 minke whales, derived from surveys conducted by NOAA and the Department of Fisheries and Oceans Canada between Labrador and central Virginia (NMFS, 2020a). There are no current population trends or net productivity rates for this species due to insufficient data. The PBR for this stock is estimated to be 170 (NMFS, 2020a). The estimated annual human-caused mortality and serious injury from 2014 to 2018 was 10.55 per year attributed to fishery interactions, vessel strikes, and non-fishery entanglement in both the U.S. and Canada (NMFS, 2020a), and a UME was declared for this species in January 2017 (NMFS, 2021c). Minke whales may also be vulnerable to climate-related changes in prey distribution, although the extent of this effect on minke whales remains uncertain (NMFS, 2020a). No designated critical habitat for this stock currently exists in the Project Area.

<u>*RWF*</u>

During previous studies conducted in the RI-MA WEA, 103 minke whales were sighted within the area (Kraus et al., 2016). Spring observations included the most individuals followed by summer, and fall (**Figure 3.1-9**). Minke whales are therefore likely to occur in the spring and summer within the RWF area.

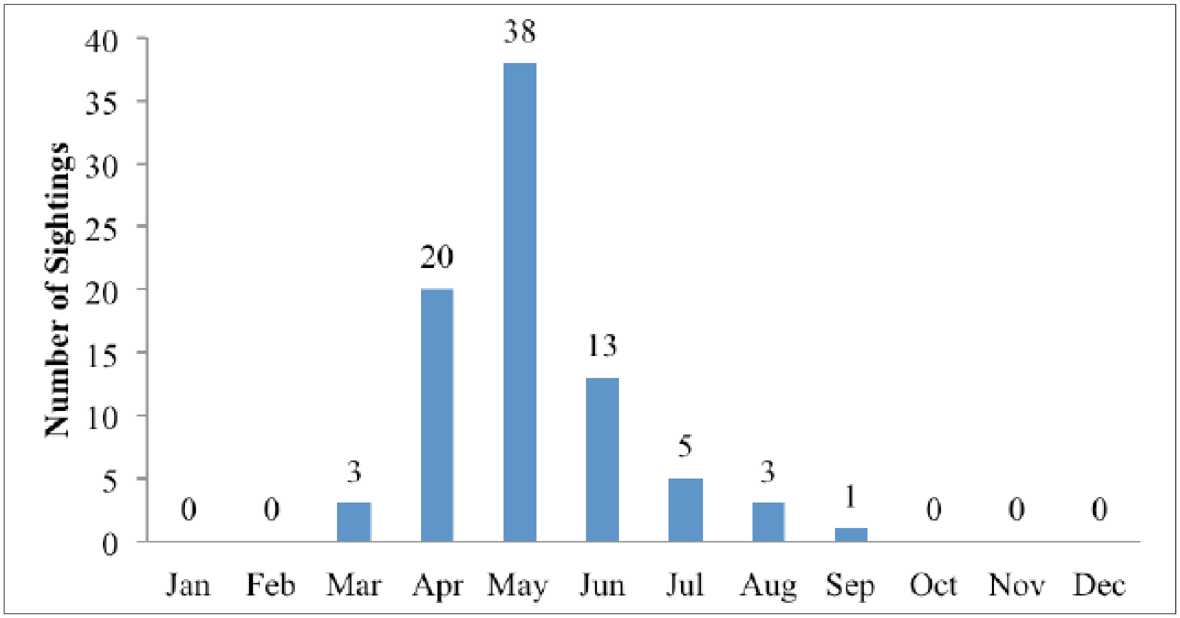

Figure 3.1-9. Visual detections of minke whales by month for all survey years between October 2011 and June 2015. From: Kraus et al. (2016).

RWEC

Minke whales have been sighted offshore Rhode Island in both state and OCS waters in all four seasons (Kenney and Vigness-Raposa, 2010). A large proportion of these sightings were reported from whale watching boats. A dense concentration was seen between Block Island, Rhode Island, and Montauk Point, New York, in the spring and summer (Kenney and Vigness-Raposa, 2010), making it likely that this species could occur within both the RWEC – OCS and RWEC – RI.

Humpback Whale

The humpback whale can be found worldwide in all major oceans from the equator to sub-polar latitudes. In the summer, humpbacks are found in higher latitudes feeding in the Gulf of Maine and Gulf of Alaska. During the winter months, humpbacks migrate to calving grounds in subtropical or tropical waters, such as the Dominican Republic in the Atlantic and Hawaiian Islands in the Pacific (Hayes et al., 2020). Humpback whales from the North Atlantic feeding areas mate and calve in the West Indies (Hayes et al., 2020). In the summer, humpback whales in the Western North Atlantic are typically observed in the Gulf of Maine and along the Scotian Shelf, and there have also been numerous winter sightings in the Southeastern U.S. (NMFS, 2020a). Feeding behavior has also been observed in New England off Long Island, New York, and survey data from NOAA suggests a potential increase in humpback whale abundance off New Jersey and New York (NMFS, 2020a).

Humpback whales are easily identified in field surveys by their long flippers, which can be up to one-third of their total body length, as well as the bumps covering their head and flippers (Jefferson et al., 1993). During migration and breeding seasons, male humpback whales are often recorded producing vocalizations arranged into repetitive sequences termed “songs” that can last for hours or even days. These songs have been well studied in the literature to document changes over time and geographic differences; generally, the bandwidth of these songs range from 20 Hz to over 24 kHz. Most of the energy is focused between 50 and 1,000 Hz and reported SLs range from 151 to 189 dB re 1 µPa m (Erbe et al., 2017). Other calls produced by humpbacks, both male and female, include pulses, moans, and grunts used for foraging and communication. These calls are lower frequency (under 2 kHz) with SLs ranging from 162 to 190 dB re 1 µPa m (Erbe et al., 2017; Thompson et al., 1986). Anatomical modeling based on humpback whale ear morphology indicate that their best hearing sensitivity is between 18 Hz and 15 kHz (Southall et al., 2019; Ketten et al., 2014).

NMFS revised the listing status for humpback whales under the ESA in 2016 (81 FR 62259). Globally, there are 14 distinct population segments (DPSs) recognized for humpback whales, four of which are listed as Endangered. The Gulf of Maine stock (formerly known as the Western North Atlantic stock) which occurs in the Project Area is not considered strategic under the MMPA and does not coincide with any ESA-list DPS (NMFS, 2020a). The global population is listed as Least Concern under the IUCN Red List, and are considered endangered by the state of Rhode Island given the previous status under the ESA and the current status of some DPSs (RI DEM, 2020; IUCN, 2021). The best available abundance estimate of the Gulf of Maine stock is 1,393, derived from modeled sighting histories constructed using photo-identification data collected through October 2016 (NMFS, 2020a). Available data indicate that this stock is characterized by a positive population trend, with an estimated increase in abundance of 2.8% per year (NMFS, 2020a). The PBR for this stock is 22, and the estimated annual human-caused mortality and serious injury between 2014 and 2018 was 15.25 whales per year (NMFS, 2020a). While the current annual mortality and serious injury is below the calculated PBR, this estimate only includes detected mortalities and serious injuries. Detected mortality is estimated to only be 20% of all mortality, which could indicate the total mortality in humpbacks has or will exceed PBR, a prediction further supported by the UME declared for this species in 2016 (NMFS, 2020a; NMFS, 2021d). Major threats to humpback whales include vessel strikes, entanglement, and climate-related shifts in prey distribution (NMFS, 2020a). There is no designated critical habitat for this stock in the Project Area.

RWF

Kraus et al. (2016) reported humpback whale sightings in the RI-MA WEA during all seasons, with peak abundance during the spring and early summer, but their presence within the region varies between years. Increased stocks of sand lance (*Ammodytes* spp.) appear to correlate with the years in which most whales were observed, suggesting that humpback whale distribution and occurrences could largely be influenced by prey availability (Kenney and Vigness-Raposa, 2010). The greatest number of sightings of humpbacks in the RI-MA WEA occurred during April (33 sightings); their presence increased starting in March and continued through July. Seasonal abundance estimates of humpback whales in the RI-MA WEA range from 0 to 41 (Kraus et al., 2016), with higher estimates observed during the spring and summer (**Figure 3.1-10**). Acoustic detections within Southern New England analyzed by Davis et al. (2020) found the greatest number of acoustic detections in the winter and spring with a similar increase in detection in March which continues through July (**Figure 3.1-11**). Based on these data, humpback whales are likely to occur in the RWF area, predominantly during winter, spring, and early summer.

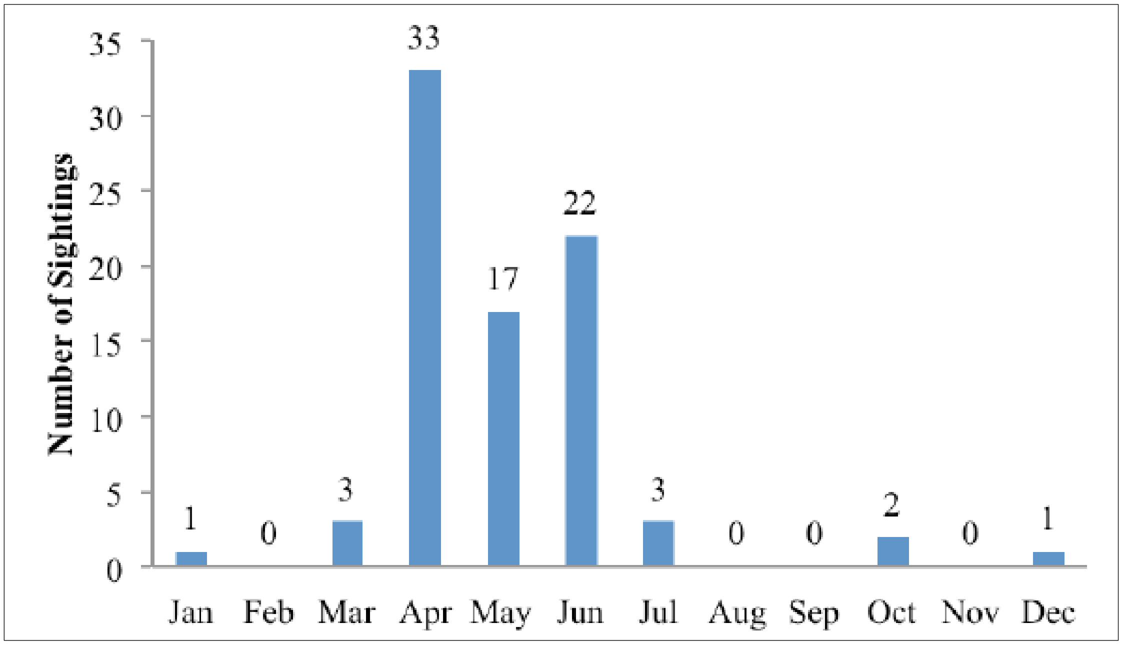

Figure 3.1-10. Visual detections of humpback whales by month for all survey years between October 2011 and June 2015. From: Kraus et al. (2016).

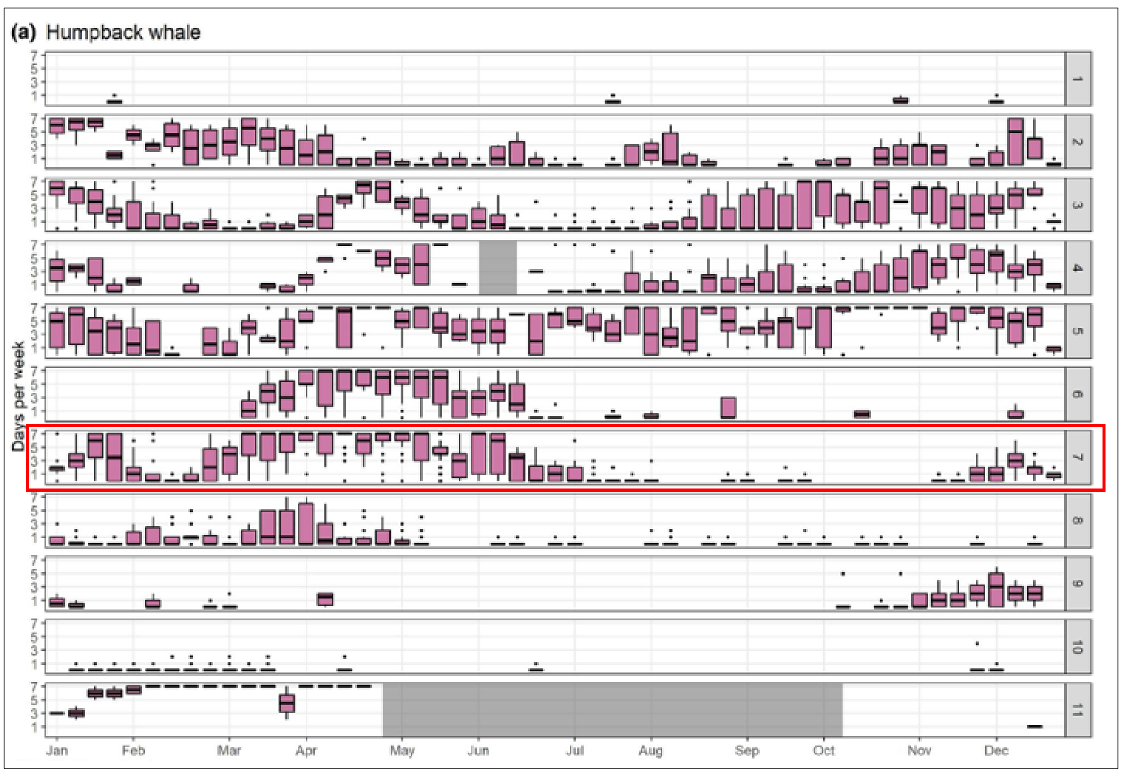

Figure 3.1-11. Acoustic detections of humpback whales from 10 years of passive acoustic data collected along the U.S. East Coast. Region 7 (red box) is Southern New England which contains the Project Area. Gray blocks indicate weeks where no data were collected. Adapted from: Davis et al. (2020).

RWEC

In the 1980s, numerous sightings of humpbacks were reported between Long Island, New York, and Martha's Vineyard, Massachusetts, by Montauk and Galilee whale watching boats. Montauk boats reported 2 sightings in 1986 and 63 sightings in 1987 (Kenney and Vigness-Raposa, 2010). Recently, multiple humpbacks were reported feeding off Long Island, New York, during July 2016 and near New York City during November and December 2016 (Hayes et al., 2020). Humpback strandings were also reported along the southern shore of eastern Long Island, New York, in February 1992, November 1992, October 1993, August 1997, and April 2004.

Humpbacks are known occur within Rhode Island state and adjacent OCS waters; however, their presence is relatively unpredictable and may be strongly influenced by prey availability (Kenney and Vigness-Raposa, 2010). They are expected to have a greater presence in the RWEC – OCS compared to the RWEC – RI, but have been observed in state waters and are therefore likely to be encountered in the RWEC – RI. During most years, their occurrence within the RWEC - RI would be uncommon; however, they may become locally abundant in certain years.

Long-finned Pilot Whale

There are two species of pilot whale in the Western North Atlantic, long-finned and short-finned (*Globicephala macrorhynchus*). Because it is difficult to differentiate between these two species in the field, sightings are usually reported to genus level only (CETAP, 1982; Hayes et al., 2020). However, short-finned pilot whales are a southern or tropical species and pilot whale sightings above approximately 42° N are most likely long-finned pilot whales. Short-finned pilot whale occurrence in the Project Area is considered rare (CETAP, 1982; Hayes et al., 2020). Long-finned pilot whales are distributed along the continental shelf waters off the Northeastern U.S. in the winter and early spring. By late spring, pilot whales migrate into more northern waters including Georges Bank and the Gulf of Maine and will remain there until fall.

Both short-finned and long-finned pilot whales are similar in coloration and body shape; however, long-finned pilot whales can be distinguished by their long flippers, which are 18 to 27% of the body length with a pointed tip and angled leading edge (Jefferson et al., 1993). Like dolphin species, long-finned pilot whales can produce whistles and burst-pulses used for foraging and communication. Whistles typically range in frequency from 1 to 11 kHz while burst-pulses cover a broader frequency range from 100 Hz to 22 kHz (Erbe et al., 2017). Auditory evoked potential (AEP) measurements conducted by Pacini et al. (2010) indicate that the hearing sensitivity for this species ranges from <4 kHz to 89 kHz.

Long-finned pilot whales are not listed under the ESA and are classified as Least Concern by the IUCN Red List (Hayes et al., 2020; IUCN, 2021). The best available estimate of long-finned pilot whales in the Western North Atlantic is 39,215 based on recent surveys covering waters between Labrador and Central Virginia (Hayes et al., 2020). A trend analysis has not been conducted for this stock due to the relatively imprecise abundance estimates (Hayes et al., 2020). The PBR for this stock is 306, and the annual human-caused mortality and serious injury was estimated to be 21 whales between 2013 and 2017 (Hayes et al., 2020). Long-finned pilot whales have a propensity to mas strand in U.S. waters, although the role of human activity in these strandings remains unknown (Hayes et al., 2020). Threats to this population include entanglement in fishing gear, contaminants, climate-related shifts in prey distribution, and anthropogenic noise (Hayes et al., 2020). There is no designated critical habitat for this stock in the Project Area.

RWF

CETAP surveys reported long-finned pilot whales as the third most commonly sighted small whale in their study area with 12,438 individuals (CETAP, 1982). Long-finned pilot whales have been observed in OCS waters off Rhode Island in all four seasons, with peak occurrences in the spring. There are 43 records of long-finned pilot whales and 226 records of non-specific pilot whales in this area. Nine sightings during the summer and three sightings in the spring were reported from whale watching data for pilot whales (Kenney and Vigness-Raposa, 2010).

Within the RI-MA WEA, no sightings of pilot whales were observed during the summer, fall, or winter (Kraus et al., 2016). Long-finned pilot whales are relatively common in the area; therefore, they may potentially occur in the RWF area. However, the likelihood of occurrences would only be in the spring.

RWEC

Long-finned pilot whales prefer deep pelagic temperate to subpolar oceanic waters; therefore, they are not likely to occur within the RWEC – OCS or RWEC – RI (Hayes et al., 2020).

Atlantic Spotted Dolphin

Atlantic spotted dolphins are found in tropical and warm temperate waters. In the Western North Atlantic, their distribution ranges from the Northeastern U.S. to the Gulf of Mexico and the Caribbean to Venezuela (Hayes et al., 2020). They are regularly seen in continental shelf and slope waters. There are two Atlantic spotted dolphin ecotypes which may be distinct sub-species. The larger heavily spotted ecotype inhabits OCS waters inside or near the 200-m isobath south of Cape Hatteras. The smaller form is less spotted and is found further offshore and only occurs in the Atlantic. Recent genetic data also suggests that they may be genetically distinct populations (Hayes et al., 2020). Both ecotypes can occur in the Northeastern U.S.; however, they are difficult to differentiate at sea and are therefore not distinguished in this assessment.

Young Atlantic spotted dolphins start out with no spotting and resemble slender bottlenose dolphins. Large spotting develops as the animals age making it easier to distinguish them in visual surveys (Jefferson et al., 1993). Atlantic spotted dolphins have an estimated auditory bandwidth of 150 Hz to 160 kHz and vocalizations typically range from 100 Hz to 130 kHz (Department of the Navy, 2007; Southall et al., 2007). No auditory sensitivity data are available for this species (Southall et al., 2019).

Atlantic spotted dolphins are not listed under the ESA and are classified as Least Concern by the IUCN Red List (Hayes et al., 2020; IUCN, 2021). The best population estimate available for this species is 39,921 based on surveys conducted in summer 2016 between the lower Bay of Fundy and Florida (Hayes et al., 2020). A population trend analysis of available abundance estimates from 2004, 2011, and 2016 indicate a linear decrease in abundance, however interannual variability in abundance is a key uncertainty in this trend analysis (Hayes et al., 2020). The PBR for this stock is 320, and the estimated annual human-caused mortality and serious injury from 2013 to 2017 was presumed to be zero (Hayes et al., 2020). Twenty-one Atlantic spotted dolphins were reported stranded between North Carolina and Florida during this period; however, no definitive evidence of human interaction was found (Hayes et al., 2020). Major threats to this population include anthropogenic noise; offshore development, particularly south of Cape Hatteras where this species inhabits inshore shelf waters; contaminants; and climate-related shifts in prey distribution (Hayes et al., 2020). There is no designated critical habitat for this stock in the Project Area.

RWF

There are few reported occurrences of general spotted dolphins (*Stenella* spp.) in the Project Area. CETAP described spotted dolphins as the seventh most commonly sighted cetaceans in the study area, with 126 sightings over the course of a 3-year study. The 1982 CETAP data observed 40 individuals south of Block Island, Rhode Island (CETAP, 1982). NMFS shipboard surveys conducted during June to August between central Virginia and the Lower Bay of Fundy reported 542 to 860 individual sightings from two separate visual teams (Palka et al., 2017). Atlantic spotted dolphins tend to be a more subtropical and offshore species, so while they may be encountered in the RWF area, this would be an uncommon occurrence.

RWEC

Atlantic spotted dolphins north of Cape Hatteras tend to be observed offshore over and beyond the continental slope; therefore, their presence in the RWEC – OCS or RWEC – RI would be uncommon.

Atlantic White-sided Dolphin

Atlantic white-sided dolphins migrate between the temperate and polar waters of the North Atlantic Ocean, but usually maintain migration routes over the deeper-sloped continental shelves. This is the most abundant dolphin in the Gulf of Maine and the Gulf of St. Lawrence; they are rarely seen off the coast of Nova Scotia (Kenney and Vigness-Raposa, 2010). Behaviorally, this species is highly social, but not as demonstrative as some other common dolphins. They typically form pods of around 30 to 150 individuals but have also been seen in very large pods of 500 to 2,000 individuals (Hayes et al., 2020). It is common to find these pods associated with the presence of other white-beaked dolphins, pilot whales, fin whales, and humpback whales.

The Atlantic white-sided dolphin gets its name from the distinctive white stripe on its side, which starts just below the dorsal fin and runs into a yellow/ochre blaze continuing onto the tailstock, which is easily seen when the animal is bow-riding or porpoising. It has a whitish lower jaw, throat, and belly to genital region, with a dark eye patch and face-flipper stripe (Cipriano, 2002; Jefferson et al., 1993). Like most dolphin species, Atlantic white-sided dolphins produce clicks, buzzes, calls, and whistles. Their clicks are broadband sounds ranging from 30 to 40 kHz that can contain frequencies over 100 kHz and are often produced during foraging and for orientation within the water column. Buzzes and calls are not as well studied, and they may be used for socialization as well as foraging. Whistles are primarily for social communication and group cohesion and are characterized by a downsweep followed by an upsweep with an approximate starting frequency of 20 kHz and ending frequency of 17 kHz (Hamran, 2014). No hearing sensitivity data are currently available for this species (Southall et al., 2019).

Atlantic white-sided dolphins are not listed under the ESA or considered a strategic stock under the MMPA and are classified as Least Concern on the IUCN Red List (Hayes et al., 2020; IUCN, 2021). The best abundance estimate currently available for the Western North Atlantic stock is 93,233 based on surveys conducted between Labrador to Florida (Hayes et al., 2020). A trend analysis is not currently available for this stock due to insufficient data (Hayes et al., 2020). The PBR for this stock is 544 and the annual rate of human-caused mortality and serious injury from 2013 to 2017 was estimated to be 26 dolphins. This estimate is based on observed fishery interactions, but Atlantic white-sided dolphins are also threatened by contaminants in their habitat, and climate-related shifts in prey distribution (Hayes et al., 2020). There is no designated critical habitat for this stock in the Project Area.

RWF

Seasonal abundances off the Northeast U.S. in spring through fall are estimated to be 38,000 to 42,000 animals (CETAP, 1982; Kenney and Vigness-Raposa, 2010). Over the course of BOEM's study in the RI-MA WEA, 185 individual Atlantic white-sided dolphins were sighted within the Lease Area; most were observed during summer (112 sightings) followed by fall (70 sightings) (Kraus et al., 2016). Atlantic white-sided dolphins are one of the most likely delphinids that would occur seasonally within the RWF area.

RWEC

Atlantic white-sided dolphins are one of the three odontocetes primarily inhabiting OCS waters shoreward of the 100-m depth contour (CETAP, 1982; Hayes et al., 2020). Most of the sightings (90%) were seen within an estimated depth range of 38 to 271 m. Sightings are concentrated in coastal waters near Cape May, New Jersey, and in shallow waters within the Gulf of Maine (CETAP, 1982). The Gulf of Maine population is commonly seen from the Hudson Canyon to Georges Bank. Sightings south of Georges Bank and Hudson Canyon occur year-round; however, at lower densities (Hayes et al., 2020).

Offshore Rhode Island, Atlantic white-sided dolphins are common in OCS waters, with a slight tendency to occur in shallower state waters in the spring (Kenney and Vigness-Raposa, 2010). Records indicate that there is an aggregation of sightings southeast of Montauk Point, New York, during the spring and summer. Strandings of white-sided dolphins in Rhode Island are relatively rare; from 2001 to 2005, there was an average of 1.2 strandings per year (Kenney and Vigness-Raposa, 2010). Atlantic white-sided dolphins occur in seasonably high numbers in nearshore areas during the spring and summer; therefore, they could potentially occur within the RWEC – OCS and RWEC – RI.

Common Dolphin

The common dolphin has a wide distribution and can be found in both tropical and temperate areas of the Pacific and Atlantic Oceans in both nearshore and offshore waters (Perrin, 2002). Two common dolphin species were previously recognized: the long-beaked common dolphin (*Delphinus capensis*) and the short-beaked common dolphin (*Delphinus delphis)*; however, Cunha et al. (2015) summarized the relevant data and analyses along with additional molecular data and analysis, and recommended that the long-beaked common dolphin not be further used for the Atlantic Ocean. This taxonomic convention was adopted by the Society or Marine Mammalogy. This highly social and energetic species usually travels in large pods consisting of 50 to >1,000 individuals (Hammond et al., 2008b). The common dolphin can frequently be seen performing acrobatics and interacting with large vessels and other marine mammals.

Common dolphins have a very distinct color pattern that takes the form of an hourglass on its side, and most individuals also have a prominent white patch on the dorsal fin (Jefferson et al., 2008). Common dolphin clicks are broadband sounds between 17 and 45 kHz with peak energy between 23 and 67 kHz. Burst-pulse sounds are typically between 2 and 14 kHz while the key frequencies of common dolphin whistles are between 3 and 24 kHz (Erbe et al., 2017). No hearing sensitivity data are available for this species (Southall et al., 2019).

The common dolphin is not listed under the ESA and is classified as Least Concern by the IUCN Red List (NMFS, 2020a; IUCN, 2021). The current best abundance estimate for the Western North Atlantic stock is 172,947 based on recent surveys conducted between Newfoundland and Florida (NMFS, 2020a). A trend analysis was not conducted for this stock because of the imprecise abundance estimate and long survey intervals (NMFS, 2020a). The common dolphin faces anthropogenic threats because of its utilization of nearshore habitat and highly social nature, but it is not considered a strategic stock under the MMPA because the average annual human-caused mortality and serious injury does not exceed the calculated PBR of 1,452 for this stock (NMFS, 2020a). Historically, this species was hunted in large numbers for food and oil. Currently, they continue to suffer incidental mortality from vessel collisions and Eastern North American fishing activities within the Atlantic, most prominently yellowfin tuna (*Thunnus albacares*) nets, driftnets, and bottom-set gillnets (Kraus et al., 2016; NMFS, 2020a). The annual estimated human-caused mortality and serious injury for 2014 to 2018 was 399, which included fishery-interactions and research takes (NMFS, 2020a). Other threats to this species include contaminants in their habitat and climate-related changes in prey distribution (NMFS, 2020a). There is no designated critical habitat for this stock in the Project Area.

RWF

Kraus et al. (2016) observed 3,896 common dolphins within the RI-MA WEA. Most were observed during summer surveys followed by fall, winter, then spring (**Figure 3.1-12**). This was the highest number of individual sightings of all the small cetaceans; therefore, it is anticipated to be one of the most frequent delphinids to occur seasonally within the RWF area.

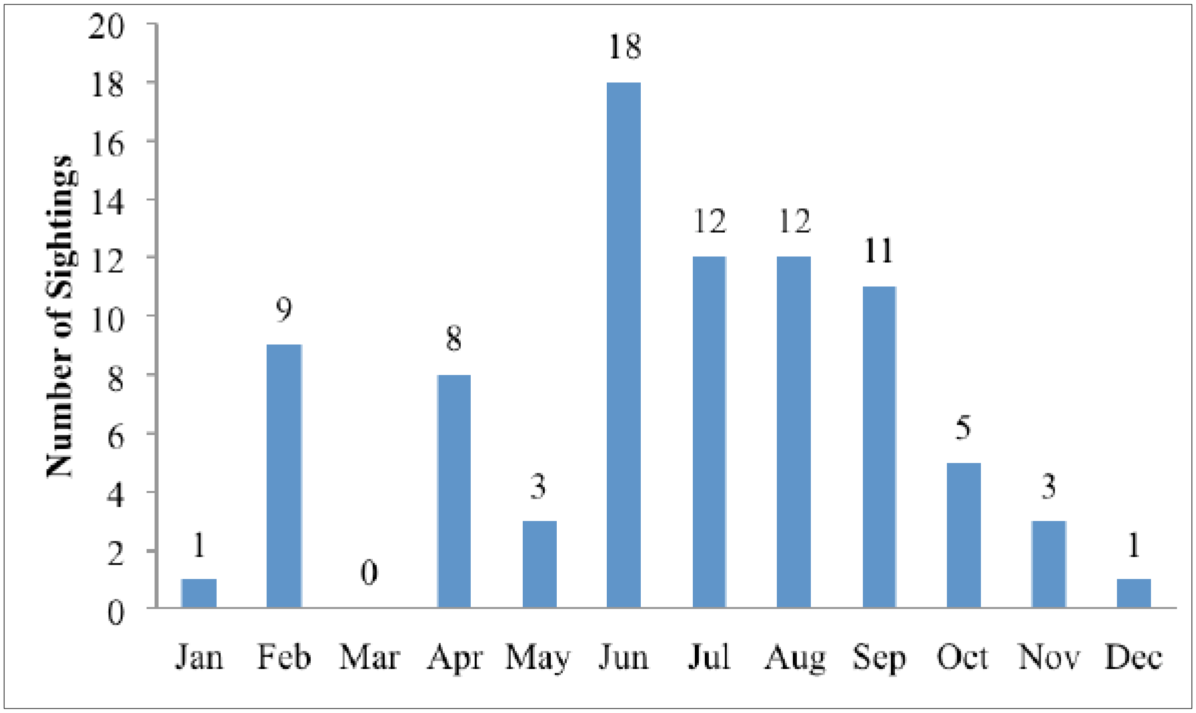

Figure 3.1-12. Visual detections of common dolphin by month for all survey years between October 2011 and June 2015. From: Kraus et al. (2016).

RWEC

Since the common dolphin has a wide distribution and can be found in both nearshore and offshore waters of the Pacific and Atlantic Oceans, they could potentially occur within both the RWEC – OCS and RWEC —RI (Perrin, 2002).

Risso's Dolphin

Risso's dolphins are found in temperate, subtropical, and tropical waters. In the Western North Atlantic, their range extends from Florida to Eastern Newfoundland. Off the Northeastern U.S. Coast, Risso's dolphins are primarily concentrated along the continental shelf edge, but they can also be found swimming in shallower waters to the mid-shelf (Hayes et al., 2020).

Unlike most other dolphins, Risso's dolphins have blunt heads without distinct beaks. Coloration for this species ranges from dark to light grey. Adult Risso's dolphins are typically covered in white scratches and spots that can be used to identify this species in field surveys (Jefferson et al., 1993). Whistles for this species have frequencies ranging from around 4 kHz to over 22 kHz with estimated SLs between 163 and 210 dB re 1 µPa m (Erbe et al., 2017). Studies using both behavioral and AEP methods have been conducted for this species, which show greatest auditory sensitivity between <4 kHz to >100 kHz (Nachtigall et al., 1995; Nachtigall et al., 2005).

Risso's dolphins are not listed under the ESA and are classified as a species of Least Concern on the IUCN Red List (Hayes et al., 2020; IUCN, 2021). The best abundance estimate in the Western North Atlantic is 35,493 based on surveys conducted from Newfoundland and Florida (Hayes et al., 2020). A trend analysis was not conducted on this species, because there are insufficient data to generate this information. PBR for this stock is 303, and the annual human-caused mortality and injury for 2013 to 2017 was estimated to be 54.3 (Hayes et al., 2020). This stock is not classified as strategic under the MMPA because mortality does not exceed the calculated PBR. Threats to this stock include fishery interactions, non-fishery related human interaction, contaminants in their habitat, and climate-related shifts in prey distribution (Hayes et al., 2020). There is no designated critical habitat for this stock in the Project Area.

RWF

Risso's dolphins have been observed in OCS waters offshore Rhode Island year-round, with most sightings during the summer. Sighting data primarily shows that this species is found along the shelf break, with only few species seen in waters shallower than 100 m. Only one sighting in the Rhode Island Ocean Special Area Management Plan study area was reported in the spring (Kenney and Vigness-Raposa, 2010). Kraus et al. (2016) only observed two Risso's dolphins in the RI-MA WEA during the spring. Risso's dolphins do occur in the area; however, because of the infrequent sightings in shallower waters and more concentrated distribution along the continental shelf, the likelihood of encountering Risso's dolphins in the RWF area is relatively low.

RWEC

Risso's dolphins are unlikely to occur within the RWEC – OCS or RWEC – RI due to their primary occurrence in deeper waters along the OCS edge (Hayes et al., 2020).

Common Bottlenose Dolphin

In the Western North Atlantic, there are two morphologically and genetically distinct common bottlenose morphotypes, the Western North Atlantic Northern Migratory Coastal stock and the Western North Atlantic Offshore stock. The offshore stock is primarily distributed along the OCS and slope from Georges Bank to Florida (Hayes et al., 2020), whereas the northern migratory coastal stock is distributed along the coast between southern Long Island, New York and Florida (NMFS, 2020a). Given their distribution, only the offshore stock is likely to occur in the Project Area and is the only stock included in this assessment.

Common bottlenose dolphins are large, relatively robust animals. The snout is stocky and set off from the head by a crease. They are typically light to dark grey in color with a white underside (Jefferson et al., 1993). Whistles produced by bottlenose dolphins can vary over geographic regions, and newborns are thought to develop “signature whistles” within the first few months of their lives that are used for intraspecific communication. Whistles generally range in frequency from 300 Hz to 39 kHz with SLs between 114 and 163 dB re 1 μPa m (Erbe et al., 2017). Bottlenose dolphins also make burst-pulse sounds and echolocation clicks, which can range from a few kHz to over 150 kHz. As these sounds are used for locating and capturing prey, they are directional calls; the recorded frequency and sound level can vary depending on whether the sound was received head-on or at an angle relative to the vocalizing dolphin. SLs for burst-pulses and clicks range between 193 and 228 dB re 1 μPa m (Erbe et al., 2017). There are sufficient available data for bottlenose dolphin hearing sensitivity using both behavioral and AEP methods as well as anatomical modeling studies, which show hearing for the species is greatest between approximately 400 Hz and 169 kHz (Southall et al., 2019).

Common bottlenose dolphins are not listed under the ESA and are classified as Least Concern on the IUCN Red List (Hayes et al., 2020; IUCN, 2021). The best abundance estimate for the Western North Atlantic offshore stock is 62,851 based on recent surveys between the lower Bay of Fundy and Florida (Hayes et al., 2020). A population trend analysis for this stock was conducted using abundance estimates from 2004, 2011, and 2016, which show no statistically significant trend (Hayes et al., 2020). The PBR for this stock is 519, and the average annual human-cause mortality and serious injury from 2013 to 2017 was estimated to be 28, attributed to fishery interactions (Hayes et al., 2020). Because annual mortality does not exceed PBR, this stock is not classified as strategic under the MMPA. In addition to fisheries, threats to common bottlenose dolphins include non-fishery related human interaction; anthropogenic noise; offshore development; contaminants in their habitat; and climate-related changes in prey distribution (Hayes et al., 2020). There is no designated critical habitat for either stock in the Project Area.

RWF

Common bottlenose dolphins were reported in the RI-MA WEA in all seasons; highest seasonal abundance estimates were during the fall, summer, and spring (**Figure 3.1-13**). Kraus et al. (2016) reports the offshore stock as only be sighted in the RI-MA WEA during the summer months. The greatest concentrations of common bottlenose dolphins were observed in the southernmost portion of the RI-MA WEA study area in the fall (Kraus et al., 2016). Therefore, common bottlenose dolphins are likely to occur in the RWF.

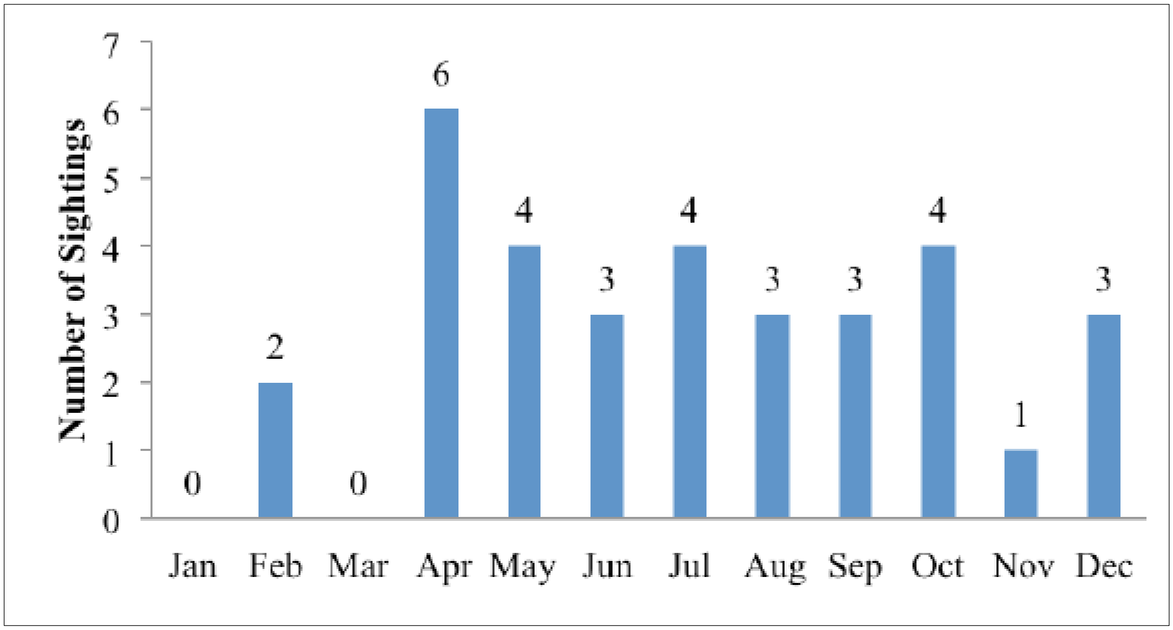

Figure 3.1-13. Visual detections of common bottlenose dolphin by month for all survey years between October 2011 and June 2015. From: Kraus et al. (2016).

RWEC

As previously discussed, common bottlenose dolphins that occur within the nearshore areas of the Project Area are likely to come from the offshore stock, despite its predominantly offshore distribution, as the seasonal stranding records match the temporal patterns of the offshore stock rather than the coastal stock (Kenney and Vigness-Raposa, 2010). Therefore, the offshore stock can be expected to occur in both the RWEC – OCS and RWEC – RI.

Harbor Porpoise

The harbor porpoise is mainly a temperate, inshore species that prefers to inhabit shallow, coastal waters of the North Atlantic, North Pacific, and Black Sea. Harbor porpoises mostly occur in shallow OCS and coastal waters. In the summer, they tend to congregate in the Northern Gulf of Maine, Southern Bay of Fundy, and around the southern tip of Nova Scotia (NMFS, 2020a). In the fall and spring, harbor porpoises are widely distributed from New Jersey to Maine (NMFS, 2020a). In the winter, intermediate densities can be found from New Jersey to North Carolina, with lower densities from New York to New Brunswick, Canada (Kenney and Vigness-Raposa, 2010). In cooler months, harbor porpoises have been observed from the coastline to deeper waters (>1,800 m), although the majority of sightings are over the continental shelf (NMFS, 2020a).

This species is among the smallest of the toothed whales and is the only porpoise species found in Northeastern U.S. waters. A distinguishing physical characteristic is the dark stripe that extends from the flipper to the eye. The rest of its body has common porpoise features; a dark gray back, light gray sides, and small, rounded flippers (Jefferson et al., 1993). Harbor porpoises produce high frequency clicks with a peak frequency between 129 and 145 kHz and an estimated SLs that ranges from 166 to 194 dB re 1 µPa m (Villadsgaard et al., 2007). Available data estimating auditory sensitivity for this species suggest that they are most receptive to noise between 300 Hz and 160 kHz (Southall et al., 2019).

This species not listed under the ESA, is listed as Least Concern by the IUCN Red List, and is considered non-strategic under the MMPA (NMFS, 2020a; IUCN, 2021). They are also not considered Endangered or Threatened by the state of Rhode Island, but they are considered a Species of Greatest Conservation Need (RI DEM, 2020). The best available abundance estimate for the Gulf of Maine/Bay of Fundy stock occurring in the Project Area is 95,543 based on combined survey data from NOAA and the Department of Fisheries and Oceans Canada between the Gulf of St. Lawrence/Bay of Fundy/Scotian Shelf and Central Virginia (NMFS, 2020a). A population trend analysis is not available because data are insufficient for this species (NMFS, 2020a). The PBR for this stock is 851, and the estimated human-caused annual mortality and serious injury from 2014 to 2018 was 150 (NMFS, 2020a). This species faces major anthropogenic effects because of its nearshore habitat. Historically, Greenland populations were hunted in large numbers for food and oil. Currently, they continue to suffer incidental mortality from Western North Atlantic fishing activities such as gillnets and bottom trawls (NMFS, 2020a). Harbor porpoises also face threats from contaminants in their habitat, vessel traffic, habitat alteration due to offshore development, and climate-related shifts in prey distribution (NMFS, 2020a). There is no designated critical habitat for this species near the Project Area.

<u>*RWF*</u>

Over the course of the study, Kraus et al. (2016) observed 121 individual harbor porpoises within the RI-MA WEA. Fall observations included the most individuals, followed by winter, spring, and summer (**Figure 3.1-14**). Vertical camera detections of all small cetaceans showed that the most commonly detected species over time was the harbor porpoise (Kraus et al., 2016). The preferred habitat of the harbor porpoise further increases the likelihood of encountering them seasonally in fall, winter, and spring within the RWF area (BOEM, 2013; NMFS, 2020a).

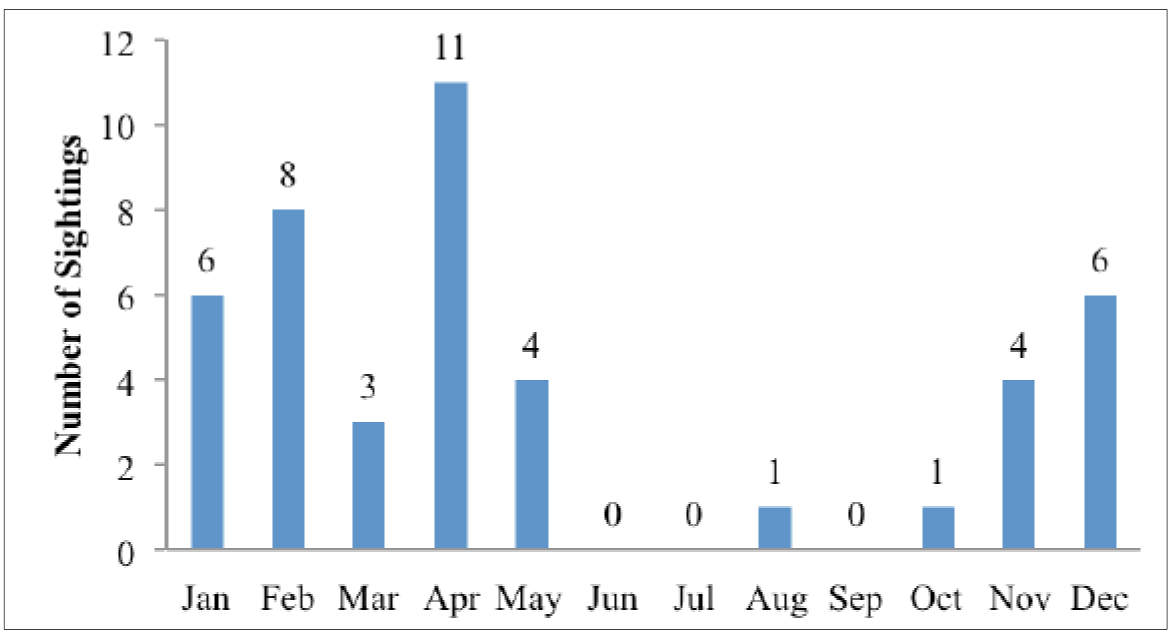

Figure 3.1-14. Visual detections of harbor porpoise by month for all survey years between October 2011 and June 2015. From: Kraus et al. (2016).

RWEC

Harbor porpoise occurrence offshore Rhode Island is highly seasonal with most sightings occurring in winter and spring and relatively few in summer and fall (Kenney and Vigness-Raposa, 2010). Strandings are reported all along the southern shore of Long Island, New York, and along both sides of Long Island Sound. They are most commonly reported in Eastern Long Island Sound, Gardiner's Bay, and Peconic Bay during the winter, west of the RWEC corridor. They have the greatest abundance in Rhode Island waters during the spring when they are known to migrate from their offshore wintering habitat in the mid-Atlantic to their summer feeding grounds in the Gulf of Maine (Kenney and Vigness-Raposa, 2010). Therefore, harbor porpoises are likely to occur within both the RWEC – OCS and RWEC – RI.

Harbor Seal

Harbor seals, also known as common seals, are one of the most widely distributed seal species in the Northern Hemisphere. They can be found inhabiting coastal and inshore waters from temperate to polar latitudes. Genetic variability from different geographic populations has led to five subspecies being recognized. Harbor seals are found in the Western Atlantic from the Mid-Atlantic U.S. to the Canadian Arctic and east to Greenland and Iceland (Rice, 1998). Peak breeding and pupping times range from February to early September, and breeding occurs in open water (Temte, 1994).

The harbor seal is one of the smaller pinnipeds, and adults are often light to dark grey or brown with a paler belly and dark spots covering the head and body (Jefferson et al., 1993; Kenney and Vigness-Raposa, 2010). Male harbor seals have been documented producing an underwater roar call which is used for competition with other males and attracting mates. These are relatively short calls with a duration of about 2 sec and a peak frequency between 1 and 2 kHz (Van Parijs et al., 2003). Behavioral audiometric studies for this species estimate peak hearing sensitivity between 100 Hz and 79 kHz (Southall et al., 2019).

Harbor seals are not listed under the ESA, are listed as Least Concern by the IUCN Red List, and are considered non-strategic because anthropogenic mortality does not exceed PBR (NMFS, 2020a; IUCN, 2021). Like the harbor porpoise they are also not listed as endangered or threatened by the state of Rhode Island but are listed as a Species of Greatest Conservation Need (RI DEM, 2020). The best available abundance estimate for harbor seals in the Western North Atlantic is 75,834, with global population estimates reaching 610,000 to 640,000 (Bjørge et al., 2010; Lowry, 2016; NMFS, 2020a). There is no population trend analysis currently available, however one is underway using 2018 survey data (NMFS, 2020a). The PBR for this population is 2,006, and the annual human-caused mortality and serious injury from 2014 to 2018 was estimated to be 365.2 seals per year. This mortality and serious injury was attributed to fishery interactions, non-fishery related human interactions, and research activities (NMFS, 2020a). Until 1972, harbor seals were commercially and recreationally hunted. Currently, only Alaska natives can hunt harbor seals for sustenance and the creation of authentic handicrafts. Other threats to harbor seals include disease and predation (NMFS, 2020a). There is no designated critical habitat for this species in the Project Area.

RWF

Harbor seals can be found along the coast of Rhode Island and the RI-MA WEA, as well as in surrounding waters. Several haul-out sites are located on Block Island, Rhode Island, which is close to the western end of the RWF area (BOEM, 2013). Survey data collected from NMFS and the Provincetown Center for Coastal Research reported 151 harbor seal sightings, a large concentration of which were observed near the coast from eastern Long Island, New York, to Buzzards Bay and Vineyard Sound. There were occurrences of harbor seal offshore; however, the level of abundance was lower than what was observed near haul-out sites (Kenney and Vigness-Raposa, 2010). Therefore, harbor seals could be potentially encountered in the RWF area.

RWEC

Harbor seals are regularly observed in coastal areas; however, there are few records from shipboard and aerial surveys. Harbor seals are difficult to detect as the only sighting cue available would be seeing the seal's head above the water. CETAP excluded seals from their data collection efforts specifically for this reason (CETAP, 1982). Most available records are of strandings and haul-out counts. Harbor seals are known to inhabit Southern New England waters year-round, although the population steadily increases in April and then abruptly declines in May.

Harbor seals are regularly observed around coastal areas throughout Rhode Island. While there are no known pupping grounds in this area, six haul-out sites have been identified in Narragansett Bay. They are most commonly observed at the Dumplings off Jamestown at Rome Point in North Kingstown, Rhode Island (Kenney and Vigness-Raposa, 2010). Nearly all the haul-outs within Narragansett Bay are rocky ledges or isolated rocks with the exception of Spar Island, which is a man-made dredge spoil (Kenney and Vigness-Raposa, 2010). Harbor seals can likely be found in the nearshore areas around the proposed RWEC corridor. Harbor seals are likely to be one of the most frequent and densely occurring marine mammal that could occur annually within both the RWEC – OCS and RWEC – RI.

Grey Seal

Gray seals inhabit temperate to sub-Arctic waters of the North Atlantic, in both nearshore and deeper OCS waters (Hall, 2002). Three different geographic populations occur; Western North Atlantic, Eastern North Atlantic, and Baltic populations (Kenney and Vigness-Raposa, 2010). Peak breeding and pupping times are January to late March, and breeding occurs in open water (Baker et al., 1995).

Gray seals are among the larger phocids found in the Western North Atlantic (Jefferson et al., 1993). Two types of underwater vocalizations have been recorded for male and female gray seals; clicks and hums. Clicks are produced in a rapid series resulting in a buzzing noise with a frequency range between 500 Hz and 12 kHz. Hums, which is described as being similar to that of a dog crying in its sleep, are lower frequency calls, with most of the energy <1 kHz (Schusterman et al., 1970). AEP studies indicate that hearing sensitivity for this species is greatest between 140 Hz and 100 kHz (Southall et al., 2019).

This species is not listed under the ESA, is listed Least Concern by the IUCN Red List, and is non-strategic because anthropogenic mortality does not exceed PBR (NMFS, 2020a; IUCN, 2021). Estimates of the entire Western North Atlantic gray seal population are not available, only estimated portions of the stock are available, although recent genetic evidence suggests that all Western North Atlantic gray seals may actually comprise a single stock (NMFS, 2020a). The best available current abundance estimate for gray seals of the Canadian gray seal stock is 424,300 and the current U.S. population estimate is 27,131 (NMFS, 2020a). The population of gray seals is likely increasing in the U.S. Atlantic EEZ; recent data show approximately 28,000 to 40,000 gray seals were observed in Southeastern Massachusetts in 2015 (NMFS, 2020a). The population trend for grey seals in the U.S. differs across all the pupping colonies, ranging from -0.2% on Green Island to 26.3% on Monomoy Island from 1988 to 2019 (NMFS, 2020a). In Canada, the total population was estimated to be increasing by 4.4% per year from 1960 to 2016. The PBR for this population is 1,389, and the annual human-caused mortality and serious injury between 2014 and 2018 was estimated to be 4,729 in both the U.S. and Canada (NMFS, 2020a). Like harbor seals, the gray seal was commercially and recreationally hunted until 1972. Mortality was attributed to fishery interactions, non-fishery related human interactions and hunting, research activities, Canadian commercial harvest, and removals of nuisance animals in Canada (NMFS, 2020a). Other threats to this population include predation, natural phenomena like storms, and disease prompting NMFS to declare a UME for pinnipeds due to phocine distemper virus in 2018 (NMFS, 2020a,b). There is no designated critical habitat for this species in the Project Area.

RWF

Overall, individuals within the RWF are relatively low; occasionally young pups have been found stranded off Long Island, New York, and Rhode Island beaches. The AMAPPS surveys identified 11 individuals during their winter aerial surveys (Palka et al., 2017). Two breeding and pupping grounds are located in Nantucket Sound at Monomoy and Muskeget Island. Gray seals live there year-round and exhibit minimal migration patterns; however, recent tagging studies observed increased movement between the U.S. and Canada. The overall time spent in U.S. waters remains uncertain, but the updated U.S. population estimates make it possible that these seals will be seen around the RWF area (NMFS, 2020a).

RWEC

Historically, gray seals were relatively absent from Rhode Island and nearby OCS waters. However, with the recent recovery of the Massachusetts and Canadian populations, their occurrence has increased in Southern New England and the Mid-Atlantic U.S. (Kenney and Vigness-Raposa, 2010). Records of gray seal strandings are primarily observed in the spring and are distributed broadly along ocean-facing beaches in Long Island, New York, and Rhode Island. In New York, gray seals are typically seen alongside harbor seal haul-outs. Two frequent sighting locations include Great Gull Island and Fisher's Island, New York (Kenney and Vigness-Raposa, 2010). Even though sightings are not as frequent as harbor seals, gray seals do occur in Rhode Island waters; therefore, these seals may be present in both the RWEC – OCS and RWEC – RI.

3.2 Sea Turtles

Four sea turtle species could potentially be present in the Project Area: green sea turtles (*Chelonia mydas*), Kemp's Ridley sea turtles (*Lepidochelys kempii*), loggerhead sea turtles, and leatherback sea turtles (*Dermochelys coriacea*). Regional Kemp's ridley and leatherback sea turtle populations are listed as Endangered under the ESA, while the green and loggerhead populations are listed as Threatened (**Table 3.2-1**). Densities for sea turtles are available from the U.S. Navy OPAREA Density Estimate database on the Strategic Environmental Research and Development Program Spatial Decision Support System (Department of the Navy, 2007, 2012) and Northeast Large Pelagic Survey Collaborative Aerial and Acoustic Surveys for Large Whales and Sea Turtles (Kraus et al., 2016) for Kemp's Ridley, loggerhead, and leatherback sea turtles for spring, summer, fall, and winter.

Table 3.2-1. Sea turtles with geographic ranges that include the Northeastern U.S. region, and the relative occurrence in the Project Area.

Common Name	Scientific Name	Stock	Current Population Status	Relative Occurrence in the RWF	Relative Occurrence in the RWEC – OCS	Relative Occurrence in the RWEC – RI
Green sea turtle	*Chelonia mydas*	North Atlantic DPS	ESA Threatened RI State Endangered	Uncommon	Uncommon	Uncommon
Kemp's Ridley sea turtle	*Lepidochelys kempii*	-	ESA Endangered RI State Endangered	Uncommon	Regular	Regular
Loggerhead sea turtle	*Caretta*	Northwest Atlantic Ocean DPS	ESA Threatened RI State Endangered	Common	Common	Common
Leatherback sea turtle	*Dermochelys coriacea*	-	ESA Endangered RI State Endangered	Common	Common	Common

DPS = Distinct Population Segment; ESA = Endangered Species Act; Project Area = includes the Revolution Wind Farm (RWF), Revolution Wind Export Cable (RWEC) – Outer Continental Shelf (OCS) and RWEC – Rhode Island (RI) state waters, and Onshore Facilities.
[1]Information based on available survey data for the region and the Wind Energy Area where Project will be located.

Sea turtle life history stages are similar in all species and include eggs, hatchling, juvenile, and adult stages. In general, sea turtles nest in tropical, subtropical, and warm-temperate beaches (Davenport, 1997). In the U.S., common nesting colonies are located in the Gulf of Mexico and Western South Atlantic Ocean; however, specific nesting distributions are described in the species-specific discussions that follow. Females mate in nearshore waters and then lay their eggs on the beach. Hatchling sea turtles move offshore in a swimming frenzy immediately after hatching (Davenport, 1997). At the surface-pelagic juvenile stage, sea turtles move to convergence zones or to *Sargassum* spp. mats and undergo passive oceanic migrations (Witherington et al., 2012). Juvenile sea turtles actively recruit to nearshore nursery habitats and move into adult foraging habitats when approaching sexual maturity. At maturity, sea turtles return to their natal beaches to lay their eggs (Davenport, 1997).

The following subsections summarize data on the status and trends, distribution and habitat preferences, behavior, and life history of sea turtles that may be found in the Project Area as available in published literature and reports, including USFWS species fact sheets.

3.2.1 Green Sea Turtle

Green sea turtles have a worldwide distribution and can be found in both tropical and subtropical waters (NatureServe, 2019; NMFS and USFWS, 1991). In the Western North Atlantic Ocean, they can be found from Massachusetts to Texas, as well as in waters off Puerto Rico and the U.S. Virgin Islands (NMFS and USFWS, 1991). Depending on the life stage, green sea turtles inhabit high-energy oceanic beaches, convergence zones in pelagic habitats, and benthic feeding grounds in shallow protected waters (NMFS and USFWS, 1991). Green sea turtles are known to make long-distance migrations between their nesting and feeding grounds. Hatchlings occupy pelagic habitats and are omnivorous. Juvenile foraging habitats include coral reefs, emergent rocky bottoms, *Sargassum* spp. mats, lagoons, and bays (USFWS, 2018a). Once mature, green sea turtles leave pelagic habitats and enter benthic foraging grounds, primarily feeding on seagrasses and algae (Bjorndal, 1997).

Major green sea turtle nesting beaches occur on Ascension Island, Aves Island, Costa Rica, and Suriname. In the U.S., green sea turtles nests in North Carolina, South Carolina, Georgia, Florida, the U.S. Virgin Islands, and Puerto Rico (USFWS, 2018a). Nesting seasons vary by region. On average, individual females nest every 2 to 4 years, laying an average of 3.3 nests per season at approximately 13-day intervals. The average clutch size is approximately 136 eggs and incubation ranges from 45 to 75 days (USFWS, 2018a).

Bartol and Ketten (2006) measured the AEPs of two Atlantic green sea turtles and six sub-adult Pacific green sea turtles. Sub-adults were found to respond to stimuli between 100 and 500 Hz, with a maximum sensitivity of 200 and 400 Hz. Juveniles responded to stimuli between 100 and 800 Hz, with a maximum sensitivity between 600 and 700 Hz. Piniak et al. (2016) confirmed similar levels, as juvenile green sea turtles responded to underwater stimuli between 50 and 1,600 Hz with maximum sensitivity between 200 and 400 Hz. Dow Piniak et al. (2012a) found that the AEPs of juvenile green sea turtles were between 50 and 1,600 Hz in water and 50 and 800 Hz in air; with ranges of maximum sensitivity between 50 and 400 Hz in water and 300 and 400 Hz in air.

There are 11 listed DPSs for green sea turtles, all of which are listed as Threatened or Endangered. The North Atlantic DPS, which is likely to occur in the Project Area, was listed as Threatened in 1978 (NMFS, 2020c). The global population is listed as Endangered under the IUCN Red List (IUCN, 2021). They are also listed as endangered by the state of Rhode Island (RI DEM, 2020). Worldwide, green sea turtle populations have declined due to past harvesting for eggs and meat (USFWS, 2018a). Currently, major risks to green sea turtles include loss of nesting and foraging habitat, nest predation, marine pollution, vessel strikes, and anthropogenic activity such as offshore dredging or fishing (USFWS, 2018a). Critical habitat was designated by NMFS for the green sea turtles in 1998 in the coastal waters of Culebra Island, Puerto Rico, and its outlying Keys (USFWS, 2018a). There is no designated critical habitat for green sea turtles in the Project Area.

RWF

There are few records of green sea turtle sightings in the RWF area. Only one confirmed sighting was reported in March 2005 south of Long Island, New York, between the 40- and 50-m isobaths (Kenney and Vigness-Raposa, 2010). NOAA's Northeast Fisheries Science Center conducted a combination of AMAPPS along the Northeast U.S. Coast from 2010 through 2015 (Palka et al., 2017). Survey waters spanned from Cape May, New Jersey, to the mouth of the Gulf of St. Lawrence, Canada. Out of five surveys that were conducted, green sea turtles were spotted only during 2010 and 2011. Six individuals were sighted south of Long Island, New York, and within the Nantucket Shoals during summer aerial surveys (17 August through 26 September 2010). Five green sea turtles were also sighted off the southern coast of Long Island, New York, during the summer aerial surveys (7 August through 26 August 2011) (Palka et al., 2017).

Digital aerial surveys conducted by the New York State Energy Research and Development Authority (NYSERDA) to gather baseline data on birds, marine mammals, turtles, and fish reported only one green sea turtle during summer 2016 surveys, and no confirmed green sea turtle sightings have been reported during 2017 or 2018 surveys (Normandeau and APEM, 2019). Based on the available sighting information of green sea turtles in this region, their occurrence would be infrequent in the RWF.

RWEC

In Southern New England, green sea turtles are known to occur in the waters around Cape Cod Bay and Block Island and Long Island Sounds (CETAP, 1982). In 2005, there was one confirmed green sea turtle sighting southwest of the RWEC corridor offshore Long Island, New York (Kenney and Vigness-Raposa, 2010). Stranding data from NMFS Sea Turtle Stranding and Salvage Network indicate that only two green sea turtles have been found stranded on Rhode Island between 2000 and 2018 (NMFS, 2019a). This species is considered uncommon in both the RWEC – OCS and RWEC – RI, and if they were to occur, it would primarily be during summer months as water temperature is a limiting factor in their distribution (BOEM, 2013).

3.2.2 Kemp's Ridley Sea Turtle

Kemp's ridley sea turtles occur off the coast of the Gulf of Mexico and along the U.S. Atlantic Coast (Turtle Expert Working Group [TEWG], 2000). Juveniles inhabit the U.S. Atlantic Coast from Florida to the Canadian Maritime Provinces. In late fall, Atlantic juveniles/sub adults travel northward to forage in the coastal waters off Georgia through New England, then return southward for the winter (New York State Department of Environmental Conservation [NYSDEC], 2019; Stacy et al., 2013). Preferred habitats include sheltered areas along the coastline, including estuaries, lagoons, and bays (NMFS, 2020d). Sixty percent of Kemp's ridley nesting occurs on beaches near Rancho Nuevo, Tamaulipas, Mexico. The nesting season spans from April through July (NMFS and USFWS, 2007). On average, individual females nest every 1 to 2 years, with an average of 1 to 3 clutches every season and an average clutch size of 110 eggs per nest (NMFS and USFWS, 2007).

Data are limited on Kemp's ridley hearing capability; however, available studies show that all sea turtle species can likely detect lower frequency noises below approximately 1 to 2 kHz. Generally, sea turtle hearing is thought to more closely resemble that of fish rather than marine mammals given their inner ear morphology and the lower frequency ranges over which sea turtle hearing has been reported (Bartol and Ketten, 2006; Dow Piniak et al., 2012a; Martin et al., 2012; Popper et al., 2014).

The Kemp's ridley sea turtle was listed as Endangered under the ESA throughout its range in 1970, and is currently listed as Critically Endangered under the IUCN Red List (IUCN, 2021; NMFS, 2020d). They are also listed as endangered by the state of Rhode Island (RI DEM, 2020). The decline in global kemp's ridley populations is the result of human activity, such as harvesting adults and eggs for food and as fisheries bycatch (USFWS, 2018b). There is no designated critical habitat for this species in the Project Area (NMFS, 2020d).

RWF

Kemp's ridley sea turtles are more common in the New York Bight region and along the Long Island, New York, coastline; there are few visual sighting data for Kemp's ridley sea turtles in the RWF (Normandeau and APEM, 2019). This could be partly be due to Kemp's ridley sea turtles' small size, which makes them difficult to detect during aerial surveys. AMAPPS surveys documented five Kemp's ridley sea turtles during aerial surveys conducted from August through September, 2010, in waters from Cape May, New Jersey, to the Gulf of St. Lawrence, Canada. No confirmed sightings were reported from 2011 through 2014 (Palka et al., 2017). Kraus et al. (2016) detected Kemp's ridley sea turtles in the RI-MA WEA using vertical camera photographs. However, only four photographic detections were confirmed in 2012 (Kraus et al., 2016). Kenney and Vigness-Raposa (2010) reported 14 observations of Kemp's ridley offshore Rhode Island around Block Island in the summer and fall. Given the available data for Kemp's ridley turtle presence in the RI-MA WEA, it is not likely that they would be encountered in the RWF area.

RWEC

Kemp's ridley sea turtles that occur in Southern New England can be seen in Long Island Sound, along the Rhode Island coastline, and in Cape Cod Bay (CETAP, 1982; Waring et al., 2012). Beginning in July, Kemp's ridley turtles begin inhabiting the Long Island Sound area. To date, all Kemp's Ridley turtles encountered in Long Island Sound have been juveniles. Between July and early October, juveniles occupy estuarine waters of the Long Island Sound, Peconic Bay, and other bays along the south shore of Long Island, New York. During this time, growth rates increase by approximately 25% per month, indicating that these waters provide an abundant food source for these turtles. The Long Island Sound has not been formally identified as critical habitat; however, research has inferred that this area could potentially provide a critical coastal developmental habitat for immature Kemp's ridley sea turtles during the early turtle life stages (2 to 5 years) (Morreale et al., 1992; NYSDEC, 2019). The main characteristics of developmental habitats are coastal areas sheltered from high winds and waves such as embayments, estuaries, and nearshore temperate waters shallower than 50 m (NMFS, 2020d).

In October, Kemp's ridley sea turtles begin to migrate out of the estuaries and back into pelagic environments. If they do not migrate out by late November, they are likely to become cold-stunned. There are many records of cold-stunned Kemp's ridley sea turtles washing ashore on Long Island, New York (Burke et al., 1993). Cold-stunned Kemp's ridley sea turtles are often found stranded on beaches of Rhode Island and Massachusetts beginning in autumn when water temperatures drop below 50°F (Stacy et al., 2013). However, strandings are more common in Massachusetts; 929 reported Kemp's ridleys between 2000 and 2018 along Massachusetts coasts versus only 8 reported for Rhode Island (NMFS, 2019a). Therefore, Kemp's ridley sea turtles may be present in low numbers in the RWEC - OCS and RWEC – RI in the spring and summer.

3.2.3 Loggerhead Sea Turtle

Loggerhead sea turtles have a worldwide distribution and inhabit temperate and tropical waters, including estuaries and continental shelves of both hemispheres. Five populations of loggerhead sea turtles exist worldwide in the Atlantic Ocean, Pacific Ocean, Indian Ocean, Caribbean Sea, and Mediterranean Sea. In the Western Atlantic Ocean, the five major nesting aggregations are: (1) a northern nesting aggregation from North Carolina to northeast Florida, approximately 20° N latitude; (2) a south Florida nesting aggregation from 29° N latitude on the east coast to Sarasota on the west coast; (3) a Florida Panhandle nesting aggregation at Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán nesting aggregation on the eastern Yucatán Peninsula, Mexico; and (5) a Dry Tortugas nesting aggregation on the islands of the Dry Tortugas, near Key West, Florida (TEWG, 2000).

Female loggerhead sea turtles mate from late April through early September. Individual females might nest several times within one season and usually nest at intervals of every 2 to 3 years. For their first 7 to 12 years, loggerhead sea turtles inhabit pelagic waters near the North Atlantic Gyre and are called pelagic immatures. When loggerhead sea turtles reach 40 to 60 cm straight-line carapace length, they begin recruiting to coastal inshore and nearshore waters of the continental shelf through the U.S. Atlantic and Gulf of Mexico and are referred to as benthic immatures. Benthic immature loggerheads have been found in waters from Cape Cod, Massachusetts, to southern Texas. Loggerhead sea turtles forage off the Northeastern U.S. and migrate south in the fall as temperatures drop. Most recent estimates indicate that the benthic immature stage ranges from ages 14 to 32 years and they mature at around ages 20 to 38 years. Prey species for omnivorous juveniles include crab, mollusks, jellyfish, and vegetation at or near the surface. Coastal subadults and adults feed on benthic invertebrates, including mollusks and decapod crustaceans (TEWG, 2000).

Based on Bartol et al. (1999), juvenile loggerhead sea turtles respond to click stimuli from tone bursts of 250 to 750 Hz. Martin et al. (2012) recorded the AEPs of one adult loggerhead sea turtle, which responded to frequencies between 100 and 1,131 Hz, with greatest sensitivity between 200 and 400 Hz.

There are nine listed DPSs for loggerhead sea turtles; the Northwest Atlantic Ocean DPS, which occurs in the Project Area, was listed as Threatened in 2011 (NMFS, 2020e). The global population is listed as Vulnerable by the IUCN Red List (IUCN, 2021). They are also listed as endangered by the state of Rhode Island (RI DEM, 2020). Major threats to this population include loss of nesting and foraging habitat, nest predation, marine pollution, vessel strikes, disease, and fisheries bycatch (USFWS, 2018c). In 2014, NMFS designated critical habitat for the Northwest Atlantic Ocean DPS in multiple locations along the U.S. East Coast and in the Gulf of Mexico. These areas include *Sargassum* spp. habitat, nearshore reproductive habitat, overwintering areas, breeding habitat, and migratory corridors located between North Carolina and Florida in the Atlantic Ocean (79 *FR* 39855). No designated critical habitat exists in the Project Area.

RWF

Loggerhead sea turtles are frequently seen in waters off the coast of Rhode Island, Massachusetts, and New York. AMAPPS surveys reported loggerhead sea turtles as the most commonly sighted sea turtles on OCS waters from New Jersey to Nova Scotia, Canada. During the December 2014 to March 2015 aerial abundance surveys, 280 individuals were recorded (Palka et al., 2017). Kraus et al. (2016) reported that loggerhead occurrence in the RI-MA WEA was highest during August and September (**Figure 3.2-1**). Across all four survey years, there were 27 sightings in August and 45 sightings in September within the RI-MA WEA. During the NYSERDA Digital Aerial Baseline Surveys, sightings were dispersed across the continental shelf offshore Long Island past Montauk, New York, and there were 649 loggerhead detections during summer 2017 surveys. Fewer individuals were observed during fall surveys, and no turtles were detected during winter surveys (Normandeau and APEM, 2019).

Because of their documented occurrence, it is likely that loggerhead sea turtles could occur within the RWF area during the summer and fall. However, it is unlikely there would be a high concentration of turtles within the RWF, because most of these observations were reported as single sightings widely distributed throughout the RI-MA WEA (Kraus et al., 2016; Palka et al., 2017).

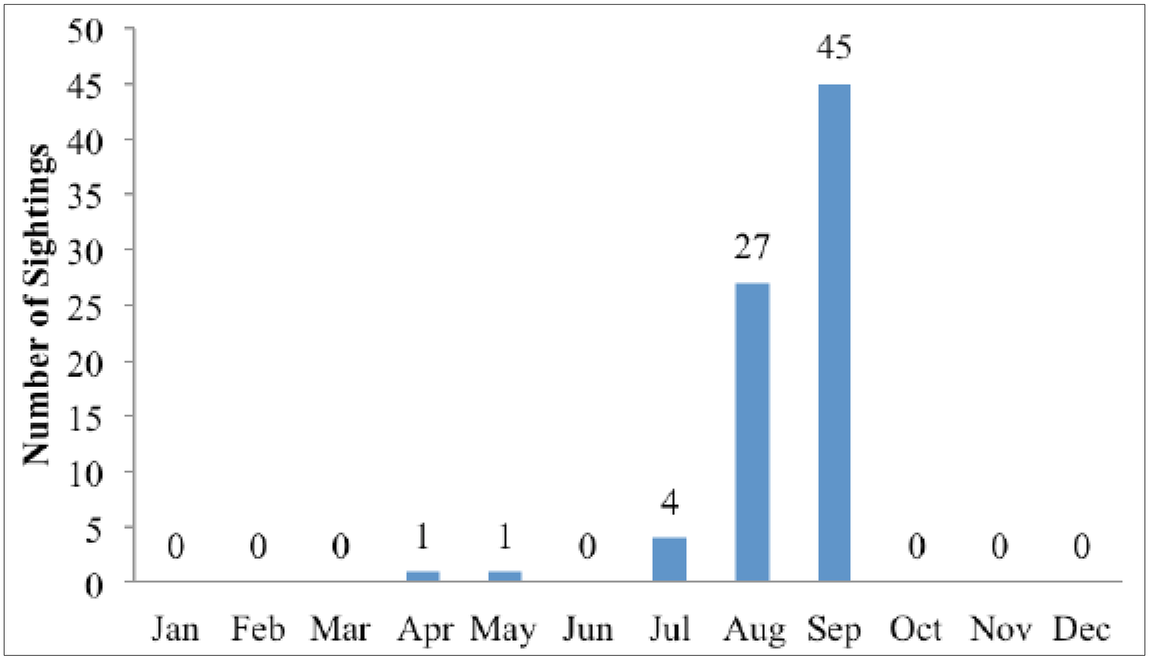

Figure 3.2-1. Visual detections of loggerhead sea turtle by month for all survey years between October 2011 and June 2015. From: Kraus et al. (2016).

<u>RWEC</u>

Loggerhead sea turtles are commonly seen off the coasts of New York and Rhode Island. CETAP conducted extensive aerial surveys from 1978 through 1982 along the coast from Cape Hatteras, North Carolina to Long Island, New York. Many loggerhead sea turtles were sighted along the continental shelf waters between Cape Hatteras, North Carolina, and Long Island, New York. A high density of loggerhead sea turtles was seen near the shore of central Long Island, New York. Loggerhead sea turtles show a northern limit at approximately 41° N latitude (CETAP, 1982), and few sightings were reported past that northern limit (Shoop and Kenney, 1992). Loggerheads are most commonly seen in June, they then begin to decrease until October as they migrate to warmer waters (Shoop and Kenney, 1992). Turtles that fall behind may succumb to cold-stunning, which usually occurs during the fall when water temperatures begin to fall. Between 1986 and 1988, 28 cold-stunned turtles were stranded in eastern Long Island, New York (Kenney and Vigness-Raposa, 2010), and recent stranding data from NMFS reported 68 loggerhead strandings in Rhode Island between 2000 and 2018 (NMFS, 2019a). Loggerhead sea turtle occurrence within both the RWEC – OCS and RWEC – RI is therefore expected to be relatively common.

3.2.4 Leatherback Sea Turtle

The leatherback sea turtle is primarily a pelagic species and is distributed in temperate and tropical waters worldwide. The leatherback is the largest, deepest diving, most migratory, widest ranging, and most pelagic of the sea turtles (NMFS, 2020f). In 2017, NMFS received a petition to identify the Northwest Atlantic subpopulation as a DPS and list it as Threatened under the ESA. In response to this petition, NMFS initiated a status review for the leatherback sea turtle to review the new information available since the original listing (82 FR 57565). This change has not yet been adopted so the global population listing remains as-is for this species. Adult leatherback sea turtles forage in temperate and subpolar regions in all oceans. Jellyfish are the major component of the leatherback diet; they are also known to feed on sea urchins, squid, crustaceans, tunicates, fish, blue-green algae, and floating seaweed (USFWS, 2018d; NMFS, 2020f).

Historically, the most important nesting ground for the leatherback was the Pacific coast of Mexico. However, because of exponential declines in leatherback nesting, French Guiana in the Western Atlantic now has the largest nesting population. Other important nesting sites for the leatherback include Papua New Guinea, Papua-Indonesia, and the Solomon Islands in the Western Pacific. In the U.S., nesting sites include the Florida east coast; Sandy Point, U.S. Virgin Islands; and Puerto Rico. U.S. nesting occurs from March through July. On average, individual females nest every 2 to 3 years, laying an average of 5 to 7 nests per season with an average clutch size of 70 to 80 eggs. Critical habitat has been designated for the leatherback sea turtle in the U.S. Virgin Islands at Sandy Point Beach, St. Croix, and the water adjacent to Sandy Point Beach (44 *FR* 17710).

Dow Piniak et al. (2012b) found that hatchling leatherback sea turtles responded to stimuli between 50 and 1,200 Hz in water and 50 and 1600 Hz in air. The maximum sensitivity was between 100 and 400 Hz in water and 50 and 400 Hz in air.

The leatherback sea turtle has been federally listed as Endangered under the ESA since 1970 and is considered Vulnerable by the IUCN Red List (IUCN, 2021; NMFS, 2020f). They are also listed as endangered by the state of Rhode Island (RI DEM, 2020). Threats to this population include fisheries bycatch, habitat loss, nest predation, and marine pollution (USFWS, 2018d). Critical habitat for this species was designated in waters adjacent to Sandy Point Beach, U.S. Virgin Islands in 1979 (44 *FR* 17710) and along the U.S. West Coast between Point Arena and Point Arguello, California, and between Cape Flattery, Washington, and Cape Blanco, Oregon, in 2012 (77 *FR* 4169).

<u>*RWF*</u>

Leatherback sea turtles were the most frequently sighted turtle species by Kraus et al. (2016) in the RI-MA WEA and were mostly observed from May through November (**Figure 3.2-2**). Leatherback sea turtles are rarely detected in the spring and not detected at all during the winter. A strong peak in leatherback sea turtle sightings is seen during August, with 71 reported sightings from Kraus et al. (2016). In the fall, there is a high concentration of sightings south of Nantucket, Massachusetts (Kraus et al., 2016). NYSERDA reported one leatherback in the RI-MA WEA during fall 2016 aerial surveys. While there were a few detections in the New York Bight region, none were detected offshore Rhode Island near the RWF during summer 2016 surveys (Normandeau and APEM, 2019). The AMAPPS surveys reported four leatherback sea turtle sightings during the summer 2011 shipboard abundance surveys (Palka et al., 2017). Because of the documented occurrence and use of Southern New England waters and within the vicinity of the RI-MA WEA, it is likely that leatherback sea turtles could occur in the RWF area during the summer and fall months. However, it is unlikely that large concentrations of these animals would be found in the RWF because observations show that their distribution is widespread, and the only concentrated occurrence was documented south of Nantucket, Massachusetts, east of the RWF.

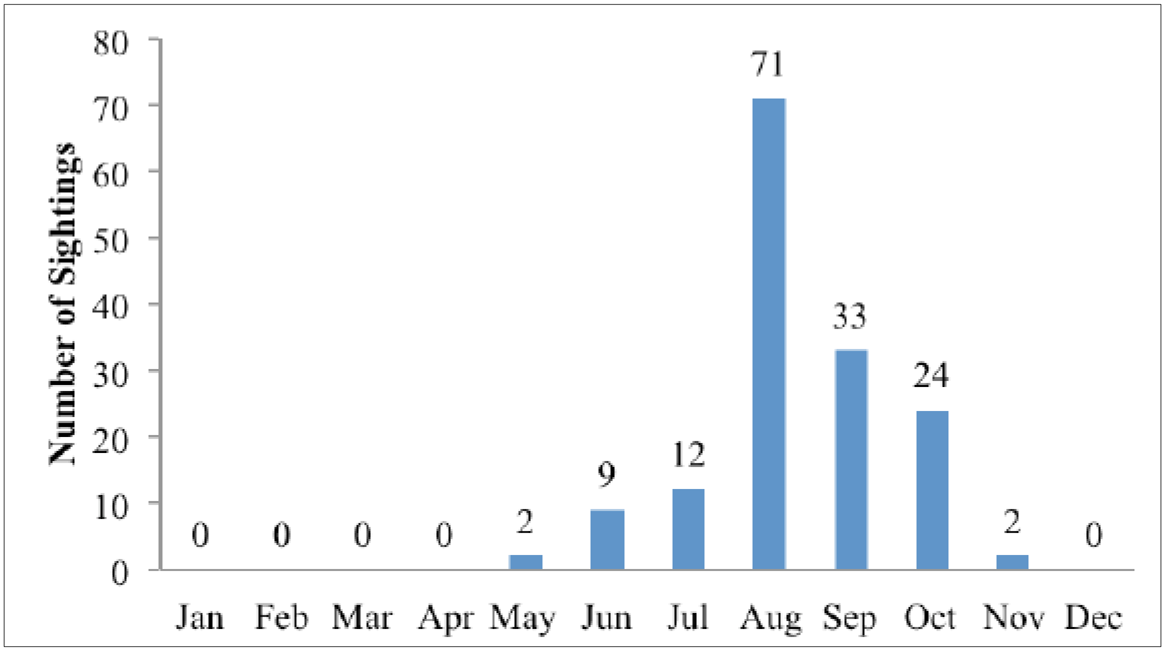

Figure 3.2-2. Visual detections of leatherback sea turtle by month for all survey years between October 2011 and June 2015. From: Kraus et al. (2016).

RWEC

Leatherback sea turtle strandings on U.S. shores are mostly of adult or near-adult size turtles (NMFS and USFWS, 1992). In relation to species occurrence, leatherback sea turtle sightings generally are fewer in number compared to loggerheads and Kemp's ridleys. Leatherback sea turtle distribution is similar to loggerhead sea turtles with occurrences from Cape Hatteras, North Carolina, to Long Island, New York, but leatherbacks are more frequently observed in the Gulf of Maine, southwest of Nova Scotia, Canada. Boaters fishing within 10 miles (16 km) of the south shore of Long Island, New York, frequently report leatherback sightings (NMFS and USFWS, 1992). Aggregations of leatherback sea turtles have been observed around Block Island, Rhode Island, and south of Long Island, New York, and strandings of this species are relatively common in Rhode Island (Kenney and Vigness-Raposa, 2010; NMFS, 2019a). Between 2000 and 2018, NMFS reported 76 leatherback sea turtle strandings in Rhode Island, the highest of the four expected sea turtle species (NMFS, 2019a). Leatherback sea turtle occurrence in both the RWEC – OCS and RWEC – RI is therefore expected to be common.

3.3 ESA-Listed Fish Species

There are three ESA-listed fish species that could potentially occur within the shelf and coastal waters of the Western North Atlantic: Atlantic sturgeon (*Acipenser oxyrhynchus*), shortnose sturgeon (*Acipenser brevirostrum*), and giant manta ray (*Mobula birostris*) (**Table 3.3-1**). These three species are listed as Endangered under the ESA so further detail is provided on their distribution, behavior, and relevant life history traits in this report.

While all three species have ranges that include the Project Area, the Atlantic sturgeon is the only species whose occurrence is common enough that they are at risk of potential impacts from Project Activities. Therefore, only this species is included in the impact assessment (**Section 5.0**). Species information and justification for excluding the shortnose sturgeon and giant manta ray from this assessment are provided in the following sections.

Table 3.3-1. Protected fish species that could potentially occur in the Project Area and their relative occurrence in the Project Area.

Common Name	Scientific Name	Stock	Federal ESA Status	Relative Occurrence in the RWF	Relative Occurrence in the RWEC – OCS	Relative Occurrence in the RWEC – RI
Atlantic sturgeon	*Acipenser oxyrhynchus*	NY Bight DPS	Endangered RI State Historical	Common	Common	Common
Shortnose sturgeon	*Acipenser brevirostrum*	-	Endangered	Rare	Rare	Rare
Giant manta ray	*Mobula birostris*	-	Endangered	Rare	Rare	Rare

DPS = Distinct Population Segment; ESA = Endangered Species Act Project Area = includes the Revolution Wind Farm (RWF), Revolution Wind Export Cable (RWEC) – Outer Continental Shelf (OCS) and RWEC – Rhode Island (RI) state waters.
[1]Information based on finfish assessment conducted in Section 4.3.3 and the Essential Fish Habitat Assessment (Inspire Environmental, 2020) provided with the Revolution Wind Construction and Operations Plan.

3.3.1 Atlantic Sturgeon

Atlantic sturgeon are found from Canada to Florida in estuarine habitats and rivers as well as in coastal and shelf marine environments. Subadults move out to estuarine and coastal waters in the fall; and adults inhabit fully marine environments and migrate through deep water when not spawning (Atlantic Sturgeon Status Review Team [ASSRT], 2007). The most recent status review for the Atlantic sturgeon was conducted in 2007. In this review, commercial bycatch was assessed, which showed that the majority (61%) of tagged sturgeon recaptures came from ocean waters within 4.8 km of shore, with the lowest ocean bycatch occurring in the summer months (July to September) (ASSRT, 2007). Atlantic sturgeon occurring within the Project Area are part of the New York Bight DPS. The Atlantic Sturgeon benchmark (SAR) (Atlantic States Marine Fisheries Commission [ASMFC], 2017) indicates that all DPS stocks are depleted but recovering. It is estimated that biomass and abundance are currently higher than that in 1998 (last year of available survey data) for the New York Bight DPS (75% average probability). The estimated abundance of age-0 to -1 Atlantic sturgeon in the Delaware River in 2014 was 3,656 individuals (Hale et al., 2016), which is similar to the age-1 estimate of 4,314 for the Hudson River in 1995 (Peterson et al., 2000). Similar estimates from the 2007 status review suggest that the Hudson River population consists of approximately 4,600 wild juveniles with a spawning stock of 870 adults.

The Atlantic sturgeon is a large (up to 4 m long), long-lived, anadromous fish that feeds on benthic invertebrates (NMFS, 2020g). Their primary hearing range falls within lower frequencies (under approximately 1 kHz), and while they do have a swim bladder, it is not involved in hearing (Popper et al., 2014).

NMFS listed the New York Bight DPS as Endangered in 2012 (77 *FR* 5879) and the critical habitat designation was finalized in 2017 (82 *FR* 3916). The IUCN lists the Atlantic sturgeon as Near Threatened (IUCN, 2021) and the Convention on International Trade in Endangered Species of Wild Fauna and Flora lists the species under *Appendix II*, which lists species that are not necessarily now threatened with extinction, but that may become so unless trade is closely controlled. Current threats to Atlantic sturgeon within critical habitat include dams and turbines, dredging, water quality, and climate change. There is critical habitat designated for the New York Bight DPS within the Connecticut, Housatonic, Hudson, and Delaware Rivers, but no offshore critical habitat designation.

RWF

Historically, this population of Atlantic sturgeon spawned in several rivers between Massachusetts and the Chesapeake Bay; currently, however, the New York Bight DPS is known to consistently spawn only within the Hudson and Delaware rivers between April and May (ASSRT, 2007). During the spring and early summer, adult Atlantic sturgeon travel upstream in spawning rivers along Southern New England and New York. Throughout the rest of the year, spawning age adults can be found in both coastal and offshore waters in this region (ASMFC, 1990). Using commercial bycatch data, Stein et al. (2004) reported numerous juvenile and adult Atlantic sturgeon caught in waters offshore Massachusetts and Rhode Island near the RWF, and therefore they can be expected to occur in the RWF area, with a peak presence between November and May.

<u>RWEC</u>

Atlantic sturgeon are not likely to use any rivers in Narragansett Bay, Rhode Island for spawning; therefore, while their occurrence within the RWEC – OCS and RWEC – RI could be expected, it would be less than that expected in the RWF area.

3.3.2 Shortnose Sturgeon

Much of the distribution information is the same for the two sturgeon species, which co-occur in habitats along the U.S. Atlantic coast. Shortnose sturgeon occurring in the Project Area are from the Northeast spawning population encompassing the Connecticut, Hudson, and Delaware Rivers.

Morphologically, the shortnose sturgeon is smaller overall with a less pronounced snout than other sturgeon species, but their hearing capabilities would be similar to those described for the Atlantic sturgeon (**Section 3.3.1**). Like the Atlantic sturgeon, the shortnose sturgeon is listed as Endangered under the ESA but is classified as Vulnerable on the IUCN Red List (IUCN, 2021; NMFS, 2020h).

<u>RWF</u>

In a 2010 Biological Assessment (Shortnose Sturgeon Status Review Team, 2010), shortnose sturgeon were described as spending less time in open ocean habitats and spawning farther upriver than Atlantic sturgeon. The Northeast spawning population in particular uses freshwater habitats more than any of the other shortnose sturgeon populations (Kynard et al., 2016). They are considered more of an amphidromous species (defined as a species that spawns and remains in freshwater for most of its lifecycle but spends some time in saline water) rather than fully anadromous. Marine migrations do occur, and individuals have been recorded traveling 140 km in 6 days when moving between rivers (Kynard et al., 2016). However, because of the shortnose sturgeon proclivity to freshwater and estuarine habitats, the potential for shortnose sturgeon to be present in both the RWF area would be considered rare.

<u>RWEC</u>

As described for the RWF, this species' preference for freshwater habitat and the fact that primary spawning rivers are located in New York and Connecticut make it unlikely that this species will occur in either the RWEC – OCS or RWEC – RI.

3.3.3 Giant Manta Ray

The giant manta ray occurs in tropical, sub-tropical, and temperate waters (NMFS, 2020i). Their distribution in the Atlantic ranges from the Carolinas to Brazil and they are very rarely found in colder waters of the Western North Atlantic. Giant manta rays undergo seasonal migrations, which are thought to coincide with the movement of zooplankton, current circulation and tidal patterns, seasonal upwelling, seawater temperature, and possibly mating behavior. The giant manta ray is a seasonal visitor to productive coastlines, oceanic island groups, and offshore pinnacles and seamounts. They are generally found at depths below 10 m and tagging studies indicate dives of up to 200 to 450 m (NMFS, 2020i). They are slow-growing, highly migratory animals with sparsely distributed and fragmented populations throughout the world. Giant manta rays may reach disc widths of over 7 m (NMFS, 2020i). Regional population sizes are small (between 100 to 1,500 individuals) (Marshall et al., 2018; NMFS, 2020i).

The giant manta ray is listed as Threatened under the ESA and Vulnerable on the IUCN Red List (IUCN, 2021; NMFS, 2020i). Commercial fishing is the primary threat to the giant manta ray (NMFS, 2020i) as it is targeted and caught as bycatch in several global fisheries throughout its range.

RWF

Giant manta rays are often observed in estuarine waters and near oceanic inlets, potentially using these habitats as nursery grounds. The giant manta ray is commonly encountered on shallow reefs and is also occasionally observed in sandy bottom areas and seagrass beds (Marshall et al., 2018). Mantas have been reported as far north as Canada in the Western North Atlantic; however, its propensity for warmer waters makes its presence unlikely in the RWF area.

RWEC

Although the giant manta ray is often observed in shallow coastal waters and estuaries, they are unlikely to occur in either the RWEC – OCS or RWEC – RI given their preference for warmer waters.

3.4 Summary

Species distribution and life history information were obtained from surveys conducted in and around the RI-MA WEA and available published literature in order to determine baseline conditions for the Project Area. This information helps determine what species are most likely to occur in the RWF and the RWEC – OCS and RWEC – RI and when they can be expected to occur. Information about their movement, behavior, feeding preferences, and reproductive characteristics help predict how vulnerable species may be to Project-related impacts, which helps determine the impact severity presented in **Sections 4.3.3.2**, **4.3.4.2**, and **4.3.5.2** of the Project's COP. Species that may occur in the Project Area include both ESA-listed Endangered and Threatened species and non-listed species. Listed species may be more vulnerable to potential population-level impacts given their lower overall abundance and thus warrants further consideration in the impact assessment process.

All 36 marine mammal species presented in **Table 3.1-1** are protected under the MMPA and have reported geographic distributions that include the Project Area. Of these species, only 15 are reasonably expected to occur in the Project Area. Four of the 15 expected species are also listed as Endangered under the ESA: the fin whale, North Atlantic right whale, sei whale, and sperm whale. The four species of sea turtle likely to occur in the Project Area are all listed as either Endangered or Threatened under the ESA. Of the three ESA-listed fish species whose ranges include the Project Area, only the Atlantic sturgeon is likely to occur in the RWF, RWEC – OCS, and RWEC – RI. The current status of these resource populations as well as the protection given to ESA- and MMPA-protected species warrants further consideration in this assessment. Using the expected distribution and known vulnerability of these species provided in the previous section, the severity of potential impacts is discussed in **Section 5.0**.

4.0 ACOUSTIC RISK ASSESSMENT

Marine mammals, sea turtles, and fish use sound for social and reproductive communication, foraging, and situational awareness which makes them susceptible to impacts from underwater noise. As discussed in **Section 2.2**, various natural and anthropogenic activities contribute to noise in the ocean creating a complex acoustic habitat. Changes in the acoustic habitat can change an animal's ability to function within its given acoustic habitat.

Marine animals can perceive underwater noise over a broad range of frequencies from about 10 Hz to more than 200 kHz, and the primary acoustic habitat for a species will be focused within their specific vocal and hearing ranges. Given the acoustic specificity of each species, noise sources present different potential impacts. Additionally, impacts will vary due to differences in the acoustic properties of the source and how it propagates through the water.

For the purposes of this acoustic assessment, noise produced by Project Activities are classified as impulsive or non-impulsive. Impulsive noises are characterized as a distinct energy pulse that has a rapid rise time and relatively high PK. Most impulsive noises are broadband and are generated by sources such as airguns, impact pile driving, and some commercial sub-bottom profilers. Non-impulsive noises do not have the characteristic energy pulse or rapid rise times seen in impulsive sources; non-impulsive sources include vessels, drilling, and vibratory pile driving (Southall et al., 2007).

Impact pile driving during Project construction is expected to pose the greatest risk of potential impact relative to other noise-producing activities. Impact pile driving could result in physiological impacts (i.e., injury in sea turtles and fish, PTS in marine mammals) for some species given the acoustic and spectral characteristics of the noise produced by the activity. However, for most noise-producing Project Activities, temporary behavioral responses by marine mammals, sea turtles, and Atlantic sturgeon are the most likely impact during construction and operation of the RWF and RWEC. The magnitude and probability of most effects generally decreases with increasing distance from a source. The potential for physiological impacts (i.e., injury, PTS,) or biologically significant behavioral impacts is further reduced by implementing active operational environmental protection measures such as use of noise mitigation systems (NMS).

The underwater acoustic analysis report (Denes et al., 2020) provides a thorough compilation of the estimated propagation distances to regulatory acoustic criteria for multiple RWF impact pile driving scenarios. Regulatory criteria are based on impact thresholds that are either regulated under the MMPA or have substantial science-based criteria and have been applied in regulatory or impact assessment under the MMPA or ESA (Fisheries Hydroacoustic Working Group [FHWG], 2008; Popper et al., 2014; Blackstock et al., 2018; NMFS, 2018, 2019b). All thresholds are based on the most current accepted threshold levels for both physiological (i.e., PTS or auditory injury) and behavioral impacts (**Section 4.1**).

For this Technical Report, noise related to Project Activities was described in detail based on Denes et al. (2020) and published literature (**Section 2.1**). A compilation of available data regarding potential impacts of underwater noise produced by sources similar to those expected during Project Activities is summarized for marine mammals, sea turtles, and Atlantic sturgeon (**Section 2.3**). Results of the underwater acoustic analysis report (Denes et al., 2020) are also summarized in this Section to further assess potential impacts that may result from Project Activities.

The following subsections provide an overview of the acoustic threshold criteria and modeling parameters used to estimate the distances to physiological and behavioral acoustic thresholds which are also summarized for reference. This information provides the basis for the impact assessment of noise-producing Project Activities (**Section 5.0**).

4.1 Acoustic Threshold Criteria

Acoustic thresholds are received sound levels that meet current scientific criteria as sufficient for eliciting the onset of a physiological effect (e.g., auditory injury, PTS) or behavioral response in a given marine species. Threshold criteria are used to identify the acoustic metrics and sound levels that may constitute an impact to a particular species and thus may require regulatory action. Acoustic threshold criteria are defined for the three faunal groups (i.e., marine mammals, sea turtles, and fish) considered in this assessment. The thresholds for each faunal group are defined with different metrics and therefore may have a different regulatory context and application.

Acoustic threshold criteria were established using two primary evaluators: 1) species' hearing sensitives; and 2) noise source characteristics. Marine mammals are divided into multiple hearing groups based on frequency-dependent hearing sensitivities (**Section 4.1.1**). Acoustic threshold criteria are the same for all sea turtle species, although there may be some distinction between hatchling and adult hearing capabilities (Lavender et al., 2014; Piniak et al., 2016) (**Section 4.1.2**). Accepted criteria for fish are dependent upon hearing mechanisms involving the swim bladder as well as the size of the fish (**Section 4.1.2**).

As discussed previously, Southall et al. (2007) identified two main types of noise sources: impulsive and non-impulsive. Non-impulsive sources can be further classified into operational categories of continuous or intermittent. Impulsive source criteria are typically presented using three metrics; PK and SEL_{24h}, which reflect the different potential exposure characteristics of the source which may cause physiological impacts; and SPL, which is used in behavioral impact assessments. Non-impulsive source criteria typically use SEL_{24h} and SPL as they do not have the characteristic peak in intensity (represented by the PK metric) that impulsive sources do. Throughout this assessment, modeling results used the most applicable physiological and behavioral threshold criterion for each affected resource for both impulsive and non-impulsive noise sources.

The noise sources of potential concern during proposed Project Activities include impact pile driving (impulsive source), geophysical surveys (both impulsive and non-impulsive sources), DP vessel thrusters, aircrafts, vibratory pile driving, and operational WTGs (non-impulsive sources). Acoustic thresholds, as defined in the following subsections, were used to establish the total ensonified area of noise received by the animal at levels that may result in either physiological or behavioral impacts, depending on the animals' hearing capability and source type.

4.1.1 Marine Mammals

Recognizing that marine mammal species do not have equal hearing capabilities, marine mammals are separated into hearing groups (Southall et al., 2007, 2019; NMFS, 2018). To account for these hearing groups, frequency weighting functions were applied when determining physiological (i.e., PTS) thresholds to scale species' sensitivities to a received noise depending on the spectral content of that noise. In effect, the sound energy contained within the frequency hearing range of an animal has the potential to affect hearing while sound energy outside an animal's frequency hearing range is unlikely to affect its hearing. The overall objective in defining hearing groups and deriving frequency weighting functions was to better define the role that frequency content plays in potential PTS.

Regulatory marine mammal hearing groups, originally identified by Southall et al. (2007) then later modified by Finneran (2016) and adopted by NMFS (2018), are categorized as LF cetaceans, mid-frequency (MF) cetaceans, HF cetaceans, phocid pinnipeds in water (PPW), and otariid pinnipeds in water (OW). Each category has a defined auditory weighting function and estimated acoustic threshold for the onset of PTS. No species from the OW hearing group (i.e., eared seals) are expected to occur in the Project Area and are not discussed further.

More recently, Southall et al. (2019) conducted a broad, structured assessment of the audiometric and physiological basis for the categorization of marine mammal hearing groups. Southall et al. (2019) kept the same frequency responses (i.e., hearing sensitivities) but re-categorized the LF, MF, and HF hearing groups to LF, HF (previously MF), and very high-frequency (VHF) (previously HF) hearing groups, and distinguished between phocid carnivores (i.e., pinnipeds) in water (PCW) and in air. Their assessment also indicated a probable distinction among baleen whales to include a very-low frequency (VLF) and a LF group, and an additional distinction among many of the odontocetes to include a distinction between an MF group containing the beaked, killer, and sperm whales and other HF cetaceans. There is insufficient evidence to support these distinctions, so the broader LF and HF hearing group categories are currently used resulting in a total of five possible groups (**Table 4.1-1**).

Southall et al. (2019) further acknowledge that there are presently insufficient direct data within the HF and VHF groups to explicitly derive distinct thresholds and weighting functions. They thus propose retaining the thresholds and functions developed by Finneran (2016) and adopted by NMFS (2018), but with slightly different categorical identifiers. The results of Southall et al. (2019) remain congruent with the current existing regulatory guidance (NMFS, 2018). A comparison of the two categorical terminologies and the general hearing ranges for each hearing group is provided in **Table 4.1-1**.

Table 4.1-1. Marine mammal hearing groups and general hearing frequency ranges as designated by the National Marine Fisheries Service (NMFS) (2018) and new hearing groups developed by Southall et al. (2019) with species that may occur in the Project Area included in each hearing group.

NMFS (2018) Hearing Group Designation and Generalized Hearing Range[1]	Southall et al. (2019) Hearing Group Designation	Species or Taxonomic Groups (species potentially occurring in the Project Area)
LF Cetacean (7 Hz to 35 kHz)	LF Cetaceans	Baleen whales (e.g., fin whale, sei whale, North Atlantic right whale, minke whale, humpback whale)
MF Cetacean (150 Hz to 160 kHz)	HF Cetaceans	Dolphins (e.g., Atlantic spotted dolphin, Atlantic white-sided dolphin, common dolphin, Risso's dolphin, common bottlenose dolphin) and toothed whales (e.g., sperm whale, long-finned pilot whale)
HF Cetacean (275 Hz to 160 kHz)	VHF Cetaceans	True porpoises (e.g., harbor porpoise)
PPW (50 Hz to 86 kHz)	PCW	True seals (e.g., harbor seal, gray seal)

HF = high-frequency; LF = low-frequency; MF = mid-frequency; PCW = phocid carnivores in water; PPW = phocid pinnipeds in water; VHF = very high-frequency.
[1]Represents the generalized hearing range for the entire group as a composite (i.e., all species within the group), where individual species' hearing ranges are typically not as broad. Generalized hearing range chosen based on an approximate 65 dB threshold from normalized composite audiogram, with the exception for lower limits for LF cetaceans (Southall et al., 2007) and PPW (approximation).

In addition to variability in marine mammal hearing sensitivities, science recognizes that different noise source types do not equally affect species in the same manner, particularly when considered in the context of accumulated sound levels. Repeated exposure to noise is potentially more damaging as it increases the accumulation of received sound necessary to elicit TTS or PTS. Within each noise source and hearing group, threshold levels are identified depending on the group-specific hearing capabilities and how they relate to the potential onset of TTS and PTS. Impulsive noise exposures result in TTS and PTS at lower accumulated sound levels than non-impulsive noises given their rapid onset and broadband nature. Consequently, they are also subject to dual thresholds (Southall et al., 2007 [adopted by Finneran (2016) and by NMFS (2018)]).

For marine mammals, acoustic thresholds are used within the context of harassment under the MMPA. The MMPA defines harassment in two levels: Level A (PTS) and Level B (behavioral). The marine mammal threshold criteria used in this assessment comprises NMFS (2018) technical guidance criteria for Level A and Level B exposure thresholds recommended by NMFS (2019b). Marine mammal species will not be equally affected by the Proposed Activities due to individual exposure patterns, the context in which noise is received, and, most prominently, individual hearing sensitivities.

Marine mammal PTS onset thresholds are frequency weighted to account for differences in hearing sensitivities among these hearing groups. Current marine mammal behavioral onset thresholds do not used frequency weighting functions to distinguish between hearing groups. However, it is common practice to apply frequency weighting functions to behavioral thresholds as they can provide valuable information regarding marine mammal behavioral responses. Therefore, to provide a more comprehensive assessment of behavioral impacts, the frequency weighted ranges to behavioral thresholds calculated by JASCO (Denes et al., 2020) were used in this assessment. The ranges in Denes et al. (2020) are provided for both the step function currently recommended by NMFS (2019b) based on work by High Energy Seismic Survey (HESS, 1999) and a range of isopleths following the probabilities of response adapted from Wood et al. (2012); however, this assessment only shows ranges to the single step function threshold of SPL 160 dB re 1 µPa following recommendation from NMFS (2019b).

4.1.2 Sea Turtles and Fish

There are three accepted references for defining acoustics thresholds in sea turtles and Atlantic sturgeon: Popper et al. (2014), criteria developed by the FHWG (2008), and a recent analysis of acoustic impacts to marine mammals and sea turtles published by the U.S. Navy (Blackstock et al., 2018). These sources present criteria for physiological effects that are categorized as injury; however, Popper et al. (2014) concedes that injury includes a very wide spectrum of physiological effects, and even those sources that have the potential for mortal injury will likely vary by context and biological conditions. The physiological thresholds indicate the received sound levels at amplitudes expected to cause physiological changes in the animal.

For sea turtles, Popper et al. (2014) provides thresholds for mortal injury or potential mortal injury only for impulsive noises, which were used in this assessment. They provide subjective criteria for recoverable injury and TTS (e.g., near, intermediate, far) rather than discrete values. The subjective nature of these criteria is not applicable to the acoustic assessment and would be highly dependent on the context of the activity. For non-impulsive noises, the only available physiological threshold criteria are from FHWG (2008). Two options are available for behavior criteria in sea turtles; FHWG (2008) and Blackstock et al. (2018). Both references base the onset of disturbed behavior on caged sea turtle studies conducted by McCauley et al. (2000) during an active seismic survey, with the difference being the assessment of the sea turtles at various received levels. Blackstock et al. (2018) noted that due to the potential caging influence, the SPL threshold of 175 dB re 1 µPa was likely a more appropriate threshold to use for the onset of behavioral disturbance in sea turtles in open water; and this threshold was used for sea turtles in this assessment.

The Popper et al. (2014) PK physiological threshold value (207 dB re 1 µPa) for fish is nearly identical to the PK physiological threshold value (206 dB re 1 µPa) for fish used by FHWG (2008). However, their reported SEL_{24h} physiological thresholds for fish differs by 27 dB, demonstrating the continued uncertainty in the understanding of acoustic criteria in fish. The fish species of primary concern in this assessment is the Atlantic sturgeon, which have a relatively primitive swim bladder with no known connection between the swim bladder and inner ear. Atlantic sturgeon are not expected to be found close enough to be impacted by pile driving activities to sustain mortal injuries; therefore, this acoustic assessment presents the Popper et al. (2014) thresholds for potential recoverable injury in fish. For impulsive sources, the threshold used in this assessment is for fish with swim bladders not involved with hearing, which is applicable to Atlantic sturgeon. For non-impulsive sources, the selected threshold was for fish with swim bladders that are involved with hearing because this is the only threshold available from Popper et al. (2014) for that source type. Popper et al. (2014) also does not provide thresholds for behavior criteria, and instead uses TTS as the onset threshold for a behavioral reaction. In order to better summarize potential injury verses behavioral impacts, the TTS criteria were not considered in this report, but are presented in the underwater acoustic analysis report (Denes et al., 2020). This assessment used the FHWG (2008) behavior criteria for sturgeon/salmon. The FHWG (2008) behavioral threshold of SPL 150 dB re 1 µPa has not been tested for biologically significant behavioral reactions in fish, and behavioral responses in fish may range from a heightened awareness of the noise to changes in movement or feeding activity (Popper and Hastings, 2009); therefore, it should be considered a highly conservative estimate for the onset of behavioral responses in Atlantic sturgeon.

The impulsive and non-impulsive thresholds used in this assessment based on the previously referenced publications are provided in the following sections. As discussed in **Section 2.3**, fish are known to be sensitive to both sound pressure and particle motion. However, there are currently no accepted thresholds for the onset of impact related to particle motion. Therefore, the thresholds and acoustic assessment provided in this Technical Report focus only on the pressure component of underwater noise.

4.1.3 Acoustic Criteria for Impulsive Sources

For impulsive sources, PK or SEL_{24h} criteria are used as the metric necessary for determining if an animal exceeds physiological auditory thresholds. These thresholds apply to impact pile driving and some equipment used during geophysical surveys. Physiological thresholds have frequency weighting functions applied for marine mammals but not for fish or sea turtles.

Impulsive sources have only a single SPL metric for behavioral criteria in each faunal group. The acoustic criteria for physiological impacts and behavioral disturbance for each faunal group are provided in **Table 4.1-2**.

Table 4.1-2. Acoustic criteria for impulsive sources used in the acoustic assessment for the Project construction scenarios.

Faunal Group	Physiological Thresholds[1]		Behavioral Thresholds[2]	
	Acoustic Metric	Threshold Value	Acoustic Metric	Threshold Value
LF Cetaceans	SEL_{24h}	183 dB re 1 µPa² s	SPL	160 dB re 1 µPa
	PK	219 dB re 1 µPa		
MF Cetaceans	SEL_{24h}	185 dB re 1 µPa² s	SPL	160 dB re 1 µPa
	PK	230 dB re 1 µPa		
HF Cetaceans	SEL_{24h}	155 dB re 1 µPa² s	SPL	160 dB re 1 µPa
	PK	202 dB re 1 µPa		
PPW	SEL_{24h}	185 dB re 1 µPa² s	SPL	160 dB re 1 µPa
	PK	218 dB re 1 µPa		
Sea Turtles	SEL_{24h}	210 dB re 1 µPa² s	SPL	175 dB re 1 µPa
	PK	207 dB re 1 µPa		
Fish	SEL_{24h}	210 dB re 1 µPa² s	SPL	150 dB re 1 µPa
	PK	207 dB re 1 µPa		

dB = decibel; HF = high-frequency; LF = low-frequency; µPa = micropascal; MF = mid-frequency; PPW = phocid pinnipeds in water; re = referenced to; SEL_{24h} = cumulative 24-h sound exposure level; PK = zero-to-peak sound pressure level; SPL = root-mean-square sound pressure level;
[1]Physiological thresholds are defined here as onset of permanent threshold shift in marine mammals (National Marine Fisheries Service [NMFS], 2018); onset of potential mortal injury in sea turtles (Popper et al, 2014); and onset of recoverable injury in fish with a swim bladder not involved in hearing (Popper et al., 2014).
[2]Behaviorial thresholds derived from the following sources: marine mammals = NMFS (2019b); sea turtles = Blackstock et al. (2018); fish = Fisheries Hydroacoustic Working Group (2008).

4.1.4 Acoustic Criteria for Non-impulsive Sources

The criteria for non-impulsive sources is somewhat simplified due to it being a singular rather than dual criteria. Non-impulsive sources are applicable for the vessels, aircrafts, some equipment used during geophysical surveys, WTG noise, and potential vibratory pile driving required for cofferdam installation in the near shore components of the RWEC. Activities with non-impulsive sources (and geophysical survey equipment, including impulsive sources) were not modeled in the underwater acoustic analysis report (Denes et al., 2020). Although non-impulsive sources were not modeled for this Project, acoustic criteria for the affected resources are available for non-impulsive sources and therefore are discussed in the context of impact assessment in this Technical Report, allowing a qualitative assessment of potential impacts relative to expected sound levels produced by these activities (**Section 2.1**).

In addition to the difference in source type, the threshold values for non-impulsive sources are different from those for impulsive sources for both physiological and behavioral impacts. Non-impulsive thresholds values are provided in **Table 4.1-3**.

Table 4.1-3. Acoustic threshold criteria for non-impulsive sources used in the acoustic assessment for Project Activities.

Faunal Group	Physiological Thresholds[1]		Behavioral Thresholds[2]	
	Acoustic Metric	Threshold Value	Acoustic Metric	Threshold Value
LF Cetaceans	SEL_{24h}	199 dB re 1 μPa^2 s	SPL	120 dB re 1 μPa
MF Cetaceans	SEL_{24h}	198 dB re 1 μPa^2 s	SPL	120 dB re 1 μPa
HF Cetaceans	SEL_{24h}	173 dB re 1 μPa^2 s	SPL	120 dB re 1 μPa
PPW	SEL_{24h}	201 dB re 1 μPa^2 s	SPL	120 dB re 1 μPa
Sea Turtles	SPL	180 dB re 1 μPa	SPL	175 dB re 1 μPa
Fish	$SPL_{,48h}$[3]	170 dB re 1 μPa	SPL	150 dB re 1 μPa

dB = decibel; HF = high-frequency; LF = low-frequency; μPa = micropascal; MF= mid-frequency; PPW = phocid pinnipeds in water; re = referenced to; SEL_{24h} = cumulative 24-h sound exposure level; SPL = root-mean-square sound pressure level;
[1]Physiological thresholds are defined here as onset of permanent threshold shift in marine mammals (National Marine Fisheries Service [NMFS],2018); onset of potential mortal injury in sea turtles (Fisheries Hydroacoustic Working Group [FHWG], 2008); and onset of recoverable injury in fish (Popper et al., 2014).
[2] Behavioral thresholds derived from the following sources: marine mammals = NMFS (2019b); sea turtles = Blackstock et al. (2018); fish = FHWG (2008).
[3]Recoverable injury threshold reported for fish with swim bladders involved in hearing. Popper et al., (2014) does not provide thresholds for fish with swim bladder not involved with hearing. Threshold assumes that the fish is exposed to the SPL value for 48 continuous hours.

4.2 Underwater Acoustic Modeling

Modeled sound fields were used to determine potential impacts to marine species based on the corresponding threshold criteria (**Section 4.1**); the methodology used for underwater acoustic modeling is fully described in Denes et al. (2020) and summarized here for reference.

Hammer energy and strikes required to reach the target pile depth are not equal throughout the period of installation of a pile. Therefore, the modeling takes into account the sequence of hammer energy and pile strikes during the course of pile installation. Modeling also considers an NMS in the form of a big bubble curtain (BBC) or similar device, which is expected to be employed during all impact pile driving events for this Project to minimize potential impact to marine species. Use of an NMS represents a measure that achieves an overall reduction of in-water sound energy resulting in smaller distances to acoustic thresholds (Denes et al., 2020). For all species, the NMS reduces the risk of impacts in two ways. First, by reducing the radial distance to a predicted threshold, the probability of an animal entering the impact area is reduced. Second, by reducing the distance to a predicted threshold level, the ability to monitor and mitigate an area of impact is improved. Based on recent information regarding the efficacy of NMSs, broadband noise attenuation of up to 10 dB is expected to be achieved during impact pile driving activities in RWF; however, attenuation levels will be dependent upon frequency (Bellman, 2014, 2020). Ranges using 0-, 6-, 10-, and 15-dB broadband attenuation are presented in the summary tables for reference (**Section 4.4**), but for the impact assessment, 10-dB attenuation is assumed. Additionally, mitigation, such as reduction in hammer energy and operational shutdowns, or aversion behavior by animals were not included in the modeling scenarios, although they warrant consideration when conducting the impact assessment.

Factors relating to the acoustic properties of the noise source and operational variables will also influence noise propagation through the water column and are described further in the following section. More importantly, certain combinations of variables will affect the distance calculations more than others. The combination of parameters to assess expected ranges to the specified threshold distances for individual faunal groups to serve as the basis for the acoustic impact assessment.

Several assumptions were applied to the presented data in order to streamline the viewing of the underwater acoustic model results (Denes et al., 2020) for use in an assessment framework. The environmental propagation conditions used in the modeled scenarios consider seasonal and geographic location variability. Generally, modeled threshold distances were larger during the winter versus summer. The actual distances created during construction are likely further influenced by *in situ* environmental conditions at different locations during construction, as seen in the variability in the model results for the two locations for the WTG and OSS foundations (Denes et al., 2020). However, for the purposes of this impact assessment, ranges modeled for each season and location are combined, and are provided as mean threshold ranges for each modeled activity. This Technical Report, where appropriate to understand the impact assessment, provides results and assumptions that are also found in Denes et al. (2020). However, fine-scale environmental as well as operational variability cannot be captured in the summary provided in this Technical Report, and readers should refer to Denes et al. (2020) for detailed modeling results and methods.

4.2.1 Impact Pile Driving Parameters

A maximum of 100 WTG monopile foundations may be installed along with two foundations for the OSSs, which may use either monopile or jacket foundations. For the WTG foundations, 12-m diameter steel monopiles were modeled at two representative locations within the RWF Lease Area (Denes et al., 2020). For the OSSs, 15-m diameter steel monopiles and 4-m diameter jacket pin piles were included in the modeling assessment, modeled at three representative locations within the RWF Lease Area. The impact pile driving parameters used in this model to calculate the ranges to prescribed physiological and behavior thresholds were based on engineering and Project design assumptions. While not expected, some of the assumptions and design criteria may change slightly up to the point of RWF construction. Modeling used the most accurate and current parameters expected for the Project, and where there is uncertainty, a conservative approach was used (Denes et al., 2020).

Operational variables specific to impact pile driving that my influence noise propagation include hammer type, pile type, pile schedule (hammer energy/number of strikes), and geographic location. To account for current uncertainty in the Project design criteria, multiple scenarios were modeled to account for variability in the anticipated pile schedule and hammer energy.

For the monopile foundations, three piling schedules were used to estimate threshold distances for each of the three foundation types proposed for this Project (Denes et al., 2020). For the modeling assessment, it was assumed that WTG monopile foundations will require up to 6,500 strikes to install, the OSS monopile foundations will require 11,500 strikes to install, and the OSS jacket foundation, which consist of four pin piles, will require 11,000 strikes to install (Denes et al., 2020). Modeling accounted for the inclusion of a soft start at the beginning of each pile. The piling scenarios for each pile type are provided in **Tables 4.2-1** through **4.2-3**.

Table 4.2-1. Piling schedule for the 12-m wind turbine generator monopile foundations (Denes et al., 2020).

Energy Level (kJ)	Strike Count	Pile Penetration (m)	Modeled strike rate (min^{-1})
1,000	500	8	30
2,000	1,000	5	
3,000	2,000	12	
4,000	3,000	15	

kJ = kilojoule.

Table 4.2-2. Piling schedule for the 15-m offshore substations monopile foundations (Denes et al., 2020).

Energy Level (kJ)	Strike Count	Pile Penetration (m)	Modeled strike rate (min^{-1})
1,000	500	12	30
2,000	1,000	8	
3,000	2,000	10	
4,000	8,000	20	

kJ = kilojoule.

Table 4.2-3. Piling schedule for the offshore substations jacket foundation consisting of four 4 m pin piles (Denes et al., 2020).

Energy Level (kJ)	Strike Count	Pile Penetration (m)	Modeled strike rate (min^{-1})
500	500	15	30
1,000	1,000	10	
1,500	1,500	13	
2,000	8,000	32	

kJ = kilojoule.

The energy output and number of blows at different pile schedules (e.g., soft-start, full driving, end set) will produce different threshold distances for each energy level. In order to better summarize the details of the model into an assessment of the installation activities, the mean threshold distances produced by all potential pile schedules and across all four hammer energies are provided in this Technical Report, representing the potential impacts produced over the course of a full pile installation (i.e., start to completion of driving a pile foundation). Multiple scenarios were modeled to estimate the linear ranges to regulatory acoustic thresholds (**Section 4.1**) for the complete pile schedule.

4.2.2 Acoustic Ranges and Exposure Ranges

Acoustic propagation through the water was modeled to produce three-dimensional sound fields around each source radiating out to a point at which sound levels reached expected ambient conditions. Noise is generally assumed to propagate out from the source to create an even spherical sound field; however, influence from local physical and oceanographic features results in sound propagating unevenly in all directions. Therefore, the radial distance that encompasses 95% of the modeled sound field is used to define the *acoustic range* from the source within which noise at or above acoustic thresholds for a marine species may be exceeded. An animal located within that range for a defined period of time is said to be exposed to the corresponding threshold. The radial distance, or acoustic range, thus relies solely on noise propagation through the environment and assumes a stationary receiver (i.e., animal) to predict the maximum distance at which that receiver could receive enough acoustic energy over the time period determined by the metric (e.g., 24-h for marine mammal SEL thresholds).

The acoustic ranges are traditionally used in the regulatory context of impact assessment and, in the case of marine mammals, are used to estimate takes as defined by the MMPA. The acoustic range can also help assess whether standard mitigation methods (e.g., visual observation) adequately reduce the risk of potential impacts from noise to a given marine species.

However, it is recognized that modeled acoustic ranges to threshold levels may overestimate the actual distances at which animals receive exposures meeting the threshold criteria and are likely not realistic, particularly for accumulating metrics like SEL. Applying animal movement and exposure models provides a more realistic indication of the distances at which acoustic thresholds are met. For this reason, *exposure ranges* were modeled to provide a realistic estimate of the ranges at which moving animals exceed the given acoustic thresholds. Notably, the exposure ranges are species-specific rather than categorized only by faunal group which affords more biological context to be considered when assessing impacts.

To determine exposure ranges, pile strikes are propagated to create an ensonified environment (**Section 4.2.1**) while simulated animals (i.e., animats) are moved about the ensonified area following known species-specific behaviors. Modeled animats that have received sound energy that exceeds the acoustic threshold criteria are registered, and the closest point of approach (CPA) recorded at any point in that animal's movement is then reported as its exposure range. This process is repeated multiple times for each animat to produce and the exposure-based ranges which comprise 95% of the CPAs for animats that exceeded the threshold (i.e., $ER_{95\%}$). The exposure range approach is used as the basis for the impact assessment in **Section 5.0**, for developing environmental protection measures, and for future MMPA assessments due to the incorporation of animal movement and behavior in the development of these ranges.

An animal being exposed to a specific threshold or occupying the waters within the propagated sound field does not alone constitute an impact for a particular species. Assessing the potential for impact needs to simultaneously consider the source, activity, environmental factors influencing propagation, frequency weighting factors, mitigation factors, and autecological characteristics of an at-risk species. Variability in each of these factors will, in turn, vary the potential risk to each species. Therefore, modeled exposure ranges are one component of the overall impact assessment process in this Technical Report.

Because accurate animal movement information is not currently available for Atlantic sturgeon to use in the model, the traditional acoustic range approach was used for the impact assessment for this species. However, it should be recognized that these are likely overestimates since Atlantic Sturgeon are not expected to remain in one location long enough to elicit potential physiological impacts or biologically significant disturbances.

The results of the modeling are summarized in **Sections 4.3** and **4.4** for acoustic ranges and exposure ranges, respectively. A wider selection of acoustic threshold criteria were modeled in the underwater acoustic analysis report (Denes et al., 2020); however, only the ranges to the threshold criteria presented in **Section 4.1.3** were summarized in the following sections and applied to the impact assessment (**Section 5.0**).

4.3 Summary of Modeled Acoustic Ranges

Summarized modeling results for acoustic ranges to physiological and behavioral thresholds are provided in **Tables 4.3-1** through **4.3-3** for each foundation type. As discussed previously, modeling was conducted for two locations for each pile type and two seasons, winter and summer (Denes et al., 2020). Ranges are provided separately for each location and season in Denes et al. (2020); however for the purposes of this report, the minimum, maximum, and mean values of the modeled ranges with 10 dB applied for both seasons and all locations are provided.

Table 4.3-1. Mean acoustic ranges (m) to physiological thresholds and frequency weighted[1] behavioral thresholds for each faunal group for a 12-m wind turbine generator monopile foundation with 10 dB noise attenuation applied (Denes et al., 2020).

Faunal Group	Physiological Threshold Ranges						Behavioral Threshold Ranges		
	PK			SEL_{24h}			SPL		
	Minimum	Maximum	Mean	Minimum	Maximum	Mean	Minimum	Maximum	Mean
LF Cetaceans	5	5	5	4,476	8,663	6,476	3,825	4,260	4,043
MF Cetaceans	-	-	-	80	102	90	2,235	3,240	2,738
HF Cetaceans	178	200	189	3,420	5,404	4,379	1,771	2,772	2,272
PPW	6	6	6	810	1,165	988	3,282	3,785	3,534
Sea Turtles[2]	95	101	98	330	512	423	481	2,741	1,465
Atlantic Sturgeon	95	101	98	330	512	423	5,805	9,758	7,782

- = threshold not reached; LF= low frequency; MF= mid frequency; HF= high frequency; PPW= phocid pinnipeds in water; SEL_{24h} = cumulative 24-h sound exposure level; PK = peak sound pressure level; SPL = root-mean-square sound pressure level.
[1]Frequency weighting applied to marine mammals only. Sea turtle and fish results are unweighted.
[2]Modeling results for SPL are only available at 170 and 180 decibels (dB) referenced to 1 micropascal; therefore, the range to the SPL 175 dB sea turtle threshold was estimated from those values.

Table 4.3-2. Mean acoustic ranges to physiological thresholds and frequency weighted[1] behavioral thresholds for each faunal group for a 15-m offshore substation monopile foundation with 10 dB noise attenuation applied (Denes et al., 2020).

Faunal Group	Physiological Threshold Ranges						Behavioral Threshold Ranges		
	PK			SEL_{24h}			SPL		
	Minimum	Maximum	Mean	Minimum	Maximum	Mean	Minimum	Maximum	Mean
LF Cetaceans	6	6	6	5,324	11,121	7,976	4,093	4,671	4,382
MF Cetaceans	-	-	-	90	142	110	2,379	3,216	2,798
HF Cetaceans	260	260	260	3,846	6,475	5,078	1,843	2,597	2,220
PPW	7	7	7	1,141	1,583	1,356	3,545	3,838	3,692
Sea Turtles[2]	90	95	93	840	1,054	945	764	3,024	1,777
Atlantic Sturgeon	90	95	93	840	1,054	945	6,921	10,888	8,905

- = threshold not reached; LF= low frequency; MF= mid frequency; HF= high frequency; PPW= phocid pinnipeds in water; SEL_{24h} = cumulative 24-h sound exposure level; PK = peak sound pressure level; SPL = root-mean-square sound pressure level.
[1]Frequency weighting applied to marine mammals only. Sea turtle and fish results are unweighted.
[2]Modeling results for SPL are only available at 170 and 180 decibels (dB) referenced to 1 micropascal; therefore, the range to the SPL 175 dB sea turtle threshold was estimated from those values.

Table 4.3-3. Mean acoustic ranges to physiological thresholds and frequency weighted[1] behavioral thresholds for each faunal group for a 4-m offshore substation jacket foundation with 10 dB noise attenuation applied (Denes et al, 2020).

Faunal Group	Physiological Threshold Ranges						Behavioral Threshold Ranges		
	PK			SEL_{24h}			SPL		
	Minimum	Maximum	Mean	Minimum	Maximum	Mean	Minimum	Maximum	Mean
LF Cetaceans	4	4	4	5,639	15,426	10,215	3,732	4,092	3,912
MF Cetaceans	-	-	-	165	277	223	2,356	3,360	2,858
HF Cetaceans	87	88	88	4,732	9,558	7,132	1,947	3,029	2,488
PPW	5	5	5	1,604	2,470	2,019	3,205	3,774	3,490
Sea Turtles[2]	42	42	42	682	888	781	368	2,253	1,187
Atlantic Sturgeon	42	42	42	682	888	781	5,871	11,345	8,608

- = threshold not reached; LF= low frequency; MF= mid frequency; HF= high frequency; PPW= phocid pinnipeds in water; SEL_{24h} = cumulative 24-h sound exposure level; PK = peak sound pressure level; SPL = root-mean-square sound pressure level.
[1]Frequency weighting applied to marine mammals only. Sea turtle and fish results are unweighted.
[2]Modeling results for SPL are only available at 170 and 180 decibels (dB) referenced to 1 micropascal; therefore, the range to the SPL 175 dB sea turtle threshold was estimated from those values.

4.4 Summary of Modeled Exposure Ranges

Applying animal movement and exposure models (Denes et al., 2020) provides a more realistic indication of the distances at which acoustic thresholds are met. As previously described, modeled exposure ranges are species-specific; however, the exposure ranges are grouped by hearing group in this report to be consistent with the approach taken for the impact assessment (**Section 5.0**).

The exposure ranges to marine mammals and sea turtle physiological and behavioral thresholds are provided in **Tables 4.4-1** through **4.4-3** for the three pile types proposed for the RWF WTG and OSS. As mentioned previously, exposure ranges are not provided for the Atlantic sturgeon because accurate animal movement information is not available to apply to the model.

Similar to the acoustic ranges (**Section 4.3**), results were provided separately for both seasons modeled (Denes et al., 2020); however, for the purposes of this report, the mean of both seasons is provided in the following tables for each level of noise attenuation modeled (0, 6, 10, and 15 dB). All levels of noise attenuation are provided for reference, but the impact assessment in **Section 5.0** only considers the ranges with 10 dB attenuation applied.

Table 4.4-1. Mean exposure ranges ($ER_{95\%}$) (m) to marine mammal and sea turtle physiological and behavioral thresholds resulting from installation of 12-m wind turbine generator monopile foundations with 0, 6, 10, and 15 dB broadband attenuation (Denes et al., 2020).

Faunal Group	Physiological Threshold Ranges								Behavioral Threshold Ranges			
	PK				SEL_{24h}				SPL			
	0 dB	6 dB	10 dB	15 dB	0 dB	6 dB	10 dB	15 dB	0 dB	6 dB	10 dB	15 dB
LF Cetaceans	89	12	5	2	7,465	3,409	1,916	770	7,650	4,928	3,891	3,169
MF Cetaceans	4	2	0	0	45	6	5	0	7,897	5,080	3,972	3,204
HF Cetaceans	850	390	205	118	5,845	3,210	2,035	955	7,830	5,040	3,960	3,225
PPW	99	15	6	3	2,453	768	195	23	7,990	5,120	4,048	3,285
Sea Turtles	460	164	110	55	688	127	17	13	3,178	1,990	1,187	520

dB=decibel; LF= low frequency; MF= mid frequency; HF= high frequency; PPW= phocid pinnipeds in water; SEL_{24h} = cumulative 24-h sound exposure level; PK = zero-to-peak sound pressure level; SPL = root-mean-square sound pressure level.
[1]Frequency weighting applied to marine mammals only. Sea turtle results are unweighted.

Table 4.4-2. Mean exposure ranges ($ER_{95\%}$) (m) to marine mammal and sea turtle physiological and behavioral thresholds resulting from installation of 15-m offshore substation monopile foundations with 0, 6, 10, and 15 dB broadband attenuation (Denes et al., 2020).

Faunal Group	Physiological Threshold Ranges								Behavioral Threshold Ranges			
	PK				SEL_{24h}				SPL			
	0 dB	6 dB	10 dB	15 dB	0 dB	6 dB	10 dB	15 dB	0 dB	6 dB	10 dB	15 dB
LF Cetaceans	77	13	6	3	7,449	3,666	2,149	868	8,530	5,519	4,196	3,436
MF Cetaceans	5	2	0	0	21	2	2	0	8,503	5,507	4,260	3,466
HF Cetaceans	580	320	260	93	5,805	3,010	1,865	885	8,640	5,585	4,260	3,470
PPW	85	88	7	4	2,395	800	305	28	8,728	5,630	4,293	3,605
Sea Turtles	360	149	81	308	1,048	273	13	0	3,417	2,317	1,500	802

dB=decibel; LF= low frequency; MF= mid frequency; HF= high frequency; PPW= phocid pinnipeds in water; SEL_{24h} = cumulative 24-h sound exposure level; PK = zero-to-peak sound pressure level; SPL = root-mean-square sound pressure level.
[1]Frequency weighting applied to marine mammals only. Sea turtle results are unweighted.

Table 4.4-3. Mean exposure ranges ($ER_{95\%}$) (m) to marine mammal and sea turtle physiological and behavioral thresholds resulting from installation of 4-m offshore substation jacket foundations with 0, 6, 10, and 15 dB broadband attenuation (Denes et al., 2020).

Faunal Group	Physiological Threshold Ranges								Behavioral Threshold Ranges			
	PK				SEL_{24h}				SPLSPL			
	0 dB	6 dB	10 dB	15 dB	0 dB	6 dB	10 dB	15 dB	0 dB	6 dB	10 dB	15 dB
LF Cetaceans	16	3	0	0	14,581	7,090	3,794	1,563	8,398	4,979	3,765	3,004
MF Cetaceans	0	0	0	0	235	41	10	2	8,511	5,106	3,824	3,041
HF Cetaceans	240	78	48	24	10,885	5,925	3,690	1,975	8,790	5,130	3,865	3,040
PPW	21	4	0	0	6,280	2,310	1,068	253	8,825	5,115	3,878	3,075
Sea Turtles	87	42	24	33	1,017	232	57	0	2,955	1,710	1,030	480

dB=decibel; LF= low frequency; MF= mid frequency; HF= high frequency; PPW= phocid pinnipeds in water; SEL_{24h} = cumulative 24-h sound exposure level; PK = zero-to-peak sound pressure level; SPLSPL = root-mean-square sound pressure level.
[1]Frequency weighting applied to marine mammals only. Sea turtle results are unweighted.

5.0 IMPACT ASSESSMENT FOR RWF AND RWEC

All potential IPFs resulting from Project Activities were assessed for marine mammals, sea turtles, and ESA-listed fish species (i.e. Atlantic Sturgeon) in **Sections 4.3.3.2**, **4.3.4.2**, and **4.3.5.2** of the Project's COP. IPFs that have the potential to have greater than negligible impacts on marine mammals, sea turtles, and Atlantic sturgeon (as defined in **Section 1.1**) include habitat alteration, underwater noise, and vessel traffic. Using the baseline information provided in **Section 3.0**, the potential for impacts from Project Activities was assessed for all affected resources and characterized as either direct or indirect, and short-term or long-term (**Sections 5.1**, **5.2**, and **5.3**) using the parameters identified in **Section 1.2** (detectability, duration, spatial extent, and severity).

The detectability of an IPF referred to whether it would be perceptible to a marine mammal, sea turtle, or fish based on published literature that documented responses to these or comparable IPFs. The duration of an impact was determined to be either short-term or long-term, and considered both the duration of the impact-producing activities (**Sections 2.0** and **3.0** of the Project's COP) and how quickly an animal would recover once the activity ceased, based on available publications. The spatial extent of the IPF was estimated using Project-specific modeling (as applicable), and information provided in **Sections 2.0** and **3.0** of the Project's COP. The severity of the potential impact was then determined based on the other three parameters, the current status of the populations under consideration, and the likelihood for population-level impacts based on published literature. These four parameters combined were used to determine if a potential impact exceeded a negligible determination. For example, a potential impact would be considered greater than negligible if it was determined an IPF was detectable to a resource, resulted from an activity occurring over a longer period or resulted in an impact that took longer for the resource to recover, and occurred over a broader spatial area which increased the risk of overlap between the IPF and the resources' geographic range.

Additionally, Project-specific modeling was conducted by JASCO to assess the potential for impact for the underwater noise IPF (Denes et al., 2020). Denes et al. (2020) defines and characterizes acoustic propagation resulting from impact pile driving activity associated with the Project for all scenarios included in the Project Envelope (**Section 3.0** of the Project's COP) and results applicable to this assessment are provided for reference. Results of the modeling provided a more quantitative estimate of the spatial extent of this IPF as it pertains to impact pile driving. Noise from DP vessels, aircraft, vibratory pile driving, geophysical survey, and WTG operations were not modeled for this Project, so the potential for impact was based predominantly on published literature and modeling conducted for other similar projects. Detectability of this IPF was based on accepted acoustic thresholds for each faunal group (**Section 4.1**), estimated source levels for each noise-producing activity (**Section 2.1**), and the description of the existing underwater acoustic habitat of the Project Area (**Section 2.0**). As stated above, the duration is based on information provided in **Sections 2.0** and **3.0** of the Project's COP. These criteria, combined with the current status of the affected populations, helped determine the severity of potential impacts. Results of the modeling, including acoustic and exposure ranges for impact pile driving are summarized in **Section 4.0** for reference.

The information provided in the following sections is intended to provide a more detailed explanation of the underwater noise IPF and any IPFs that may result in greater than negligible impacts on marine mammals, sea turtles, and ESA-listed fish, specifically Atlantic sturgeon.

5.1 Summary of Impacts

Based on the list of affected species identified in **Section 2.2**, the potential for impacts resulting from Project activities during construction, O&M, and decommissioning were assessed using the methodology described in **Section 1.2**. All potential IPFs are discussed in **Section 4.1** of the COP; only habitat alteration, underwear noise, and vessel traffic were discussed in this Technical Report as they are the only IPFs with the potential to result in greater than negligible impacts to affected resources (**Section 1.3**). As previously discussed in **Section 3.3**, the only ESA-listed fish species likely to occur in the Project Area is the Atlantic sturgeon, so potential impacts were only assessed for this species. A summary of anticipated impacts to marine mammals, sea turtle, and Atlantic sturgeon discussed in this report is provided in **Table 5.1-1**.

Table 5.1-1. Summary of anticipated impacts on marine mammals, sea turtles, and Atlantic sturgeon from underwater noise, vessel traffic, and habitat alteration resulting from Project Activities during construction, operation and maintenance (O&M), and decommissioning.

IPF	Marine Mammals	Sea Turtles	Atlantic Sturgeon
DP Vessel Noise	Direct, Short-term	Direct, Short-term	Direct, Short-term
Aircraft Noise	Direct, Short-term	Direct, Short-term	Direct, Short-term
Geophysical Surveys	Direct, Short-term	Direct, Short-term	Direct, Short-term
Impact Pile Driving	Direct, Short-term	Direct, Short-term	Direct, Short-term
Vibratory Pile Driving	Direct, Short-term	Direct, Short-term	Direct, Short-term
WTG Noise	Direct, Long-term	Direct, Long-term	Direct, Long-term
Vessel Traffic	Direct, Short-term (construction/decommissioning) and Long-term (O&M)	Direct, Short-term (construction/decommissioning) and Long-term (O&M)	Direct, Short-term (construction/decommissioning) and Long-term (O&M)
Habitat Alteration	Direct, Short-term (construction and decommissioning) and Long-term (O&M)	Direct (construction and decommissioning), Direct and Indirect (O&M), Short-term (construction and decommissioning) and Long-term (O&M)	-

- indicates no impact expected; DP = dynamic positioning; ESA = Endangered Species Act; IPF = impact producing factor; WTG = wind turbine generator.

The primary IPF expected to impact all potentially affected resources is underwater noise. Project Activities that will produce noise include impact pile driving during construction, the use of DP vessels and aircraft, vibratory pile driving used for the installation of a cofferdam, geophysical surveys, and WTG operations. Impact pile driving is likely to have the greatest risk of impact due to the impulsive characteristics and high noise levels produced by this source (**Section 4.2**). No injury is anticipated for any resource with the application of the environmental protection measures outlined in **Section 5.5**, but some level of behavioral response is anticipated for all resources (**Section 5.0**).

Project-related vessel traffic will contribute a nominal amount to the overall volume of existing traffic in this region. Although the risk of a strike is low, in the unlikely event a strike were to occur, the consequences of an individual mortality in a population that is listed as Threatened or Endangered is countered by their overall resilience to population-level impacts. The implementation of vessel strike avoidance measures (**Section 5.5**) will reduce the risk of strikes for potentially affected species.

Marine mammals and sea turtles are the only resources expected to receive greater than negligible impacts as a result of habitat alteration caused by the presence of the RWF foundations and associated scour protection. Studies have shown that marine mammals may forage around the foundations (**Section 5.1.3**) and sea turtles use artificial structures offshore for foraging and shelter from ocean currents and vessel traffic (**Section 5.2.3**). However, the habitat alteration resulting from the installation of the foundations and scour protection may have inadvertent impacts on these resources, such as wakes disrupting zooplankton prey species and increased susceptibility of sea turtles to cold stunning if they remain in the RWF area longer than typically expected (**Sections 5.1.3** and **5.2.3**). Sea turtles may also become habituated to the habitat created by the foundations and scour protection and may be impacted by the removal of foraging and sheltering habitat when the RWF is decommissioned (**Section 5.2.3**).

5.2 Marine Mammals

As shown in **Table 1.2-1**, IPFs that could have greater than negligible impacts on marine mammals include underwater noise, vessel traffic, and habitat alteration. These IPFs are discussed further in the following subsections.

5.2.1 Underwater Noise

As discussed in **Section 2.3**, the range of potential effects from noise includes hearing threshold shift; auditory injury; masking; and stress and disturbance, including behavioral responses (NRC, 2003; 2005; Nowacek et al., 2004; Richardson et al., 1995; Southall et al., 2007). The severity of potential impacts increases when the exposure occurs close to a noise source and with the duration of the exposure. Impact pile driving was identified as the activity that would likely have the greatest potential for auditory impact, including PTS, on marine mammals; however, through the use of NMSs and other mitigation measures, no acoustic injury is expected to any marine mammal species. DP vessel noise, aircraft activities, vibratory pile driving, geophysical surveys, and WTG noise may also affect the acoustic habitat of marine mammals and in some cases result in behavioral disturbance. Impact and vibratory pile driving, geophysical surveys, and aircraft activities would occur during construction of the RWF and RWEC, WTG noise would occur during RWF operations, and DP vessel activity could occur during any Project phase.

5.2.1.1 DP Vessel Noise

Impacts on marine mammals from vessel noise have been documented and include temporary disruptions of communication or echolocation from auditory masking; behavior disruptions of individual or localized groups of marine mammals; and limited, localized, and short-term displacement of individuals of any species, including strategic stocks, from localized areas around the vessels. Aguilar-Soto et al. (2006) reported that the noise from a passing vessel masked ultrasonic vocalizations of a Cuvier's beaked whale (*Ziphius cavirostris*) and reduced the maximum communication range by 82% when exposed to a 15-dB increase in ambient noise levels at the vocalization frequencies, resulting in a 58% reduction in the effective detection distance of the Cuvier's beaked whale's echolocation clicks. Hatch et al. (2012) estimated that calling North Atlantic right whales may have lost 63% to 67% of their communication "space" due to shipping noise. LF (20 to 200 Hz) noise from large ships overlaps the frequency range of some mysticete vocalizations, and increased levels of ambient noise have been documented in areas with high shipping traffic, causing responses in some mysticetes that have included habitat displacement; changes in behavior; and alterations in the intensity, frequency, and intervals of their calls (Rolland et al., 2012).

Marine mammals are able to compensate, to a limited extent, for auditory masking through a variety of mechanisms, including increasing SLs (i.e., the Lombard effect) or durations of their vocalizations or by changing spectral and temporal properties of their vocalizations (Hotchkin and Parks, 2013; Parks et al., 2010). North Atlantic right whales in high-noise conditions have been documented to lower their call rate and produce calls with a higher average fundamental frequency (Parks et al., 2007). In the presence of ship noise, beluga whales produced whistles at higher frequencies and longer durations (Lesage et al., 1999). Di Iorio and Clark (2009) found that blue whales increased their rate of social calling in the presence of sub-bottom exploration equipment, which was presumed to represent a compensatory behavior to elevated ambient noise levels during the surveys. Several marine mammal species are also known to increase the SLs of their calls in the presence of elevated noise levels (Dahlheim, 1987; Lesage et al., 1999; Terhune, 1999). Holt et al. (2008) studied the effects of anthropogenic noise exposure on Endangered southern resident killer whales in Puget Sound, reporting that they increased their call amplitude by 1 dB for every 1 dB increase in ambient noise in the 1 to 40 kHz frequency band. Castellote et al. (2012) reported that male fin whales from two different subpopulations not only modified their song characteristics during increased ambient noise conditions, but also left the area and did not return for 14 days. Castellote et al. (2012) hypothesized that the fin whales modified their acoustic communications to compensate for the increased ambient noise levels and that the animals had a lower tolerance for seismic airgun noise than for shipping noise.

Modeling was not conducted for DP vessel noise for this Project, but a qualitative discussion of noise produced by DP vessels can be found in Denes et al. (2020). No acoustic injury impacts are expected to occur to marine mammals as a result of vessel noise due to the non-impulsive nature of the sources and relatively low SLs produced (BOEM, 2013; McPherson et al., 2016). Because vessel noise is perceptible and can temporarily alter a mammal's acoustic habitat, it has the potential for disrupting or interfering with normal biological activities that could constitute behavioral disturbance. Behavioral impacts resulting from vessel noise would be expected only from vessels that use DP thrusters. DP vessels will predominately be used during the approximate 18-month construction period and during the decommissioning phase. During the 20 to 35 year O&M period, DP vessels operating in a station-keeping mode, which produce the greatest sound levels, will be used intermittently; however, DP thrusters may also be used for propulsion on some vessels during transits between ports and the RWF and RWEC. For those few individuals that are present in the region during DP vessel operations, behavioral disturbances may be consequential if the response results in the interruption of critical behavior. However, the anticipated noise associated with DP vessel operations throughout the Project would be temporary and is not expected to be a significant contribution to cumulative vessel noise already present in the region. With the added presumption that individual or groups of marine mammals in the Project Area are familiar with vessel-related noises, particularly within trafficked areas around the RWF and nearby shipping lanes, behavioral impacts on marine mammals from Project-related DP vessel noise are expected but would not be extensive or biologically significant. Impacts are expected to be temporary, and marine mammal behavior would return to baseline conditions when DP vessel activity ceases. Therefore, the effects of Project-related DP vessel noise on marine mammals are considered ***direct*** and ***short-term***.

5.2.1.2 Aircraft Noise

Noise produced from aircrafts used during Project construction have the potential to propagate underwater at levels that could be detectable to marine mammals. Received SPL measured from a helicopter at 18 m depth were approximately 106 dB re 1 μPa and were shown to generally increase with decreasing water depth, decreasing altitude of the aircraft, and increasing flight speed (Patenaude et al., 2002). Additionally, behavioral responses to aircraft noise have been observed in bowhead whales (*Balaena mysticetus*) in response to both helicopters and planes (Patenaude et al., 2002). However, helicopters would only be used intermittently to support crew transfers during construction and O&M (**Section 4.1.4.1** of the Project's COP), and given the relatively short duration of construction activities (approximately 18 months), only temporary changes in behavior are expected to occur. Impacts from aircraft noise are considered ***direct*** and ***short-term***.

5.2.1.3 Geophysical Surveys

As discussed in **Section 2.1.5**, geophysical surveys will be conducted prior to construction of the RWF and RWEC to identify any seabed obstructions or potential MEC/UXOs. The likelihood of encountering MEC/UXOs within the Project Area is low, and should one be identified it will be disposed of using methods designed to avoid potential detonation of the device. The preferred approach for MEC/UXO is avoidance, but in a situation where avoidance is not possible, low-noise methods of removal or relocation will be employed (**Section 3.3.3.2** of the Project's COP). Therefore, explosive decommissioning of MEC/UXOs is not considered in this assessment, and only noise from the geophysical survey equipment used to locate potential obstructions was analyzed.

Equipment used during these surveys has the potential to produce noise that would exceed physiological and behavioral thresholds for marine mammals (**Section 4.1**). However, previous assessments estimated ranges to physiological thresholds of <50 m, and ranges to behavioral thresholds were all <200 m (CSA Ocean Sciences Inc., 2018, 2020). With the implementation of the environmental protection measures outlined in **Section 5.5**, the risk of impact is low and would be limited to temporary disturbances. Furthermore, due to the relatively short duration of these activities which would only occur during a portion of the full 18-month construction period, impacts are considered ***direct*** and ***short-term***.

5.2.1.4 Impact Pile Driving

Potential acoustic impacts from impact pile driving include noise levels that can elicit direct injury to or behavioral responses in marine mammals and have the potential to cause displacement from critical habitat (Brandt et al., 2011; Bailey et al., 2010), alteration of acoustic habitat availability, and masking (Madsen et al., 2006). Within 10 m of the source, impact pile driving can generate SLs expressed as PK ranging from 233 to 245 dB re 1 µPa m and SLs expressed as SEL_{24h} ranging from 218 to 249 dB re 1 µPa^2 m^2 s with a predominant frequency content below 1,000 Hz (Amaral et al., 2018). During the 2015 Block Island impact pile driving activities, distances to measured behavior SPL threshold isopleths (160 dB re 1 µPa, unweighted) ranged from 2.7 to 4.6 km from the pile source (Amaral et al., 2018). However, physiological threshold distance calculations during the 2015 Block Island impact pile driving measurements used pre-2016 NOAA acoustic guidance criteria (SPL of 180 dB re 1 µPa, unweighted). Recently, BOEM (2018) detailed best management practices designed to minimize pile driving impacts on marine mammals, which will be applied during RWF WTG and OSS installation activities. The application of these practices will minimize the potential for impact ranges by reducing the distances to physiological and behavioral thresholds, and by allowing for the effective application of environmental protection measures (**Section 5.4**).

Results of acoustic modeling conducted for this Project are fully described in Denes et al. (2020) and summarized in **Sections 4.3** and **4.4** for reference. Modeled impact pile driving was conducted for three pile types; 12-m monopile foundations used for the RWF WTGs, 15-m monopile foundations being considered for the RWF OSS, and 4-m jacket pin pile foundations also being considered for the RWF OSS (**Section 3.0** of the Project's COP). Results of the exposure range modeling (**Section 4.4**) indicate that sound levels generated during impact pile driving for all pile types and scenarios with 10 dB attenuation applied will exceed the biological thresholds associated with behavioral disturbance in marine mammals; and could exceed thresholds for the potential onset of physiological effects in some species beyond 3 km if the duration of exposure approached 24 h (**Section 4.4**). The $ER_{95\%}$ for PK physiological thresholds for all pile types and scenarios were generally small (<10 m) with 10 dB attenuation applied for all marine mammal hearing groups except HF cetaceans whose $ER_{95\%}$ for PK reached up to 260 m (**Section 4.4**). $ER_{95\%}$ for SEL_{24h} with 10 dB attenuation for all pile types and scenarios ranged from 1,916 to 3,794 m for LF cetaceans; 0 to 10 m for MF cetaceans; 1,865 to 3,690 for HF cetaceans; and 195 to 1,068 for PPW for all pile types and scenarios (**Section 4.4**). Estimated $ER_{95\%}$ to behavioral thresholds ranged from approximately 3 to 4 km for all hearing groups (**Section 4.4**).

Physiological exposures based on the PK metric are not expected for any marine mammal hearing group due to the small propagation distances and use of an NMS that not only reduces propagation ranges but acts as a physical barrier excluding many species from PK threshold exposures. Based on the modeled $ER_{95\%}$ for SEL_{24h}, only LF and HF cetaceans have large enough ranges to result in a reasonable potential to receive sound levels that exceed physiological thresholds; and this potential primarily exists during periods when species presence is greatest (**Section 3.1**). Additionally, receiving sound levels that exceed thresholds does not equate to PTS, and auditory injury is not expected to occur from impact pile driving activities. Implementation of environmental protection measures in the form of an NMS and monitoring programs (**Section 5.5**) applied during impact pile driving will further reduce the risk of physiological exposures. However, because the potential for PTS exists it is necessary to assess the effect of such an impact should it occur. PTS occurring to species with very low populations such as the North Atlantic right whale has the potential to cause population-level effects should an individual be functionally removed from that population (e.g., loss of communication with conspecifics). Therefore, ESA-listed species with already low population estimates would face a higher risk of population-level effects compared to non-ESA-listed species that have a greater capacity to absorb and recover from potential impact without incurring population-level effects.

There is a greater likelihood of behavioral disturbances to all marine mammal species because the metric for such exposures is based on an instantaneous received SPL, rather than an accumulated metric (e.g., SEL_{24h}). The $ER_{95\%}$ to behavioral thresholds range from approximately 3 to 4 km for all hearing groups. At these ranges, the ability to monitor and mitigate becomes challenging in an operational setting. As discussed in **Section 2.3**, behavioral disturbances are contextual, and disturbance from the relatively short pile installation period is not expected to have any population-level effects and would likely result in only brief disruptions in species' activities. Because impacts would only occur during the 18-month duration of construction activities, impacts from impact pile driving are considered ***direct*** and ***short-term*** for all marine mammal species.

5.2.1.5 Vibratory Pile Driving

Based on previous assessments of vibratory pile driving, sound levels may reach physiological threshold criteria for marine mammals at relatively small distances. *In situ* measurements conducted by the California Department of Transportation during bridge construction vibratory pile driving of sheet piles along the U.S. West Coast and Alaska reported a 162 dB re 1 μPa^2 s SEL over 1 s of vibratory pile driving measured 10 m from the source (Buehler et al., 2015). However, given the relatively short duration of vibratory pile driving activities (up to 3 days) and the location of the proposed cofferdam installation in Narragansett Bay, Rhode Island (**Section 3.0** of the Project's COP), it is unlikely species will be present within proximity of this noise source for durations sufficient to result in the onset of PTS in marine mammals.

While physiological thresholds consider exposure time, current behavioral metrics do not consider the duration of the animal's exposure to noise above the threshold. Therefore, the traditional assessment for behavioral exposures is dependent solely on the presence or absence of a species within the ensonified area. Animals are less likely to respond to sound levels when distant from a source, even when those levels elicit responses at closer ranges; both proximity and received levels are important factors in aversion responses (Dunlop et al., 2017). While vibratory pile driving activities may produce noise which exceeds the behavioral thresholds for marine mammals (**Section 2.1.4**), exposure to an SPL at a specified threshold level does not equate to a behavioral response or a biological consequence. Furthermore, the low abundance of marine mammal species in the nearshore location of the proposed cofferdam and the short period of vibratory pile driving activities significantly reduces the risk of behavioral exposures. There is a low potential for some dolphin, porpoise, and seal species to be present in the region around the cofferdam in Narragansett Bay (**Section 3.1**), and for those species vibratory pile driving presents a behavioral disturbance risk but not a physiological risk. Because impacts would only occur during the approximate 3-day installation period over which vibratory pile driving will occur, impacts to all marine mammals are considered ***direct*** and ***short-term***.

5.2.1.6 WTG Operations

WTGs primarily produce two types of noise: aerodynamic WTG blade noise and mechanical noise. The mechanical noise type can be transmitted underwater via the WTG towers and foundations. As described in **Section 2.1.4**, underwater noise generated by WTGs is concentrated below 500 Hz (Tougaard et al., 2009); and therefore, poses the greatest risk to the LF cetacean hearing group. However, Tougaard et al. (2009) stated that it was unlikely that auditory masking would occur due to the low noise levels produced by operational WTGs. They showed that WTG produced SPL ranging from 100 to 120 dB re 1 μPa at roughly 100 m from the foundation, although the MW size was not identified. Noise measurements taken at 50 m away from a 3.6 MW WTG reported peak power spectral density levels of 126 dB re 1 μPa^2 Hz^{-1} with frequencies centered at 162 Hz and noise levels that varied by wind speed. Acoustic monitoring at the Block Island Wind Farm showed that WTG blades turning at maximum speed (12 rpm) increased noise in lower frequency bands by 3 to 10 dB (HDR, 2019). However, the WTG proposed for the RWF range in size from 8 to 12 MW, and measurements of operational noise for WTGs above 6 MW are not available in the published literature. Madsen et al. (2006) noted that there seemed to be only a weak relationship between the size of the WTG and the emitted noise levels, but cautions that this may not be valid for large WTGs of several megawatts.

Even with the larger WTGs proposed for this Project, noise levels are unlikely to exceed physiological onset thresholds, and impacts would be limited to audibility and perhaps some degree of responsiveness, such as avoidance (MMS, 2007). There is no published information about long term sound exposures to marine mammals from offshore wind farms. Animals such as seals and dolphins display some attraction to prey increases at wind farms, which may suggest that noise levels produced are insufficient to elicit behavioral disturbances in those groups (Teilmann and Carstensen, 2012). There is no published literature assessing long-term movement or acoustic exposure of LF cetaceans in or around offshore wind farms. Additionally, WTG noise will persist for longer periods of time and could impact more species compared to noise produced by construction and installation activities (MMS, 2007).

LF cetaceans are the most likely to perceive and potentially react to the LF noise produced by the WTGs; however, such responses have not been documented. However, due to the large uncertainty regarding the noise propagated by large-scale wind farms with >6 MW WTGs, additional considerations were made for LF cetaceans. Should avoidance behaviors due to noise produced by the wind farm result in reduced access to feeding areas that intersect or are adjacent to the RWF, impact severity could be greater for these species. While this impact is not anticipated, the lack of documented activity of LF cetaceans around operational wind farms requires that such impacts be considered a possibility.

Given the relatively low sound levels that would be produced during WTG operations, only temporary changes in marine mammal behavior would be expected to occur, and no measurable impacts are expected to MF and HF cetaceans or PPW. Due to the anticipated operation of the RWF of 20 to 35 years, impacts to marine mammals are considered ***direct*** and ***long-term***.

5.2.2 Vessel Traffic

Marine mammals may be vulnerable to collisions with moving vessels (Douglas et al., 2008; Laist et al., 2001; Pace, 2011). Vessel strikes happen when either marine mammals or vessels fail to detect one another in time to avoid the collision. Variables that contribute to the likelihood of a vessel strike include vessel speed, vessel size and type, and visibility. Marine mammal strikes have been reported at vessel speeds of 2 to 51 kn, and lethal or severe injuries are most likely to occur at speeds of 14 kn or more (MMS, 2007). Most reports of collisions involve large whales, but collisions with smaller species have also been reported (Van Waerebeek et al., 2007). Laist et al. (2001) provided records of the vessel types associated with collisions with marine mammals; most severe and lethal marine mammal injuries involved large ships (80 m or more in length). Vessel speed was found to be a significant factor as well, with 89% of the records involving vessels moving at 14 kn or more (MMS, 2007).

All large marine mammals are potentially at risk of a vessel strike. Whale species that are most frequently involved in vessel collisions include the fin whale, North Atlantic right whale, humpback whale, minke whale, sperm whale, sei whale, gray whale (*Eschrichtius robustus*), and blue whale (Dolman et al., 2006). Smaller cetaceans and pinnipeds are also at risk of vessel strikes; however, these species tend to be more agile, power swimmers and are more capable of avoiding collisions with oncoming vessels (MMS, 2007).

For some species, like the North Atlantic right whale, vessel strikes pose a significant risk mainly due to behavioral characteristics and habitat preferences. Vessel strikes are consistently one of the most common causes of North Atlantic right whale mortality annually (Hayes et al., 2020). Slow-moving and deep diving species that rest while on the surface or species that traverse or occupy shipping lanes are at highest risk.

Annual large whale mortality records include a vessel strike assessment. A high number of mortalities prompted NMFS to declare a UME from January 2016 through September 2020 for Atlantic coast humpbacks (NMFS, 2020d); from January 2017 through September 2020 for minke whales (NMFS, 2020c); and from January 2017 through October 2020 for North Atlantic right whales (NMFS, 2020a). A total of 133 humpback whales and 97 minke whales were found dead between Maine and North Carolina since 2016, and 42 North Atlantic right whales were found dead or seriously injury between Newfoundland and North Carolina (NMFS, 2020a,c,d). Necropsy examinations were conducted on approximately half the humpback whales observed, of which 50% showed evidence of human interaction such as a vessel strike (NMFS, 2020d). More than 60% of the mink whales were able to be examined, of which several showed signs of human interaction, but findings were not consistent and further research is needed (NMFS, 2020c). Necropsies were able to be conducted on 20 of the 31 dead North Atlantic right whales, and although results are still pending approximately 50% of the whales examined showed evidence of vessel strikes (NMFS, 2020a). Between 2013 through 2017, there was 0.8 records of annual vessel strikes of fin whales and 0.8 records annual vessel strikes of sei whales which resulted in serious injury or mortality (Hayes et al., 2020).

Most fast-moving cetacean species, including several delphinids such as the bottlenose and common dolphin, actively approach vessels to swim within the pressure wave produced by the vessel's bow and are at lower risk of vessel strike (Glass et al., 2009; Jensen and Silber, 2003; Laist et al., 2001; van der Hoop et al., 2015).

Project vessel traffic will result in a relatively short-term increase in the volume and movement of vessels in the Project Area during construction and decommissioning. Larger work vessels will generally transit to the work location and remain in the area until installation is complete. These large vessels will move slowly over a short distance between work locations. Transport vessels will travel between ports in Massachusetts, Rhode Island, Connecticut, New York, New Jersey, Virginia, and Maryland and the offshore construction area (**Section 3.0** of the COP). During O&M, Project vessel traffic will be present over a longer duration, but the general size and number of vessels used for routine maintenance will be smaller than that of construction and decommissioning, except in the event major maintenance is required in which case traffic will be similar to construction and decommissioning. Depending on the time of year, the Project-related increase in vessel traffic would be nominal compared to other vessel operations within the area. For this analysis, it is expected that the proposed additional volume of vessel traffic associated with Project Activities would not constitute a significant increase to existing vessel traffic within the relatively heavy trafficked RI-MA WEA due to the close proximity of shipping lanes. To mitigate marine mammal vessel strikes, BOEM and NOAA require vessel strike avoidance measures that are based on NMFS's Vessel Strike Avoidance Measures and Reporting for Mariners (NMFS, 2008). Adherence to these provisions would further reduce the risk of associated vessel strikes or disturbance to marine mammals that might result from the proposed RWF construction activities or subsequent decommissioning activities.

The temporary increase in traffic during the construction and decommissioning phases pose the highest risk of vessel strikes to marine mammals. As previously discussed, not all marine mammal species are uniformly affected by vessel strikes. Some species have a higher risk of collision with vessels given their size, mobility, and surface behavior. Due to the low populations estimates for Endangered whale species, vessel strikes that may result in injury or mortality would result in the removal of that animal from the population; however, the severity of a mortality in a population that is listed as Endangered is countered by their overall resilience to population-level impacts. Vessel traffic during the activity is not expected to result in vessel strikes. Adherence to all NOAA and lease-stipulated speed restrictions and watch requirements by Project-related vessels reduces the risk of vessel strikes. Due to the relatively short duration of construction and decommissioning activities (approximately 18 months each), only ***direct***, ***short-term*** impacts are anticipated for all marine mammals. Vessel traffic during O&M will use vessels which will be generally smaller in size but will make more transits between the port and the RWF on a regular basis for maintenance and repairs throughout the operational life of the Project; therefore, impacts on all marine mammal species during this phase are therefore considered ***direct*** and ***long-term***.

5.2.3 Habitat Alteration

As introduced in **Section 4.3.4.2** of the Project's COP, impacts of habitat alteration on marine mammals during construction of the RWF are expected to be ***direct*** and ***short-term***. Seafloor preparation, installation of the foundations, vessel anchoring, and installation of the IAC and OSS-Link Cable will temporarily displace existing communities both on and in the sediment in the RWF, which is expected to alter the existing benthic habitat. Marine mammals foraging in the RWF area may experience a temporary loss in prey availability, and those species that forage on benthic species will encounter reduced foraging opportunities where soft-bottom communities are displaced by the placement of the foundations and scour protection. This is not anticipated to produce measurable impacts on marine mammals because the area altered by the RWF foundations represent a portion of available habitat for benthic communities in the region, and pelagic species are expected to return to the area following construction.

Impacts on marine mammals due to habitat alteration are expected to occur primarily during the O&M phase. During O&M the presence of the WTG and OSS foundations and scour protection, and the IAC and OSS-Link Cable protection in the RWF will alter the existing sandy-bottom habitat and provide structural relief that may act as an artificial reef, a phenomenon termed the "reef effect." The reef effect caused by the introduction of a new hard bottom habitat in this area is expected to attract numerous species of algae, shellfish, and finfish to this site (Langhamer, 2012; Reubens et al., 2013; Wilhelmsson et al., 2006). Colonization of these structures often follows a characteristic sequence, starting with settlement of smaller planktonic organisms such as algae and zooplankton followed by barnacles and other organisms that live on the seafloor or on structures in the water column (Langhamer, 2012). Fish and invertebrate species are also likely to aggregate around the foundations and scour protection, which could provide increased prey availability and structural habitat (Boehlert and Gill, 2010; Bonar et al., 2015). This can have a positive side effect, by creating a sanctuary area for trawled organisms where higher survival of larger fish species is an expected outcome that can extend to outer areas (Langhamer, 2012).

Long-term studies of artificial reefs in European seas indicate that it takes approximately 5 years before stable communities are established (Jensen et al., 2000; Petersen and Malm, 2006). The Project is anticipated to operate over a 20- to 35-year period, making it likely that colonization of the foundations and scour protection will occur. This will result in an increase in the availability of marine mammal prey species, thus providing beneficial foraging opportunities for some marine mammals in this region. Projects to restore artificial reefs noted an increase in the presence of harbor porpoises at the new artificial reef site compared to surrounding habitats, and it was hypothesized they were following prey species (Mikkelsen et al., 2013). Other studies have observed seals concentrating their foraging efforts around wind farms and oil and gas platforms, often returning to these areas, which suggests successful foraging behavior around the foundations (Arnould et al., 2015; Russell et al., 2014). Another benefit for some species is that windfarms are not just a single structure, but a series of many located relatively closely to each other. This presents many feeding opportunities for smaller species of dolphins with low body fat percentages (that require multiple feedings) or mother/calf pairs (that have been observed repeatedly at structures in the literature) (Lindeboom et al., 2011; Hammar et al., 2010).

However, this effect will not be universal across marine mammal species. Currently, there are no quantitative data on the responses of large whale species (i.e., mysticete species) to the presence of offshore wind farms. It is uncertain whether large whale species will avoid or be attracted to the RWF structures, and Kraus et al. (2019) indicated that this potential shift in large whale distribution is a critical issue to consider as offshore wind farms are developed. It is possible that they may face similar beneficial foraging opportunities as smaller odontocetes and seals; however, differences in prey preference will result in differences in impacts on marine mammal species. The presence of the foundations in the water column could create wakes that may disrupt aggregations of zooplankton prey species within the RWF. This could impact species such as the North Atlantic right whale who primarily feed on zooplankton, but benthic and pelagic fish and shellfish would not be affected by the wakes, so whales foraging on these prey species would not be impacted (Kraus et al., 2019).

Large whale species could also be impeded by the presence of the foundations in the water column. As discussed in **Section 3.0** of the Project's COP, up to 100 foundations spaced approximately 1.85 km may be installed. Larger marine mammal species and those that engage in foraging behaviors, such as bubble-net feeding performed by humpback whales or surface active groups observed for North Atlantic right whales, may be affected by the foundations in the water column compared to smaller species or species that forage independently.

While limited data are available on the long-term effects of habitat alteration due to the installation of an offshore wind farm, the primary impact on marine mammals would be from altered prey distribution. For some species, this impact could be beneficial due to increase foraging opportunities, while other species may experience difficulties foraging within the RWF area due to the presence of the foundations. Because the three-dimensional habitat introduced by the RWF foundation will be present throughout the 20-35 year life of the Project, impacts from habitat alteration due to the installation of the RWF are considered ***direct*** and ***long-term*** for marine mammals during O&M.

5.3 Sea Turtles

Sea turtles are primarily present in the Project Area during summer and fall months and can occur in the RWF and RWEC corridor depending on the species and age class. As shown in **Table 1.2-1**, IPFs for sea turtles include underwater noise, vessel traffic (i.e., physical disturbance, risk of strikes), and habitat alteration due to the presence of RWF foundations and scour protection.

5.3.1 Underwater Noise

Few studies have examined the role of acoustic cues in relation to sea turtle ecology (Cook and Forrest, 2005; Mrosovsky, 1972; Samuel et al., 2005). Sea turtles may use noise for navigation, locating prey, avoiding predators, and environmental awareness (Dow Piniak et al., 2012a). The few vocalizations described for sea turtles are restricted to the grunts and gular (throat) pumps of nesting females, which are LF sounds and are relatively loud when compared to ambient noise, leading to speculation that nesting females may use these sounds to communicate within species (Cook and Forrest, 2005; Mrosovsky, 1972). Very little is known about the extent to which sea turtles use their auditory environment ("soundscape") for navigation, assessment of their environment, or identification of predators and prey, and the acoustic habitat for sea turtles change with each life stage as the preferred habitat shifts (**Section 3.2**). For example, the inshore acoustic habitat where juvenile and adult sea turtles generally reside is dominated by LF noise and generally has higher ambient noise levels than the open ocean environment where hatchlings reside (Hawkins and Myrberg, 1983). Moreover, in highly trafficked inshore areas, nearly constant LF noises from shipping, recreational boating, and seismic surveys increase the potential for acoustic impact (Hildebrand, 2005, 2009) and masking of biologically important sounds (Fay, 2009).

Popper et al. (2014) made a distinction between "mortal injury" and "recoverable injury," with the latter defined as an injury that is not likely to result in mortality such as sensory hair cell damage, minor internal or external hematoma. The definition of "recoverable injury" in this context implicitly includes PTS due to permanent inner-ear hair cell damage because the term "recoverable injury" is defined as any injury that is not a mortal injury. Therefore, PTS could be considered a threshold for injury, as it has been used for marine mammals (NMFS, 2018).

Due to the lack of data on sea turtle hearing and auditory impacts, no quantitative TTS criteria for sea turtles have been developed. Some previous environmental analyses have applied cetacean TTS criteria to sea turtles (BOEM, 2013; U.S. Department of the Navy, 2001). Finneran and Jenkins (2012) developed TTS criteria for sea turtles based on criteria for LF cetaceans, with the inclusion of an auditory weighting function for sea turtles. However, Popper et al. (2014) concluded that sea turtle hearing is better represented by data from fishes than from marine mammals because the functioning of the inner ear of sea turtles is dissimilar to that of mammals. Popper et al. (2014) used data from fishes exposed to impact pile driving to develop criteria for death or mortal injury of sea turtles exposed to impulsive noises.

The potential for masking impacts on sea turtles is difficult to evaluate because the role of noise in their ecology is not known. Sea turtles can hear LF noises. It has been hypothesized that the natural noise of the surf zone may help nesting sea turtles find their nesting site (Nunny et al., 2011) and that grunts made by nesting sea turtles may be for terrestrial communication (Cook and Forrest, 2005). Ferrara et al. (2014) identified four types of sounds in leatherback sea turtle nests during incubation and hypothesized that sounds are used to coordinate group behavior in hatchlings. Recent studies of a freshwater turtle species identified 11 types of sounds that are used to synchronize behavior among hatchlings and coordinate the movements of hatchlings and adult females (Ferrara et al., 2013).

Sources of noise resulting from Project Activities that have the potential to impact sea turtles include both impact and vibratory pile driving during the construction phase, DP vessel thrusters throughout all Project phases, and WTG noise during the O&M phase. Construction activities, specifically impact pile driving, are likely to generate the greatest noise levels, which can result in physiological injury or behavioral disturbances to sea turtles. Severity of impacts depends on the level and frequency characteristics of the noise as well as anticipated presence of sea turtle species.

5.3.1.1 DP Vessel Noise

Underwater noise generated by Project-related vessels, including those using DP thrusters, and equipment noise could disturb sea turtles or contribute to auditory masking throughout all phases of the Project. The intensity of this noise is largely related to vessel size and speed as well as thruster operations on DP vessels. Quantitative modeling was not conducted for this Project, a qualitative discussion of DP vessel noise is provided in Denes et al. (2020).

The most likely effects of vessel noise on sea turtles would include behavioral changes and auditory masking. Vessel noise is transitory, and the SLs are too low to cause death or injuries such as auditory threshold shifts. Based on existing studies on the role of hearing in sea turtle ecology, it is unclear whether masking resulting from vessel noise would have biologically significant impacts on sea turtles. Behavioral responses to vessels have been observed but are difficult to attribute exclusively to noise rather than to visual or other vessel cues. Studies of sea turtles are also inconclusive as to whether they may habituate to a continuous noise source. Nevertheless, it is conservative to assume that noise associated with Project DP vessels may elicit behavioral changes in individual sea turtles near the vessels. It is assumed that these behavioral changes would be limited to evasive maneuvers such as diving, changes in swimming direction, or changes in swimming speed to distance themselves from vessels. Also, as indicated in **Section 5.1.2**, the low volume of Project-related vessel traffic relative to existing traffic would contribute a nominal amount to the overall noise levels in an already heavily trafficked area. Given that impacts would only occur while the limited number of DP vessels are operating during construction and decommissioning, and DP vessels operating in a station-keeping mode, which produces the greatest sound levels, are expected to occur infrequently during O&M, it is expected that impacts to sea turtles from vessel noise are considered ***direct*** and ***short-term***.

5.3.1.2 Aircraft Noise

Noise produced from aircrafts used during Project construction have the potential to propagate underwater at levels that could be detectable to sea turtles. Received SPL measured from a helicopter at 18 m depth were approximately 106 dB re 1 μPa and were shown to generally increase with decreasing water depth, decreasing altitude of the aircraft, and increasing flight speed (Patenaude et al., 2002). Additionally, sea turtles are known to be able to detect lower frequency noises and recordings of helicopter noise show primary frequencies below approximately 400 Hz (Patenaude et al., 2002; Dow Piniak et al., 2012a; Dow Piniak et al., 2012b; Martin et al., 2012; Popper et al., 2014). However, helicopters would only be used intermittently to support crew transfers during construction and O&M (**Section 4.1.4.1** of the Project's COP), and given the relatively short duration of construction activities (approximately 18 months), only temporary changes in behavior are expected to occur. Impacts from aircraft noise are considered ***direct*** and ***short-term***.

5.3.1.3 Geophysical Surveys

As discussed in **Section 2.1.5**, geophysical surveys will be conducted prior to construction of the RWF and RWEC to identify any seabed obstructions or potential MEC/UXOs. The likelihood of encountering MEC/UXOs within the Project Area is low, and should one be identified it will be disposed of using methods designed to avoid potential detonation of the device. The preferred approach for MEC/UXO is avoidance, but in a situation where avoidance is not possible, low-noise methods of removal or relocation will be employed (**Section 3.3.3.2** of the Project's COP). Therefore, explosive decommissioning of MEC/UXOs is not considered in this assessment, and only noise from the geophysical survey equipment used to locate potential obstructions was analyzed.

Equipment used during these surveys has the potential to produce noise that would exceed physiological and behavioral thresholds for sea turtles (**Section 4.1**). However, based on previous assessments conducted for marine mammals (CSA Ocean Sciences Inc., 2018, 2020) estimated ranges to physiological thresholds are not expected to exceed more than a few meters, and behavioral thresholds would be <200 m. With the implementation of the environmental protection measures outlined in **Section 5.5**, the risk of impact is low and would be limited to temporary disturbances. Furthermore, due to the relatively short duration of these activities which would only occur during a portion of the full 18-month construction period, impacts are considered ***direct*** and ***short-term***.

5.3.1.4 Impact Pile Driving

Available data indicate that adult sea turtles in water can hear frequencies ranging from 50 Hz to 1,200 Hz and juveniles can hear frequencies up to 1,600 Hz, a range that overlaps with the main energy output from impact pile driving (Bartol and Ketten, 2006; Bartol et al., 1999; Dow Piniak et al., 2012a; Lavender et al., 2014; Martin et al., 2012; Ridgway et al., 1969). Reported hearing ranges and thresholds differ somewhat among species and life stages, but the data are too limited to be definitive because of the small numbers of individuals tested. Death or injury can occur from exposure to high intensity impulsive noises (Popper et al., 2014). Sea turtle deaths and injuries have been documented in proximity to underwater explosions (Gitschlag and Herczeg, 1994; Klima et al., 1988; Viada et al., 2008), but those impacts were attributed primarily to barotrauma resulting from exposure to the high energy of the shock wave generated by the explosions. Based on an extensive review of current scientific literature and studies, no sea turtle deaths or injuries are documented to have been caused by impact pile driving. Because of their rigid external anatomy, it is possible that sea turtles may be protected to some degree from the impacts of lower energy impulsive noises (Ketten and Bartol, 2005; Popper et al., 2014).

Avoidance of impulsive noise sources by sea turtles has also been inferred from field observations of sea turtle behavior during seismic surveys (DeRuiter and Doukara, 2012; Holst et al., 2006; Weir, 2007). Based on the best available data, it is assumed that sea turtle behavioral responses to impulsive noise may begin to occur at a received SPL between 166 and 175 dB re 1 µPa (Blackstock et al., 2018; FHWG, 2008; Popper et al., 2014).

Modeled impact pile driving at RWF WTG for the 12-m WTG monopiles with 10 dB attenuation resulted in a $ER_{95\%}$ distance of 110 m to the sea turtle PK physiological threshold and 20 m to the SEL_{24h} threshold (**Table 4.4-1**). For the 15-m monopiles used in the RWF OSS, mean $ER_{95\%}$ were 81 m to the PK physiological thresholds and 13 m to the SEL_{24h} thresholds, and for the 4-m jacket foundations, mean modeled distances were 24 and 57 m for the PK and SEL_{24h} thresholds, respectively (**Tables 4.4-2** and **4.4-3**). Sea turtles are not expected to linger within this distance for durations that would elicit a physiological impact. The maximum distance to PK thresholds represents the greatest potential for instantaneous injury to sea turtles and would be reached only at the highest hammer energy near the end of pile installation (Denes et al., 2020). Due to the placement of noise attenuation devices and general construction activities combined with smaller impact isopleths for the majority of hammer strikes, sea turtles are not expected to encroach any of the PK isopleths and, therefore, no physiological exposures are expected for sea turtles from impact pile driving.

Modeled $ER_{95\%}$ for sea turtle behavioral thresholds ranged from 1,030 to 1,500 m for all pile types and scenarios (**Section 4.4**). There is a likelihood of behavioral threshold exposure and general activity in the area that could result in sea turtles temporarily vacating the RWF construction area. Exposures to behavioral thresholds are expected to be temporary and not biologically significant. Because impacts are only expected during the 18-month duration of construction activities, it is expected that impact pile driving will result in ***direct***, ***short-term*** impacts on sea turtles.

5.3.1.5 Vibratory Pile Driving

Vibratory pile driving associated with RWEC construction, while within the estimated hearing range of sea turtles, is expected to produce lower noise levels relative to impact pile driving. Modeling was not conducted for cofferdam installation for RWEC; however, no injury or mortality is expected, and behavioral exposures are unlikely due to the relatively low SLs produced by this activity (**Section 2.1.4**) and the nearshore location of the proposed cofferdam installation (**Section 3.0** of the Project's COP). If behavioral exposures were to occur, behavioral responses are expected to be temporary, short-term, and would not affect the reproduction, survival, or recovery of Threatened or Endangered species. Additionally, vibratory pile driving would only occur during a 3-day period between October and January, and winter and spring have very low densities of sea turtles in the area (**Section 3.2**) and would have a lower potential for any exposure risk. Vibratory pile driving is therefore anticipated to have ***direct***, ***short-term*** impacts on sea turtles.

5.3.1.6 WTG Operations

Sea turtle hearing is within the frequency range (<1,200 Hz) for operational WTG (Popper et al., 2014; Thomsen et al., 2006). Thus, it is possible that WTG noise may influence sea turtle behavior. Potential responses to WTG noise generated during normal operations may be expected to be behavioral and include avoidance of the noise source, disorientation, and disturbance of normal behaviors such as feeding (MMS, 2007). Noise generated during normal operations might affect many individuals and for a much longer time period (MMS, 2007). As discussed in **Section 5.1.1.4**, operational WTGs can produce SPL ranging from 100 to 120 dB re 1 µPa at roughly 100 m from the foundation, which is higher than the ambient levels measured within the RI-MA WEA (Kraus et al., 2016; Tougaard et al., 2009).

Although operational WTGs could potentially increase ambient noise levels around the RWF, the sound levels produced are not high enough to result in potential injury to sea turtles. Only behavioral disturbances such as long-term avoidance of the RWF and surrounding vicinity are likely to occur. Sea turtles are known to occur in areas of higher ambient noise given their preference for coastal habitats, and therefore are more likely to habituate to increases in ambient noise. Additionally, as discussed in **Section 5.2.3**, sea turtles will likely be attracted to the RWF foundations due to beneficial foraging and sheltering opportunities, which further indicate the potential effects of operation WTG noise will not be biologically significant. Based on this, the anticipated behavioral impacts on to sea turtles from WTG noise is not expected to be biologically significant, but will be present throughout the 20 to 35-year life of the Project and are therefore considered ***direct*** and ***long-term***.

5.3.2 Vessel Traffic

Sea turtles may be able to actively maneuver within the water column to avoid collisions with approaching slow-moving (<5 kn) construction vessels; however, construction support vessels may travel at faster speeds and sea turtles may not be able to avoid them. Based on knowledge of their sensory biology (Bartol and Ketten, 2006; Bartol and Musick, 2003; Levenson et al., 2004), sea turtles may detect objects such as vessels, prey, and predators in the water column by means of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows that they may rely more on visual than auditory cues (Hazel et al., 2007). Sea turtle collisions with commercial vessels are not well-documented, but many rescued or stranded sea turtles show evidence of vessel strikes (Singel et al., 2007). From 1997 to 2005, 14.9% of all stranded loggerhead turtles in the U.S. Atlantic and Gulf of Mexico were documented as having sustained some type of propeller or collision injury. This study did not indicate what proportion of these injuries was post- or ante-mortem (NMFS and USFWS, 2008). It is likely that collisions with small or submerged sea turtles, or collisions during nighttime or periods of poor visibility, may go undetected and undocumented. Sea turtles are negatively buoyant and remains will sink in deep water, making them very unlikely to drift to shore or be recovered.

The potential for collisions between vessels and sea turtles increases at night and during inclement weather. Sea turtles spend at least 20% to 30% of their time at the surface for respiration, basking, feeding, orientation, and mating, during which time they are more susceptible to vessel strikes (Lutcavage et al., 1997). Temporary vessel traffic during all Project phases would slightly increase vessel traffic within the area; however, it represents a very small contribution in overall vessel traffic in the already heavily trafficked region. Large construction and decommissioning vessels will generally transit to the work location and remain in the area until installation is complete. These large vessels will move slowly and over short distance between work locations. Transport vessels will travel between ports in Massachusetts, Rhode Island, Connecticut, New York, New Jersey, Virginia, and Maryland and the RWF throughout all Project phases (**Section 3.0** of the COP). These vessels will range in size from smaller crew transport boats to tug and barge vessels.

While mortality from vessel collision is frequently documented in sea turtle stranding data, the issue is most prevalent in shallow inshore and near-coastal waters where there are high densities of high-speed vessel traffic (Singel et al., 2007). In the unlikely event of a sea turtle vessel strike that results in injury or mortality, the risk of population-level consequences would be greater due to the removal of an individual(s) from a population or DPS that is considered already at risk. However, considering that Project-related vessel traffic will comprise slower moving work vessels and a relatively low volume of support vessels, and that vessel strike avoidance measures including speed restrictions and minimum separation distances following guidance from NMFS (2008) will be implemented for all Project vessels, the risk of a strike is expected to be low. Therefore, potential impacts on sea turtles from vessel traffic during construction and decommissioning are considered ***direct*** and ***short-term*** due to the relatively short duration of these activities (approximately 18 months each). As discussed briefly in **Section 5.2.2**, vessel traffic during the O&M phase is expected to comprise smaller vessels but a higher number of transits compared to the construction and decommissioning phases throughout the 20-35 year life of the Project, and impacts are therefore considered ***direct*** and ***long-term***.

5.3.3 Habitat Alteration

The presence of the RWF foundations and scour protection and IAC and OSS-Link Cable protection throughout the 20 to 35 year life of the Project will alter the existing sandy-bottom habitat and structural relief that may act as an artificial reef, a phenomenon termed the "reef effect". The reef effect caused by the introduction of a new hard bottom habitat in this area is expected to attract numerous species of algae, shellfish, finfish, and sea turtles to this site (Langhamer, 2012; Reubens et al., 2013; Wilhelmsson et al., 2006). For sea turtles, artificial reefs have been shown to provide a number of ecological functions such as foraging and sheltering habitat and structures are used to remove biological build-up from their carapace (Barnette, 2017; NRC, 1996). In the Gulf of Mexico, both loggerhead and leatherback turtles were often observed resting at oil and gas platforms, making it likely that these species will behave similarly at the proposed windfarm structures (Gitschlag and Herczeg, 1994; NRC, 1996). The increased abundance of benthic species such as mussels and crabs, as well as the pelagic fish species attracted to this site would provide foraging opportunities for sea turtles transiting this site. Colonization of offshore structures often follows a characteristic succession starting with lower trophic level species such as diatoms and algae followed by upper trophic level species (Langhamer, 2012). Long-term studies indicate that it takes approximately 5 years for a stable community to be established, but biomass coverage of mussel species at these artificial structures has been shown to dramatically increase within the first 2 years (Joschko et al., 2008; Petersen and Malm, 2006). Particularly in areas with minimal hard bottom habitat or structural relief, these artificial reefs may supply important inter-nesting habitats for sea turtles (Barnette, 2017). Multiple species like green, hawksbill (*Eretmochelys imbricata*), and loggerhead sea turtles have also been observed using anthropogenic structures and submerged rocks to clean their flippers and carapace (Barnette, 2017). With the proposed foundations and scour protection, it is likely this will be result in a beneficial impact to sea turtles due to increased structural habitat and foraging opportunities.

The habitat conversion is also expected to attract commercial and recreational fishing to the area, which could pose a threat to sea turtles through entanglement or ingestion of fishing gear. Greater fishing effort around RWF area would increase the amount of equipment in the water, particularly monofilament line, which has been identified as a major hazard for all sea turtle species. Additionally, the beneficial foraging and sheltering opportunities for sea turtles could cause them to remain in the area longer than they typically would, making them more susceptible to cold stunning. Wakes created by the presence of the foundations may also influence distributions of drifting jellyfish aggregations; however, since other prey species available to sea turtles will not be affected by these wakes, impacts on sea turtle foraging are not expected to be substantial (Kraus et al., 2019). Given the available data that suggests an attraction of sea turtles to offshore structures and because the newly created habitat by the RWF foundations will be present throughout the 20-35 year life of the Project, impacts on sea turtles are considered ***direct*** and ***indirect***, and ***long-term*** during O&M.

Limited information is available related to the effect of decommissioning these structures after artificial reef habitat has been formed. The majority of research examining the impacts of decommissioning offshore structures focuses on methods involving explosives, which will not be used for this Project. Revolution Wind plans to fully dismantle the RWF components and either remove them from the seabed completely or cut the foundations at an appropriate depth below the mudline, enabling the environment to return to near baseline conditions. Sea turtles using these structures for foraging and shelter will be negatively impacted; however, the level of impact from removal of this habitat is uncertain. Studies of manatees at power plants in Florida indicate that they become dependent on these structures as habitat and struggle to adapt when they are decommissioned (Laist, 2005; Sattelberger, 2017). Given the propensity for sea turtles to utilize artificial reef habitats created by offshore structures, the current listing status of local sea turtles, and the expected loss of beneficial habitat used for foraging and shelter, potential negative impacts from decommissioning of the RWF are expected. However, because of the relatively short duration of decommissioning activities, and the anticipated return to baseline once the Project components are removed, impacts would be considered ***direct*** and ***short-term***.

5.4 Atlantic Sturgeon

Potential impacts on the Atlantic sturgeon would not be substantially different from impacts on other fish species and species with designated Essential Fish Habitat. No spawning habitat will be affected as Atlantic surgeon spawn in hard-bottom, freshwater habitats. Seasonal migratory patterns present the potential for Atlantic sturgeon to be present in the RWF area; however, it is not expected to be a regular visitor or occupant in large numbers. As shown in **Table 1.2-1**, IPFs for Atlantic sturgeon that could reach greater than negligible determinations include underwater noise and vessel traffic (i.e., physical disturbance, risk of strikes).

5.4.1 Underwater Noise

Atlantic sturgeon have a primitive swim bladder that is not connected to the inner ear. Anatomical and physiological variations make it difficult to generalize about the impacts of noise on individual species (Thomsen et al., 2006). There are few studies specific to sturgeon hearing; however, Popper (2005) estimated that noise detection in sturgeon ranged from <100 Hz up to 1,000 Hz and indicated that sturgeon may be able to localize noise sources (i.e., determine the direction from which it comes). Sturgeon produce vocalizations during spawning, indicating some level of acoustic dependence for critical biological functions.

A workshop report is available, which contains a summary of research on fish hearing and physiology and presents audiograms for fish that have been measured under appropriate acoustic conditions (Normandeau, 2011). However, as discussed in **Section 2.3.2**, there is a gap in the understanding of particle motion sensitivity in fish, as few studies examined both the effects of pressure and particle motion simultaneously. It is expected that particle motion associated with impulsive noise sources such as impact pile driving will have similar effects as pressure waves with fish exhibiting behavioral responses such as temporarily vacating the impact area. Excess particle motion may also mask communication and could cause permanent or temporary damage to sensory structures.

There are only limited data on mortality in response to anthropogenic noise, and it is not clear whether death or injury only occurs in close proximity to a noise source (Hawkins et al., 2014). Overall, it is more likely that fish will experience sub lethal impacts that increase the possibility for delayed mortality when exposure occurs near a source (Hawkins et al., 2014). Because the majority of Project activities produce non-impulsive LF noise that is within the sensitive hearing range of most fish, the potential for fish to experience TTS, masking, and behavioral impacts are a higher likelihood than auditory injury or mortality.

Behavioral responses (e.g., fleeing, avoidance) to active acoustic noise sources are the most likely direct effect for Atlantic sturgeon exposed to noise during Project activities. Fewtrell and McCauley (2012) found that fish exhibited alarm responses to airgun noises exceeding SEL_{24h} between 147 and 151 dB re 1 μPa^2 s. The potential for masking or behavioral response may exist at a distance of many kilometers from a noise source, depending on the ambient noise levels in the region and the frequency and amplitude characteristics of the noise source.

5.4.1.1 DP Vessel Noise

Research indicates that the direct effects of DP vessel noise will not cause mortality or barotraumatic injuries in adult fish (Hawkins et al., 2014). DP vessel SLs have been shown to cause several different behavioral responses, TTS, auditory masking, and changes in blood chemistry. The most common behavioral responses are avoidance, alteration of swimming speed and direction, and alteration of schooling behavior (Becker et al., 2013; Handegard and Tjøstheim, 2005; Sarà et al., 2007; Vabø et al., 2002).

Laboratory and field studies have demonstrated several other behaviors that are influenced by DP vessel noise. For example, several studies noted changes in the time spent burrowing or using a refuge, time spent defending or tending to nests and eggs (Bruintjes and Radford, 2013; Picciulin et al., 2010), intraspecific aggression and territoriality interactions (Bruintjes and Radford, 2013; Sebastianutto et al., 2011), foraging behavior (Bracciali et al., 2012; Purser and Radford, 2011; Voellmy et al., 2014a,b), vocalization patterns (Picciulin et al., 2008, 2012), and overall frequency of movement (Buscaino et al., 2010). These studies also demonstrated that behavioral changes were generally temporary or that fish habituated to the noises. Some studies noted changes in the blood chemistry of several fish species (e.g., European sea bass [*Dicentrarchus labrax*], gilthead seabream [*Sparus aurata*], red drum [*Sciaenops ocellatus*], spotted sea trout [*Cynoscion nebulosus*]) in response to vessel noise (Buscaino et al., 2010; Spiga et al., 2012).

Auditory masking and TTS in fish exposed to vessel noise has been demonstrated in a few studies. Auditory thresholds have been shown to increase by as much as 40 dB when fish are exposed to vessel noise playbacks (Codarin et al., 2009; Wysocki and Ladich, 2005; Vasconcelos et al., 2007). The degree of auditory masking or TTS generally depends on the hearing sensitivity of the fish, the frequency, and the noise levels tested (Wysocki and Ladich, 2005). The impact of auditory masking and TTS indicate that vessel noise can lower the ability of fish to detect biologically relevant sounds, but the effects were found to be temporary and hearing abilities returned to normal after cessation of the vessel noise.

Modeling was not conducted for DP vessel noise for this Project, but a qualitative discussion of noise produced by DP vessels can be found in Denes et al., 2020. It is unlikely that Atlantic sturgeon would be exposed to DP vessel noise associated with the Project because of their sparse spatial distribution in the Project Area and habitat preference of estuaries and rivers adjacent to, and occasionally in, coastal and shelf waters. Given these factors, and because impacts would only occur while the limited number of DP vessels are operating during construction and decommissioning, and DP vessels operating in a station-keeping mode, which produces the greatest sound levels, are expected to occur infrequently during O&M, impacts of DP vessel noise on Atlantic sturgeon are considered ***direct*** and ***short-term***.

5.4.1.2 Aircraft Noise

Noise produced from aircrafts used during Project construction have the potential to propagate underwater at levels that could be detectable to Atlantic sturgeon. Received SPL measured from a helicopter at 18 m depth were approximately 106 dB re 1 µPa and were shown to generally increase with decreasing water depth, decreasing altitude of the aircraft, and increasing flight speed (Patenaude et al., 2002). Additionally, most fish species are known to be able to detect lower frequency noises and recordings of helicopter noise show primary frequencies below approximately 400 Hz (Patenaude et al., 2002; Dow Piniak et al., 2012a,b; Martin et al., 2012; Popper et al., 2014). However, helicopters would only be used intermittently to support crew transfers during construction and O&M (**Section 4.1.4.1** of the Project's COP), and given the relatively short duration of construction activities (approximately 18 months), only temporary changes in behavior are expected to occur. Impacts from aircraft noise are considered ***direct*** and ***short-term***.

5.4.1.3 Geophysical Surveys

As discussed in **Section 2.1.5**, geophysical surveys will be conducted prior to construction of the RWF and RWEC to identify any seabed obstructions or potential MEC/UXOs. The likelihood of encountering MEC/UXOs within the Project Area is low, and should one be identified it will be disposed of using methods designed to avoid potential detonation of the device. The preferred approach for MEC/UXO is avoidance, but in a situation where avoidance is not possible, low-noise methods of removal or relocation will be employed (**Section 3.3.3.2** of the Project's COP). Therefore, explosive decommissioning of MEC/UXOs is not considered in this assessment, and only noise from the geophysical survey equipment used to locate potential obstructions was analyzed.

Equipment used during these surveys has the potential to produce noise that would exceed physiological and behavioral thresholds for fish (**Section 4.1**). However, based on previous assessments conducted for marine mammals (CSA Ocean Sciences Inc., 2018, 2020) estimated ranges to physiological thresholds are not expected to exceed more than a few meters, and behavioral thresholds would be <200 m. With the implementation of the environmental protection measures outlined in **Section 5.5**, the risk of impact is low and would be limited to temporary disturbances. Furthermore, due to the relatively short duration of these activities which would only occur during a portion of the full 18-month construction period, impacts are considered ***direct*** and ***short-term***.

5.4.1.4 Impact Pile Driving

Impact pile driving is an impulsive noise source that has the potential to cause barotrauma at close ranges (Halvorsen et al., 2012a,b). Because the effect of changing pressure on the swim bladder is the underlying cause of barotrauma, fish without swim bladders like elasmobranchs (i.e., sharks, skates, rays) and flatfish are not as vulnerable to underwater noise impacts as those with swim bladders. Atlantic sturgeon have a relatively small swim bladder which is not directly connected to the inner ear, and they are able to voluntarily release gas from their swim bladder. Therefore, the risk of barotrauma due to exposure to impulsive signals from impact pile driving is lower relative to fish species that cannot release swim bladder gas.

Anticipated noise levels during RWF construction may exceed behavioral thresholds for fish, including Atlantic sturgeon, and may elicit a behavioral avoidance response as observed for some fish species (Becker et al., 2013). A physiological stress response or TTS may also occur due to exposure to impact pile driving noise. The stress response may involve elevated levels of stress hormones (i.e., corticosteroids) as documented for fish exposed to continuous SPL of 153 to 170 dB re 1 µPa (Smith et al., 2004; Wysocki et al., 2006) or increased heart rate following exposure to elevated SPL (Graham and Cooke, 2008).

Elevated noise levels are expected to cause Atlantic sturgeon to temporarily vacate the area (Krebs et al., 2016), resulting in a temporary disruption of feeding, mating, and other essential activities. Atlantic sturgeon have been shown to avoid impact pile driving activities in the Hudson River, and based on this, they were not expected to be exposed to the SEL_{24h} produced by this activity (Krebs et al., 2016). The same avoidance response is expected should Atlantic sturgeon be present during impact pile driving activities at the RWF given the highly mobile nature of this species.

Mean modeled acoustic ranges to Atlantic sturgeon SEL_{24h} thresholds with 10 dB attenuation were approximately 423 m for the 12-m WTG monopile foundations, 945 m for the 15-m OSS monopile foundations, and 781 m for the 4-m OSS jacket foundations (**Section 4.3**). PK ranges were generally smaller, ranging from 42 to 98 m for all pile types and scenarios with 10 dB attenuation applied (**Section 4.3**). Average acoustic ranges for behavioral thresholds were 7 to 8 km for all pile types and scenarios (**Section 4.3**). As discussed in earlier sections, exposure to behavioral thresholds does not constitute behavioral responses, nor are they expected to create any biologically significant consequences.

Atlantic sturgeon are an anadromous species that primarily utilize rivers, bays, estuaries, coastal, and shallow continental shelf waters. However, since Atlantic sturgeon are a demersal species that could potentially be present in the RWF area during impact pile driving activities, behavioral impacts could occur. Because impacts to Atlantic sturgeon from impact pile driving would only occur during the approximate 18-month construction period, impacts are considered ***direct*** and ***short-term***.

5.4.1.5 Vibratory Pile Driving

Vibratory pile driving generally poses less risk of an acoustic impact to fish than impact pile driving because of the non-impulsive nature of the noise produced by vibratory hammers. Unlike impact hammers, which are classified as an impulsive noise source, the sound energy produced by vibratory hammers rises more gradually and SLs are typically 10 to 20 dB lower than those for impact hammers (Buehler et al., 2015).

Vibratory pile driving is not known to produce noise levels that cause mortality in fish due to the non-impulsive nature of this noise source. As such, there are no biological thresholds for mortality associated with non-impulsive noise sources. Modeling was not conducted for cofferdam installation for RWEC; however, information regarding the acoustic properties of DP vessels is provided in Denes et al. (2020). Atlantic sturgeon that are present within the area ensonified at levels exceeding the behavioral threshold are expected to move away from the noise source and avoid the area where the physiological threshold would be exceeded during vibratory pile driving.

Underwater noise produced during vibratory pile driving for the installation and removal of temporary cofferdams would be intermittent and short term, after which, the potential acoustic impacts to Atlantic sturgeon posed by cofferdam installation would no longer be present. Based on these factors and the results of previous acoustic modeling for the South Fork Wind Farm, which demonstrate the relatively small spatial extent of acoustic impacts as well as the likely avoidance of this activity by Atlantic sturgeon, there is a low risk of acoustic impacts to this species. Because impacts are only expected during the approximate 3-day period anticipated for vibratory pile driving for installation of temporary cofferdams at RWEC, impacts are considered ***direct*** and ***short-term***.

5.4.1.6 WTG Operations

Noise produced by WTGs is within the hearing range of Atlantic sturgeon. Depending on the noise intensity, such noises could disturb or displace fish within the surrounding area or cause auditory masking (MMS, 2007). However, with generally low noise levels, fish would be impacted only at close ranges (within 100 m) (Thomsen et al., 2006). Thomsen et al. (2006) reviewed the observations of fish behaviors in proximity to an operational WTG and found varying results, from no perceived changes in swimming behavior of European eels (*Anguilla anguilla*) and both increased and decreased catch rates of cod within 100 m of the operational WTGs. Additionally, Atlantic sturgeon are an anadromous species that primarily utilize rivers, bays, estuaries, coastal, and shallow continental shelf waters, and their occurrence in the RWF is expected to be seasonal in very low numbers (**Section 3.3.1**). While there may be some behavioral modifications, these would be localized and would not represent any population-level changes. Therefore, impacts from WTG noise on Atlantic sturgeon are considered ***direct*** and ***long-term***, given the anticipated 20 to 35-year life of the Project.

5.4.2 Vessel Traffic

The potential for Atlantic sturgeon to be struck by a vessel is high and vessel strikes are a fairly common occurrence. Between 2005 and 2008, surveys in the Delaware estuary reported a total of 28 Atlantic sturgeon mortalities, of which 50% were the result of an apparent vessel strike (Brown and Murphy, 2010). Similarly, five Atlantic sturgeon were reported to have been struck by commercial vessels within the James River, Virginia, in 2005, and one strike per 5 years is reported for the Cape Fear River, North Carolina. The majority of strikes occurred near busy ports where entrance channels narrow, or a significant portion of estuary and river habitat is transited by commercial vessels entering a port (Brown and Murphy, 2010).

As previously mentioned, vessel traffic during construction and decommissioning of the RWF would result in a temporary increase vessel traffic within the area; however, it represents a very small contribution in overall vessel traffic in the already heavily trafficked region. Larger construction vessels will generally transit to the work location and remain in the area until installation is complete. These large vessels will move slowly and over short distances between work locations.

Transport vessels will travel between several ports and the RWF over the course of Project construction and decommissioning. These vessels will range in size from smaller crew transport boats to tug and barge vessels. Smaller vessels will also be used for routine maintenance trips during the O&M phase.

The Project-related increase in vessel traffic during all phases is not expected to be significant when compared to other vessel traffic within the region, and most vessels will be slow moving. Additionally, the implementation of vessel strike avoidance measures such as speed restrictions will further reduce the risk of collisions with Atlantic sturgeon. In the unlikely event that an Atlantic sturgeon is struck and injury or mortality occurs, the risk of population-level impacts would be greater given the Endangered status of this population. However, as previously stated, Atlantic sturgeon occurrence in the RWF is expected to be seasonal, and occurrence in the RWEC would be less common than the RWF (**Section 3.3.1**), making it unlikely they would incur population-level impacts due to vessel strikes. Impacts from vessel strikes are considered ***direct*** and ***short-term*** for Atlantic sturgeon during the construction and decommissioning phases, given the relatively short, 18-month duration anticipated for each. As discussed in **Sections 5.2.2** and **5.3.2**, vessels used during the O&M phase will be generally smaller, but will require more trips between the port and the RWF throughout the 20-35 year operational life of the Project, so impacts during this phase are considered ***direct*** and ***long-term***.

5.5 Avoidance, Minimization and Mitigation

Revolution Wind will implement the avoidance, minimization, and environmental protection measures considered to reduce potential impacts resulting from exposure to underwater noise and vessel traffic during construction and operation of the RWF and RWEC. Revolution Wind, through Orsted NA, is developing a comprehensive Protected Species Mitigation and Monitoring Plan (PSMMP) across all Orsted NA wind leases. The RWF PSMMP will align with all regulatory requirements from BOEM and NMFS by the time necessary for approval of the mitigation and monitoring plans. Details and implementation parameters of each mitigation measure will be provided in the final PSMMP. Additional environmental protection measures beyond those summarized here may be implemented during construction and operations of the RWF and RWEC; and those will be fully detailed in the PSMMP. The mitigation categories that will be used for RWF and REC construction include:

- Noise attenuation through use of a noise mitigation system;
- Establishment of exclusion zones;
- Visual and passive acoustic monitoring;
- Area clearance prior to start of hammer;
- Operational shutdowns and delays;
- Soft start procedures; and
- Vessel strike avoidance and other precautionary procedures.

Project-specific training will be conducted for all Project crews prior to the start of construction activities. Confirmation of the training and understanding of the requirements will be documented on a training course log sheet. Signing the log sheet will certify that the crew members understand and will comply with the necessary requirements throughout the construction activities.

5.5.1 Noise Attenuation

A noise mitigation system is any device or suite of devices that reduces pile driving sound levels that are transmitted through the water. Primary systems reduce the source levels produced by the pile and secondary systems reduce the propagated sound levels of the piling. A noise mitigation system, such as a bubble curtain, hydro damper, or similar, will be used during impact pile driving to decrease the sound levels in the water near the source and thus reduce the impact on marine mammals. Attenuation levels vary by type of system, frequency band, and location. Small bubble curtains have been measured to reduce sound levels from approximately 10 dB to more than 20 dB, but they are highly dependent on water depth, current, and configuration and operation of the curtain (Austin et al., 2016; Bellmann, 2014; Koschinksi and Lüdemann, 2013; Bellmann et al., 2020).

No noise attenuation will be used at the cofferdam due to its location, the activities occurring at the cofferdam, the short time period involved with installation and removal, and very low risk of physiological exposures when other mitigations, as descried in the following sections, are employed.

5.5.2 Establishment of Exclusion Zones

Exclusion zones (EZs) and monitoring zones (MZs) will be established within which Protected Species Observers (PSOs) will monitor for the presence of marine protected species in the vicinity of activities. The size of the EZs and MZs will be based on the type of activity being conducted and the various protected species or species groups expected within the region.

5.5.3 Visual and Acoustic Monitoring

Visual and acoustic monitoring of the established MZs will be performed by qualified and NMFS-approved PSOs. PSOs will be responsible for detecting and identifying marine mammals and sea turtles approaching the established EZs; notifying Project personnel to the presence of species as well as communicating and enforcing the action(s) that are necessary to ensure mitigation and monitoring requirements are implemented as appropriate.

5.5.4 Area Clearance

At the start of each impact pile driving activity, PSOs (and/or PAM operators) will clear the EZ before initiation of soft start procedures. A soft start may not be initiated if any marine mammal or sea turtle is observed within the EZ. If a marine mammal or sea turtle is observed within the EZ during the pre-clearance period, a soft start may not begin until the animal(s) has been observed exiting its respective zone or until a designated time period has elapsed with no further sightings.

5.5.5 Soft Start Procedures

Soft start procedures are applicable to impact pile driving only. Every pile installation will begin with a soft start procedure. The soft start procedure is detailed in **Section 3.2.4.2**. A soft start procedure is used to allow animals potentially in the Project Area to detect the presence of the noise-producing activities and depart the area before full power impact pile driving activity begins. A soft start of impact pile driving will not begin until the EZ has been cleared by the PSOs (and PAM operators when applicable), as described above.

5.5.6 Vessel Strike Avoidance and Other Protective Measures

Vessel operators and crew will maintain a vigilant watch for marine mammals and sea turtles, and slow down or stop their vessels if either are sighted to minimize the potential for a vessel strike. Survey vessel crew members responsible for navigation duties will receive site-specific training on marine mammal sighting/reporting and vessel strike avoidance measures. All vessel crew members will undergo Project-specific marine mammal and compliance training and all vessels will adhere to NOAA vessel guidelines, Lease stipulations, and additional restrictions in management areas as necessary. Vessels will maintain Lease-stipulated separation distances and safe maneuvering when in the proximity of marine mammals. Vessels will monitor NMFS North Atlantic right whale reporting systems daily. Additional measures will also be implemented to minimize non-acoustic impacts including:

- Vessels will follow NOAA guidelines for marine mammal strike avoidance measures, including vessel speed restrictions;
- All personnel working offshore will receive training on marine mammal awareness and marine debris awareness;
- All construction and operations vessels will comply with regulatory requirements related to the prevention and control of spills and discharges;
- Accidental spill or release of oils or other hazardous materials will be managed through and Oil Spill Response Plan (OSRP); and
- The IAC, OSS-Link Cable, and RWEC will be buried to a target depth of 1.2 to 1.8 m to the extent feasible. Actual burial depths and the potential need for cable protection measures will be based on a Cable Burial Risk Assessment, which will evaluate seabed conditions, seabed mobility, and risk of interaction with external hazards such as fishing gear and vessel anchors.

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