ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
BIOLOGICAL OPINION

Action Agency: Bureau of Ocean Energy Management
Army Corps of Engineers, New England District

Activity: Commercial Wind Lease Issuance and Site Assessment Activities on the Atlantic Outer Continental Shelf in Massachusetts, Rhode Island, New York and New Jersey Wind Energy Areas
NER-2012-9211

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1.0 INTRODUCTION

The Bureau of Ocean Energy Management (BOEM) regulates the development of wind energy resources on the Outer Continental Shelf (OCS). BOEM is proposing to issue leases and approve site assessment activities on the OCS in BOEM-identified wind energy areas (WEAs) off the coasts of Massachusetts, Rhode Island and New Jersey and unsolicited proposed development areas off New York. A complete description of the proposed actions is provided below. This Opinion is based on information provided in a Biological Assessment (BA) prepared by BOEM dated October 2012 and other available information as cited herein. We will keep a complete administrative record of this consultation at our Northeast Regional Office.

1.1 Programmatic Consultations

NMFS has developed a range of techniques to streamline the procedures and time involved in consultations for broad agency programs or numerous similar activities with predictable effects on listed species and critical habitat. Some of the more common of these techniques and the requirements for ensuring that streamlined consultation procedures comply with section 7 of the ESA and its implementing regulations are discussed in the October 2003 joint Services memorandum, *Alternative Approaches for Streamlining Section 7 Consultation on Hazardous Fuels Treatment Projects* (http://www.fws.gov/endangered/pdfs/MemosLetters/streamlining.pdf; see also, 68 FR 1628 (January 13, 2003)). Pursuant to this guidance, programmatic consultations may be conducted on any Federal agency's proposal to apply specified standards or design criteria to future proposed actions. Programmatic consultations can be used to evaluate the anticipated effects of groups of related agency actions expected to be implemented in the future, where specifics of individual projects such as project location are not definitively known. A programmatic consultation must identify project design criteria and/or standards that will be applicable to all future projects implemented under the consultation document. These criteria and standards serve to prevent adverse effects to listed species (informal consultation), or to limit adverse effects to predictable levels that will not jeopardize the continued existence of listed species or destroy or adversely modify critical habitat, at the individual project level or in the aggregate from all projects implemented under a programmatic Biological Opinion (formal consultation). Programmatic consultations allow for streamlined project-specific consultations because the effects analysis is completed up front in the programmatic consultation document. At the project-specific consultation stage, a proposed project is reviewed to determine if it can be implemented according to the criteria or standards under the programmatic consultation. Consistent with the 2003 memo referenced above, the following elements should be included in a programmatic consultation to ensure its consistency with ESA section 7 and its implementing regulations.

1. Project design criteria (PDC) to prevent or limit future adverse effects on listed species and critical habitat;

2. Description of the manner in which projects to be implemented under the programmatic consultation may affect listed species and critical habitat and evaluation of expected level of effects from covered projects;

3. Process for evaluating expected, and tracking actual aggregate or net additive effects of all projects expected to be implemented under the programmatic consultation. The

programmatic consultation document must demonstrate that when the PDCs or standards are applied to each project, the aggregate effect of all projects will not adversely affect listed species or their critical habitat (for informal consultations) or are not likely to jeopardize the continued existence of listed species or destroy or adversely modify critical habitat (for formal consultations);

4. Procedures for streamlined project-specific consultation. As discussed above, if an approved programmatic consultation document is sufficiently detailed, project-specific consultations ideally will consist of certifications between action agency biologists and consulting agency biologists, respectively. An action agency biologist or team will provide a description of a proposed project, or batched projects, and a certification that the project(s) will be implemented in accordance with the criteria or standards. The consulting agency biologist reviews the submission and provides certification, or adjustments to the project(s) necessary to bring it (them) into compliance with the programmatic consultation document.

5. Procedures for monitoring projects and validating effects predictions; and,

6. Comprehensive review of the program, generally conducted annually.

All of these procedures and criteria are included in this consultation.

2.0 CONSULTATION HISTORY

In May 2012, BOEM requested concurrence with their determination that the activities associated with the issuance of leases and approval of site assessment activities at the MA, RI, NY and NJ wind energy areas were not likely to adversely affect any NMFS listed species. During the summer of 2012, BOEM developed new sound modeling information in support of the ongoing section 7 consultation being carried out between BOEM and NMFS' Office of Protected Resources (OPR; located at NMFS Headquarters in Silver Spring, MD) regarding the effects of geophysical and geotechnical (G&G) activities in the Mid and South Atlantic Planning Areas. The new sound models are more conservative in many respects than previous models and indicate that increased underwater noise from certain site characterization surveys may be experienced in a larger area than previously modeled. This new information on effects of certain sound producing activities led BOEM to withdraw their request for informal consultation and request formal consultation under Section 7 of the ESA.

This formal programmatic consultation is independent of the consultation being carried out by OPR and BOEM on G&G activities and the action areas do not overlap. In March 2011, BOEM initiated informal consultation with us for the issuance of leases, site assessment, and site characterization activities on the Atlantic Outer Continental Shelf Offshore New Jersey, Delaware, Maryland, and Virginia (BOEM 2012a). The consultation was concluded in a September 20, 2011, letter from us concurring with the determination that the issuance of leases associated with site characterization and subsequent site assessment activities for siting of wind energy facilities in the identified WEAs may affect but is not likely to adversely affect any listed species under our jurisdiction. This programmatic formal consultation will replace the effects analysis in the September 2011 letter as it relates to lease issuance and site characterization activities in New Jersey. The formal consultation currently being conducted by OPR is expected to replace the effects analysis in the September 2011 letter as it relates to G&G activities in the Delaware, Maryland and Virginia WEAs.

3.0 DESCRIPTION OF THE PROPOSED ACTION

The proposed action is the issuance of commercial wind energy leases by BOEM for the four WEAs and the carrying out of site characterization activities (i.e., G&G surveys) in these lease areas. For the RI/MA and MA WEAs, the action also includes the approval of site assessment plans within all or some of the RI/MA WEA and the MA WEA. This Opinion will consider the effects to listed species associated with reasonably foreseeable site characterization scenarios associated with leasing (including geophysical, geotechnical, archeological and biological surveys), and for the RI/MA and MA WEAs site assessment activities (including the installation, operation and decommissioning of meteorological towers and buoys). The installation of meteorological towers and buoys in the MA and RI/MA WEAs will also require authorization from the U.S. Army Corps of Engineers (USACE), New England District. For purposes of this consultation, BOEM is acting as the lead Federal agency. All activities considered in this consultation are expected to occur in the next five years (2013-2018).

Under BOEM's renewable energy regulations, the issuance of leases and subsequent approval of wind energy development on the OCS is a staged decision-making process. BOEM's wind energy program occurs in four distinct phases:

1) **Planning and Analysis.** The first phase is to identify suitable areas to be considered for wind energy project leases through collaborative, consultative, and analytical processes using the state's task forces, public information meetings, input from the states, Native American Tribes, and other stakeholders.
2) **Lease Issuance.** The second phase is the issuance of a commercial wind energy lease. The competitive lease process is set forth at 30 CFR 585.210 to 585.225, and the noncompetitive process is set forth at 30 CFR 585.230 to 585.232. A commercial lease gives the lessee the exclusive right to subsequently seek BOEM approval for the development of the leasehold. The lease does not grant the lessee the right to construct any facilities; rather, the lease grants the right to use the leased area to develop its plans, which must be approved by BOEM before the lessee can move on to the next stage of the process (30 CFR 585.600 and 585.601).
3) **Approval of a Site Assessment Plan (SAP).** The third stage of the process is the submission of a SAP, which contains the lessee's detailed proposal for the construction of a meteorological tower and/or the installation of meteorological buoys on the leasehold (30 CFR 585.605 to 585.618). The lessee's SAP must be approved by BOEM before it conducts these "site assessment" activities on the leasehold. BOEM may approve, approve with modification, or disapprove a lessee's SAP (30 CFR 585.613).
4) **Approval of a Construction and Operation Plan (COP).** The fourth and final stage of the process is the submission of a COP, a detailed plan for the construction and operation of a wind energy project on the lease (30 CFR 585.620 to 585.638). BOEM approval of a COP is a precondition to the construction of any wind energy facility on the OCS (30 CFR 585.628). As with a SAP, BOEM may approve, approve with modification, or disapprove a lessee's COP (30 CFR 585.628).

The regulations also require that a lessee provide the results of surveys with its SAP or COP, including a shallow hazards survey (30 CFR 585.626 (a)(1)), geological survey (30 CFR 4 585.616(a)(2)), geotechnical survey (30 CFR 585.626(a)(4)), and an archaeological resource survey (30 CFR 585.626(a)(5)). BOEM refers to these surveys as "site characterization" activities. Although BOEM does not issue permits or approvals for these site characterization activities, it will not consider approving a lessee's SAP or COP if the required survey information is not included. *See* "Guidelines for Providing Geological and Geophysical, Hazards, and Archaeological Information Pursuant to 30 CFR Part 585," referred to herein as the 'GGARCH guidelines' (USDOI, BOEMRE, OAEP, 2011a).

The actions being evaluated as a part of this consultation are the issuance of renewable energy leases and subsequent site assessment activities to aid in the siting of potential wind turbine generators in the OCS in the BOEM North Atlantic Planning Area. BOEM has stated that the issuance of a lease does not constitute an irreversible commitment of the resources toward full development of the lease area. Thus, the issuance of a lease or approval of a SAP does not authorize, and this consultation does not evaluate, the construction of any commercial electricity generating facilities or transmission cables with the potential to export electricity. Any such proposals for BOEM approval of installation of electricity generating facilities (*i.e.*, installation of wind turbines) or transmission cables would be a separate federal action requiring a separate section 7 consultation.

Summary of the Proposed Action

The type of activities evaluated for this consultation includes the following:

1) GGARCH assessment (MA, MA/RI, NJ and NY)

    a) High resolution geophysical surveys (surface and subsurface seismic profiling, extent/intensity determined by the area being considered for development (primarily high to mid frequency sonar (i.e., side scan sonar, echo sounder, sub-bottom profilers)). The use of airguns is not being considered as a part of this activity.

    b) Geotechnical sub-bottom sampling (includes cone penetrometer tests (CPTs), geologic borings, vibracores, etc.).

2) Wind resource assessment (MA, MA/RI only)

    a) Construction of meteorological towers

    b) Installation of LIDAR buoys

3) Biological resource assessment (MA, MA/RI, NJ and NY)

    a) Presence/absence of threatened and endangered species

    b) Presence/absence of sensitive biological resources/habitats

4) Archaeological resource assessment (MA, MA/RI, NJ and NY)

5) Assessment of coastal and marine use (MA, MA/RI, NJ and NY).

Project Location

The four WEAs under consideration in the North Atlantic Planning Area comprise a total area of approximately 2,100 square statute miles (1,344,000 acres) and contain 178 whole OCS lease blocks and 94 partial OCS lease blocks. These areas are collectively referred to as the Project Area (see Figures 1a and 1b).

Figure 1a. MA and MA/RI Wind Energy Areas

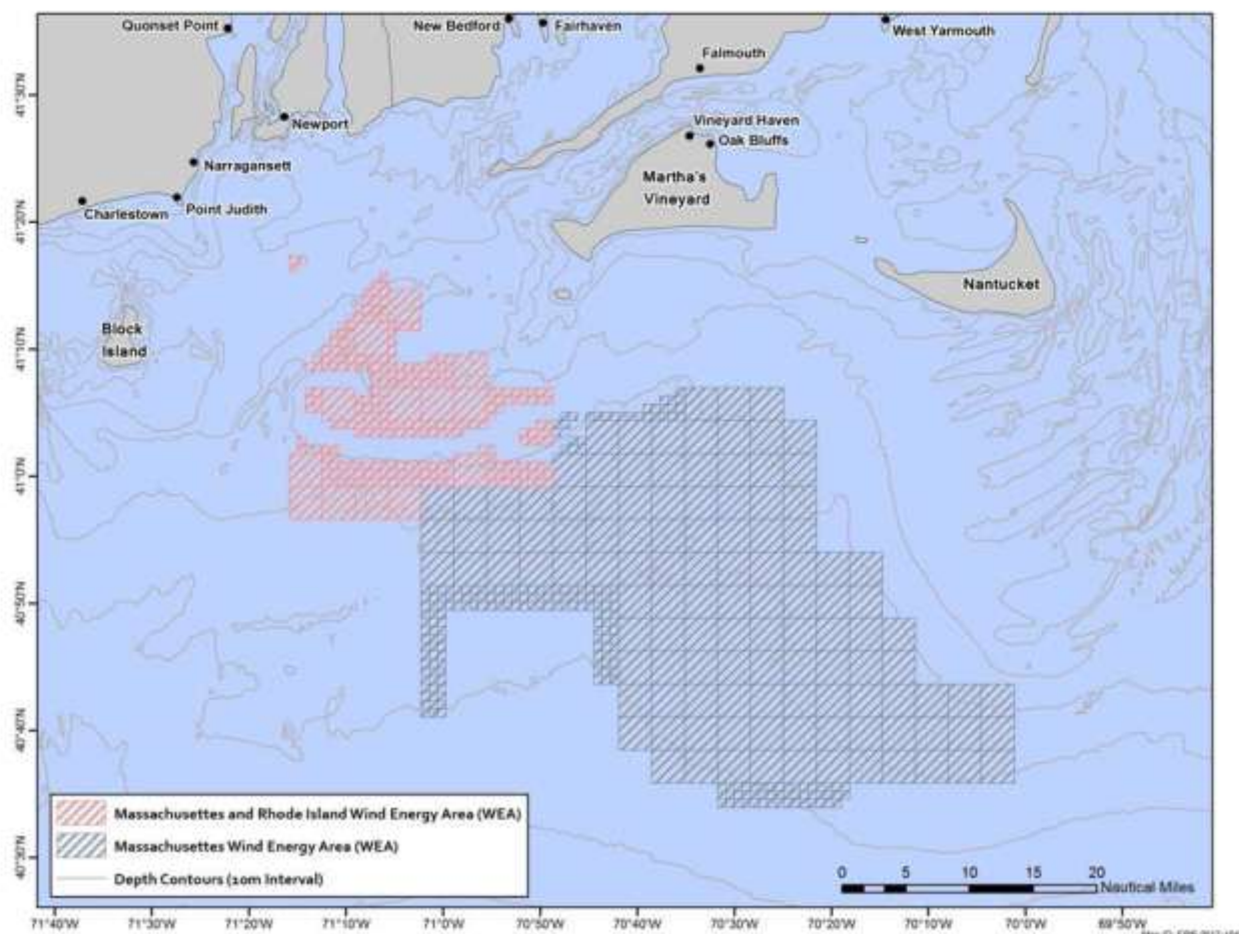
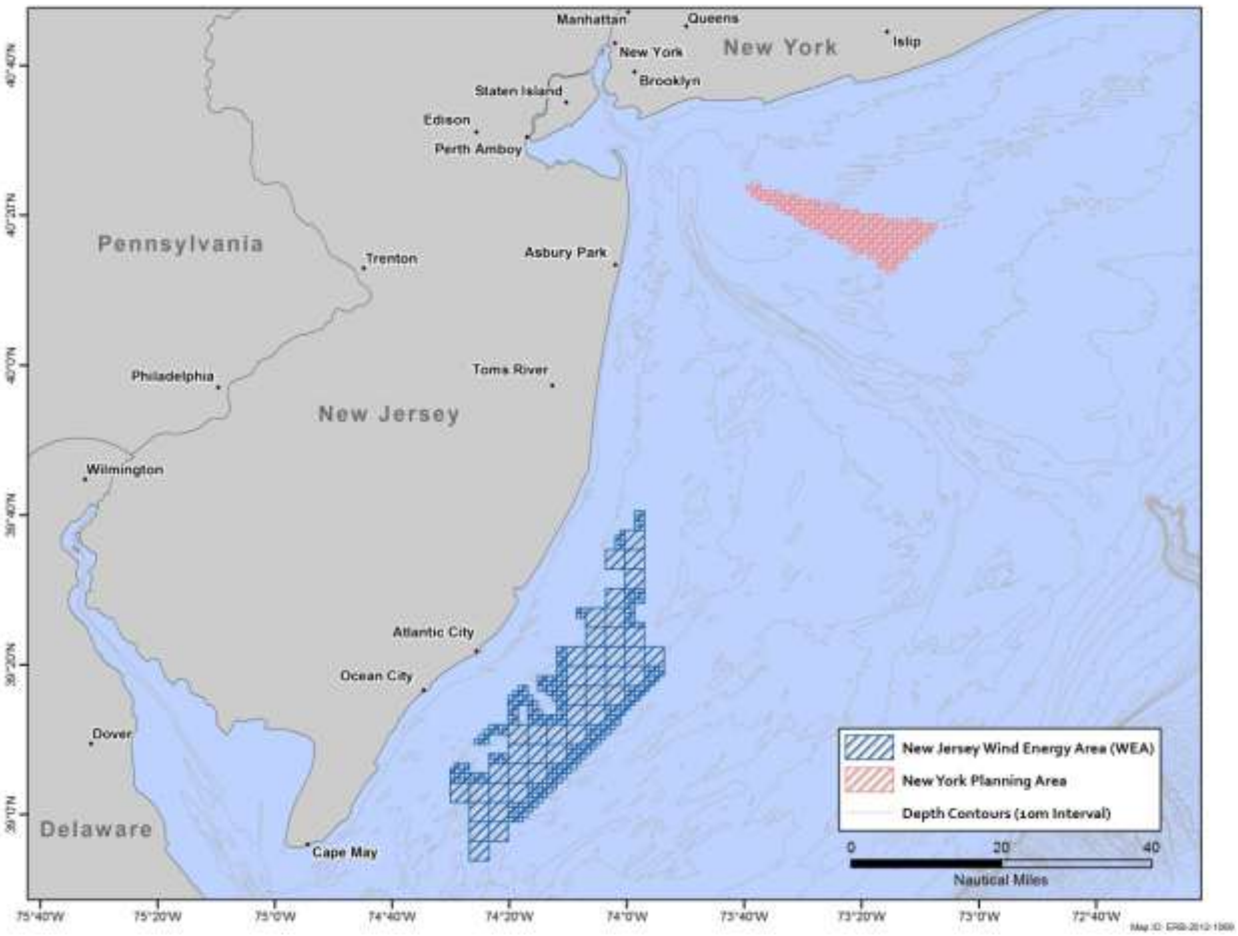


Figure 1b. NY Planning Area and NJ Wind Energy Area



The proposed action consists of the issuance of commercial wind energy leases in the Project Area and implementation of BOEM-approved site characterization activities on those leaseholds. The effects of site assessment activities are assessed for the RI/MA and MA WEAs in addition to the effects of site characterization activity. Because of the expressions of commercial wind energy interests, BOEM assumes that the entire Project Area would be leased. The New Jersey and New York areas only include impacts from site characterization activities. For the New York WEA, BOEM has received an unsolicited lease application but has yet to determine if there is competitive interest in leasing the area. Thus, they do not currently anticipate any site assessment activities within that area. If in the future, there is a site assessment plan involving activities not assessed herein and submitted to BOEM for the New York WEA, this would be a separate federal action requiring a separate section 7 consultation.

3.1 Site Characterization Surveys (RI/MA, MA, NY and NJ WEAs)

Site characterization surveys include a number of activities that allow the lessee to locate shallow hazards, physical restrictions and cultural and biological resources in the area where a project may take place. The activities are described below.

3.1.1 High-resolution Geophysical (HRG) Survey

Data obtained from the HRG surveys will provide information on geophysical shallow hazards, the presence or absence of archaeological resources, biological resources and to conduct bathymetric charting. This information is used in the design construction and operations of meteorological towers and future wind turbine placement to mitigate the potential impacts to installations, operations and production activities, and structure integrity. The scope of HRG surveys will be sufficient to reliably cover any portion of the site that may be affected by the renewable energy project's construction, operation, and decommissioning. This includes the project area encompassing all seafloor/bottom-disturbing activities. The maximum project area includes but is not limited to the footprint of all seafloor/bottom-disturbing activities (including the areas in which installation vessels, barge anchorages, and/or appurtenances may be placed) associated with construction, installation, inspection, operation, maintenance, and removal of structures.

The geophysical survey grid(s) for the proposed transmission cable route(s) to shore would be oriented with respect to the bathymetry, shallow geologic structures, and renewable energy structure locations. The grid pattern for each survey would cover the project area for all anticipated physical disturbances from construction and operation of a wind facility. Parameters for line spacing include:

- For collection of geophysical data for shallow hazard assessments using side scan-sonar/sub-bottom profilers, spacing would not likely exceed 492 feet (150 meter) throughout the project area.
- For collecting geophysical data for archaeological resource assessment using magnetometers, side-scan sonar, and all sub-bottom profilers, lines are to be flown at approximately 98 feet (30 meter) throughout the project area.
- For bathymetric charting using a multi-beam echo-sounder or side-scan sonar mosaic, construction may vary based on water depth but will provide full coverage of the seabed plus suitable overlap and resolution of small discrete targets of 1.6 to 3.3 feet (0.5 to 1.0 meters) in diameter. This is also necessary for the identification of potential archaeological resources.

3.1.1.1 HRG Survey Instrumentation

Table 1 gives an overview of the types of instrumentation that could be used during HRG survey work in the Project Area.

Table 1. Summary of Peak Source Levels for HRG Survey Activities and Operating Frequencies within Cetacean, Sea Turtle and Atlantic sturgeon Hearing Range

Source	Pulse Length	Broadband Source Level (dB re 1 µPa at 1 m)	Operating Frequencies	Within Hearing Range		
				Cetaceans	Sea Turtles	Atlantic sturgeon
Boomer	180µs	212	200 Hz – 16kHz	**Yes**	**Yes**	**Yes**
Side-scan sonar	20 ms	226	100 kHz	**Yes**	No	No
			400 kHz	No	No	No
Chirp sub-bottom Profiler	64 ms	222	3.5 kHz	**Yes**	No	No
			12 kHz	**Yes**	No	No
			200kHz	No	No	No
Multi-beam depth sounder	225 µs	213	240kHz	No	No	No

Bathymetry/Depth Sounder. The depth sounder system would record with a sweep appropriate to the range of depths expected in the survey area. Lessees can use multi-beam and/or single-beam bathymetry systems. The use of a multi-beam bathymetry system may be more appropriate for characterizing those lease areas containing complex topography or fragile habitats.

Magnetometer. Magnetometer surveys would be used to detect the identification of ferrous, ferric, or other objects having a distinct magnetic signature. The magnetometer sensor is typically towed as near as possible to the seafloor, which is anticipated to be approximately 20 feet (6 meters) above the seafloor.

Seafloor Imagery / Side-Scan Sonar. A typical side-scan sonar system consists of a top-side processor, tow cable, and towfish with transducers (or ‘pingers’) located on the sides, which generate and record the returning sound that travels through the water column at a known speed. BOEM assumes that lessees would use a digital dual-frequency side-scan sonar system with frequencies of 445 and 900 kiloHertz (kHz) and no less than 100 and 500 kHz to record continuous planimetric images of the seafloor. The data would be processed in a mosaic form to allow for a true plan view and 100 percent coverage of the project area. The side-scan sonar sensor would be towed above the seafloor at a distance that is 10 to 20 percent of the range of the instrument.

Shallow and Medium Penetration Sub-bottom Profilers. A high-resolution Compressed High-Intensity Radar Pulse (CHIRP) System sub-bottom profiler is used to generate a profile view below the bottom of the seabed, which is interpreted to develop a geologic cross-section of subsurface sediment conditions under the track line surveyed. A boomer sub-bottom profiler system is capable of penetrating depth ranges of 32 to 328 feet (10 to 100 meters) depending on frequency and bottom composition. The sub-bottom profiler would deliver a simple, stable, and repeatable signature that is near to minimum phase output with usable frequency content.
HRG survey method source levels and pulse lengths were used to model threshold radii for the various profiler methods for the Atlantic OCS Proposed Geological and Geophysical (G&G) Activities Mid-Atlantic and South Atlantic Planning Areas Draft Programmatic Environmental Impact Statement (DPEIS) (USDOI, BOEM 2012a). These profilers include a boomer, side-scan sonar, chirp sub-bottom profiler, and a multi-beam depth sounder. Three of the four profiler methods have operating frequencies that are within the range of cetacean and sea turtle hearing (Table 1). The pulse length and peak source level that were used for each profiler method modeling scenario and can be assumed to representative of profiler sources that could be used for HRG surveys during the proposed action.

3.1.1.2 Proposed HRG Survey Action Scenario

It is assumed that the HRG survey would cover the entire Project Area, and geophysical surveys for shallow hazards (approximately 492 feet [150 meters] line spacing) and archaeological resources (approximately 98 feet [30 meters] line spacing) would be conducted at the same time on the same vessels conducting sweeps at the finer line spacing. This would result in about 500 NM of HRG surveys per OCS block (3 statute miles by 3 statute miles [approximately 5 kilometers by 5 kilometers]), not including turns. Assuming a vessel speed of 4.5 knots and 10 hour days (daylight hours minus transit time to the site), it would take about 11 days to survey one OCS block or about 100 days to survey an average-size lease of eight OCS blocks. To survey all of the Project Area, HRG surveys would have to be conducted by multiple vessels and/or over multiple years and potential cable routes. Assuming 100 percent coverage of the Project Area, the proposed action would result in a total of approximately 117,200 nautical miles or 25,990 hours of HRG surveys. BOEM’s predicted scenario is summarized in Table 2.

Table 2. Projected Site Characterization and Assessment Activities for the Proposed Action

WEA	Leaseholds	Site Characterization Activities		Site Assessment Activities	
		High-Resolution Geophysical (HRG) Surveys (max NM/hours)	Geotechnical Sampling (min-max)	Installation of Meteorological Towers (max)	Installation of Meteorological Buoys (max)
New Jersey	Up to 7	31,000/6,900	900-2,500	-	-
New York	Up to 1	7,200/1,600	200-600	-	-
RI/MA	Up to 4	17,500/4,000	500 - 1,400	4	8
MA	Up to 5	61,500/13,490	708 – 2,900	5	10
Total	Up to 17	117,200/25,990	2,308 – 7,400	9	18

3.1.2 Biological Resources Surveys

Vessel and/or aerial surveys would need to characterize three primary biological resources categories: (1) benthic habitats; (2) avian resources; and (3) marine fauna. Sub-marine surveys such as the shallow hazard and geological and geotechnical surveys described earlier would be able to capture all the salient features of the benthic habitat on the leasehold. These surveys would acquire information suggesting the presence or absence of exposed hard bottoms of high, moderate, or low relief; hard bottoms covered by thin, ephemeral sand layers; seagrass patches; and other algal beds, all of which are key characteristics of benthic habitat. The various remote sensing activities used in the biological resource survey will likely occur simultaneously with the HRG survey activity and is thus not repeated here. Protected Species Observers (PSOs) would follow mitigation protocols and monitor and document sightings of marine mammals, sea turtles and birds within the lease area. Surveys that would result in interactions or capture of listed species (e.g., trawls, gillnets, etc.) are not contemplated as part of the proposed action. If a Lessee proposed this type of survey, additional consultation would likely be necessary.

3.1.3 Geotechnical Sampling

Geotechnical sampling is used to determine site specific geology profile of a specific site within the lease area. In order to achieve this, geotechnical sampling is typically conducted using cone penetration tests (CPT) or deep sediment boring / drilling at the location of the proposed meteorological tower or wind turbine. The purpose of this work is to assess the suitability of shallow foundation sediments to support a structure or transmission cable under any operational or environmental conditions that may be encountered, and document the soil characteristics necessary for design and installation of all structures. Vibracores may be taken when there are known or suspected archaeological/and or cultural resources present (identified through the HRG survey or other work) or for some limited geological sampling.

Vibracores would likely be deployed from a small (less than 45 foot) gasoline powered vessel. The diameter of a typical vibracore barrel is approximately 4 inches (10.15 centimeters) and the cores are advanced up to a maximum of 15 feet (4.5 meters). Deep borings would be advanced from a truck-mounted drill rig placed upon a jack-up barge that rests on spuds lowered to the seafloor. Each of the four spuds would be approximately 4 feet (1.2 meters) in diameter, with a pad approximately 10 feet (3.05 meters) on a side on the bottom of the spud. The barge would be towed from boring location to location by a tugboat. The drill rig would be powered using a gasoline or diesel powered electric generator. Crew would access the boring barge daily from port using a small boat. Geologic borings generally can be advanced to the target depth (100 to 200 feet [30.5 to 70 meters]) within 1 to 3 days, subject to weather and substrate conditions. Drive and wash drilling techniques would be used; the casting would be approximately 6 inches (15.24 centimeters) in diameter. The CPT or an alternative subsurface evaluation technique would supplement or be used in place of deep borings. A CPT rig would be mounted on a jack-up barge similar to that used for the borings. The top of a CPT drill probe is typically up to 3 inches (7.6 centimeters) in diameter, with connecting rods less than 6 inches (15.24 centimeters) in diameter.

It is anticipated that the majority of the work will be accomplished by CPT which does not require deep borehole drilling. However, should CPT be found an inappropriate technique given the conditions encountered, borehole drilling may be required. Previous estimates submitted to BOEM for geotechnical drilling have sound source levels at around 118-145 dB re 1 μPa at a frequency of 120 Hertz (Hz) (MMS, 2009b).

3.1.3.1 Geotechnical Sampling Scenario

In order to estimate the number of geotechnical samples per leasehold it is necessary to estimate the number of turbine foundations on each leasehold. As discussed in the Programmatic EIS (USDOI, MMS 2007), spacing between turbines is typically determined on a case-by-case basis to minimize wake effect and is based on rotor diameter associated with turbine size. In Denmark's offshore applications, for example, a spacing of seven rotor diameters between units has been used (USDOI, MMS 2007). Spacing of 6 by 9 rotor diameters, or six rotor diameters between turbines in a row and nine rotor diameters between rows was approved for the Cape Wind project (USDOI, MMS 2009b). In some land-based settings, turbines are separated by much greater distances, as much as 10 rotor diameters from each other (USDOI, MMS 2007). Based on this spacing range for a 3.6-megawatt (MW) (110 meter rotor diameter) turbine and a 5 MW (130 meter rotor diameter) turbine, it would be possible to place anywhere from 14 to 40 turbines in one OCS block (3 statute miles by 3 statute miles [approximately 5 kilometers by 5 kilometers]).

A total of 2,308 to 7,400 geotechnical surveys could occur as a result of the proposed action (see Table 2, above). This is based on the information presented above and assuming:

1) "maximum" scenario of wind development on every OCS block (which is extremely unlikely, but the lower amount of samples associated with less development would result in lower environmental impacts);

2) geotechnical sampling (vibracore, CPT, and/or deep boring) would be conducted at every potential wind turbine location throughout the Project Area;

3) geotechnical sampling would be conducted every nautical mile along the projected transmission corridors to shore; and,

4) geotechnical sampling would be conducted at the foundation of each meteorological tower and/or buoy.

3.2 Site Assessment (RI/MA and MA WEAs)

"Site assessment" describes the assessment of wind resources and ocean conditions to allow the lessee to determine: (1) whether the lease area is suitable for wind energy development; (2) where on the lease it would propose development; and (3) what form of development to propose in a COP. To determine this, a meteorological tower or buoy would be installed or deployed in the lease area to measure wind speeds and collect other relevant data necessary to assess the viability of a potential commercial wind facility. This scenario is only described and assessed in relation to the RI/MA and MA WEAs. BOEM does not currently have enough information to reasonably predict the potential site assessment scenarios for the NY WEA. Site assessment activities in the NJ WEA were considered in our September 2011 informal programmatic consultation. The analysis and conclusions of that consultation, as they relate to the installation of meteorological buoys and towers, remain valid. If, in the future, site assessment activities are proposed in the NY WEA, additional consultation will be necessary.

To obtain meteorological data, scientific measurement devices, consisting of anemometers, vanes, barometers, and temperature transmitters, would be mounted either directly on the tower or buoy or on instrument support arms. In addition to conventional data collection methods, buoys and/or bottom-founded structures could use LIDAR, Sonic Detecting and Ranging (SODAR) and Coastal Ocean Dynamic Applications Radar (CODAR) technologies for collecting wind resource data. At this time, no proposals have been submitted meteorological towers (towers in this case being up to the estimated hub height for a commercial wind turbine) mounted on a floating platform (e.g., spar, semi-submersible, or tension leg). BOEM assumes full-size met towers will utilize a fixed, pile-supported platform (monopile, jackets, or gravity bases) and that buoys would use the floating designs (e.g., boat-shaped, spar-type, tension-leg, disc-shaped or similar).

The following scenario addresses the reasonably foreseeable range of data collection devices that lessees may install under an approved SAP. The actual tower and foundation type and/or buoy type and anchoring system would be included in a detailed SAP submitted to BOEM, along with the results of site characterization surveys. This would be done prior to the installation of any device(s).

It is assumed that each of the nine leaseholds projected for the RI/MA and MA WEAs would result in zero or one meteorological tower, zero or two buoys or a combination, being constructed or deployed. This would result in a maximum of 9 meteorological towers and 18 meteorological buoys within the RI/MA and MA WEAs. Total installation time for a single meteorological tower would take eight days to ten weeks depending on the type of structure installed and the weather and sea state conditions. It is anticipated that an average meteorological buoy installation would likely take one to two days. Installation of meteorological towers and buoys would likely occur in the spring and summer months during calmer weather, however, installation could potentially occur at any time of year when weather permits. Pile installation, however, is prohibited between November 1 and April 30 (see section 3.6 below).

3.2.1 Meteorological Towers

The only meteorological tower currently installed on the OCS for the purposes of renewable energy site assessment is located on Horseshoe Shoal, in Nantucket Sound. A monopile mast was used for this meteorological tower. The tower was installed in 2003 and consists of three pilings supporting a single steel pile that supports the deck. The overall height of the structure is 197 feet (60 meters) above the mean lower low water datum. The Cape Wind meteorological tower represents the smaller end of the range of structures anticipated in southern New England. It is located in shallower water (8 to 10 feet [2.4 to 3 meters]) and nearer to shore (approximately 6 miles [9.7 kilometers]) than the RI/MA and MA WEAs.

At a maximum, a single meteorological tower would be installed per lease area. The foundation structure and a scour control system, if required based on potential seabed scour anticipated at the site, would occupy less than 2 acres. Once installed, the top of a meteorological tower would be 295 to 328 feet (90 to 100 meters) above mean sea level.

A meteorological tower consists of a mast mounted on a foundation anchored to the seafloor. The mast may be either a monopole such as that used in the Cape Wind project mentioned above or a lattice (i.e. jacket foundation). The mast and data-collection devices would be mounted on a fixed or pile-supported platform (monopile, jackets, or gravity bases) or floating platform (spar, semi-submersible, or tension-leg).

In the case of fixed platforms, it is assumed that a deck would be supported by a single 10 foot-diameter (approximately 3 meter diameter) monopile, tripod, or a steel jacket with three to four 36-inch-diameter piles. The monopile or piles would be driven anywhere from 25 to 100 feet

(7.6 to 30.5 meters) into the seafloor depending on subsea geotechnical properties. The foundation structure and a scour-control system, if required based on potential seabed scour anticipated at the site, would occupy less than 2 acres (0.81 hectare). Once installed, the top of a meteorological tower would be 295 to 328 feet (90 to 100 meters) above mean sea level. The area of ocean bottom affected by a meteorological tower would range from about 200 square feet (approximately 18.6 square meters), if supported by a monopile, to 2,000 square feet (approximately 184.1 meters) if supported by a jacket foundation.

3.2.1.1 Installation of the Foundation Structure

A jacket or monopile foundation and deck would be fabricated onshore, transferred to barge(s) and carried or towed to the offshore site. This equipment would typically be deployed from two barges, one containing the pile-driving equipment and a second containing a small crane, support equipment, and the balance of materials needed to erect the platform deck. These barges would be tended by appropriate tugs and workboats, as needed.

The foundation pile(s) for a fixed platform could range from either a single 10-foot (3 meter)-diameter monopile or three to four 36-inch (0.9-meter)-diameter piles (jacket). These piles would be driven anywhere from 25 to 100 feet (7.6 to 30.5 meters) below the seafloor with a pile-driving hammer typically used in marine construction operations. After approximately three days, when the pile-driving is complete, the pile-driver barge would be removed. In its place, a jack-up barge equipped with a crane would be used to assist in the mounting of the platform decking, tower, and instrumentation onto the foundation. Depending on the type of structure installed and the weather and sea conditions, the in-water construction of the foundation pilings and platform would range from several days (monopile construction in good weather) to six weeks (jacket foundation in bad weather) (USDOI, MMS 2009a). The mast sections would be raised using a separate barge-mounted crane; installation would likely be complete within a few weeks.

Piles are generally driven into the substrate using one of two methods: impact hammers or vibratory hammers (Nedwell and Howell 2004; Hansen *et al.*, 2003). Impact hammers use a heavy weight to repeatedly strike the pile and dive it into the substrate. Vibratory hammers use a combination of vibration and a heavy weight to force the pile into the sediment. Impact hammers produce sharp striking sounds, whereas vibratory hammers produce more continuous, low frequency sounds (Nedwell and Howell 2004; Hanson *et al.*, 2003). The type of hammer used depends on a variety of factors, such as the material the pile is composed of, and the sediment the pile will be driven into. Impact hammers can be used for any type of pile, and can drive piles into most all substrates. Vibratory hammers are more useful when driving a pile that has a sharp edge that can cut into the sediment (i.e. an open ended steel pile); as opposed to one that displaces the sediment (i.e. closed ended steel pile, wood, or cement). Also, vibratory hammers are most useful in softer sediments such as sand or mud (Hanson *et al.*, 2003). A combination of vibratory hammers and impact hammers can also be used, again, depending on the substrate. This method can be used when there is softer substrate in the upper layers, where the vibratory hammer is more useful at positioning the pile while hammering. The impact hammer can then be used to drive the pile the remainder of the depth when harder, more resistance substrates are encountered (Hanson *et al.*, 2003). This method may also be useful in the case of meteorological towers which must meet seismic stability criteria, which required that the supporting piles are either attached to, or driven into, the underlying hard sediment (Hanson *et al.*, 2003).

During installation, a radius of approximately 1,500 feet (457 meters) around the site would be needed for the movement and anchoring of support vessels. Total installation time for one meteorological tower would take eight days to ten weeks, depending on the type of structure to be installed and the weather and ocean conditions (USDOI, MMS 2009a).

3.2.1.2 Scour Control Systems

Wave action, tidal circulation, and storm waves interact with sediments on the surface of the OCS, inducing sediment reworking and/or transport. Episodic sediment movement caused by ocean currents and waves can cause erosion or scour around the tower bases. Erosion caused by scour may undermine meteorological tower structural foundations leading to potential failure. BOEM assumes that scour control systems would be installed, based on potential seabed scour anticipated at sites. There are several methods for minimizing scour around piles, such as the placement of rock armoring and mattresses of artificial (polypropylene) seagrass.

Artificial grass mats have been found to be effective in both shallow and deep waters, therefore this is the most likely scour control system to be used for the proposed meteorological towers. These mats are made of synthetic fronds that mimic seafloor vegetation to trap sediment and become buried over time. If used, these mats would be installed by divers or underwater remotely operated vehicle (ROV). Each mat would be anchored at 8 to 16 locations, about 1 foot into the sand. Once installed the mats would not require future maintenance. Monitoring of scouring at the Cape Wind meteorological tower found that at one pile where two artificial seagrass scour mats were installed, there was a net increase of 12 inches (30.5 centimeters) of sand, and at another pile with artificial seagrass scour mats, there was a net scour of 7 inches (18 centimeter); both occurred over a three-year timeframe (Ocean and Coastal Consultants Inc. 2006).

It is anticipated that for a pile-supported platform, four mats each of about 16.4 by 8.2 feet (5 by 2.5 meters) would be placed around each pile. Including the extending sediment bank, a total area disturbance of about 5,200 to 5,900 square feet (approximately 483 to 548 square meters) for a three-pile structure and 5,900 to 7,800 square feet (approximately 548 to 724.6 square meters) for a four-pile structure is estimated. For a monopile, it is anticipated that eight mats 16.4 feet by 16.4 feet (5 meters by 5 meters) would be used, and thus there would be a total disturbance area of about 3,700 to 4,000 square feet (343.74 by 371.61 square meters) per foundation.

A rock armor scour protection system may also be used to stabilize a structure's foundation area. Rock armor and filter layer material would be placed on the seabed using a clamshell bucket or a chute. The filter layer would help prevent the loss of underlying sediments and sinking of the rock armor (ESS Group, Inc. 2006). In water depths greater than 15 feet (4.5 meters), the median stone size would be about 50 pounds (approximately 22.6 kilograms) with a stone layer thickness of about 3 feet (approximately 0.9 meters). The rock armor for a monopile foundation for a wind turbine has been estimated to occupy 16,000 square feet (0.37 acre [0.15 hectares]) of the seabed

(ESS Group, Inc. 2006). While the piles of meteorological tower would be much smaller than those of a wind turbine, a meteorological tower may be supported by up to four piles. Therefore, the maximum area of the seabed impacted by rock armor for a single meteorological tower is estimated to also be 16,000 square feet (0.37 acre [0.15 hectares]).

A scour control system would be monitored throughout the lease term. It is expected that the foundation would be visually inspected monthly for the first year of installation, and then every year after that or after each significant storm activity. Inspections would be carried out by divers or ROVs. Removal of the scour control system is discussed in Section 3.5, below.

3.2.1.3 Meteorological Tower Operation and Maintenance Activities

The length of time a meteorological tower may be present on a leasehold would be influenced by a number of factors, including how long it takes to install the tower, whether the lessee has submitted a COP, and/or how long the subsequent BOEM review of the COP takes. For the proposed action, BOEM anticipates that a tower may be present for approximately five years before the final decision is made to either allow the tower to remain or be decommissioned. During the life of the meteorological tower, the structure and instrumentation would be accessible by boat for routine maintenance. As indicated in previous site assessment proposals submitted to BOEM, lessees with towers powered by solar panels or small wind turbines would conduct monthly or quarterly vessel trips for operation and maintenance activity over the five-year life of a meteorological tower (USDOI, MMS 2009a). However, if a diesel generator is used to power the meteorological tower's lighting and equipment, a maintenance vessel would make a trip at least once every other week, if not weekly, to provide fuel, change oil, and perform maintenance on the generator. Depending on the frequency of the trips, support for the meteorological towers in the RI/MA and MA WEAs would result in anywhere from 36 quarterly to 468 weekly round trips per year for up to nine meteorological towers. No additional onshore facilities, or expansion of existing facilities, would be required to conduct these tasks. It is projected that crew boats 51 to 57 feet in length with 400 to 1,000 horsepower engines and 1,800-gallon fuel capacity would be used for routine maintenance and generator refueling if diesel generators are used.

3.2.1.4 Meteorological Tower Lighting

All meteorological towers and buoys, regardless of height, would have lighting and marking for aviation and navigational purposes. Meteorological towers and buoys would be considered Private Aids to Navigation, and are required to be maintained by the individual owner under the regulations of the USCG. The USCG lighting for navigation safety would consist of two amber lights (USCG Class C) mounted on the platform deck. In accordance with FAA guidelines, the tower would be equipped with a light system consisting of a low intensity flashing red light (FAA designated L-864) for night use.

3.2.2 Meteorological Buoys

While a meteorological tower has been the traditional device for characterizing wind conditions, several companies have expressed their interest in installing one or two meteorological buoys per lease instead. Meteorological buoys can be used as an alternative to a meteorological tower in the offshore environment for meteorological resource data collection (i.e., wind, wave, and current).

These meteorological buoys would be anchored at fixed locations and would regularly collect observations from many different atmospheric and oceanographic sensors.

These meteorological buoys, of varying designs, utilize LIDAR and/or SODAR. These may be used instead of, or in addition to, anemometers to obtain meteorological data. LIDAR is a surface-based remote sensing technology that operates via the transmission and detection of light. SODAR is also a surface-based remote sensing technology; however it operates via the transmission and detection of sound.

A meteorological buoy can vary in height, hull type, and anchoring method. NOAA has successfully used discus-shaped hull buoys and boat-shaped hull buoys for weather data collection for many years. In addition, spar buoy and tension-leg platform buoy designs have been recently submitted to BOEM for approval. All of these buoy types will likely be utilized for offshore wind data collection. A large discus buoy has a circular hull range between 32 and 39 feet (10 and 12 meters) in diameter and is designed for many years of service (USDOC, NOAA, National Data Buoy Center [NBDC], 2008). The boat-shaped hull buoy (known as a 'NOMAD' [Naval Oceanographic and Meteorological Automated Device]) is an aluminum-hulled, boat-shaped buoy that provides long-term survivability in severe seas (USDOC, NOAA, NBDC, 2008). This buoy design could be utilized to mount a LIDAR wind assessment system. A typical NOMAD is a 19.6 feet by 10.2 feet (6 meters by 3.1 meters) aluminum hulled buoy with a draft of 10.5 feet (3.2 m). Originally designed by the U.S. Navy in the 1940s, the NOMAD has since been adopted and widely used by researchers, including NOAA's National Data Buoy Center. The following description is from Fishermen's Energy SAP (Fishermen's Energy 2011 *as cited in* USDOI, BOEM, OREP, 2012a).

Primary electrical (DC) power for all equipment on a NOMAD-type buoy could be provided by four deep cycle 12 volt batteries. Batteries will be charged by renewable sources which include two wind generators and four 40-watt solar panels. In the event that the renewable power sources fail to keep the batteries adequately charged (extended heavy cloud cover with little wind), the power monitoring system could prompt an onboard diesel fuel powered generator to start and run until the batteries reach the required charge level. The system would revert back to renewable charging once these systems return to proper operation (Fishermen's Energy 2011 *as cited in* USDOI, BOEM, OREP, 2012a). Up to 500 gallons of diesel fuel could be stored on board the buoy to operate the generator.

The anchoring system for the NOMAD-type buoy could be a via a standard ¾ inch steel chain to a 10,000 pounds (4,536 kilograms) steel or concrete block (s). The footprint of the anchor itself is conservatively estimated at 16 square feet (1.49 square meters). Fishermen's Energy conservatively estimates the total bottom-disturbing footprint from the anchor and anchor chain sweep of a disc-shaped or a boat-shaped buoy to range from 121,613 square feet (approximately 11,298 square meters) to 372440 square feet (approximately 34,600 square meters) assuming approximately 100 feet (30.5 meters) of slack chain at low tide.

Because of its size, a buoy of the NOMAD design would likely be towed by a single vessel to the site in the lease area at speeds of around 3 knots. Although USCG buoy tending vessels greater than or equal to 180 feet (approximately 55 meters) are known to be able to transport and deploy a buoy of this size from its deck, a wind developer may not have access to a vessel of this size. Buoys can use a wide range of moorings to attach to the seabed. On the OCS, a larger discus-type or boat-shaped hull buoy may require a combination of a chain, nylon, cable and/or buoyant polypropylene materials designed for many years of ocean service. Some deep-ocean moorings have operated without failure for over 10 years (USDOC, NOAA, NBDC 2008).

A spar-type buoy can be stabilized through an on-board ballasting mechanism approximately 60 feet (18.3 meters) below the sea surface. Approximately 30 to 40 feet (approximately 9 to 12 meters) of the spar-type buoy would be above the ocean surface where meteorological and other equipment would be located. A spar buoy is a long, thin, typically cylindrical buoy, ballasted at one end so that it floats in a vertical position. This design maintains tension in the anchor chain between the buoy and the anchor, thus eliminating slack in the chain that results in chain sweep around the anchor. Tension-leg platforms use the same tension in the mooring chain, but may utilize a more traditional discus-shaped buoy with a larger mast for mounting data collection instrumentation.

3.2.2.1 Buoy Installation

Boat-shaped, spar-type and discus-shaped buoys are typically towed or carried aboard a vessel to the installation location. Once at the location site, the buoy would be either lowered to the surface from the deck of the transport vessel or placed over the final location, and then the mooring anchor dropped. A boat-shaped buoy in shallower waters of the RI/MA and MA WEAs may be moored using an all-chain mooring, while a larger discus-type buoy would use a combination of chain, nylon, and buoyant polypropylene materials (USDOC, NOAA, NBDC, 2008). Based on previous proposals, anchors for boat-shaped and discus-shaped buoys would weigh about 6,000 to 10,000 pounds (2,721 to 4,536 kilograms) with a footprint of about 16 square feet (approximately 1.49 square meters) and an anchor sweep of about 8.5 acres (approximately 3.4 hectares). After installation, the transport vessel would remain in the area for several hours while technicians configure proper operation of all systems. Boat-shaped and discus-shaped buoys would typically take one day to install. Transport and installation vessel anchoring for one day is anticipated for these types of buoys (Fishermen's Energy 2011 *as cited in* USDOI, BOEM, OREP 2012).

Typically, a spar-type buoy would take two days to install. It would be towed to the installation location by a transport vessel after assembly at a land-based facility. Deployment would occur in two phases: deployment of a clump anchor to the seabed as a pre-set anchor (Phase 1) and deployment of the spar buoy and connection to the clump anchor (Phase 2). Phase 1 would take approximately one day and would include placement of the clump anchor on a barge and transporting it to the installation site. The monitoring buoy would be anchored to the seafloor using a clump weight anchor and mooring chain. Installation could take approximately two days. Spar-type buoys may have all-chain moorings or cables. Moorings for a spar-type buoy tension leg anchoring system may weigh up to 165 tons with a 26 by 26 foot (7.9 by 7.9 meter) footprint. The total area of bottom disturbance associated with buoy and vessel anchors would be 28 by 28 feet (8.5 by 8.5 meters), with a total area of 784 square feet (73 square meters) to a 1,200-foot (356.7 meter) radius anchor sweep for the installation vessel with a total of just over 100 acres of disturbance. The maximum area of disturbance to benthic sediments would occur during anchor deployment and removal (e.g., sediment resettlement, sediment extrusion, etc.) for this type of buoy.

3.2.3 Other Ocean Monitoring Equipment

In addition to the meteorological buoys described above, a small tethered buoy (typically 3 meters [approximately 10 feet] or less in diameter) and/or other instrumentation also could be installed on, or tethered to, a meteorological tower to monitor oceanographic parameters and to collect baseline information on the presence of certain marine life.

To measure the speed and direction of ocean currents, Acoustic Doppler Current Profilers (ADCPs) would likely be installed on each meteorological tower or buoy. The ADCP is a remote sensing technology that transmits sound waves at a constant frequency and measures the ricochet of the sound wave off fine particles or zooplanktons suspended in the water column.

The ADCPs may be mounted independently on the seafloor or to the legs of the platform, or attached to a buoy. A seafloor-mounted ADCP would likely be located near the meteorological tower (within approximately 500 feet [152 meters]) and would be connected by a wire that is hand-buried into the ocean bottom. A typical ADCP has three to four acoustic transducers that emit and receive acoustical pulses from different directions, with frequencies ranging from 300 to 600 kHz with a sampling rate of 1 to 60 minutes. A typical ADCP is about 1 to 2 feet tall (approximately 0.3 to 0.6 meters) and 1 to 2 feet wide (approximately 0.3 to 0.6 meters). Its mooring, base, or cage (surrounding frame) would be several feet wider.

A meteorological tower or buoy also could accommodate environmental monitoring equipment, such as avian monitoring equipment (e.g., radar units, thermal imaging cameras), acoustic monitoring for marine mammals, data-logging computers, power supplies, visibility sensors, water measurements (e.g., temperature, salinity), communications equipment, material hoist, and storage containers.

3.3 Vessel Traffic (RI/MA, MA, NY, and NJ Areas)

Vessel traffic, both by air and by sea, occurs during all phases of the site characterization and assessment activities. In an effort to reduce ship strikes to endangered right whales, NOAA issued regulations requiring ships 65 feet (19.8 meters) or longer to travel at 10 knots or less in certain areas where right whales gather (Effective December 9, 2008 to December 9, 2013) (73 FR 60173). The Seasonal Management Areas (SMAs) aim to reduce the likelihood of deaths and serious injuries to endangered North Atlantic right whales that result from collisions with ships, which also benefits other marine mammal species. These restrictions extend out to 20 NM (37 kilometers) around major mid-Atlantic ports. The Block Island Sound SMA includes all of the RI/MA WEA and a small portion of the MA WEA. The Delaware Bay SMA does not fully overlap with the NJ WEA, and the New York SMA partially overlaps with the NY WEA. Except for crew boats, which are typically smaller than 65 feet (19.8 meters), these restrictions would be applicable to most vessels associated with the proposed action. Speed restrictions are in effect from November 1st to April 30th. In addition to the seasonal restrictions, Dynamic Management Areas (DMAs) created by NMFS and based on recent right whale sightings (when a group of three or more right whales is confirmed) may be present within the Project Area or surrounding waters. Should a DMA become active encompassing all or a portion of the Project Area, NMFS would encourage vessel operators to voluntarily adhere to the seasonal restrictions, or, if possible, re-route their path outside of the designated DMA. Lessees in the RI/MA, MA, NY, and NJ areas would be required to abide by these otherwise voluntary restrictions (See Project Design Criteria in Section 3.6, below).

3.3.1 HRG Survey Traffic

As detailed above, it is assumed that the HRG survey would cover the entire Project Area, and geophysical surveys for shallow hazards (492 feet [150 meters] line spacing) and archaeological resources (98 feet [30 meters] line spacing) would be conducted at the same time on the same vessels conducting sweeps at the finer line spacing array. This would result in about 500 NM of HRG surveys per OCS block (3 statute miles by 3 statute miles [approximately 5 kilometers by 5 kilometers]), not including turns. Assuming a vessel speed of 4.5 knots and 10-hour days (daylight hours minus transit time to the site), it would take about 11 days to survey one OCS block or about 100 days to survey an average-size lease of eight OCS blocks. To survey all of the Project Area, HRG surveys would have to be conducted by multiple vessels and/or over multiple years. Assuming 100 percent coverage of the Project Area, the proposed action would result in a total of approximately 117,200 NM or 25,990 hours/ 2,750 round trips of HRG surveys (see Tables above).

Vessels would be required to maintain a vigilant watch for marine mammals and sea turtles during transit to and from the survey area, as well as during the HRG survey itself. Section 3.6 details the standard operating conditions that would be required for vessels.

3.3.2 Geotechnical Sampling Vessel Traffic

As described in the geotechnical sampling activity scenario, it is anticipated that there would be approximately 2,308 – 7,400 geotechnical samples taken within the Project Area. The amount of effort and vessel trips vary greatly by the type of technology used to retrieve the sample, and each work day would be associated within one round trip. The following details the type of vessels and collection time per sample:

Vibracores: Likely to be advanced from a single small vessel (~45 feet [~14.7 meters]), and collect one sample per day.

CPT: Depending on the size of the CPT, it could be advanced from medium vessel (~65 feet [~19.8 meters]), a jack-up barge, a barge with a 4-point anchoring system, or a vessel with a dynamic positioning system. Each barge scenario would include a support vessel. This range of vessels could sample one location per day.

Geologic boring: Would be advanced from a jack-up barge, a barge with a 4-point anchoring system, or a vessel with a dynamic positioning system. Each barge scenario would include a support vessel. Each deep geologic boring could take one day.

Based on the expected number of both HRG surveys and geotechnical samples, as well as, presumed independent biological surveys, approximately 2,750 vessel trips (round trips) associated with site characterization surveys are projected to occur as a result of the proposed action over five years (2013 to 2018).

3.3.3 Meteorological Tower Construction and Operation Traffic (RI/MA and MA WEAs)

The proposed action scenario estimates a maximum of nine meteorological towers to be constructed within the RI/MA and MA WEAs. During installation, a radius of approximately 1,500 feet (457.2 meters) around the site would be needed for the movement and anchoring of support vessels. A maximum of 40 round trip vessel trips are expected during construction of each meteorological tower or 360 rounds trips for up to nine meteorological towers. Several vessels would be involved in installing and constructing a meteorological tower. Vessels delivering construction material or crews to the site will be present in the area between the mainland and the construction site, as well as vessel being present at the site during installation. The barges, tugs and vessels delivering construction materials will typically be 65 to 270 feet (19.8 to 82.3 meters) in length, while the vessel carrying construction crews will typically be 51 to 57 feet (15.5 to 17.4 meters) in length.

After installation, data would be monitored and processed. The structure and instrumentation would be accessed by boat for routine maintenance. Assuming a single maintenance trip to each meteorological tower quarterly to weekly, the proposed action would result in an additional 40 to 520 vessel trips per year for up to nine meteorological towers, or 180 to 2,340 vessel trips over a five-year period. These vessel trips would not require any additional or expansion of onshore facilities. It is projected that crew boats 51 to 57 feet (15.5 to 17.4 meters) in length would be used to service the structure.

Vessel usage during decommissioning will be similar to that during construction. Up to approximately 40 round trips by various vessels are expected during decommissioning of each meteorological tower. Similar to construction, this yields an average of 360 round trips for the decommissioning of up to nine meteorological towers.

3.3.4 Meteorological Buoy Deployment and Operation Traffic (RI/MA and MA WEAs)

The proposed action scenario estimates a maximum of 18 meteorological buoys could be deployed throughout the RI/MA and MA WEAs. As described in Section 4.3.5.3, the installation of each buoy could utilize 1-2 round trips per buoy deployment. The types of vessels involved in the deployment include barge/tug (for buoy and/or anchoring system), large work vessel (for towing and/or carrying the buoy), and an additional support vessel (for crew and other logistical needs).

Similar to the meteorological towers, it is expected that maintenance for the buoy would be required on a quarterly to weekly basis resulting in maximum of 80-1,040 to round-trips per year for up to 18 buoys, or 360-4,680 vessel trips over a five year period. It should be noted that it is unlikely that all 18 meteorological buoys would be in service at the same time over the entire period. For meteorological buoys, the decommissioning is expected to be the reverse of the deployment, with one round trip required to retrieve each buoy.

Table 4. Total Number of Estimated Vessel Trips for Project Area Over a Five Year Period

WEA	HRG Survey	Geotechnical sample	Met tower install	Met buoy install	Met tower ops	Met buoy ops	Met tower decom.	Met buoy decom.
New Jersey	690	900-2,500	-	-	-	-	-	-
New York	160	200-600	-	-	-	-	-	-
Rhode Island / Massachusetts	400	500 – 1,400	160	8-16	80-1,040	160-2,080	160	8-16
Massachusetts	1,500	708 –2,900	200	10-20	100-1,300	200-2,600	200	10-20
Total	2,750	2,308 –7,400	360	18-26	180-2,340	360-4,680	360	18-26

Note:
Met = Meteorological
ops = operations
decom = decommissioning

3.4 Onshore Activity (RI/MA and MA WEAs)

For site assessment-related activity in the RI/MA and MA WEAs there are several southern New England ports that could be used as a fabrication sites, staging areas and crew/cargo launch sites. Existing ports or industrial areas are expected to be used. The fabrication facilities in the relevant major port areas are large and have high capacities, therefore BOEM does not anticipate that the fabrication of meteorological towers or buoys associated with the proposed action would have any substantial effect on the operations of, transportation to or from, or conditions at these facilities.

Several major ports exist near the RI/MA and MA WEAs that are suitable to support the fabrication and staging of meteorological towers and buoys, including the ports of New Bedford, Massachusetts and Quonset Point, Rhode Island.

A meteorological tower platform or meteorological buoy would be constructed or fabricated onshore at an existing fabrication yard or final assembly of the tower could be completed offshore. The location of these fabrication yards is directly tied to the availability of a large enough channel that would allow the towing of these structures. The average bulkhead depth needed for water access to fabrications yards is 15 to 20 feet (4.6 to 6.1 meters).

3.5 Decommissioning (RI/MA and MA WEAs)

No later than two years after the cancellation, expiration, relinquishment, or other termination of the lease, the lessee would be required to remove all devices, works, and structures from the site and restore the leased area to its original condition before issuance of the lease (30 CFR 585, Subpart I). Decommissioning is only being assessed for the RI/MA and MA WEAs.

It is estimated that the entire removal process of a meteorological tower would take one week or less. Decommissioning activities would begin with the removal of all meteorological instrumentation from the tower, typically using a single vessel. A derrick barge would be transported to the offshore site and anchored next to the structure. The mast would be removed from the deck and loaded onto the transport barge. The deck would be cut from the foundation structure and loaded onto the transport barge. The same number of vessels necessary for installation would likely be required for decommissioning. The sea bottom area beneath installed structures would be cleared of all materials that have been introduced to the area in support of the lessee's project.

Buoy decommissioning is the reverse of the installation process. Equipment recovery would be performed with support of a vessel(s) equivalent in size and capability to those used for installation. For small buoys, a crane lifting hook would be secured to the buoy. A water/air pump system would de-ballast the buoy into the horizontal position. The mooring chain(s)/cable(s) and anchor would be recovered to the deck using a winching system. The buoy would then be towed to shore by the barge. All buoy decommissioning is expected to be completed within one or two days. Buoys would be returned to shore and disassembled or reused in other applications. It is anticipated that the mooring devices and hardware would be reused or disposed of as scrap iron for recycling (Fishermen's Energy 2011 *as cited in* USDOI, BOEM, OREP, 2012a).

3.5.1 Cutting and Removing Piles

As required by BOEM, the lessee would sever bottom-founded structures and their related components at least 15 feet (5 meters) below the mud line to ensure that nothing would be exposed that could interfere with future lessees and other activities in the area (30 CFR 585.910(a)). The choice of severing tool depends on the target size and type, water depth, economics, environmental concerns, tool availability, and weather conditions (USDOI, MMS 2005). Meteorological tower piles in the RI/MA and MA WEAs would be removed using non-explosive severing methods.

Common non-explosive severing tools that may be used consist of abrasive cutters (e.g., sand cutters and abrasive water jets), mechanical (carbide) cutters, diver cutting (e.g., underwater arc cutters and oxyacetylene/oxyhydrogen torches), and diamond wire cutters. Of these, the most likely tools to be employed would be an internal cutting tool, such as a high-pressure water jet-cutting tool that would not require the use of divers to set up the system or jetting operations to access the required mud line (Kaiser *et al.,* 2005). To cut a pile internally, the sand that had been forced into the hollow pile during installation would be removed by hydraulic dredging/pumping and stored on a barge. Once cut, the steel pile would then be lifted onto a barge and transported to shore. Following the removal of the cut pile and the adjacent scour control system, the sediments would be returned to the excavated pile site using a vacuum pump and diver-assisted hoses. As a result, no excavation around the outside of the monopile or piles prior to the cutting is anticipated. Cutting and removing piles would take anywhere from several hours to one day per pile. After the foundation is severed, it would be lifted on the transport barge and towed to a decommissioning site onshore (USDOI, MMS 2009a).

3.5.2 Removal of Scour Control System

Any scour control system would be removed during the decommissioning process. Scour mats would be removed by divers or ROV and a support vessel in a similar manner to installation. Removal is expected to result in the suspension of sediments that were trapped in the mats. If rock armoring is used, armor stones would be removed using a clamshell dredge or similar equipment and placed on a barge. It is estimated that the removal of the scour control system would take a half-day per pile. Therefore, depending on the foundation structure, removal of the scour system would take from one half to two days to complete (USDOI, MMS 2009a).

3.6 Project Design Criteria

This section outlines the standard operating conditions that BOEM will require in order to minimize or eliminate potential impacts to protected species, including ESA-listed species. These conditions are divided into five sections: (1) those required during all project activity associated with SAP and/or COP submittal or activity under a SAP; (2) those required during geological and geophysical (G&G) survey activity in support of plan (i.e., SAP and/or COP) submittal; (3) those required during pile driving of a meteorological tower foundation; (4) reporting requirements; and (5) other requirements. These project design criteria will be required as part of any activity considered in this consultation and are considered part of the proposed action.

3.6.1 General Requirements

3.6.1.1 Vessel Strike Avoidance Measures

The vessel strike avoidance measures required above are based on the Joint BOEM-BSEE Notice To Lessees and Operators (NTL) of Federal Oil, Gas, and Sulphur Leases in the OCS, Gulf of Mexico of Mexico OCS Region on “Vessel Strike Avoidance and Injured/Dead Protected Species Reporting” (NTL 2012-JOINT-G01) (see http://www.bsee.gov/Regulations-and-Guidance/Notices-to-Lessees-and-Operators.aspx), which in turn is based upon the NMFS’ Vessel Strike Avoidance Measures and Reporting for Mariners. These measures have become standard means to protect marine mammals and sea turtles by maintaining a vigilant watch for these species and reducing speed and/or course to reduce or eliminate the potential for injury. A single cetacean at the surface may indicate the presence of submerged animals in the vicinity of the vessel thus requiring the precautionary vessel-strike avoidance measures. Given that delphinoid cetaceans often bow ride and are far more quick to react to vessel movement than large non-delphinoid cetaceans, the requirement to shift the engine into neutral is not required for those species.

The seasonal temporal speed restriction is based upon vessel strike reduction measures implemented through the Seasonal Management Areas (SMAs) for North Atlantic right whales by NMFS; however, for the actions considered in this consultation, BOEM will require that these measures be implemented in all WEAs.

The Lessee must ensure that all vessels conducting activity in support of a plan (i.e., SAP and/or COP) comply with the vessel strike avoidance measures specified below except under extraordinary circumstances when the safety of the vessel or crew are in doubt or the safety of life at sea is in question:

1) The Lessee must ensure that vessel operators and crews maintain a vigilant watch for cetaceans, pinnipeds, and sea turtles and slow down or stop their vessel to avoid striking protected species.

2) The lessee must ensure that all vessels 65 feet in length or greater, operating from November 1 through July 31, operate at speeds of 10 knots (<18.5 km/h) or less. In addition, vessel operators must comply with 10 knot (<18.5 km/h) speed restrictions in any Dynamic Management Area (DMA). Vessel operators may send a blank email to ne.rw.sightings@noaa.gov for an automatic response listing all current SMAs and DMAs.

3) North Atlantic right whales

    (a) The Lessee must ensure all vessels maintain a separation distance of 500 meters (1,640 feet) or greater from any sighted North Atlantic right whale(s) pursuant to 50 CFR 224.103.

    b) The Lessee must ensure that the following avoidance measures are taken if a vessel comes within 500 meters (1,640 feet) of a right whale(s):

        (i) The Lessee must ensure that while underway, any vessel must steer a course away from the right whale(s) at 10 knots (< 18.5 km/h) or less until the 500 meters (1,640 feet) minimum separation distance has been established (unless (ii) below applies).

        (ii) The Lessee must ensure that when a North Atlantic right whale is sighted in a vessel's path, or within 100 meters (328 feet) to an underway vessel, the underway vessel must reduce speed and shift the engine to neutral. The Lessee must not engage the engines until the right whale(s) has moved outside of the vessel's path and beyond 100 meters (328 feet).

        (iii) The Lessee must ensure that if a vessel is stationary, the vessel must not engage engines until the North Atlantic right whale(s) has moved beyond 100 meters (328 feet), at which time refer to point 3(b)(i).

        (iv) The Lessee must ensure that any vessel must reduce vessel speed to 10 knots (<18.5 km/h) or less within any Dynamic Management Area (DMA).

4) Non-delphinoid cetaceans other than the North Atlantic right whale

    a) The Lessee must ensure all vessels maintain a separation distance of 100 meters (328 feet) or greater from any sighted non-delphinoid cetacean (s):
    b) The Lessee must ensure that the following avoidance measures are taken if a vessel comes within 100 meters (328 feet) of a non-delphinoid cetacean:
        i) The Lessee must ensure that when a non-delphinoid cetacean(s) (other than a North Atlantic right whale) is sighted, the vessel underway must reduce speed and shift the

engine to neutral, and must not engage the engines until the non-delphinoid cetacean(s) has moved outside of the vessel's path and beyond 100 meters (328 feet).

ii) The Lessee must ensure that if a vessel is stationary, the vessel must not engage engines until the non-delphinoid cetacean(s) has moved out of the vessel's path and beyond 100 meters (328 feet).

5) Delphinoid cetaceans

a) The Lessee must ensure all vessels maintain a separation distance of 50 meters (164 feet) or greater from any sighted delphinoid cetacean(s).
b) The Lessee must ensure that the following avoidance measures are taken if the vessel comes within 50 meters (164 feet) of a delphinoid cetacean(s):
   i) The Lessee must ensure that any vessel underway remain parallel to a sighted delphinoid cetacean's course whenever possible, and avoid excessive speed or abrupt changes in direction. Course and speed may be adjusted once the delphinoid cetacean(s) has moved beyond50 meters (164 feet) and/or abeam of the underway vessel.
   ii) In addition, the Lessee must ensure that any vessel underway reduce vessel speed to 10 knots (<18.5 km/h) or less when pods (including mother/calf pairs) or large assemblages of delphinoid cetaceans are observed. Course and speed may be adjusted once the minimum separation distance (50 meters (164 feet)) has been established and/or the delphinoid cetaceans have moved abeam of the underway vessel.

6) Sea turtles. The Lessee must ensure all vessels maintain a separation distance of 50 meters (164 feet) or greater from any sighted sea turtle.

7) The Lessee must ensure that vessel operators are briefed to ensure they are familiar with the above requirements.

3.6.1.2 Marine Debris Awareness

Marine debris awareness measures are intended to reduce the risk marine debris poses to protected species from ingestion and entanglement. These simple measures will reduce the potential for debris ending up in the marine environment.

The lessee must ensure that vessel operators, employees and contractors engaged in activity in support of a plan (i.e., SAP and/or COP) are briefed on marine trash and debris awareness elimination as described in the BSEE NTL No. 2012-G01 ("Marine Trash and Debris Awareness and Elimination"). BOEM (the Lessor) will not require the lessee to undergo formal training or post placards, as described under this NTL. Instead, the lessee must ensure that its employees and contractors are made aware of the environmental and socioeconomic impacts associated with marine trash and debris and their responsibilities for ensuring that trash and debris are not intentionally or accidentally discharged into the marine environment. The above referenced NTL provides information the lessee may use for this awareness training.

3.6.2 Geological and Geophysical (G&G) Survey Requirements

1) Visibility. The Lessee must not conduct G&G surveys in support of plan (i.e., SAP and/or COP) submittal at any time when lighting or weather conditions (e.g., darkness, rain, fog, sea state) prevents visual monitoring of the exclusion zones for HRG surveys and geotechnical surveys as specified below. This requirement may be modified as specified below.

2) Modification of Visibility Requirement. If the Lessee intends to conduct G&G survey operations in support of a plan at night or when visual observation is otherwise impaired, an alternative monitoring plan detailing the alternative monitoring methodology (e.g. active or passive acoustic monitoring technologies) must be submitted to the Lessor for consideration. The Lessor may, after consultation with NMFS, decide to allow the Lessee to conduct G&G surveys in support of a plan at night or when visual observation is otherwise impaired using the proposed alternative monitoring methodology.

3) Protected-Species Observer (PSO). The Lessee must ensure that the exclusion zone for all G&G surveys performed in support of plan (i.e., SAP and/or COP) submittal is monitored by a NMFS-approved PSO. The Lessee must provide to the Lessor a list of observers and their résumés no later than forty-five (45) calendar days prior to the scheduled start of surveys performed in support of plan submittal. The résumés of any additional observers must be provided fifteen (15) calendar days prior to each observer's start date. The Lessor will send the observer information to NMFS for approval.

4) Optical Device Availability. The Lessee must ensure that reticulated binoculars or other suitable equipment are available to each observer so that they can adequately perceive and monitor distant objects within the exclusion zone during surveys conducted in support of plan (i.e., SAP and/or COP) submittal.

3.6.2.1 High Resolution Geophysical Survey Requirements

1) Clearance Period and Sea Turtle Exclusion Zone. BOEM is requiring that the Lessee maintain a 200 meter exclusion zone during the surveys where one or more acoustic sound sources is operating at frequencies below 200 kHz and that this exclusion zone be monitored for at least 60 minutes prior to ramp up of the survey equipment.

2) Modification of Exclusion Zone. The modification of the exclusion zone reflects several principles: a) the lessee may utilize a type of survey equipment whose sound profile was not captured by BOEM's model and modification of the exclusion zone is appropriate; b) equipment specifications submitted to BOEM with the lessee's plan documents indicate a sound profile that exceeds BOEM's modeled area of ensonification at the 180 dB level; and c) the lessee may wish to expand the exclusion zone to encompass the 160 dB level if it can be effectively monitored in order to reduce potential for needing an incidental harassment authorization issued under the Marine Mammal Protection Act.

3) Shutdown Provisions. Prior to beginning either HRG or geotechnical surveys the exclusion zone must be clear of all cetaceans, pinnipeds, and sea turtles. This will ensure that these species are far enough from the sound source prior to the activity that harassment does not occur. After the initial startup of the sound source, shutdown of

either electromechanical or geotechnical survey equipment is only required for non-delphinoid cetaceans and sea turtles. This is primarily a precautionary measure targeted at endangered species. Incursion of the exclusion zone after the start of the sound source by pinnipeds and delphinoid cetaceans must be recorded by the observer, but -especially in the case of delphinoid cetaceans- because of their documented curiosity and voluntary approach of seismic sound sources (air guns) in the Gulf of Mexico (Barkaszi et al 2012) it was determined that a shutdown of the active sound source was not appropriate for these species.

(a) Establishment of Exclusion Zone. The lessee must ensure that a 200 meter default exclusion zone for cetaceans, pinnipeds, and sea turtles will be monitored by a protected species observer around a survey vessel actively using electromechanical survey equipment where one or more acoustic sound sources is operating at frequencies below 200 kHz. In the case of the North Atlantic right whale, the minimum separation distance of 500 m (1,640 feet) is in effect when the vessel is underway as described in the vessel-strike avoidance measures.

(i) If the Lessor determines that the exclusion zone does not encompass the 180-dB Level A harassment radius calculated for the acoustic source having the highest source level, the Lessor will consult with NMFS about additional requirements.

(ii) The Lessor may authorize surveys having an exclusion zone larger than 200 m (656 feet) to encompass the 160-dB Level B harassment radius if the Lessee can demonstrate the zone can be effectively monitored.

(b) Modification of Exclusion Zone. The Lessee may use the field-verification method described below to modify the HRG survey exclusion zone for specific HRG survey equipment being utilized. Any new exclusion zone radius must be based on the most conservative measurement (i.e., the largest safety zone configuration) of the 160 dB or 180 dB zone. This modified zone must be used for all subsequent use of field-verified equipment and may be periodically reevaluated based on the regular sound monitoring described below. The Lessee must obtain Lessor approval of any new exclusion zone before it may be implemented.

(c) Field Verification of Exclusion Zone. If the Lessee wishes to modify the exclusion zone as described above, the Lessee must conduct field verification of the exclusion zone for specific HRG survey equipment. The results of the sound measurements from the survey equipment must be used to establish a new exclusion zone which may be greater than or less than the 200-meter default exclusion zone depending on the results of the field tests. The Lessee must take acoustic measurements at a minimum of two reference locations. The first location must be at a distance of 200 meters from the sound source and the second location must be as close to the sound source as technically feasible. Sound measurements must be taken at the reference locations at two depths (i.e., a depth at mid-water and a depth at approximately 1 meter above the seafloor). Sound pressure levels must be measured and reported in the field in dB re 1 μPa rms (impulse). An infrared range finder may be used to determine distance from the sound source to the reference location.

(d) Clearance of Exclusion Zone. The lessee must ensure that active acoustic sound sources must not be activated until the protected species observer has reported the exclusion zone clear of all cetaceans, pinnipeds, and sea turtles for 60 minutes.

(e) Electromechanical Survey Equipment Ramp-Up. The lessee must ensure that when technically feasible a "ramp-up" of the electromechanical survey equipment occur at the start or re-start of HRG survey activities. A ramp-up would begin with the power of the smallest acoustic equipment for the HRG survey at its lowest power output. The power output would be gradually turned up and other acoustic sources added in a way such that the source level would increase in steps not exceeding 6 dB per 5-min period.

(f) Shut Down for Non-Delphinoid Cetaceans and Sea Turtles. If a non-delphinoid cetacean or sea turtle is sighted at or within the exclusion zone, an immediate shutdown of the electromechanical survey equipment is required. The vessel operator must comply immediately with such a call by the observer. Any disagreement or discussion should occur only after shut-down. Subsequent restart of the electromechanical survey equipment must use the ramp-up provisions described above and may only occur following clearance of the exclusion zone of all cetaceans, pinnipeds, and sea turtles for 60 minutes.

(g) Power Down for Delphinoid Cetaceans and Pinnipeds. If a delphinoid cetacean or pinniped is sighted at or within the exclusion zone, the electromechanical survey equipment must be powered down to the lowest power output that is technically feasible. The vessel operator must comply immediately with such a call by the observer. Any disagreement or discussion should occur only after power-down. Subsequent power up of the electromechanical survey equipment must use the ramp-up provisions described above and may occur after (1) the exclusion zone is clear of a delphinoid cetacean and/or pinniped or (2) a determination by the protected species observer after a minimum of 10 minutes of observation that the delphinoid cetacean and/or pinniped is approaching the vessel or towed equipment at a speed and vector that indicates voluntary approach to bow-ride or chase towed equipment. An incursion into the exclusion zone by a non-delphinoid cetacean or sea turtle during a power-down requires implementation of the shut-down procedures described above.

(h) Pauses in Electromechanical Survey Sound Source. The lessee must ensure that if the electromechanical sound source shuts down for reasons other than encroachment into the exclusion zone by a non-delphinoid cetacean or sea turtle, including, but not limited to, mechanical or electronic failure, resulting in the cessation of the sound source for a period greater than 20 minutes, the lessee must the lessee must restart the electromechanical survey equipment using the full ramp-up procedures and clearance of the exclusion zone of all cetaceans, pinnipeds, and sea turtles for 60 minutes. If the pause is less than 20 minutes the equipment may be re-started as soon as practicable at its operational level as long as visual surveys were continued diligently throughout the silent period and the exclusion zone remained clear of cetaceans, pinnipeds, and sea turtles. If visual surveys were not continued diligently during the pause of 20-minutes or less, the lessee must restart the electromechanical survey equipment using the full ramp-up procedures and clearance of the exclusion zone of all cetaceans, pinnipeds, and sea turtles for 60 minutes.

3.6.2.2 Geotechnical Survey Requirements

1) Establishment of Exclusion Zone. The lessee must ensure that a 200 meter radius exclusion zone for all cetaceans, pinnipeds, and sea turtles will be monitored by a protected species observer around any vessel conducting geotechnical surveys (i.e. drilling, cone penetrometer tests, etc.).

2) Modification of Exclusion Zone. The Lessee may use the field-verification method as described below to modify the geotechnical survey exclusion zone for specific geotechnical sampling equipment being utilized. Any new exclusion zone radius must be based on the most conservative measurement (i.e., the largest safety zone configuration) of the 120 dB zone. This modified zone must be used for all subsequent use of field-verified equipment and may be periodically reevaluated based on the regular sound monitoring described below. The Lessee must obtain Lessor approval of any new exclusion zone before it may be implemented.

3) Field Verification of Exclusion Zone. If the Lessee wishes to modify the exclusion zone as described above, the Lessee must conduct field verification of the exclusion zone for specific geotechnical sampling equipment. The results of the measurements from the equipment must be used to establish a new exclusion zone, which may be greater than or less than the 200-meter default exclusion zone depending on the results of the field tests. The Lessee must take acoustic measurements at a minimum of two reference locations. The first location must be at a distance of 200 meters from the sound source and the second location must be as close to the sound source as technically feasible. Sound measurements must be taken at the reference locations at two depths (i.e., a depth at mid-water and a depth at approximately 1 meter above the seafloor). Sound pressure levels must be measured and reported in the field in dB re 1 μPa rms (impulse). An infrared range finder may be used to determine distance from the sound source to the reference location.

4) Clearance of Exclusion Zone. The lessee must ensure that geotechnical sound source must not be activated until the protected species observer has reported the exclusion zone clear of all cetaceans, pinnipeds, and sea turtles for 60 minutes.

5) Shut Down for Non-Delphinoid Cetaceans and Sea Turtles. If any non-delphinoid cetaceans or sea turtles are sighted at or within the exclusion zone, an immediate shutdown of the geotechnical survey equipment is required. The vessel operator must comply immediately with such a call by the observer. Any disagreement or discussion should occur only after shut-down. Subsequent restart of the geotechnical survey equipment may only occur following clearance of the exclusion zone for 60 minutes.

6) Pauses in Geotechnical Survey Sound Source. The lessee must ensure that if the geotechnical sound source shuts down for reasons other than encroachment into the exclusion zone by a non-delphinoid cetacean or sea turtle, including, but not limited to, mechanical or electronic failure, resulting in the cessation of the sound source for a period greater than 20 minutes, the lessee must ensure clearance of the exclusion zone of all cetaceans, pinnipeds and sea turtles for 60 minutes. If the pause is less than 20 minutes the equipment may be re-started as soon as practicable as long as visual surveys were continued diligently throughout the silent period and the exclusion zone remained clear of cetaceans, pinnipeds, and sea turtles. If visual surveys were not continued diligently during the pause of 20-minutes or less, the lessee must restart the geotechnical survey equipment only after the clearance of the exclusion zone of all cetaceans, pinnipeds, and sea turtles for 60 minutes.

3.6.3 Requirements for Pile Driving of a Meteorological Tower Foundation

The 3281 feet (1,000 meters) default exclusion zone is based upon the field of ensonification at the 180 dB level and based upon previous reports to BOEM on modeled areas of ensonification from pile driving activities. Because of the greater risk of injury to cetaceans, pinnipeds, and sea turtles from pile driving, BOEM has adopted a very conservative shutdown requirement that would apply to all incursions into the exclusion zone during pile driving.

1) Visibility. The Lessee must not conduct pile driving for a meteorological tower foundation at any time when lighting or weather conditions (e.g., darkness, rain, fog, sea state) prevents visual monitoring of the exclusion zones for meteorological tower foundation pile driving as specified below. This requirement may be modified as specified below.

2) Modification of Visibility Requirement. If the Lessee intends to conduct pile driving for a meteorological tower foundation at night or when visual observation is otherwise impaired, an alternative monitoring plan detailing the alternative monitoring technologies (e.g. active or passive acoustic monitoring technologies) must be submitted to the Lessor for consideration. The Lessor may, after consultation with NMFS, decide to allow the Lessee to conduct pile driving for a meteorological tower foundation at night or when visual observation is otherwise impaired.

3) Protected-Species Observer (PSO). The Lessee must ensure that the exclusion zone for all pile driving for a meteorological tower foundation is monitored by a NMFS-approved

PSO. The Lessee must provide to the Lessor a list of observers and their résumés no later than forty-five (45) calendar days prior to the scheduled start of meteorological tower construction activity. The résumés of any additional observers must be provided fifteen (15) calendar days prior to each observer's start date. The Lessor will send the observer information to NMFS for approval.

4) Optical Device Availability. The Lessee must ensure that reticuled binoculars or other suitable equipment are available to each observer to adequately perceive and monitor distant objects within the exclusion zone during meteorological tower construction activities.

5) Pre-Construction Briefing. Prior to the start of construction, the lessee must hold a briefing to establish responsibilities of each involved party, define the chains of command, discuss communication procedures, provide an overview of monitoring purposes, and review operational procedures. This briefing must include construction supervisors and crews, and the protected species observer(s) (see further below). The Resident Engineer (or other authorized individual) will have the authority to stop or delay any construction activity, if deemed necessary by the Resident Engineer. New personnel must be briefed as they join the work in progress.

3.6.3.1 Requirements for Pile Driving

1) Prohibition on Pile Driving. The lessee must ensure that no pile-driving activities (e.g. pneumatic, hydraulic, or vibratory installation of foundation piles) occur from November 1 – April 30 nor during an active Dynamic Management Area (DMA) if the pile driving location is within the boundaries of the DMA as established by the National Marine Fisheries Service or within 1 kilometer of the boundaries of the DMA.

2) Establishment of Exclusion Zone. The lessee must ensure the establishment of a default 3281-foot (1,000-meter) radius exclusion zone for cetaceans, sea turtles, and pinnipeds around each pile driving site. The 3,281 feet (1,000 meter) exclusion zone must be monitored from two locations. One observer must be based at or near the sound source and will be responsible for monitoring out to 1,640 feet (500 meters) from the sound source. An additional observer must be located on a separate vessel navigating approximately 3,281 feet (1,000 meters) around the pile hammer and will be responsible for monitoring the area between 500 m to 1,000 m from the sound source.

3) Modification of Exclusion Zone. If multiple piles are being driven, the lessee may use the field verification method described below to modify the default exclusion zone provided above for pile driving activities. Any new exclusion zone radius must be based on the most conservative measurement (i.e., the largest safety zone configuration) of the 180 dB zone.

4) Field Verification of Exclusion Zone. If the lessee wishes to modify the exclusion zone the lessee must conduct a field verification of the exclusion zone during pile driving of the first pile if the meteorological tower foundation design includes multiple piles. The results of the measurements from the first pile must be used to establish a new exclusion

zone which may be greater than or less than the 3281-foot (1,000-meter) default exclusion zone, depending on the results of the field tests. Acoustic measurements must take place during the driving of the last half (deepest pile segment) for any given open-water pile. A minimum of two reference locations must be established at a distance of 1,640 feet (500 meters) and 3281-foot (1,000-meter) from the pile driving. Sound measurements must be taken at the reference locations at two depths (a depth at mid-water and a depth at approximately 1m above the seafloor). Sound pressure levels must be measured and reported in the field in dB re 1 μPa rms (impulse). An infrared range finder may be used to determine distance from the pile to the reference location.

5) Clearance of Exclusion Zone. The lessee must ensure that visual monitoring of the exclusion zone must begin no less than 60 minutes prior to the beginning of soft start and continue until pile driving operations cease or sighting conditions do not allow observation of the sea surface (e.g., fog, rain, darkness). If a cetacean, pinniped, or sea turtle is observed, the observer must note and monitor the position, relative bearing and estimated distance to the animal until the animal dives or moves out of visual range of the observer. The observer must continue to observe for additional animals that may surface in the area, as often there are numerous animals that may surface at varying time intervals.

6) Implementation of Soft Start. The lessee must ensure that a “soft start” be implemented at the beginning of each pile installation in order to provide additional protection to cetaceans, pinnipeds, and sea turtles near the project area by allowing them to vacate the area prior to the commencement of pile driving activities. The soft start requires an initial set of 3 strikes from the impact hammer at 40 percent energy with a one minute waiting period between subsequent 3 strike sets.

7) Shut Down for Cetaceans, Pinnipeds, and Sea Turtles. The lessee must ensure that any time a cetacean, pinniped, and/or sea turtle is observed within the exclusion zone, the observer must notify the Resident Engineer (or other authorized individual) and call for a shutdown of pile driving activity. The pile driving activity must cease as soon as it is safe to do so. Any disagreement or discussion should occur only after shut-down, unless such discussion relates to the safety of the timing of the cessation of the pile driving activity. Subsequent restart of the pile driving equipment may only occur following clearance of the exclusion zone of any cetacean, pinniped, and/or sea turtle for 60 minutes.

8) Pauses in Pile Driving Activity. The lessee must ensure that if pile driving ceases for 30 minutes or more and a cetacean, pinniped, and/or sea turtle is sighted within the exclusion zone prior to re-start of pile driving, the observer(s) must notify the Resident Engineer (or other authorized individual) that an additional 60 minute visual and acoustic observation period must be completed, as described above, before restarting pile driving activities. A pause in pile driving for less than 30 minutes must still begin with soft start but will not require the 60 minute clearance period as long as visual surveys were continued diligently throughout the silent period and the exclusion zone remained clear of cetaceans, pinnipeds, and sea turtles. If visual surveys were not continued diligently

during the pause of 30-minutes or less, the lessee must clear the exclusion zone of all cetaceans, pinnipeds, and sea turtles for 60 minutes.

3.6.3 Protected Species Reporting Requirements

The Lessee must ensure compliance with the following reporting requirements for site characterization activities performed in support of plan (i.e., SAP and/or COP) submittal and must use contact information provided by the Lessor, to fulfill these requirements:

1. Reporting Injured or Dead Protected Species. The Lessee must ensure that sightings of any injured or dead protected species (e.g., marine mammals, sea turtles or sturgeon) are reported to the NMFS Northeast Region's Stranding Hotline ((866) 755-6622 or current) within 24 hours of sighting, regardless of whether the injury or death is caused by a vessel. In addition, if the injury or death was caused by a collision with a project-related vessel, the Lessee must ensure that the Lessor is notified of the strike within 24 hours. The notification of such strike must include the date and location (latitude/longitude) of the strike, the name of the vessel involved, and the species identification or a description of the animal, if possible. If the Lessee's activity is responsible for the injury or death, the Lessee must ensure that the vessel assist in any salvage effort as requested by NMFS.

2. Reporting Observed Impacts to Protected Species. The observer must report any observations concerning impacts on Endangered Species Act listed marine mammals or sea turtles to the Lessor and NMFS within 48 hours. Any injuries or mortalities must be documented on the form provided as Appendix A to this Opinion. Any observed Takes of listed marine mammals or sea turtles resulting in injury or mortality must be reported within 24 hours to the Lessor and NMFS.

3. Report Information. Data on all protected-species observations must be recorded based on standard marine mammal observer collection data by the protected-species observer. This information must include: dates, times, and locations of survey operations; time of observation, location and weather; details of marine mammal sightings (e.g., species, numbers, and behavior); and details of any observed Taking (e.g., behavioral disturbances or injury/mortality).

4. Final Report of G&G Survey Activities and Observations. The lessee must provide the Lessor and NMFS with a report within ninety (90) calendar days following the commencement of HRG and/or geotechnical sampling activities that includes a summary of the survey activities and an estimate of the number of listed marine mammals and sea turtles observed or Taken during these survey activities.

5. Final Technical Report for Meteorological Tower Construction and Observations. The lessee must provide the Lessor and NMFS a report within 120 days after completion of the pile driving and construction activities. The report must include full documentation of methods and monitoring protocols, summarizes the data recorded during monitoring, estimates the number of listed marine mammals and sea turtles that may have been taken during construction activities, and provides an interpretation of the results and effectiveness of all monitoring tasks.

Reports must be sent to:
Bureau of Ocean Energy Management
Environment Branch for Renewable Energy
Phone: 703-787-1340
Email: renewable_reporting@boem.gov

National Marine Fisheries Service
Northeast Regional Office, Protected Resources Division
Section 7 Coordinator
Phone: 978-281-9328
Email: incidental.take@noaa.gov

3.6.4 Other Requirements

3.6.4.1 Requirements for Meteorological Tower Decommissioning
The vessel mitigation measures outlined above will be required. Foundation structures must be removed by cutting at least 15 feet (4.6 meters) below mudline (see 30 CFR 585.910(a). BOEM assumes the meteorological towers to be constructed in southern New England can be removed using non-explosive severing methods. As detailed in 30 CFR Part 585.902, before the lessee decommissions the facilities under their SAP, the lessee must submit a decommissioning application and receive approval from the BOEM. Furthermore, the approval of the decommissioning concept/methodology in the SAP is not an approval of a decommissioning application.

3.6.4.2 Other Non-ESA Related Standard Operating Conditions
The regulations for site assessment plans found at 30 CFR Part 585.610 specify the requirements of a site assessment plan. These include a description of the measures the lessee will use to avoid or minimize adverse effects and any potential incidental take of endangered species before conducting activities on the lease, and how the lessee will mitigate environmental impacts from their proposed activities. 30 CFR 585 Subpart F also specifies measures the lease must take to comply with the Endangered Species Act and the Marine Mammal Protection Act.

3.6.4.3 Site Characterization Data Collection
In addition to the collection of meteorological and oceanographic data, the purpose of these meteorological towers/buoys and site characterization surveys are to also collect biological and archaeological data. This data will assist in future analysis of proposed wind facilities. In addition to required reports, all site characterization data will be shared with NMFS, USFWS, and appropriate State agencies, upon request.

3.7 Action Area
The action area for Section 7 consultations is defined as all of the areas directly or indirectly affected by the Federal action, and not merely the immediate area involved in the action. Therefore, for the purpose of this consultation, the action area for the proposed action is defined as the wind energy areas under consideration, where all surveys and met tower construction or

buoy operation will occur as well as the transit routes to be used by vessels moving from shore based facilities to the offshore wind areas.

4.0 STATUS OF THE SPECIES

This section presents biological and ecological information relevant to formulating the Biological Opinion. Information on species' life history, its habitat and distribution, and other factors necessary for its survival are included to provide background for analyses in later sections of this Opinion.

4.1 Listed Species in the Action Area that are not likely to be adversely affected by the action

We have determined that the actions being considered in the Opinion are not likely to adversely affect shortnose sturgeon (*Acipenser brevirostrum*), the Gulf of Maine DPS of Atlantic salmon (*Salmo salar*), hawksbill sea turtles (*Eretmochelys imbricata*), blue whales (*Balaenoptera musculus*), and sperm whales (*Physeter macrocephalus*), all of which are listed as endangered species under the ESA. Thus, after a brief discussion below these species will not be considered further in this Opinion. Below, we present our rationale for these determinations.

4.1.1 Shortnose sturgeon

Shortnose sturgeon are benthic fish that occur in large coastal rivers of eastern North America. They range from as far south as the St. Johns River, Florida (possibly extirpated from this system) to as far north as the Saint John River in New Brunswick, Canada. Shortnose sturgeon occur in 19 rivers along the U.S. Atlantic coast. Limited information is available on intrabasin movements. Within the Gulf of Maine, some shortnose sturgeon have been documented to make coastal migrations from one river to another. At this time, it is unclear whether this is common in other areas outside of the Gulf of Maine. We do not anticipate that shortnose sturgeon will be present in the action area and therefore, any effects to shortnose sturgeon are extremely unlikely to occur.

4.1.2 Gulf of Maine DPS of Atlantic salmon

The GOM DPS of Atlantic salmon is listed as endangered. The DPS includes all naturally spawned and conservation hatchery populations of anadromous Atlantic salmon whose freshwater range occurs in the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River (NMFS 2009a, 2009b). These populations include those in the Dennys, East Machias, Machias, Pleasant, Narraguagus, Ducktrap, Sheepscot, Penobscot, Androscoggin, and Kennebec Rivers as well as Cove Brook. Juvenile salmon in New England rivers typically migrate to sea in May after a two- to three-year period of development in freshwater streams, and remain at sea for two winters before returning to their U.S. natal rivers to spawn. GOM DPS Atlantic salmon are extremely unlikely to occur in the action area; therefore, any effects to this species are extremely unlikely to occur. The action area is also outside of the area designated as critical habitat for this species; therefore, there will be no effects to critical habitat for the GOM DPS of Atlantic salmon.

4.1.3 Hawksbill sea turtle

The hawksbill sea turtle is listed as endangered. This species is uncommon in the waters of the continental U.S. Hawksbills prefer coral reef habitats, such as those found in the Caribbean and Central America. Mona Island (Puerto Rico) and Buck Island (St. Croix, U.S. Virgin Islands) contain especially important foraging and nesting habitat for hawksbills. Within the continental U.S., nesting is restricted to the southeast coast of Florida and the Florida Keys, but nesting is rare in these areas. Hawksbills have been recorded from all the Gulf States and along the east coast of the U.S. as far north as Massachusetts, but sightings north of Florida are rare. Aside from Florida, Texas is the only other U.S. state where hawksbills are sighted with any regularity. Since hawksbill sea turtles are not expected to be present in the action area, it is extremely unlikely that the proposed action will affect this sea turtle species.

4.1.4 Blue Whales

Blue whales are listed as endangered. These species are unlikely to occur in the action area. Although blue whales are occasionally seen in U.S. waters, they are more commonly found in Canadian waters and are rare in continental shelf waters of the eastern U.S. (Waring *et al.* 2000). Given the predominantly far offshore distribution of this species, it is highly unlikely to occur in the action area or to be affected by the proposed action. Since this species is not expected to be present in the action area, it is extremely unlikely that the proposed action will affect this species.

4.2 Listed Species in the Action Area that may be Affected by the Proposed Action

NMFS has determined that the actions being considered in the Opinion may adversely affect the following listed species:

Cetaceans	
North Atlantic Right whale (*Eubalaena glacialis*)	Endangered
Humpback whale (*Megaptera novaeangliae*)	Endangered
Fin whale (*Balaenoptera physalus*)	Endangered
Sei whale (*Balaenoptera borealis*)	Endangered
Sperm whale (*Physeter macrocephalus*)	Endangered
Sea Turtles	
Northwest Atlantic DPS of Loggerhead sea turtle (*Caretta caretta*)	Threatened
Leatherback sea turtle (*Dermochelys coriacea*)	Endangered
Kemp's ridley sea turtle (*Lepidochelys kempi*)	Endangered
Green sea turtle (*Chelonia mydas*)	Endangered/Threatened[1]
Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*)	
Gulf of Maine DPS	Threatened
New York Bight DPS	Endangered
Chesapeake Bay DPS	Endangered

[1] Pursuant to NMFS regulations at 50 CFR 223.205, the prohibitions of Section 9 of the Endangered Species Act apply to all green turtles, whether endangered or threatened.

South Atlantic DPS	Endangered
Carolina DPS	Endangered

4.2.1 North Atlantic Right whales

Historically, right whales have occurred in all the world's oceans from temperate to subarctic latitudes (Perry *et al.* 1999). In both hemispheres, they are observed at low latitudes and in nearshore waters where calving takes place in the winter months, and in higher latitude foraging grounds in the summer (Clapham *et al.* 1999; Perry *et al.* 1999).

The North Atlantic right whale (*Eubalaena glacialis*) has been listed as endangered under the ESA since 1973. It was originally listed as the "northern right whale" as endangered under the Endangered Species Conservation Act, the precursor to the ESA in June 1970. The species is also designated as depleted under the Marine Mammal Protection Act (MMPA).

In December 2006, NMFS completed a comprehensive review of the status of right whales in the North Atlantic and North Pacific Oceans. Based on the findings from the status review, NMFS concluded that right whales in the northern hemisphere exist as two species: North Atlantic right whale (*Eubalaena glacialis*) and North Pacific right whale (*Eubalaena japonica*). NMFS determined that each of the species is in danger of extinction throughout its range. In 2008, based on the status review, NMFS listed the endangered northern right whale (*Eubalaena spp.*) as two separate endangered species: the North Atlantic right whale (*E. glacialis*) and North Pacific right whale (*E. japonica*) (73 FR 12024; March 6, 2008).

Habitat and Distribution
Western North Atlantic right whales generally occur from the Southeast U.S. to Canada (*e.g.*, Bay of Fundy and Scotian Shelf) (Kenney 2002; Waring *et al.* 2011). Like other right whale species, they follow an annual pattern of migration between low latitude winter calving grounds and high latitude summer foraging grounds (Perry *et al.* 1999; Kenney 2002).

The distribution of right whales seems linked to the distribution of their principal zooplankton prey, calanoid copepods (Winn *et al.* 1986; NMFS 2005; Baumgartner and Mate 2005; Waring *et al.* 2011). Right whales are most abundant in Cape Cod Bay between February and April (Hamilton and Mayo 1990; Schevill *et al.* 1986; Watkins and Schevill 1982) and in the Great South Channel in May and June (Kenney *et al.* 1986; Payne *et al.* 1990; Kenney *et al.* 1995; Kenney 2001) where they have been observed feeding predominantly on copepods of the genera *Calanus* and *Pseudocalanus* (Baumgartner and Mate 2005; Waring *et al.* 2011). Right whales also frequent Stellwagen Bank and Jeffrey's Ledge, as well as Canadian waters including the Bay of Fundy, Browns, and Baccaro Banks in the summer through fall (Mitchell *et al.* 1986; Winn *et al.* 1986; Stone *et al.* 1990). The consistency with which right whales occur in such locations is relatively high, but these studies also highlight the high interannual variability in right whale use of some habitats. Calving is known to occur in the winter months in coastal waters off Georgia and Florida (Kraus *et al.* 1988). Calves have also been sighted off the coast of North Carolina during winter months suggesting the calving grounds may extend as far north as Cape Fear, NC. In the North Atlantic, it appears that not all reproductively active females return to the calving grounds each year (Kraus *et al.*, 1986; Payne, 1986). Patrician *et al.* (2009) analyzed photographs of a right whale calf sighted in the Great South Channel in June 2007 and determined the calf appeared too young to have been born in the known southern calving area. Although it is possible the female traveled south to New Jersey or Delaware to give birth, evidence suggests that calving in waters of the northeastern U.S. is possible. The location of some portion of the population during the winter months remains unknown (NMFS 2005a). However, recent aerial surveys conducted under the North Atlantic Right Whale Sighting Survey (NARWSS) program have indicated that some individuals may reside in the northern Gulf of Maine during the winter. In 2008, 2009, and 2010, right whales were sighted on Jeffreys and Cashes Ledges, Stellwagen Bank, and Jordan Basin from December to February (Khan *et al*. 2009, 2010, 2011).

While right whales are known to congregate in the aforementioned areas, much is still not understood about their seasonal distribution, and movements within and between these areas (Waring *et al.* 2011). In the winter, only a portion of the known right whale population is seen on the calving grounds. The winter distribution of the remaining right whales remains uncertain (NMFS 2005, Waring *et al.* 2011). Results from winter surveys and passive acoustic studies suggest that animals may be dispersed in several areas including Cape Cod Bay (Brown *et al.* 2002) and offshore waters of the southeastern U.S. (Waring *et al.* 2011). On multiple days in December 2008, congregations of more than 40 individual right whales were observed in the Jordan Basin area of the Gulf of Maine, leading researchers to believe this may be a wintering ground (NOAA 2008). Telemetry data have shown lengthy and somewhat distant excursions into deep water off the continental shelf (Mate *et al.* 1997) as well as extensive movements over the continental shelf during the summer foraging period (Mate *et al.* 1992; Mate *et al.* 1997; Bowman 2003; Baumgartner and Mate 2005). Knowlton *et al.* (1992) reported several long-distance movements as far north as Newfoundland, the Labrador Basin, and southeast of Greenland; in addition, resightings of photographically identified individuals have been made off Iceland, arctic Norway, and in the old Cape Farewell whaling ground east of Greenland. The Norwegian sighting (September 1999) is one of only two sightings in the 20th century of a right whale in Norwegian waters, and the first since 1926. Together, these long-range matches indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not presently well described. Similarly, records from the Gulf of Mexico (Moore and Clark, 1963; Schmidly *et al.*, 1972) represent either geographic anomalies or a more extensive historic range beyond the sole known calving and wintering ground in the waters of the southeastern United States. The frequency with which right whales occur in offshore waters in the southeastern U.S. remains unclear (Waring *et al.* 2011).

Abundance estimates and trends

An estimate of the pre-exploitation population size for the North Atlantic right whale is not available. As is the case with most wild animals, an exact count of North Atlantic right whales cannot be obtained. However, abundance can be reasonably estimated from the extensive study of western North Atlantic right whale population. IWC participants from a 1999 workshop agreed to a minimum direct-count estimate of 263 right whales alive in 1996 and noted that the true population was unlikely to be greater than this estimate (Best *et al.* 2001). Based on a census of individual whales using photo-identification techniques and an assumption of mortality for those whales not seen in seven years, a total of 299 right whales was estimated in 1998 (Kraus *et al*. 2001), and a review of the photo-ID recapture database on July 6, 2010, indicated that 396 individually recognized whales were known to be alive during 2007 (Waring *et al.* 2011). Because this 2009 review was a nearly complete census, it is assumed this estimate represents a minimum population size. The minimum number alive population index for the years 1990-2007 suggests a positive trend in numbers. These data reveal a significant increase in the number of catalogued whales alive during this period, but with significant variation due to apparent losses exceeding gains during 1998-1999. Mean growth rate for the period was 2.4% (Waring *et al*. 2011).

A total of 297 right whale calves were born during the years 1993-2009 (Waring *et al*. 2011). The mean calf production for this 15-year period is estimated to be 17.2/year (Waring *et al.* 2011). Calving numbers have been variable, with large differences among years, including a second largest calving season in 2000/2001 with 31 right whale births (Waring *et al.* 2011). The three calving years (97/98; 98/99; 99/00) prior to this record year provided low recruitment levels with only 11 calves born. The last nine calving seasons (2000-2009) have been remarkably better with 31, 21, 19, 17, 28, 19, 23, 23, and 39 births, respectively (Waring *et al.* 2011). However, the western North Atlantic stock has also continued to experience losses of calves, juveniles, and adults.

As is the case with other mammalian species, there is an interest in monitoring the number of females in this western North Atlantic right whale population since their numbers will affect the population trend (whether declining, increasing or stable). Kraus *et al.* (2007) reported that as of 2005, 92 reproductively active females had been identified and Schick *et al.* (2009) estimated 97 breeding females. From 1983 to 2005, the number of new mothers recruited to the population (with an estimated age of 10 for the age of first calving), varied from 0-11 each year with no significant increase or decline over the period (Kraus *et al.* 2007). By 2005, 16 right whales had produced at least six calves each, and four cows had at least seven calves. Two of these cows were at an age that indicated a reproductive life span of at least 31 years (Kraus *et al.* 2007). As described above, the 2000/2001 through 2006/2007 calving seasons had relatively high calf production and have included several first-time mothers (*e.g.*, eight new mothers in 2000/2001). However, over the same time period, there have been continued losses to the western North Atlantic right whale population, including the death of mature females, as a result of anthropogenic mortality (including that described in Henry *et al.* 2011, below). Of the 12 serious injuries and mortalities between 2005 and 2009, at least six were adult females, three of which were carrying near-term fetuses and four of which were just starting to bear calves (Waring *et al.* 2011). Since the average lifetime calf production is 5.25 calves (Fujiwara and Caswell 2001), the deaths of these six females represent a loss of reproductive potential of as many as 32 animals. However, it is important to note that not all right whale mothers are equal with regards to calf production. Right whale #1158 had only one calf over a 25-year period (Kraus *et al.* 2007). In contrast, one of the largest right whales on record was a female nicknamed "Stumpy," who was killed in February 2004 of an apparent ship strike (NMFS 2006a). She was first sighted in 1975 and known to be a prolific breeder, successfully rearing calves in 1980, 1987, 1990, 1993, and 1996 (Moore *et al.* 2007). At the time of her death, she was estimated to be 30 years of age and carrying her sixth calf; the near-term fetus also died (NMFS 2006a).

Abundance estimates are an important part of assessing the status of the species. However, for section 7 purposes, the population trend (*i.e.*, whether increasing or declining) provides better information for assessing the effects of a proposed action on the species. As described in previous biological opinions, data collected in the 1990s suggested that right whales were experiencing a slow but steady recovery (Knowlton *et al.* 1994). However, Caswell *et al.* (1999) used photo-identification data and modeling to estimate survival and concluded that right whale survival decreased from 1980 to 1994. Modified versions of the Caswell *et al.* (1999) model as well as several other models were reviewed at a 1999 IWC workshop (Best *et al.* 2001). Despite differences in approach, all of the models indicated a decline in right whale survival in the 1990s compared to the 1980s, with female survival, in particular, apparently affected (Best *et al.* 2001). In 2002, NMFS' Northeast Fisheries Science Center (NEFSC) hosted a workshop to review right whale population models to examine: (1) potential bias in the models and (2) changes in the subpopulation trend based on new information collected in the late 1990s (Clapham *et al.* 2002). Three different models were used to explore right whale survivability and to address potential sources of bias. Although biases were identified that could negatively affect the results, all three modeling techniques resulted in the same conclusion; survival has continued to decline and seems to be focused on females (Clapham *et al.* 2002). Increased mortalities in 2004 and 2005 were cause for serious concern (Kraus *et. al* 2005). Calculations indicate that this increased mortality rate would reduce population growth by approximately 10% per year (Kraus *et. al* 2005). Despite the preceding, examination of the minimum number alive population index calculated from the individual sightings database (as it existed on July 6, 2010) suggest a positive trend in numbers for the years 1990-2007 (Waring *et al.* 2011). These data reveal a significant increase in the number of catalogued right whales alive during this period, but with significant variation due to apparent losses exceeding gains during 1998-1999 (Waring *et al.* 2011). Recently, NMFS NEFSC developed a population viability analysis (PVA) to examine the influence of anthropogenic mortality reduction on the recovery prospects for the species (Pace, unpublished). The PVA evaluated several scenarios on how the populations would fare without entanglement mortalities compared to the status quo. Only two of 1,000 projections (with the status quo simulation) ended with a smaller total population size than the starting population size, and no projections resulted in extinction. As described above, the mean growth rate estimated in the latest stock assessment report, for the period 1990-2007, was 2.4% (Waring *et al.* 2011). The potential biological removal (PBR)[2] for the Western Atlantic stock of North Atlantic right whale is 0.8

Reproduction

Healthy reproduction is critical for the recovery of the North Atlantic right whale (Kraus *et al.* 2007). Researchers have suggested that the population has been affected by a decreased reproductive rate (Best *et al.* 2001; Kraus *et al.* 2001). Kraus *et al.* (2007) reviewed reproductive parameters for the period 1983-2005, and estimated calving intervals to have changed from 3.5 years in 1990 to more than 5 years between 1998-2003, and then decreased to just over 3 years in 2004 and 2005.

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[2] Potential biological removal is the product of minimum population size, one-half the maximum net productivity rate and a "recovery" factor for endangered, depleted, threatened stocks, or stocks of unknown status relative to optimum sustainable population.

Factors that have been suggested as affecting the right whale reproductive rate include reduced genetic diversity (and/or inbreeding), contaminants, biotoxins, disease, and nutritional stress. Although it is believed that a combination of these factors is likely affecting right whales (Kraus *et al.* 2007), there is currently no evidence to support this. The dramatic reduction in the North Atlantic right whale population due to commercial whaling may have resulted in a loss of genetic diversity, which could affect the ability of the current population to successfully reproduce (*i.e.*, decreased conceptions, increased abortions, and increased neonate mortality). One hypothesis is that the low level of genetic variability in this species produces a high rate of mate incompatibility and unsuccessful pregnancies (Frasier *et al.* 2007). Analyses are currently under way to assess this relationship further as well as the influence of genetic characteristics on the potential for species recovery (Frasier *et al.* 2007). Studies by Schaeff *et al.* (1997) and Malik *et al.* (2000) indicate that western North Atlantic right whales are less genetically diverse than southern right whales. While contaminant studies have confirmed that right whales are exposed to and accumulate contaminants, researchers could not conclude that these contaminant loads were negatively affecting right whale reproductive success since concentrations were lower than those found in other marine mammals proven to be affected by PCBs and DDT (Weisbrod *et al.* 2000). Another suite of contaminants (i.e. antifouling agents and flame retardants) that disrupt reproductive patterns and have been found in other marine animals, raises new concerns (Kraus *et al.* 2007). Recent data also support a hypothesis that chromium, an industrial pollutant, may be a concern for the health of the North Atlantic right whales and that inhalation may be an important exposure route (Wise *et al.* 2008).

A number of diseases could be also affecting reproduction, although tools for assessing disease factors in free-swimming large whales currently do not exist (Kraus *et al*. 2007). Once developed, such methods may allow for the evaluation of diseases on right whales. Impacts of biotoxins on marine mammals are also poorly understood, yet there are some data showing that marine algal toxins may play significant roles in mass mortalities of large whales (Rolland *et al.* 2007). Although there are no published data concerning the effects of biotoxins on right whales, researchers are certain that right whales are being exposed to measurable quantities of paralytic shellfish poisioning (PSP) toxins and domoic acid via trophic transfer through the presence of these biotoxins in their prey (Durbin *et al.* 2002, Rolland *et al.* 2007).

Data on food-limitation are difficult to evaluate (Kraus *et al.* 2007). North Atlantic right whales seem to have thinner blubber than right whales from the South Atlantic (Kenney 2002; Miller *et al.* 2011). Miller *et al*. 2011 suggest that lipids in the blubber are used as energetic support for reproduction in female right whales. In the same study, blubber thickness was also compared among years of differing prey abundances. During a year of low prey abundances, right whales had significantly thinner blubber than during years of greater prey abundances. The results suggest that blubber thickness is indicative of right whale energy balance and that the marked fluctuations in the North Atlantic right whale reproduction have a nutritional component (Miller *et al*. 2011).

Modeling work by Caswell *et al.* (1999) and Fujiwara and Caswell (2001) suggests that the North Atlantic Oscillation (NAO), a naturally occurring climatic event, affects the survival of mothers and the reproductive rate of mature females, and it also seems to affect calf survival (Clapham *et al.* 2002). Greene *et al.* (2003) described the potential oceanographic processes linking climate variability to the reproduction of North Atlantic right whales. Climate-driven changes in ocean circulation have had a significant impact on the plankton ecology of the Gulf of Maine, including effects on *Calanus finmarchicus*, a primary prey resource for right whales. Researchers found that during the 1980s, when the NAO index was predominately positive, *C. finmarchicus* abundance was also high; when a record drop occurred in the NAO index in 1996, *C. finamarchicus* abundance levels also decreased significantly. Right whale calving rates since the early 1980s seem to follow a similar pattern, where stable calving rates were noted from 1982 to 1992, but then two major, multi-year declines occurred from 1993 to 2001, consistent with the drops in copepod abundance. It has been hypothesized that right whale calving rates are thus a function of food availability as well as the number of females available to reproduce (Greene *et al.* 2003, Greene and Pershing 2004). Such findings suggest that future climate change may emerge as a significant factor influencing the recovery of right whales. Some believe the effects of increased climate variability on right whale calving rates should be incorporated into future modeling studies so that it may be possible to determine how sensitive right whale population numbers are to variable climate forcing (Greene and Pershing 2004).

Anthropogenic Mortality

Right whale recovery is negatively affected by anthropogenic mortality. From 2005 to 2009, right whales had the highest proportion of reported entanglement and ship strike events for a species (Waring *et al.* 2011). Given the small population size and low annual reproductive rate of right whales, human sources of mortality may have a greater effect on population growth rate than for other large whale species (Waring *et al.* 2011). For the period 2005-2009, the annual human-caused mortality and serious injury rate for the North Atlantic right whale averaged 2.4 per year (2.0 in U.S. waters; 0.4 in Canadian waters) (Waring *et al*. 2011). Twenty confirmed right whale mortalities were reported along the U.S. East Coast and adjacent Canadian Maritimes from 2005 to 2009 (Henry *et al*. 2011). These numbers represent the minimum values for serious injury and mortality for this period. Given the range and distribution of right whales in the North Atlantic, and the fact that positively buoyant species like right whales may become negatively buoyant if injury prohibits effective feeding for prolonged periods, it is highly unlikely that all carcasses will be observed (Moore *et. al.* 2004, Glass *et al.* 2009). Moreover, carcasses floating at sea often cannot be examined sufficiently and may generate false negatives if they are not towed to shore for further necropsy (Glass *et al*. 2009). Decomposed and/or unexamined animals represent lost data, some of which may relate to human impacts (Waring *et al.* 2011).

Considerable effort has been made to examine right whale carcasses for the cause of death (Moore *et al*. 2004). Examination is not always possible or conclusive because carcasses may be discovered floating at sea and cannot be retrieved, or may be in such an advanced stage of decomposition that a complete examination is not possible. Wave action and post-mortem predation by sharks can also damage carcasses, and preclude a thorough examination of all body parts. It should also be noted that mortality and serious injury event judgments are based upon the best available data and later information may result in revisions (Henry *et al.* 2011). Of the 20 total, confirmed right whale mortalities (2005-2009) described in Henry *et al.* (2011), 2 were confirmed to be entanglement mortalities (1 female calf, 1 male calf) and 6 were confirmed to be ship strike mortalities (3 adult females, 1 female of unknown age, 1 male calf, and 1 yearling male). Serious injury involving right whales was documented for 2 entanglement events (juvenile sex unknown, juvenile male) and 2 ship strike events (1 adult female and 1 yearling male).

Although disentanglement is either unsuccessful or not possible for the majority of cases, during the period of 2005-2009, there were at least three documented cases of entanglements for which the intervention of disentanglement teams likely averted a serious injury (Waring *et al.* 2011). Even when entanglement or vessel collision does not cause direct mortality, it may weaken or compromise individuals so that subsequent injury or death is more likely (Waring *et. al* 2011). Some right whales that have been entangled were later involved in ship strikes (Hamilton *et al.* 1998), suggesting that the animal may have become debilitated by the entanglement to such an extent that it was less able to avoid a ship. Similarly, skeletal fractures and/or broken jaws sustained during a vessel collision may heal, but then compromise a whale's ability to efficiently filter feed (Moore *et al.* 2007). A necropsy of right whale #2143 ("Lucky") found dead in January 2005 suggested the animal (and her near-term fetus) died after healed propeller wounds from a ship strike re-opened and became infected as a result of pregnancy (Moore *et al.* 2007, Glass *et al.* 2008). Sometimes, even with a successful disentanglement, an animal may die of injuries sustained by fishing gear (e.g. RW #3107) (Waring *et al.* 2011).

NMFS' entanglement records from 1990 to 2009 include 94 confirmed right whale entanglement events (Waring *et al.* 2011). Because whales often free themselves of gear following an entanglement event, scarification analysis of living animals may provide better indications of fisheries interactions than entanglement records (Waring *et al.* 2011). Data presented in Knowlton *et al.* (2008) indicate the annual rate of entanglement interaction remains at high levels. Four hundred and ninety-three individual, catalogued right whales were reviewed and 625 separate entanglement interactions were documented between 1980 and 2004. Approximately 358 out of 493 animals (72.6% of the population) were entangled at least once; 185 animals bore scars from a single entanglement, however one animal showed scars from 6 different entanglement events. The number of male and female right whales bearing entanglement scars was nearly equivalent (142/202 females, 71.8%; 182/224 males, 81.3%), indicating that both sexes are equally vulnerable to entanglement. However, juveniles appear to become entangled at a higher rate than expected if all age groups were equally vulnerable. For all years but one (1998), the proportion of juvenile, entangled right whales exceeded their proportion within the population. Based on photographs of catalogued animals from 1935 through 1995, Hamilton *et al.* (1998) estimated that 6.4% of the North Atlantic right whale population exhibit signs of injury from vessel strikes. Reports received from 2005 to 2009 indicate that right whales had the greatest number of ship strike mortalities (n=6) and serious injuries (n=2) compared to other large whales in the Northwest Atlantic (Henry *et al.* 2011). In 2006 alone, four reported mortalities and one serious injury of right whales resulted from ship strikes (Henry *et al.* 2011).

Right whales are expected to be affected by climate change; however, no significant climate change-related impacts to right whales have been observed to date. The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats, and the potential decline of forage.

The North Atlantic right whale currently has a range of sub-polar to sub-tropical waters. An increase in water temperature would likely result in a northward shift of range, with both the northern and southern limits moving poleward. The northern limit, which may be determined by feeding habitat and the distribution of preferred prey, may shift to a greater extent than the southern limit, which requires ideal temperature and water depth for calving. This may result in an unfavorable effect on the North Atlantic right whale due to an increase in the length of migrations (Macleod 2009) or a favorable effect by allowing them to expand their range.
The indirect effects to right whales, that may be associated with sea level rise, are the construction of sea-wall defenses and protective measures for coastal habitats, which may impact coastal marine species and may interfere with migration (Learmonth *et al.* 2006). The effect of sea level rise to cetaceans is likely negligible.

The direct effects of increased CO_2 concentrations, and associated decrease in pH (ocean acidification), on marine mammals are unknown (Learmonth *et al.* 2006). Marine plankton is a vital food source for many marine species. Studies have demonstrated adverse impacts from ocean acidification on the ability of free-swimming zooplankton to maintain protective shells as well as a reduction in the survival of larval marine species. A decline in marine plankton could have serious consequences for the marine food web.

Summary of Right Whale Status
In March 2008, NMFS listed the North Atlantic right whale as a separate, endangered species (*Eubalaena glacialis*) under the ESA. This decision was based on an analysis of the best scientific and commercial data available. The decision took into consideration current population trends and abundance, demographic risk factors affecting the continued survival of the species, and ongoing conservation efforts. NMFS determined that the North Atlantic right whale is in danger of extinction throughout its range because of: (1) overutilization for commercial, recreational, scientific or educational purposes; (2) the inadequacy of existing regulatory mechanisms; and (3) other natural and manmade factors affecting its continued existence.

Previous models estimated that the right whale population in the Atlantic numbered 300 (+/- 10%) (Best *et al*. 2001). However, a review of the photo-ID recapture database on July 6, 2010, indicated that 396 individually recognized right whales were known to be alive in 2007 (Waring *et al.* 2011). The 2000/2001-2008/2009 calving seasons have had relatively high calf production (31, 21, 19, 17, 28, 19, 23, 23, and 39 calves, respectively) and have included additional first time mothers (*e.g.*, eight new mothers in 2000/2001) (Waring *et al*. 2009, 2011).

Over the five-year period 2005-2009, right whales had the highest proportion of reported entanglements and ship strikes for a species: of 60 reports involving right whales, 29 were confirmed entanglements and 17 were confirmed ship strikes. There were 20 verified right whale mortalities, 2 due to entanglements, and 6 due to ship strikes (Henry *et al.* 2011). This represents an absolute minimum number of the right whale mortalities for this period. Given the range and distribution of right whales in the North Atlantic, it is highly unlikely that all carcasses will be observed. Scarification analysis indicates that some whales do survive encounters with ships and fishing gear. However, the long-term consequences of these interactions are unknown. Right whale recovery is negatively affected by human causes of mortality. This mortality appears to, have a greater impact on the population growth rate of right whales, compared to other baleen whales in the western North Atlantic, given the small population size and low annual reproductive rate of right whales (Waring *et al.* 2011).

A variety of modeling exercises and analyses indicate that survival probability declined in the 1990s (Best *et al.* 2001), and mortalities in 2004-2005, including a number of adult females, also suggested an increase in the annual mortality rate (Kraus *et al*. 2005). Nonetheless, a census of the minimum number alive population index calculated from the individual sightings database as of July 6, 2010, for the years 1990-2007 suggest a positive trend in the population growth rate of right whales (Waring *et al.* 2011). In addition, calving intervals appear to have declined to three years in recent years (Kraus *et al.* 2007), and calf production has been relatively high over the past several seasons. The most recent stock assessment report for right whales indicates that examination of the minimum number alive population index calculated from the individual sightings database, as it existed on July 6, 2010, for the years 1990-2007, suggests a positive trend in population size; mean growth rate for the period was 2.4% (Waring et al. 2011).

4.2.2 Humpback Whales

Humpback whales inhabit all major ocean basins from the equator to subpolar latitudes. With the exception of the northern Indian Ocean population, they generally follow a predictable migratory pattern in both hemispheres, feeding during the summer in the higher near-polar latitudes and migrating to lower latitudes in the winter where calving and breeding takes place (Perry *et al.* 1999). Humpbacks are listed as endangered under the ESA at the species level and are considered depleted under the MMPA. Therefore, information is presented below regarding the status of humpback whales throughout their range.

North Pacific, Northern Indian Ocean, and Southern Hemisphere
Humpback whales in the North Pacific feed in coastal waters from California to Russia and in the Bering Sea. They migrate south to wintering destinations off Mexico, Central America, Hawaii, southern Japan, and the Philippines (Carretta *et al*. 2011). Although the IWC only considered one stock (Donovan 1991) there is evidence to indicate multiple populations migrating between their summer/fall feeding areas to winter/spring calving and mating areas within the North Pacific Basin (Angliss and Outlaw 2007, Carretta *et al.* 2011).

NMFS recognizes three management units within the U.S. EEZ in the Pacific for the purposes of managing this species under the MMPA. These are: the California-Oregon-Washington stock (feeding areas off the U.S. West Coast), the central North Pacific stock (feeding areas from Southeast Alaska to the Alaska Peninsula) and the western North Pacific stock (feeding areas from the Aleutian Islands, the Bering Sea, and Russia) (Carretta *et al.* 2011). Because fidelity appears to be greater in feeding areas than in breeding areas, the stock structure of humpback whales is defined based on feeding areas (Carretta *et al.* 2011). Recent research efforts via the Structure of Populations, Levels of Abundance, and Status of Humpback Whales (SPLASH) Project estimate the abundance of humpback whales to be just under 20,000 whales for the entire North Pacific, a number that doubles previous population predictions (Calambokidis *et al.* 2008). There are indications that the California-Oregon-Washington stock was growing in the 1980s and early 1990s with a best estimate of 8% growth per year (Carretta *et al.* 2011). The best available estimate for the California-Oregon-Washington stock is 2,043 whales (Carretta *et al*. 2011). The central North Pacific stock is estimated at 4,005 (Allen and Angliss 2011), and various studies report that it appears to have increased in abundance at rates of 6.6%-10% per year (Allen and Angliss 2011). Although there is no reliable population trend data for the western North Pacific stock, as surveys of the known feeding areas are incomplete and many feeding areas remain unknown, minimum population size is currently estimated at 732 whales (Allen and Angliss 2011).

The Northern Indian Ocean population of humpback whales consists of a resident stock in the Arabian Sea, which apparently does not migrate (Minton *et al.* 2008). The lack of photographic matches with other areas suggests this is an isolated subpopulation. The Arabian Sea subpopulation of humpback whales is geographically, demographically, and genetically isolated, residing year round in sub-tropical waters of the Arabian Sea (Minton *et al.* 2008). Although potentially an underestimate due to small sample sizes and insufficient spatial and temporal coverage of the population's suspected range, based on photo-identification, the abundance estimate off the coast of Oman is 82 animals [60-111 95% confidence interval (CI)](Minton *et al.* 2008).

The Southern Hemisphere population of humpback whales is known to feed mainly in the Antarctic, although some have been observed feeding in the Benguela Current ecosystem on the migration route west of South Africa (Reilly *et al.* 2008). The IWC Scientific Committee recognizes seven major breeding stocks, some of which are tentatively further subdivided into substocks. The seven major breeding stocks, with their respective breeding ground estimates in parenthesis, include Southwest Atlantic (6,251), Southeast Atlantic (1,594), southwestern Indian Ocean (5,965), southeastern Indian Ocean (10,032), Southwest Pacific (7,472), Central South Pacific (not available), and Southeast Pacific (2,917) (Reilly *et al.* 2008). The total abundance estimate of 36,600 humpback whales for the Southern Hemisphere is negatively biased due to no available abundance estimate for the central South Pacific subpopulation and only a partial estimate for the Southeast Atlantic subpopulation. Additionally, these abundance estimates have been obtained on each subpopulation's wintering grounds, and the possibility exists that the entire population does not migrate to the wintering grounds (Reilly *et al.* 2008).

Like other whales, southern hemisphere humpback whales were heavily exploited for commercial whaling. Although they were given protection by the IWC in 1963, Soviet whaling data made available in the 1990s revealed that 48,477 southern hemisphere humpback whales were taken from 1947 to 1980, contrary to the original reports to the IWC that accounted for the take of only 2,710 humpbacks (Zemsky *et al.* 1995, IWC 1995, Perry *et al.* 1999).

Gulf of Maine (North Atlantic)

Humpback whales from most Atlantic feeding areas calve and mate in the West Indies and migrate to feeding areas in the northwestern Atlantic during the summer months. Most of the humpbacks that forage in the Gulf of Maine visit Stellwagen Bank and the waters of Massachusetts and Cape Cod Bay. Previously, the North Atlantic humpback whale population was treated as a single stock for management purposes; however due to the strong fidelity to the region displayed by many whales, the Gulf of Maine stock was reclassified as a separate feeding stock (Waring *et al.* 2011). The Gulf of St. Lawrence, Newfoundland/Labrador, western Greenland, Iceland, and northern Norway are the other regions that represent relatively discrete subpopulations. Sightings are most frequent from mid-March through November between 41°N and 43°N, from the Great South Channel north along the outside of Cape Cod to Stellwagen Bank and Jeffreys Ledge (CeTAP 1982) and peak in May and August. Small numbers of individuals may be present in this area year-round, including the waters of Stellwagen Bank. They feed on a number of species of small schooling fishes, particularly sand lance and Atlantic herring, targeting fish schools and filtering large amounts of water for their associated prey. Humpback whales may also feed on euphausiids (krill) as well as capelin (Waring *et al*. 2011, Stevick *et al.* 2006).

In winter, whales from waters off New England, Canada, Greenland, Iceland, and Norway, migrate to mate and calve primarily in the West Indies, where spatial and genetic mixing among these groups occurs (Waring *et al.* 2011). Various papers (Clapham and Mayo 1990; Clapham 1992; Barlow and Clapham 1997; Clapham *et al.* 1999) summarize information gathered from a catalogue of photographs of 643 individuals from the western North Atlantic population of humpback whales. These photographs identified reproductively mature western North Atlantic humpbacks wintering in tropical breeding grounds in the Antilles, primarily on Silver and Navidad Banks north of the Dominican Republic. The primary winter range also includes the Virgin Islands and Puerto Rico (NMFS 1991).

Humpback whales use the Mid-Atlantic as a migratory pathway to and from the calving/mating grounds, but it may also be an important winter feeding area for juveniles. Since 1989, observations of juvenile humpbacks in the Mid-Atlantic have been increasing during the winter months, peaking January through March (Swingle *et al.* 1993). Biologists theorize that non-reproductive animals may be establishing a winter feeding range in the Mid-Atlantic since they are not participating in reproductive behavior in the Caribbean. Swingle *et al.* (1993) identified a shift in distribution of juvenile humpback whales in the nearshore waters of Virginia, primarily in winter months. Identified whales using the Mid-Atlantic area were found to be residents of the Gulf of Maine and Atlantic Canada (Gulf of St. Lawrence and Newfoundland) feeding groups, suggesting a mixing of different feeding populations in the Mid-Atlantic region. Strandings of humpback whales have increased between New Jersey and Florida since 1985 consistent with the increase in Mid-Atlantic whale sightings. Strandings were most frequent September through April in North Carolina and Virginia waters, and were composed primarily of juvenile humpback whales of no more than 11 meters in length (Wiley *et al.* 1995).

Abundance Estimates and Trends

Photographic mark-recapture analyses from the Years of the North Atlantic Humpback (YONAH) project gave an ocean-basin-wide estimate of 11,570 animals during 1992/1993 and an additional genotype-based analysis yielded a similar but less precise estimate of 10,400 whales (95% c.i. = 8,000-13,600) (Waring *et al.* 2011). For management purposes under the MMPA, the estimate of 11,570 individuals is regarded as the best available estimate for the North Atlantic population (Waring *et al.* 2011). The best recent estimate for the Gulf of Maine stock is 847 whales, derived from a 2006 line-transect aerial sighting survey (Waring *et al.* 2011).

Population modeling, using data obtained from photographic mark-recapture studies, estimates the growth rate of the Gulf of Maine stock to be 6.5% for the period 1979-1991 (Barlow and Clapham 1997). More recent analysis for the period 1992-2000 estimated lower population growth rates ranging from 0% to 4.0%, depending on calf survival rate (Clapham *et al.* 2003 in Waring *et al.* 2011). However, it is unclear whether the apparent decline in growth rate is a biased result due to a shift in distribution documented for the period 1992-1995, or whether the population growth rates truly declined due to high mortality of young-of-the-year whales in U.S. Mid-Atlantic waters (Waring *et al.* 2011). Regardless, calf survival appears to have increased since 1996, presumably accompanied by an increase in population growth (Waring *et al.* 2011). Stevick *et al.* (2003) calculated an average population growth rate of 3.1% in the North Atlantic population overall for the period 1979-1993. The PBR for the Gulf of Maine stock of humpback whale is 1.1.

Anthropogenic Injury and Mortality
As with other large whales, the major known sources of anthropogenic mortality and injury of humpback whales occur from fishing gear entanglements and ship strikes. For the period 2005-2009, the minimum annual rate of human-caused mortality and serious injury to the Gulf of Maine humpback whale stock averaged 5.2 animals per year (U.S. waters, 4.8; Canadian waters, 0.4) (Waring *et al.* 2011). Between 2005 and 2009, humpback whales were involved in 94 confirmed entanglement events and 18 confirmed ship strike events (Henry *et al.* 2011). Over the five-year period, humpback whales were the most commonly observed entangled whale species; entanglements accounted for 6 mortalities and 12 serious injuries (Henry *et al.* 2011). Of the 18 confirmed ship strikes, 7 of the events were fatal (Henry *et al*. 2011). It was assumed that all of these events involved members of the Gulf of Maine stock of humpback whales unless a whale was confirmed to be from another stock. In reports prior to 2007, only events involving whales confirmed to be members of the Gulf of Maine stock were included. There were also many carcasses that washed ashore or were spotted floating at sea for which the cause of death could not be determined. Decomposed and/or unexamined animals (e.g., carcasses reported but not retrieved or no necropsy performed) represent 'lost data' some of which may relate to human impacts (Henry *et al.* 2011, Waring *et al.* 2011).

Based on photographs taken from 2000 to 2002 of the caudal peduncle and fluke of humpback whales, Robbins and Mattila (2004) estimated that at least half (48-57%) of the sample (187 individuals) was coded as having a high likelihood of prior entanglement. Evidence suggests that entanglements have occurred at a minimum rate of 8-10% per year. Scars acquired by Gulf of Maine humpback whales between 2000 and 2002 suggest a minimum of 49 interactions with gear. Based on composite scar patterns, male humpback whales appear to be more vulnerable to entanglement than females. Males may be subject to other sources of injury that could affect scar pattern interpretation. Of the images obtained from a humpback whale breeding ground, 24% showed raw injuries, presumably a result from agonistic interactions. However, current evidence suggests that breeding ground interactions alone cannot explain the higher frequency of healed scar patterns among Gulf of Maine male humpback whales (Robbins and Matilla 2004).

Humpback whales, like other baleen whales, may also be adversely affected by habitat degradation, habitat exclusion, acoustic trauma, harassment, or reduction in prey resources resulting from a variety of activities including fisheries operations, vessel traffic, and coastal development. Currently, there is no evidence that these types of activities are affecting humpback whales. However, Geraci *et al*. (1989) provide strong evidence that a mass mortality of humpback whales in 1987-1988 resulted from the consumption of mackerel whose livers contained high levels of saxitoxin, a naturally occurring red tide toxin, the origin of which remains unknown. The occurrence of a red tide event may be related to an increase in freshwater runoff from coastal development, leading some observers to suggest that such events may become more common among marine mammals as coastal development continues (Clapham *et al.* 1999). There have been three additional known cases of a mass mortality involving large whale species along the East Coast between 1998 and 2008. In the 2006 mass mortality event, 21 dead humpback whales were found between July 10 and December 31, 2006, triggering NMFS to declare an unusual mortality event (UME) for humpback whales in the Northeast United States. The UME was officially closed on December 31, 2007 after a review of 2007 humpback whale strandings and mortality showed that the elevated numbers were no longer being observed. The cause of the 2006 UME has not been determined to date, although investigations are ongoing.

Changes in humpback whale distribution in the Gulf of Maine have been found to be associated with changes in herring, mackerel, and sand lance abundance associated with local fishing pressures (Stevick *et al.* 2006, Waring *et al.* 2011). Shifts in relative finfish species abundance correspond to changes in observed humpback whale movements (Stevick *et al.* 2006). However, there is no evidence that humpback whales were adversely affected by these trophic changes.

Humpback whales are expected to be affected by climate change; however, no significant climate change-related impacts to humpback whales have been observed to date. The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats, and the potential decline of forage.

Of the main factors affecting distribution of cetaceans, water temperature appears to be the main influence on geographic ranges of cetacean species (Macleod 2009). Humpback whales are distributed in all water temperature zones, therefore, it is unlikely that their range will be directly affected by an increase in water temperature.

The indirect effects to humpback whales, that may be associated with sea level rise, are the construction of sea-wall defenses and protective measures for coastal habitats, which may impact coastal marine species and may interfere with migration (Learmonth *et al.* 2006). Cetaceans are unlikely to be directly affected by sea level rise, although important coastal bays for humpback breeding could be affected (IWC 1997).

The direct effects of increased CO_2 concentrations, and associated decrease in pH (ocean acidification), on marine mammals are unknown (Learmonth *et al.* 2006). Marine plankton is a vital food source for many marine species. Studies have demonstrated adverse impacts from ocean acidification on the ability of free-swimming zooplankton to maintain protective shells as well as a reduction in the survival of larval marine species.

Summary of Humpback Whale Status

The best available population estimate for humpback whales in the North Atlantic Ocean is 11,570 animals, and the best, recent estimate for the Gulf of Maine stock is 847 whales (Waring *et al.* 2011). Anthropogenic mortality associated with fishing gear entanglements and ship strikes remains significant. In the winter, mating and calving occurs in areas located outside of the U.S. where the species is afforded less protection. Despite all of these factors, current data suggest that the Gulf of Maine humpback stock is steadily increasing in size (Waring *et al.* 2011). This is consistent with an estimated average trend of 3.1% in the North Atlantic population overall for the period 1979-1993 (Stevick *et al*. 2003). With respect to the species overall, there are also indications of increasing abundance for the California-Oregon-Washington, central North Pacific, and Southern Hemisphere stocks: Southwest Atlantic, Southeast Atlantic, Southwest Indian Ocean, Southeast Indian Ocean, and Southwest Pacific. Trend data is lacking for the western North Pacific stock, the central South Pacific and Southeast Pacific subpopulations of the southern hemisphere humpback whales, and the northern Indian Ocean humpbacks.

4.2.3 Fin Whales

The fin whale (*Balaenoptera physalus*) is listed as endangered under the ESA and also is designated as depleted under the MMPA. Fin whales inhabit a wide range of latitudes between 20-75°N and 20-75°S (Perry *et al.* 1999). The fin whale is ubiquitous in the North Atlantic and occurs from the Gulf of Mexico and Mediterranean Sea northward to the edges of the Arctic ice pack (NMFS 1998a). The overall pattern of fin whale movement is complex, consisting of a less obvious north-south pattern of migration than that of right and humpback whales. Based on acoustic recordings from hydrophone arrays, Clark (1995) reported a general southward flow pattern of fin whales in the fall from the Labrador/Newfoundland region, south past Bermuda, and into the West Indies. The overall distribution may be based on prey availability, as this species preys opportunistically on both invertebrates and fish (Watkins *et al*. 1984). Fin whales feed by gulping prey concentrations and filtering the water for the associated prey. Fin whales are larger and faster than humpback and right whales and are less concentrated in nearshore environments.

Pacific Ocean

Within U.S. waters of the Pacific, fin whales are found seasonally off the coast of North America and Hawaii and in the Bering Sea during the summer (Allen and Angliss 2010). Although stock structure in the Pacific is not fully understood, NMFS recognizes three fin whale stocks in U.S. Pacific waters for the purposes of managing this species under the MMPA. These are: Alaska

(Northeast Pacific), California/Washington/Oregon, and Hawaii (Carretta *et al.* 2011). Reliable estimates of current abundance for the entire Northeast Pacific fin whale stock are not available (Allen and Angliss 2010). A provisional population estimate of 5,700 was calculated for the Alaska stock west of the Kenai Peninsula by adding estimates from multiple surveys (Allen and Angliss 2010). This can be considered a minimum estimate for the entire stock because the surveys covered only a portion of its range (Allen and Angliss 2010). An annual population increase of 4.8% between 1987 and 2003 was estimated for fin whales in coastal waters south of the Alaska Peninsula (Allen and Angliss 2010). This is the first estimate of population trend for North Pacific fin whales; however, it must be interpreted cautiously due to the uncertainty in the initial population estimate and the population structure (Allen and Angliss 2010). The best available estimate for the California/Washington/Oregon stock is 3,044, which is likely an underestimate (Carretta *et al*. 2011). The best available estimate for the Hawaii stock is 174, based on a 2002 line-transect survey (Carretta *et al*. 2011).

Stock structure for fin whales in the southern hemisphere is unknown. Prior to commercial exploitation, the abundance of southern hemisphere fin whales was estimated at 400,000 (IWC 1979, Perry *et al.* 1999). There are no current estimates of abundance for southern hemisphere fin whales. Since these fin whales do not occur in U.S. waters, there is no recovery plan or stock assessment report for the southern hemisphere fin whales.

North Atlantic
NMFS has designated one population of fin whales in U.S. waters of the North Atlantic (Waring *et al.* 2011). This species is commonly found from Cape Hatteras northward. Researchers have suggested the existence of fin whale subpopulations in the North Atlantic based on local depletions resulting from commercial overharvesting (Mizroch and York 1984) or genetics data (Bérubé *et al.* 1998). Photo-identification studies in western North Atlantic feeding areas, particularly in Massachusetts Bay, have shown a high rate of annual return by fin whales, both within years and among years (Seipt *et al.* 1990), suggesting some level of site fidelity. The Scientific Committee of the International Whaling Commission (IWC) has proposed stock boundaries for North Atlantic fin whales. Fin whales off the eastern U.S., Nova Scotia, and southeastern coast of Newfoundland are believed to constitute a single stock of fin whales under the present IWC scheme (Donovan 1991). However, it is uncertain whether the proposed boundaries define biologically isolated units (Waring *et al.* 2011).

During the 1978-1982 aerial surveys, fin whales accounted for 24% of all cetaceans and 46% of all large whales sighted over the continental shelf between Cape Hatteras and Nova Scotia (Waring *et al.* 2011). Underwater listening systems have also demonstrated that the fin whale is the most acoustically common whale species heard in the North Atlantic (Clark 1995). The single most important area for this species appeared to be from the Great South Channel, along the 50 meter isobath past Cape Cod, over Stellwagen Bank, and past Cape Ann to Jeffreys Ledge (Hain *et al.*1992).

Like right and humpback whales, fin whales are believed to use North Atlantic waters primarily for feeding, and more southern waters for calving. However, evidence regarding where the majority of fin whales winter, calve, and mate is still scarce. Clark (1995) reported a general pattern of fin whale movements in the fall from the Labrador/Newfoundland region, south past Bermuda and into the West Indies, but neonate strandings along the U.S. Mid-Atlantic coast from October through January suggest the possibility of an offshore calving area (Hain *et al.* 1992).

Fin whales achieve sexual maturity at 6-10 years of age in males and 7-12 years in females (Jefferson *et al.* 2008), although physical maturity may not be reached until 20-30 years (Aguilar and Lockyer 1987). Conception is believed to occur in tropical and subtropical areas during the winter, with the birth of a single calf after an 11-12 month gestation (Jefferson *et al.* 2008). The calf is weaned 6-11 months after birth (Perry *et al.* 1999). The mean calving interval is 2.7 years (Agler *et al.* 1993).

The predominant prey of fin whales varies greatly in different geographical areas depending on what is locally available (IWC 1992). In the western North Atlantic, fin whales feed on a variety of small schooling fish (*i.e.*, herring, capelin, sand lance) as well as squid and planktonic crustaceans (Wynne and Schwartz 1999).

Population Trends and Status

Various estimates have been provided to describe the current status of fin whales in western North Atlantic waters. One method used the catch history and trends in Catch Per Unit Effort (CPUE) to obtain an estimate of 3,590 to 6,300 fin whales for the entire western North Atlantic (Perry *et al.* 1999). Hain *et al.* (1992) estimated that about 5,000 fin whales inhabit the Northeastern U.S. continental shelf waters. The 2011 Stock Assessment Report (SAR) gives a best estimate of abundance for fin whales in the western North Atlantic of 3,985 (CV = 0.24). However, this estimate must be considered extremely conservative in view of the incomplete coverage of the known habitat of the stock and the uncertainties regarding population structure and whale movements between surveyed and unsurveyed areas (Waring *et al*. 2011). The minimum population estimate for the western North Atlantic fin whale is 3,269 (Waring *et al.* 2011). However, there are insufficient data at this time to determine population trends for the fin whale (Waring *et al.* 2011). The PBR for the western North Atlantic fin whale is 6.5.

Other estimates of the abundance of fin whales in the North Atlantic are presented in Pike *et al.* (2008) and Hammond *et al.* (2011). Pike *et al.* (2008) estimates the abundance of fin whales to be 27,493 (CV 0.2) in waters around Iceland and the Denmark Strait. Hammond *et al.* (2008) estimates the abundance of 19,354 (CV 0.24) fin whales in the eastern North Atlantic.

Anthropogenic Injury and Mortality

The major known sources of anthropogenic mortality and injury of fin whales include entanglement in commercial fishing gear and ship strikes. The minimum annual rate of confirmed human-caused serious injury and mortality to North Atlantic fin whales from 2005 to 2009 was 2.6 (U.S. waters, 2.0; Canadian waters, 0.6) (Waring *et al*. 2011). During this five-year period, there were 14 confirmed entanglements (2 fatal; 2 serious injuries) and 12 ship strikes (9 fatal) (Henry *et al.* 2011). Fin whales are believed to be the cetacean most commonly struck by large vessels (Laist *et al.* 2001). In addition, hunting of fin whales continued well into the 20th century. Fin whales were given total protection in the North Atlantic in 1987, with the exception of an aboriginal subsistence whaling hunt for Greenland (Gambell 1993, Caulfield 1993). However, Iceland has increased its whaling activities in recent years and reported a catch of 136 whales in the 1988/89 and 1989/90 seasons (Perry *et al.* 1999), 7 in 2006/07, and 273 in 2009/2010. Fin whales may also be adversely affected by habitat degradation, habitat exclusion, acoustic trauma, harassment, or reduction in prey resources resulting from a variety of activities.

Fin whales are expected to be affected by climate change; however, no significant climate change-related impacts to fin whales have been observed to date. The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats, and the potential decline of forage.

Of the main factors affecting distribution of cetaceans, water temperature appears to be the main influence on geographic ranges of cetacean species (Macleod 2009). Fin whales are distributed in all water temperature zones, therefore, it is unlikely that their range will be directly affected by an increase in water temperature.

The indirect effects to fin whales, that may be associated with sea level rise, are the construction of sea-wall defenses and protective measures for coastal habitats, which may impact coastal marine species and may interfere with migration (Learmonth *et al.* 2006). The effect of sea level rise to fin whales is likely negligible.

The direct effects of increased CO_2 concentrations, and associated decrease in pH (ocean acidification), on marine mammals are unknown (Learmonth *et al.* 2006). Marine plankton is a vital food source for many marine species. Studies have demonstrated adverse impacts from ocean acidification on the ability of free-swimming zooplankton to maintain protective shells as well as a reduction in the survival of larval marine species. A decline in marine plankton could have serious consequences for the marine food web.

Summary of Fin Whale Status
Information on the abundance and population structure of fin whales worldwide is limited. NMFS recognizes three fin whale stocks in the Pacific for the purposes of managing this species under the MMPA. Reliable estimates of current abundance for the entire Northeast Pacific fin whale stock are not available (Angliss *et al.* 2001). Stock structure for fin whales in the southern hemisphere is unknown and there are no current estimates of abundance for southern hemisphere fin whales. As noted above, the best population estimate for the western North Atlantic fin whale is 3,985 and the minimum population estimate is 3,269. The 2011 SAR indicates that there are insufficient data at this time to determine population trends for the fin whale. Fishing gear appears to pose less of a threat to fin whales in the North Atlantic than to North Atlantic right or humpback whales. However, commercial whaling for fin whales in the North Atlantic has resumed and fin whales continue to be struck by large vessels. Based on the information currently available, for the purposes of this Opinion, NMFS considers the population trend for fin whales to be undetermined.

4.2.4 Sei Whales

The sei whale (*Balaenoptera borealis*) has been listed as endangered under the ESA. The species is also designated as depleted under the MMPA. Sei whales are a widespread species in the world's temperate, subpolar, subtropical, and even tropical marine waters. Sei whales reach sexual maturity at 5-15 years of age. The calving interval is believed to be two to three years (Perry *et al.* 1999).

North Pacific and Southern Hemisphere
The IWC only considers one stock of sei whales in the North Pacific (Donovan 1991), but for NMFS management purpose under the MMPA, sei whales within the Pacific U.S. EEZ are divided into three discrete non-contiguous areas: 1) waters around Hawaii, 2) California, Oregon, and Washington waters, and 3) Alaskan waters (Carretta *et al.* 2011). There are no abundance estimates for sei whales in the entire eastern North Pacific. The best estimate of abundance for California, Oregon, and Washington waters out to 300 nautical miles is 126 (CV=0.53) sei whales (Barlow and Forney 2007; Forney 2007; Carretta *et al.* 2011). No fishery related serious injuries or mortalities have been documented from 2004 through 2008 in the eastern North Pacific stock of sei whales (Carretta *et al*. 2011). During 2002-2008, there was one reported ship strike mortality in Washington in 2003 (NMFS Northwest Regional Office, unpublished data). The Hawaiian stock includes animals found both within the Hawaiian Islands EEZ and in adjacent international waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for international waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (Carretta *et al.* 2011). The best estimate of abundance for the Hawaiian stock of sei whales is 77 (CV=1.06). Between 2004 and 2008, no human-caused serious injury or mortality was documented in the Hawaiian stock of sei whales (Carretta *et al.* 2011).

The stock structure of sei whales in the southern hemisphere is unknown. Like other whale species, sei whales in the southern hemisphere were heavily impacted by commercial whaling, particularly in the mid-20th century as humpback, fin, and blue whales became scarce. Sei whales were protected by the IWC in 1977 after their numbers had substantially decreased and they also became more difficult to find (Perry *et al.* 1999). Since southern hemisphere sei whales do not occur in U.S. waters, there is no stock assessment report for southern hemisphere sei whales.

North Atlantic
NMFS considers sei whales in the North Atlantic as one stock known as the Nova Scotia stock (formerly known as the Western North Atlantic stock). Sei whales occur in deep water throughout their range, typically over the continental slope or in basins situated between banks (NMFS 1998a). In the Northwest Atlantic, it is speculated that the whales migrate from south of Cape Cod along the eastern Canadian coast in June and July, and return on a southward migration again in September and October (Waring *et al.* 2011). Olsen *et al.* (2009) tracked a tagged sei whale that moved from the Azores to off eastern Canada; however, such a migration remains unverified. Within the U.S. Atlantic EEZ, the sei whale is most common on Georges Bank and into the Gulf of Maine/Bay of Fundy region during spring and summer, primarily in

deeper waters. Recent springtime research in the Southwestern Gulf of Maine, suggests sei whales are reasonably common in this area in most years (Baumgartner *et al.* 2011).

Although sei whales may prey upon small schooling fish and squid, available information suggests that calanoid copepods and euphausiids are the primary prey of this species (Flinn *et al*. 2002). Sei whales are occasionally seen feeding in association with right whales in the southern Gulf of Maine and in the Bay of Fundy. However, there is no evidence to demonstrate interspecific competition between these species for food resources.

There is limited information on the stock identity of sei whales in the North Atlantic (Waring *et al*. 2011). For purposes of the Marine Mammal Stock Assessment Reports, and based on a proposed IWC stock definition, NMFS recognizes the sei whales occurring from the U.S. East Coast to Cape Breton, Nova Scotia, and east to 42°W longitude as the "Nova Scotia stock" of sei whales (Waring *et al*. 2011).

Abundance Estimates and Trends

The abundance estimate of 386 sei whales (CV=0.85), obtained from a line-transect sighting survey conducted during June 12 to August 4, 2004, by a ship and a plane, covering 10,761 kilometers of trackline in the region from the 100 meter depth contour on the southern edge of Georges Bank to the lower Bay of Fundy, is considered the best available for the Nova Scotia stock of sei whales according to the 2011 SAR (Waring *et al*. 2011). This estimate is considered extremely conservative in view of the known range of the sei whale in the western North Atlantic, and the uncertainties regarding population structure and whale movements between surveyed and unsurveyed areas. Hammond *et al*. (2011) estimates the abundance of sei whales in European Atlantic waters to be 619 (CV of 0.34) for identified sightings identified to species. The minimum population estimate for this sei whale stock is 208 (Waring *et al*. 2011). Current and maximum net productivity rates are unknown for this stock. There are insufficient data to determine trends of the sei whale population (Waring *et al.* 2011). The PBR for the Nova Scotia stock sei whale is 0.4.

Anthropogenic Injury and Mortality

Few instances of injury or mortality of sei whales due to entanglement or vessel strikes have been recorded in U.S. waters, possibly because sei whales typically inhabit waters farther offshore than most commercial fishing operations, or perhaps entanglements do occur but are less likely to be observed. The mean annual rate of confirmed human-caused serious injury and mortality to Nova Scotian sei whales from 2005 to 2009 was 1.2 (Waring *et al*. 2011), which includes 0.6 fishery interaction records and 0.6 vessel collision records. During this five-year period, there were three confirmed entanglements (one fatal; two serious injuries) and three ship strikes (all fatal) (Waring *et al.* 2011). Other impacts noted above for other baleen whales may also occur in this species (e.g., habitat degradation, etc.).

Sei whales are expected to be affected by climate change; however, no significant climate change-related impacts to sei whales have been observed to date. The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats and the potential decline of forage.

Of the main factors affecting distribution of cetaceans, water temperature appears to be the main influence on geographic ranges of cetacean species (Macleod 2009). Sei whales currently range from sub-polar to tropical waters. An increase in water temperature may be a favorable affect on sei whales, allowing them to expand their range into higher latitudes (Macleod 2009).

The indirect effects to sei whales, that may be associated with sea level rise, are the construction of sea-wall defenses and protective measures for coastal habitats, which may impact coastal marine species and may interfere with migration (Learmonth *et al.* 2006). The effect of sea level rise to sei whales is likely negligible.

The direct effects of increased CO_2 concentrations, and associated decrease in pH (ocean acidification), on marine mammals are unknown (Learmonth *et al.* 2006). Marine plankton is a vital food source for many marine species. Studies have demonstrated adverse impacts from ocean acidification on the ability of free-swimming zooplankton to maintain protective shells as well as a reduction in the survival of larval marine species. A decline in marine plankton could have serious consequences for the marine food web.

Summary of Sei Whale Status

The best estimate of abundance for the Nova Scotia stock of sei whales is 386 (Waring *et al.* 2011). There are insufficient data to determine trends of the Nova Scotian sei whale population. Two sei whale serious injuries and one mortality from fisheries interactions and three mortalities from ship strikes have been recorded in U.S. waters between 2005 and 2009 (Waring *et al.* 2011). Information on the status of sei whale populations worldwide is similarly lacking. There are no abundance estimates for sei whales in the entire eastern North Pacific, however the best estimate of abundance for California, Oregon, and Washington waters out to 300 nautical miles is 126 (Carretta *et al.* 2011). The stock structure of sei whales in the southern hemisphere is unknown. Based on the information currently available, for the purposes of this Opinion, NMFS considers the population trend for sei whales to be undetermined.

4.2.5 Sperm Whales

Sperm whales are the largest of the odontocetes (toothed whales) and the most sexually dimorphic cetaceans, with males considerably larger than females. Sperm whales are found throughout the world's oceans in deep waters between about 60° N and 60° S latitudes. During the past two centuries, commercial whalers took about 1,000,000 sperm whales. Despite this high level of take, the sperm whale remains the most abundant of the large whale species. Currently, there is no reliable estimate for the total number of sperm whales worldwide. The best estimate, that there are between 300,000 and 450,000 sperm whales, is based on extrapolations from only a few areas that have useful estimates (Whitehead 2002). The sperm whale was listed as endangered throughout its range on June 2, 1970 under the Endangered Species Conservation Act of 1969. The most recent Recovery Plan was published in 2010 (NMFS 2010).

Species Description, Distribution and Population Structure
Sperm whales tend to inhabit areas with a water depth of 600 meters or more, and are uncommon in waters less than 300 meters deep. Female sperm whales are generally found in deep waters (at least 1000 m) of low latitudes (less than 40°, except in the North Pacific where they are found as high as 50°). These conditions generally correspond to sea surface temperatures greater than 15°C, and while female sperm whales are sometimes seen near oceanic islands, they are typically far from land. Immature males will stay with female sperm whales in tropical and subtropical waters until they begin to slowly migrate towards the poles, anywhere between ages 4 and 21 years old. Older, larger males are generally found near the edge of pack ice in both hemispheres. On occasion, however, these males will return to the warm water breeding area.

In winter, sperm whales are concentrated east and northeast of Cape Hatteras. In spring, the center of distribution shifts northward to east of Delaware and Virginia, and is widespread throughout the central portion of the mid-Atlantic bight and the southern portion of Georges Bank. In summer, the distribution is similar but also includes the areas east and north of Georges Bank and into the Northeast Channel region, as well as the continental shelf (inshore of the 100 m isobath) south of New England. In the fall, sperm whale occurrence south of New England on the continental shelf if at its highest levels, and there remains a continental shelf edge occurrence in the mid-Atlantic bight.

While they may be encountered almost anywhere on the high seas their distribution shows a preference for continental margins, sea mounts, and areas of upwelling, where food is abundant (Leatherwood and Reeves 1983). Waring *et al.* (2005) suggest sperm whale distribution is closely correlated with the Gulf Stream edge. Sperm whales migrate to higher latitudes during summer months, when they are concentrated east and northeast of Cape Hatteras. Bull sperm whales migrate much farther poleward than the cows, calves, and young males. Because most of the breeding herds are confined almost exclusively to warmer waters many of the larger mature males return in the winter to the lower latitudes to breed.

For management purposes, sperm whales inhabiting U.S. waters have been divided into five stocks: California-Oregon-Washington Stock, North Pacific (Alaska), Hawaii, Northern Gulf of Mexico Stock, and North Atlantic Stock. Only whales from the North Atlantic stock are likely to occur in the action area. In the western North Atlantic the species ranges from Greenland to the Gulf of Mexico and the Caribbean. The sperm whales that occur in the eastern U.S. EEZ are believed to represent only a portion of the total stock. The best available abundance estimate for the North Atlantic stock is 4,804 with a minimum population estimate of 3,539 (Waring et al. 2005).

Sperm whale sightings recorded from the NOAA vessel Oregon II from 1991 - 1997 are concentrated just beyond the 100 m depth contour in the northern Gulf of Mexico, east of the Mississippi River Delta. Recent studies conducted jointly by researchers from NMFS and Texas A&M indicate that these offshore waters are an important area for Gulf sperm whales. This is the only known breeding and calving area in the Gulf, for what is believed to be an endemic population.

Sperm whales feed primarily on medium to large-sized mesopelagic squids *Architeuthis* and *Moroteuthis*. Sperm whales, especially mature males in higher latitude waters, also take significant quantities of large demersal and mesopelagic sharks, skates, and bony fishes (Clarke 1962, 1980). Sperm whale populations are organized into two types of groupings: breeding schools and bachelor schools. Older males are often solitary (Best 1979). Breeding schools consist of females of all ages and juvenile males. The mature females ovulate April through August in the Northern Hemisphere. During this season one or more large mature bulls temporarily join each breeding school. A single calf is born at a length of about 4 meters after a 15 month gestation period. A mature female will produce a calf every 3-6 years. Females attain sexual maturity at the mean age of nine years and a length of about nine meters. Males have a prolonged puberty and attain sexual maturity at about age 20 and a body length of 12 meters. Bachelor schools consist of maturing males who leave the breeding school and aggregate in loose groups of about 40 animals. As the males grow older they separate from the bachelor schools and remain solitary most of the year (Best 1979).

Threats to Sperm Whale Recovery
Sperm whales were hunted in America from the 17th century through the early 1900's. The International Whaling Commission (IWC) estimates that nearly a quarter-million sperm whales were killed worldwide in whaling activities between 1800 and 1900 (IWC 1971). With the advent of modern whaling the larger rorqual whales were targeted. However as their numbers decreased, greater attention was paid to smaller rorquals and sperm whales. From 1910 to 1982 there were nearly 700,000 sperm whales killed worldwide from whaling activities (Clarke 1954; Committee for Whaling Statistics 1959 -1983). In recent years the catch of sperm whales has been drastically reduced as a result of the imposition of catch quotas.

Because of their generally more offshore distribution and their benthic feeding habits, sperm whales are less subject to entanglement than right or humpback whales. However, sperm whales have been taken in the pelagic drift gillnet fishery for swordfish. Also, interactions between sperm whales and longlines for sable fish have been noted in Alaska waters. Three sperm whale entanglements in the North Atlantic have been documented from August 1993 to May 1998.

Due to their offshore distribution, sperm whales tend to strand infrequently. During 1994-2000, eighteen sperm whale strandings have been documented along the U.S. Atlantic coast between Maine and Miami, Florida (NMFS unpublished data in Waring et al. 2007). One 1998 and one 2000 stranding off Florida showed signs of human interactions. The 1998 animal’s head was severed, but it is unknown if it occurred pre- or postmortem. The 2000 animal had fishing gear in the blowhole. In October 1999, a live sperm whale calf stranded on eastern Long Island, and was subsequently euthanized. Also, a dead calf was found in the surf off Florida in 2000.

During 2001 to 2005, fifteen sperm whale strandings were documented along the U.S. Atlantic coast and in Puerto Rico and the EEZ according the NER and SER strandings databases (see Waring et al. 2007). Except for a sperm whale struck by a naval vessel in the EEZ in 2001, there were no confirmed documented signs of human interactions on the other animals.

Ship strikes are another source of human- induced mortality. In May 1994 a ship-struck sperm whale was observed south of Nova Scotia (Reeves and Whitehead 1997) and in May 2000 a merchant ship reported a strike in Block Canyon (NMFS, unpublished data, in Waring et al. 2007). Waring et al. (2007), reports that based on stranding and entanglement data, during 2001-2005, one sperm whale was confirmed struck by a ship, thus, there is an annual average of 0.2 sperm whales per year struck by ships. No sperm whale stranding mortalities during this period were confirmed fishery interactions.

Total numbers of sperm whales off the U.S. or Canadian Atlantic coast are unknown. The best recent abundance estimate for sperm whales is the sum of the estimates from the two 2004 U.S. Atlantic surveys, 4,804 (CV =0.38), where the estimate from the northern U.S. Atlantic is 2,607 (CV =0.57), and from the southern U.S. Atlantic is 2,197 (CV =0.47) (Waring et al. 2007). This joint estimate is considered best because together these two surveys have the most complete coverage of the species' habitat. However, because these estimates were not corrected for dive-time, they are likely downwardly biased and an underestimate of actual abundance. The collective 1990- 2004 data suggest that, seasonally, at least several thousand sperm whales are occupying suspected high-use habitats off the northeastern U.S. coast (Waring et al. 2007). As noted in the most recent stock assessment report (Waring et al. 2007), there are insufficient data to determine the population trends for this species.

4.2.6 Status of Sea Turtles

With the exception of loggerheads, sea turtles are listed under the ESA at the species level rather than as subspecies or distinct population segments (DPS). Therefore, information on the range-wide status of leatherback, Kemp's ridley and green sea turtles is included to provide the status of each species overall. Information on the status of loggerheads will only be presented for the DPS affected by this action. Additional background information on the range-wide status of these species can be found in a number of published documents, including sea turtle status reviews and biological reports (NMFS and USFWS 1995; Hirth 1997; Marine Turtle Expert Working Group [TEWG] 1998, 2000, 2007, 2009; NMFS and USFWS 2007a, 2007b, 2007c, 2007d; Conant *et al.* 2009), and recovery plans for the loggerhead sea turtle (NMFS and USFWS 2008), Kemp's ridley sea turtle (NMFS *et al.* 2011), leatherback sea turtle (NMFS and USFWS 1992, 1998), Kemp's ridley sea turtle (NMFS *et al.* 2011)and green sea turtle (NMFS and USFWS 1991, 1998b).

2010 BP Deepwater Horizon Oil Spill
The April 20, 2010, explosion of the Deepwater Horizon oil rig affected sea turtles in the Gulf of Mexico. There is an on-going assessment of the long-term effects of the spill on Gulf of Mexico marine life, including sea turtle populations. Following the spill, juvenile Kemp's ridley, green, and loggerhead sea turtles were found in *Sargassum* algae mats in the convergence zones, where currents meet and oil collected. Sea turtles found in these areas were often coated in oil and/or had ingested oil. Approximately 536 live adult and juvenile sea turtles were recovered from the Gulf and brought into rehabilitation centers; of these, 456 were visibly oiled (these and the following numbers were obtained from http://www.nmfs.noaa.gov/pr/health/oilspill/). To date, 469 of the live recovered sea turtles have been successfully returned to the wild, 25 died during rehabilitation, and 42 are still in care but will hopefully be returned to the wild eventually.

During the clean-up period, 613 dead sea turtles were recovered in coastal waters or on beaches in Mississippi, Alabama, Louisiana, and the Florida Panhandle. As of February 2011, 478 of these dead turtles had been examined. Many of the examined sea turtles showed indications that they had died as a result of interactions with trawl gear, most likely used in the shrimp fishery, and not as a result of exposure to or ingestion of oil.

During the spring and summer of 2010, nearly 300 sea turtle nests were relocated from the northern Gulf to the east coast of Florida with the goal of preventing hatchlings from entering the oiled waters of the northern Gulf. From these relocated nests, 14,676 sea turtles, including 14,235 loggerheads, 125 Kemp's ridleys, and 316 greens, were ultimately released from Florida beaches.

A thorough assessment of the long-term effects of the spill on sea turtles has not yet been completed. However, the spill resulted in the direct mortality of many sea turtles and may have had sublethal effects or caused environmental damage that will impact other sea turtles into the future. The population level effects of the spill and associated response activity are likely to remain unknown for some period into the future.

4.2.7 Northwest Atlantic DPS of loggerhead sea turtle

The loggerhead is the most abundant species of sea turtle in U.S. waters. Loggerhead sea turtles are found in temperate and subtropical waters and occupy a range of habitats including offshore waters, continental shelves, bays, estuaries, and lagoons. They are also exposed to a variety of natural and anthropogenic threats in the terrestrial and marine environment.

Listing History
Loggerhead sea turtles were listed as threatened throughout their global range on July 28, 1978. Since that time, several status reviews have been conducted to review the status of the species and make recommendations regarding its ESA listing status. Based on a 2007 5-year status review of the species, which discussed a variety of threats to loggerheads including climate change, NMFS and FWS determined that loggerhead sea turtles should not be delisted or reclassified as endangered. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPSs should be identified for the loggerhead (NMFS and USFWS 2007a). Genetic differences exist between loggerhead sea turtles that nest and forage in the different ocean basins (Bowen 2003; Bowen and Karl 2007). Differences in the maternally inherited mitochondrial DNA also exist between loggerhead nesting groups that occur within the same ocean basin (TEWG 2000; Pearce 2001; Bowen 2003; Bowen *et al*. 2005; Shamblin 2007; TEWG 2009; NMFS and USFWS 2008). Site fidelity of females to one or more nesting beaches in an area is believed to account for these genetic differences (TEWG 2000; Bowen 2003).

In part to evaluate those genetic differences, in 2008, NMFS and FWS established a Loggerhead Biological Review Team (BRT) to assess the global loggerhead population structure to determine whether DPSs exist and, if so, the status of each DPS. The BRT evaluated genetic data, tagging and telemetry data, demographic information, oceanographic features, and geographic barriers to determine whether population segments exist. The BRT report was completed in August 2009 (Conant *et al.* 2009). In this report, the BRT identified the following nine DPSs as being discrete from other conspecific population segments and significant to the species: (1) North Pacific Ocean, (2) South Pacific Ocean, (3) North Indian Ocean, (4) Southeast Indo-Pacific Ocean, (5) Southwest Indian Ocean, (6) Northwest Atlantic Ocean, (7) Northeast Atlantic Ocean, (8) Mediterranean Sea, and (9) South Atlantic Ocean.

The BRT concluded that although some DPSs are indicating increasing trends at nesting beaches (Southwest Indian Ocean and South Atlantic Ocean), available information about anthropogenic threats to juveniles and adults in neritic and oceanic environments indicate possible unsustainable additional mortalities. According to an analysis using expert opinion in a matrix model framework, the BRT report stated that all loggerhead DPSs have the potential to decline in the foreseeable future. Based on the threat matrix analysis, the potential for future decline was reported as greatest for the North Indian Ocean, Northwest Atlantic Ocean, Northeast Atlantic Ocean, Mediterranean Sea, and South Atlantic Ocean DPSs (Conant *et al.* 2009). The BRT concluded that the North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Southeast Indo-Pacific Ocean, Northwest Atlantic Ocean, Northeast Atlantic Ocean, and Mediterranean Sea DPSs were at risk of extinction. The BRT concluded that although the Southwest Indian Ocean and South Atlantic Ocean DPSs were likely not currently at immediate risk of extinction, the extinction risk was likely to increase in the foreseeable future.

On March 16, 2010, NMFS and USFWS published a proposed rule (75 FR 12598) to divide the worldwide population of loggerhead sea turtles into nine DPSs, as described in the 2009 Status Review. Two of the DPSs were proposed to be listed as threatened and seven of the DPSs, including the Northwest Atlantic Ocean DPS, were proposed to be listed as endangered. NMFS and the USFWS accepted comments on the proposed rule through September 13, 2010 (75 FR 30769, June 2, 2010). On March 22, 2011 (76 FR 15932), NMFS and USFWS extended the date by which a final determination on the listing action will be made to no later than September 16, 2011. This action was taken to address the interpretation of the existing data on status and trends and its relevance to the assessment of risk of extinction for the Northwest Atlantic Ocean DPS, as well as the magnitude and immediacy of the fisheries bycatch threat and measures to reduce this threat. New information or analyses to help clarify these issues were requested by April 11, 2011.

On September 22, 2011, NMFS and USFWS issued a final rule (76 FR 58868), determining that the loggerhead sea turtle is composed of nine DPSs (as defined in Conant *et al.*, 2009) that constitute species that may be listed as threatened or endangered under the ESA. Five DPSs were listed as endangered (North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Northeast Atlantic Ocean, and Mediterranean Sea), and four DPSs were listed as threatened (Northwest Atlantic Ocean, South Atlantic Ocean, Southeast Indo-Pacific Ocean, and Southwest Indian Ocean). Note that the Northwest Atlantic Ocean (NWA) DPS and the Southeast Indo-Pacific Ocean DPS were originally proposed as endangered. The NWA DPS was determined to be threatened based on review of nesting data available after the proposed rule was published, information provided in public comments on the proposed rule, and further discussions within the agencies. The two primary factors considered were population abundance and population trend. NMFS and USFWS found that an endangered status for the NWA DPS was not warranted given the large size of the nesting population, the overall nesting population remains widespread, the trend for the nesting population appears to be stabilizing, and substantial conservation efforts are underway to address threats. This final listing rule became effective on October 24, 2011.

The September 2011 final rule also noted that critical habitat for the two DPSs occurring within the U.S. (NWA DPS and North Pacific DPS) will be designated in a future rulemaking. Information from the public related to the identification of critical habitat, essential physical or biological features for this species, and other relevant impacts of a critical habitat designation was solicited. Currently, no critical habitat is designated for any DPS of loggerhead sea turtles, and therefore, no critical habitat for any DPS occurs in the action area.

Presence of Loggerhead Sea Turtles in the Action Area
The effects of this proposed action are only experienced within the Atlantic Ocean. NMFS has considered the available information on the distribution of the 9 DPSs to determine the origin of any loggerhead sea turtles that may occur in the action area. As noted in Conant *et al.* (2009), the range of the four DPSs occurring in the Atlantic Ocean are as follows: NWA DPS – north of the equator, south of 60° N latitude, and west of 40° W longitude; Northeast Atlantic Ocean (NEA) DPS – north of the equator, south of 60° N latitude, east of 40° W longitude, and west of 5° 36' W longitude; South Atlantic DPS – south of the equator, north of 60° S latitude, west of 20° E longitude, and east of 60° W longitude; Mediterranean DPS – the Mediterranean Sea east of 5° 36' W longitude. These boundaries were determined based on oceanographic features, loggerhead sightings, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. While adults are highly structured with no overlap, there may be some degree of overlap by juveniles of the NWA, NEA, and Mediterranean DPSs on oceanic foraging grounds (Laurent *et al.* 1993, 1998; Bolten *et al.* 1998; LaCasella *et al.* 2005; Carreras *et al.* 2006, Monzón-Argüello *et al.* 2006; Revelles *et al.* 2007). Previous literature (Bowen *et al.* 2004) has suggested that there is the potential, albeit small, for some juveniles from the Mediterranean DPS to be present in U.S. Atlantic coastal foraging grounds. These conclusions must be interpreted with caution however, as they may be representing a shared common haplotype and lack of representative sampling at Eastern Atlantic rookeries rather than an actual presence of Mediterranean DPS turtles in US Atlantic coastal waters. A re-analysis of the data by the Atlantic loggerhead Turtle Expert Working Group has found that that it is unlikely that U.S. fishing fleets are interacting with either the Northeast Atlantic loggerhead DPS or the Mediterranean loggerhead DPS (Peter Dutton, NMFS, Marine Turtle Genetics Program, Program Leader, personal communication, September 10, 2011). Given that the action area is a subset of the area fished by US fleets, it is reasonable to assume that based on this new analysis, no individuals from the Mediterranean DPS or Northeast Atlantic DPS would be present in the action area. Sea turtles of the South Atlantic DPS do not inhabit the action area of this consultation (Conant *et al.* 2009). As such, the remainder of this consultation will only focus on the NWA DPS, listed as threatened.

Distribution and Life History
Ehrhart *et al.* (2003) provided a summary of the literature identifying known nesting habitats and foraging areas for loggerheads within the Atlantic Ocean. Detailed information is also provided in the 5-year status review for loggerheads (NMFS and USFWS 2007a), the TEWG report

(2009), and the final revised recovery plan for loggerheads in the Northwest Atlantic Ocean (NMFS and USFWS 2008), which is a second revision to the original recovery plan that was approved in 1984 and subsequently revised in 1991.

In the western Atlantic, waters as far north as 41° N to 42° N latitude are used for foraging by juveniles, as well as adults (Shoop 1987; Shoop and Kenney 1992; Ehrhart *et al.* 2003; Mitchell *et al.* 2003). In U.S. Atlantic waters, loggerheads commonly occur throughout the inner continental shelf from Florida to Cape Cod, Massachusetts and in the Gulf of Mexico from Florida to Texas, although their presence varies with the seasons due to changes in water temperature (Shoop and Kenney 1992; Epperly *et al.* 1995a, 1995b; Braun and Epperly 1996; Braun-McNeill *et al.* 2008; Mitchell *et al.* 2003). Loggerheads have been observed in waters with surface temperatures of 7°C to 30°C, but water temperatures ≥11°C are most favorable (Shoop and Kenney 1992; Epperly *et al.* 1995b). The presence of loggerhead sea turtles in U.S. Atlantic waters is also influenced by water depth. Aerial surveys of continental shelf waters north of Cape Hatteras, North Carolina indicated that loggerhead sea turtles were most commonly sighted in waters with bottom depths ranging from 22 m to 49 m deep (Shoop and Kenney 1992). However, more recent survey and satellite tracking data support that they occur in waters from the beach to beyond the continental shelf (Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; Mansfield 2006; Blumenthal *et al.* 2006; Hawkes *et al.* 2006; McClellan and Read 2007; Mansfield *et al.* 2009).

Loggerhead sea turtles occur year round in ocean waters off North Carolina, South Carolina, Georgia, and Florida. In these areas of the South Atlantic Bight, water temperature is influenced by the proximity of the Gulf Stream. As coastal water temperatures warm in the spring, loggerheads begin to migrate to inshore waters of the Southeast United States (*e.g.*, Pamlico and Core Sounds) and also move up the U.S. Atlantic coast (Epperly *et al.* 1995a, 1995b, 1995c; Braun-McNeill and Epperly 2004), occurring in Virginia foraging areas as early as April/May and on the most northern foraging grounds in the Gulf of Maine in June (Shoop and Kenney 1992). The trend is reversed in the fall as water temperatures cool. The large majority leave the Gulf of Maine by mid-September but some turtles may remain in Mid-Atlantic and Northeast areas until late fall. By December, loggerheads have migrated from inshore and more northern coastal waters to waters offshore of North Carolina, particularly off of Cape Hatteras, and waters further south where the influence of the Gulf Stream provides temperatures favorable to sea turtles (Shoop and Kenney 1992; Epperly *et al.* 1995b).

Recent studies have established that the loggerhead's life history is more complex than previously believed. Rather than making discrete developmental shifts from oceanic to neritic environments, research is showing that both adults and (presumed) neritic stage juveniles continue to use the oceanic environment and will move back and forth between the two habitats (Witzell 2002; Blumenthal *et al.* 2006; Hawkes *et al.* 2006; McClellan and Read 2007; Mansfield *et al.* 2009). One of the studies tracked the movements of adult post-nesting females and found that differences in habitat use were related to body size with larger adults staying in coastal waters and smaller adults traveling to oceanic waters (Hawkes *et al.* 2006). A tracking study of large juveniles found that the habitat preferences of this life stage were also diverse with some remaining in neritic waters and others moving off into oceanic waters (McClellan and Read

2007). However, unlike the Hawkes *et al.* (2006) study, there was no significant difference in the body size of turtles that remained in neritic waters versus oceanic waters (McClellan and Read 2007).

Pelagic and benthic juveniles are omnivorous and forage on crabs, mollusks, jellyfish, and vegetation at or near the surface (Dodd 1988; NMFS and USFWS 2008). Sub-adult and adult loggerheads are primarily coastal dwelling and typically prey on benthic invertebrates such as mollusks and decapod crustaceans in hard bottom habitats (NMFS and USFWS 2008).

As presented below, Table 3 from the 2008 loggerhead recovery plan (Table 1 in this Opinion) highlights the key life history parameters for loggerheads nesting in the United States.

Table 3. Typical values of life history parameters for loggerheads nesting in the U.S.

Life History Parameter	Data
Clutch size	100-126 eggs[1]
Egg incubation duration (varies depending on time of year and latitude)	42-75 days[2,3]
Pivotal temperature (incubation temperature that produces an equal number of males and females)	29.0°C[5]
Nest productivity (emerged hatchlings/total eggs) x 100 (varies depending on site specific factors)	45-70%[2,6]
Clutch frequency (number of nests/female/season)	3-5.5 nests[7]
Internesting interval (number of days between successive nests within a season)	12-15 days[8]
Juvenile (<87 cm CCL) sex ratio	65-70% female[4]
Remigration interval (number of years between successive nesting migrations)	2.5-3.7 years[9]
Nesting season	late April-early September
Hatching season	late June-early November
Age at sexual maturity	32-35 years[10]
Life span	>57 years[11]

1 Dodd 1988.
2 Dodd and Mackinnon (1999, 2000, 2001, 2002, 2003, 2004).
3 Blair Witherington, FFWCC, personal communication, 2006 (information based on nests monitored throughout Florida beaches in 2005, n=865).
4 National Marine Fisheries Service (2001); Allen Foley, FFWCC, personal communication, 2005.
5 Mrosovsky (1988).
6 Blair Witherington, FFWCC, personal communication, 2006 (information based on nests monitored throughout Florida beaches in 2005, n=1,680).
7 Murphy and Hopkins (1984); Frazer and Richardson (1985); Ehrhart, unpublished data; Hawkes *et al*. 2005; Scott 2006; Tony Tucker, Mote Marine Laboratory, personal communication, 2008.
8 Caldwell (1962), Dodd (1988).
9 Richardson *et al*. (1978); Bjorndal *et al*. (1983); Ehrhart, unpublished data.
10 Melissa Snover, NMFS, personal communication, 2005; see Table A1-6.
11 Dahlen *et al*. (2000).

Population Dynamics and Status

By far, the majority of Atlantic nesting occurs on beaches of the southeastern United States (NMFS and USFWS 2007a). For the past decade or so, the scientific literature has recognized five distinct nesting groups, or subpopulations, of loggerhead sea turtles in the Northwest Atlantic, divided geographically as follows: (1) a northern group of nesting females that nest from North Carolina to northeast Florida at about 29° N latitude; (2) a south Florida group of nesting females that nest from 29° N latitude on the east coast to Sarasota on the west coast; (3) a Florida Panhandle group of nesting females that nest around Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán group of nesting females that nest on beaches of the eastern Yucatán Peninsula, Mexico; and (5) a Dry Tortugas group that nests on beaches of the islands of the Dry Tortugas, near Key West, Florida and on Cal Sal Bank (TEWG 2009). Genetic analyses of mitochondrial DNA, which a sea turtle inherits from its mother, indicate that there are genetic differences between loggerheads that nest at, and originate from, the beaches used by each of the five identified nesting groups of females (TEWG 2009). However, analyses of microsatellite loci from nuclear DNA, which represents the genetic contribution from both parents, indicates little to no genetic differences between loggerheads originating from nesting beaches of the five Northwest Atlantic nesting groups (Pearce and Bowen 2001; Bowen 2003; Bowen *et al.* 2005; Shamblin 2007). These results suggest that female loggerheads have site fidelity to nesting beaches within a particular area, while males provide an avenue of gene flow between nesting groups by mating with females that originate from different nesting groups (Bowen 2003; Bowen *et al.* 2005). The extent of such gene flow, however, is unclear (Shamblin 2007).

The lack of genetic structure makes it difficult to designate specific boundaries for the nesting subpopulations based on genetic differences alone. Therefore, the Loggerhead Recovery Team recently used a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries, in addition to genetic differences, to reassess the designation of these subpopulations to identify recovery units in the 2008 recovery plan.

In the 2008 recovery plan, the Loggerhead Recovery Team designated five recovery units for the Northwest Atlantic population of loggerhead sea turtles based on the aforementioned nesting groups and inclusive of a few other nesting areas not mentioned above. The first four of these recovery units represent nesting assemblages located in the Southeast United States. The fifth recovery unit is composed of all other nesting assemblages of loggerheads within the Greater Caribbean, outside the United States, but which occur within U.S. waters during some portion of their lives. The five recovery units representing nesting assemblages are: (1) the Northern Recovery Unit (NRU: Florida/Georgia border through southern Virginia), (2) the Peninsular Florida Recovery Unit (PFRU: Florida/Georgia border through Pinellas County, Florida), (3) the Dry Tortugas Recovery Unit (DTRU: islands located west of Key West, Florida), (4) the Northern Gulf of Mexico Recovery Unit (NGMRU: Franklin County, Florida through Texas), and (5) the Greater Caribbean Recovery Unit (GCRU: Mexico through French Guiana, Bahamas, Lesser Antilles, and Greater Antilles).

The Recovery Team evaluated the status and trends of the Northwest Atlantic loggerhead population for each of the five recovery units, using nesting data available as of October 2008 (NMFS and USFWS 2008). The level and consistency of nesting coverage varies among recovery units, with coverage in Florida generally being the most consistent and thorough over time. Since 1989, nest count surveys in Florida have occurred in the form of statewide surveys (a near complete census of entire Florida nesting) and index beach surveys (Witherington *et al.* 2009). Index beaches were established to standardize data collection methods and maintain a constant level of effort on key nesting beaches over time.

Note that NMFS and USFWS (2008), Witherington *et al.* (2009), and TEWG (2009) analyzed the status of the nesting assemblages within the NWA DPS using standardized data collected over periods ranging from 10-23 years. These analyses used different analytical approaches, but found the same finding that there had been a significant, overall nesting decline within the NWA DPS. However, with the addition of nesting data from 2008-2010, the trend line changes showing a very slight negative trend, but the rate of decline is not statistically different from zero (76 FR 58868, September 22, 2011). The nesting data presented in the Recovery Plan (through 2008) is described below, with updated trend information through 2010 for two recovery units.

From the beginning of standardized index surveys in 1989 until 1998, the PFRU, the largest nesting assemblage in the Northwest Atlantic by an order of magnitude, had a significant increase in the number of nests. However, from 1998 through 2008, there was a 41% decrease in annual nest counts from index beaches, which represent an average of 70% of the statewide nesting activity (NMFS and USFWS 2008). From 1989-2008, the PFRU had an overall declining nesting trend of 26% (95% CI: -42% to -5%; NMFS and USFWS 2008). With the addition of nesting data through 2010, the nesting trend for the PFRU does not show a nesting decline statistically different from zero (76 FR 58868, September 22, 2011). The NRU, the second largest nesting assemblage of loggerheads in the United States, has been declining at a rate of 1.3% annually since 1983 (NMFS and USFWS 2008). The NRU dataset included 11 beaches with an uninterrupted time series of coverage of at least 20 years; these beaches represent approximately 27% of NRU nesting (in 2008). Through 2008, there was strong statistical data to suggest the NRU has experienced a long-term decline, but with the inclusion of nesting data through 2010, nesting for the NRU is showing possible signs of stabilizing (76 FR 58868, September 22, 2011). Evaluation of long-term nesting trends for the NGMRU is difficult because of changed and expanded beach coverage. However, the NGMRU has shown a significant declining trend of 4.7% annually since index nesting beach surveys were initiated in 1997 (NMFS and USFWS 2008). No statistical trends in nesting abundance can be determined for the DTRU because of the lack of long-term data. Similarly, statistically valid analyses of long-term nesting trends for the entire GCRU are not available because there are few long-term standardized nesting surveys representative of the region. Additionally, changing survey effort at monitored beaches and scattered and low-level nesting by loggerheads at many locations currently precludes comprehensive analyses (NMFS and USFWS 2008).

Sea turtle census nesting surveys are important in that they provide information on the relative abundance of nesting each year, and the contribution of each nesting group to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually. The 2008 recovery plan compiled information on mean number of loggerhead nests and the approximated counts of nesting females per year for four of the five identified recovery units (*i.e*., nesting groups). They are: (1) for the NRU, a mean of 5,215 loggerhead nests per year (from 1989-2008) with approximately 1,272 females nesting per year; (2) for the PFRU, a mean of 64,513 nests per year (from 1989-2007) with approximately 15,735 females nesting per year; (3) for the DTRU, a mean of 246 nests per year (from 1995-2004, excluding 2002) with approximately 60 females nesting per year; and (4) for the NGMRU, a mean of 906 nests per year (from 1995-2007) with approximately 221 females nesting per year. For the

GCRU, the only estimate available for the number of loggerhead nests per year is from Quintana Roo, Yucatán, Mexico, where a range of 903-2,331 nests per year was estimated from 1987-2001 (NMFS and USFWS 2007a). There are no annual nest estimates available for the Yucatán since 2001 or for any other regions in the GCRU, nor are there any estimates of the number of nesting females per year for any nesting assemblage in this recovery unit. Note that the above values for average nesting females per year were based upon 4.1 nests per female per Murphy and Hopkins (1984).

Genetic studies of juvenile and a few adult loggerhead sea turtles collected from Northwest Atlantic foraging areas (beach strandings, a power plant in Florida, and North Carolina fisheries) show that the loggerheads that occupy East Coast U.S. waters originate from these Northwest Atlantic nesting groups; primarily from the nearby nesting beaches of southern Florida, as well as the northern Florida to North Carolina beaches, and finally from the beaches of the Yucatán Peninsula, Mexico (Rankin-Baransky *et al.* 2001; Witzell *et al.* 2002; Bass *et al.* 2004; Bowen *et al.* 2004). The contribution of these three nesting assemblages varies somewhat among the foraging habitats and age classes surveyed along the east coast. The distribution is not random and bears a significant relationship to the proximity and size of adjacent nesting colonies (Bowen *et al.* 2004). Bass *et al.* (2004) attribute the variety in the proportions of sea turtles from loggerhead turtle nesting assemblages documented in different east coast foraging habitats to a complex interplay of currents and the relative size and proximity of nesting beaches.

Unlike nesting surveys, in-water studies of sea turtles typically sample both sexes and multiple age classes. In-water studies have been conducted in some areas of the Northwest Atlantic and provide data by which to assess the relative abundance of loggerhead sea turtles and changes in abundance over time (Maier *et al.* 2004; Morreale *et al.* 2005; Mansfield 2006; Ehrhart *et al.* 2007; Epperly *et al.* 2007). The TEWG (2009) used raw data from six in-water study sites to conduct trend analyses. They identified an increasing trend in the abundance of loggerheads from three of the four sites located in the Southeast United States, one site showed no discernible trend, and the two sites located in the northeast United States showed a decreasing trend in abundance of loggerheads. The 2008 loggerhead recovery plan also includes a full discussion of in-water population studies for which trend data have been reported, and a brief summary will be provided here.

Maier *et al.* (2004) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the southeast coast of the United States (Winyah Bay, South Carolina to St. Augustine, Florida) during the period 2000-2003. A comparison of loggerhead catch data from this study with historical values suggested that in-water populations of loggerhead sea turtles along the southeast U.S. coast appear to be larger, possibly an order of magnitude higher than they were 25 years ago, but the authors caution a direct comparison between the two studies given differences in sampling methodology (Maier *et al.* 2004). A comparison of catch rates for sea turtles in pound net gear fished in the Pamlico-Albemarle Estuarine Complex of North Carolina between the years 1995-1997 and 2001-2003 found a significant increase in catch rates for loggerhead sea turtles for the latter period (Epperly *et al.* 2007). A long-term, on-going study of loggerhead abundance in the Indian River Lagoon System of Florida found a significant increase in the relative abundance of loggerheads over the last 4 years of the study (Ehrhart *et al.*

2007). However, there was no discernible trend in loggerhead abundance during the 24-year time period of the study (1982-2006) (Ehrhart *et al.* 2007). At St. Lucie Power Plant, data collected from 1977-2004 show an increasing trend of loggerheads at the power plant intake structures (FPL and Quantum Resources 2005).

In contrast to these studies, Morreale *et al.* (2005) observed a decline in the percentage and relative numbers of loggerhead sea turtles incidentally captured in pound net gear fished around Long Island, New York during the period 2002-2004 in comparison to the period 1987-1992, with only two loggerheads (of a total 54 turtles) observed captured in pound net gear during the period 2002-2004. This is in contrast to the previous decade's study where numbers of individual loggerheads ranged from 11 to 28 per year (Morreale *et al.* 2005). No additional loggerheads were reported captured in pound net gear in New York through 2007, although two were found cold-stunned on Long Island bay beaches in the fall of 2007 (Memo to the File, L. Lankshear, December 2007). Potential explanations for this decline include major shifts in loggerhead foraging areas and/or increased mortality in pelagic or early benthic stage/age classes (Morreale *et al.* 2005). Using aerial surveys, Mansfield (2006) also found a decline in the densities of loggerhead sea turtles in Chesapeake Bay over the period 2001-2004 compared to aerial survey data collected in the 1980s. Significantly fewer loggerheads ($p<0.05$) were observed in both the spring (May-June) and the summer (July-August) of 2001-2004 compared to those observed during aerial surveys in the 1980s (Mansfield 2006). A comparison of median densities from the 1980s to the 2000s suggested that there had been a 63.2% reduction in densities during the spring residency period and a 74.9% reduction in densities during the summer residency period (Mansfield 2006). The decline in observed loggerhead populations in Chesapeake Bay may be related to a significant decline in prey, namely horseshoe crabs and blue crabs, with loggerheads redistributing outside of Bay waters (NMFS and USFWS 2008).

As with other turtle species, population estimates for loggerhead sea turtles are difficult to determine, largely given their life history characteristics. However, a recent loggerhead assessment using a demographic matrix model estimated that the loggerhead adult female population in the western North Atlantic ranges from 16,847 to 89,649, with a median size of 30,050 (NMFS SEFSC 2009). The model results for population trajectory suggest that the population is most likely declining, but this result was very sensitive to the choice of the position of the parameters within their range and hypothesized distributions. The pelagic stage survival parameter had the largest effect on the model results. As a result of the large uncertainty in our knowledge of loggerhead life history, at this point predicting the future populations or population trajectories of loggerhead sea turtles with precision is very uncertain. It should also be noted that additional analyses are underway which will incorporate any newly available information.

As part of the Atlantic Marine Assessment Program for Protected Species (AMAPPS), line transect aerial abundance surveys and turtle telemetry studies were conducted along the Atlantic coast in the summer of 2010. AMAPPS is a multi-agency initiative to assess marine mammal, sea turtle, and seabird abundance and distribution in the Atlantic. Aerial surveys were conducted from Cape Canaveral, Florida to the Gulf of St. Lawrence, Canada. Satellite tags on juvenile loggerheads were deployed in two locations – off the coasts of northern Florida to South Carolina (n=30) and off the New Jersey and Delaware coasts (n=14). As presented in NMFS

NEFSC (2011), the 2010 survey found a preliminary total surface abundance estimate within the entire study area of about 60,000 loggerheads (CV=0.13) or 85,000 if a portion of unidentified hard-shelled sea turtles were included (CV=0.10). Surfacing times were generated from the satellite tag data collected during the aerial survey period, resulting in a 7% (5%-11% inter-quartile range) median surface time in the South Atlantic area and a 67% (57%-77% inter-quartile range) median surface time to the north. The calculated preliminary regional abundance estimate is about 588,000 loggerheads along the U.S. Atlantic coast, with an inter-quartile range of 382,000-817,000 (NMFS NEFSC 2011). The estimate increases to approximately 801,000 (inter-quartile range of 521,000-1,111,000) when based on known loggerheads and a portion of unidentified turtle sightings. The density of loggerheads was generally lower in the north than the south; based on number of turtle groups detected, 64% were seen south of Cape Hatteras, North Carolina, 30% in the southern Mid-Atlantic Bight, and 6% in the northern Mid-Atlantic Bight. Although they have been seen farther north in previous studies (*e.g.*, Shoop and Kenney 1992), no loggerheads were observed during the aerial surveys conducted in the summer of 2010 in the more northern zone encompassing Georges Bank, Cape Cod Bay, and the Gulf of Maine. These estimates of loggerhead abundance over the U.S. Atlantic continental shelf are considered very preliminary. A more thorough analysis will be completed pending the results of further studies related to improving estimates of regional and seasonal variation in loggerhead surface time (by increasing the sample size and geographical area of tagging) and other information needed to improve the biases inherent in aerial surveys of sea turtles (*e.g.*, research on depth of detection and species misidentification rate). This survey effort represents the most comprehensive assessment of sea turtle abundance and distribution in many years. Additional aerial surveys and research to improve the abundance estimates are anticipated in 2011-2014, depending on available funds.

Threats

The diversity of a sea turtle's life history leaves them susceptible to many natural and human impacts, including impacts while they are on land, in the neritic environment, and in the oceanic environment. The 5-year status review and 2008 recovery plan provide a summary of natural as well as anthropogenic threats to loggerhead sea turtles (NMFS and USFWS 2007a, 2008). Amongst those of natural origin, hurricanes are known to be destructive to sea turtle nests. Sand accretion, rainfall, and wave action that result from these storms can appreciably reduce hatchling success. Other sources of natural mortality include cold-stunning, biotoxin exposure, and native species predation.

Anthropogenic factors that impact hatchlings and adult females on land, or the success of nesting and hatching include: beach erosion, beach armoring, and nourishment; artificial lighting; beach cleaning; beach pollution; increased human presence; recreational beach equipment; vehicular and pedestrian traffic; coastal development/construction; exotic dune and beach vegetation; removal of native vegetation; and poaching. An increased human presence at some nesting beaches or close to nesting beaches has led to secondary threats such as the introduction of exotic fire ants, feral hogs, dogs, and an increased presence of native species (*e.g.*, raccoons, armadillos, and opossums), which raid nests and feed on turtle eggs (NMFS and USFWS 2007a, 2008). Although sea turtle nesting beaches are protected along large expanses of the Northwest Atlantic coast (in areas like Merritt Island, Archie Carr, and Hobe Sound National Wildlife Refuges), other areas along these coasts have limited or no protection. Sea turtle nesting and hatching success on unprotected high density East Florida nesting beaches from Indian River to Broward County are affected by all of the above threats.

Loggerheads are affected by a completely different set of anthropogenic threats in the marine environment. These include oil and gas exploration, coastal development, and transportation; marine pollution; underwater explosions; hopper dredging; offshore artificial lighting; power plant entrainment and/or impingement; entanglement in debris; ingestion of marine debris; marina and dock construction and operation; boat collisions; poaching; and fishery interactions.

A 1990 National Research Council (NRC) report concluded that for juveniles, subadults, and breeding adults in coastal waters, the most important source of human caused mortality in U.S. Atlantic waters was fishery interactions. The sizes and reproductive values of sea turtles taken by fisheries vary significantly, depending on the location and season of the fishery, and size-selectivity resulting from gear characteristics. Therefore, it is possible for fisheries that interact with fewer, more reproductively valuable turtles to have a greater detrimental effect on the population than one that takes greater numbers of less reproductively valuable turtles (Wallace *et al.* 2008). The Loggerhead Biological Review Team determined that the greatest threats to the NWA DPS of loggerheads result from cumulative fishery bycatch in neritic and oceanic habitats (Conant *et al.* 2009). Attaining a more thorough understanding of the characteristics, as well as the quantity of sea turtle bycatch across all fisheries is of great importance.

Finkbeiner *et al.* (2011) compiled cumulative sea turtle bycatch information in U.S. fisheries from 1990 through 2007, before and after implementation of bycatch mitigation measures. Information was obtained from peer reviewed publications and NMFS documents (e.g., Biological Opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, occurred annually (since implementation of bycatch mitigation measures). Kemp's ridleys interacted with fisheries most frequently, with the highest level of mean annual mortality (2,700), followed by loggerheads (1,400), greens (300), and leatherbacks (40). The Southeast/Gulf of Mexico shrimp trawl fishery was responsible for the vast majority of U.S. interactions (up to 98%) and mortalities (more than 80%). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations.

Of the many fisheries known to adversely affect loggerheads, the U.S. South Atlantic and Gulf of Mexico shrimp fisheries were considered to pose the greatest threat of mortality to neritic juvenile and adult age classes of loggerheads (NRC 1990, Finkbeiner *et al.* 2011). Significant changes to the South Atlantic and Gulf of Mexico shrimp fisheries have occurred since 1990, and the effects of these shrimp fisheries on ESA-listed species, including loggerhead sea turtles, have been assessed several times through section 7 consultation. There is also a lengthy regulatory history with regard to the use of Turtle Excluder Devices (TEDs) in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries (Epperly and Teas 2002; NMFS 2002a; Lewison *et al.* 2003). The current section 7 consultation on the U.S. South Atlantic and Gulf of Mexico shrimp fisheries was completed in 2002 and estimated the total annual level of take for loggerhead sea turtles to be 163,160 interactions (the total number of turtles that enter a shrimp trawl, which may then escape through the TED or fail to escape and be captured) with 3,948 of those takes being lethal (NMFS 2002a).

In addition to improvements in TED designs and TED enforcement, interactions between loggerheads and the shrimp fishery have also been declining because of reductions in fishing effort unrelated to fisheries management actions. The 2002 Opinion take estimates are based in part on fishery effort levels. In recent years, low shrimp prices, rising fuel costs, competition with imported products, and the impacts of recent hurricanes in the Gulf of Mexico have all impacted the shrimp fleets; in some cases reducing fishing effort by as much as 50% for offshore waters of the Gulf of Mexico (GMFMC 2007). As a result, loggerhead interactions and mortalities in the Gulf of Mexico have been substantially less than projected in the 2002 Opinion. Currently, the estimated annual number of interactions between loggerheads and shrimp trawls in the Gulf of Mexico shrimp fishery is 23,336, with 647 (2.8%) of those interactions resulting in mortality (Memo from Dr. B. Ponwith, Southeast Fisheries Science Center to Dr. R. Crabtree, Southeast Region, PRD, December 2008). Section 7 consultation on the Shrimp FMP has recently been reinitiated and a new Biological Opinion is forthcoming.

Loggerhead sea turtles are also known to interact with non-shrimp trawl, gillnet, longline, dredge, pound net, pot/trap, and hook and line fisheries. The NRC (1990) report stated that other U.S. Atlantic fisheries collectively accounted for 500 to 5,000 loggerhead deaths each year, but recognized that there was considerable uncertainty in the estimate. The reduction of sea turtle captures in fishing operations is identified in recovery plans and 5-year status reviews as a priority for the recovery of all sea turtle species. In the threats analysis of the loggerhead recovery plan, trawl bycatch is identified as the greatest source of mortality. While loggerhead bycatch in U.S. Mid-Atlantic bottom otter trawl gear was previously estimated for the period 1996-2004 (Murray 2006, 2008), a recent bycatch analysis estimated the number of loggerhead sea turtle interactions with U.S. Mid-Atlantic bottom trawl gear from 2005-2008 (Warden 2011a). Northeast Fisheries Observer Program data from 1994-2008 were used to develop a model of interaction rates and those predicted rates were applied to 2005-2008 commercial fishing data to estimate the number of interactions for the trawl fleet. The number of predicted average annual loggerhead interactions for 2005-2008 was 292 (CV=0.13, 95% CI=221-369), with an additional 61 loggerheads (CV=0.17, 95% CI=41-83) interacting with trawls but being released through a TED. Of the 292 average annual observable loggerhead interactions, approximately 44 of those were adult equivalents. Warden (2011b) found that latitude, depth and SST were associated with the interaction rate, with the rates being highest south of 37°N latitude in waters < 50 m deep and SST > 15°C. This estimate is a decrease from the average annual loggerhead bycatch in bottom otter trawls during 1996-2004, estimated to be 616 sea turtles (CV=0.23, 95% CI over the 9-year period: 367-890) (Murray 2006, 2008).

There have been several published estimates of the number of loggerheads taken annually as a result of the dredge fishery for Atlantic sea scallops, ranging from a low of zero in 2005 (Murray 2007) to a high of 749 in 2003 (Murray 2004). Murray (2011) recently re-evaluated loggerhead sea turtle interactions in scallop dredge gear from 2001-2008. In that paper, the average number of annual observable interactions of hard-shelled sea turtles in the Mid-Atlantic scallop dredge fishery prior to the implementation of chain mats (January 1, 2001 through September 25, 2006) was estimated to be 288 turtles (CV = 0.14, 95% CI: 209-363) [equivalent to 49 adults], 218 of which were loggerheads [equivalent to 37 adults]. After the implementation of chain mats, the average annual number of observable interactions was estimated to be 20 hard-shelled sea turtles (CV = 0.48, 95% CI: 3-42), 19 of which were loggerheads. If the rate of observable interactions from dredges without chain mats had been applied to trips with chain mats, the estimated number of observable and inferred interactions of hard-shelled sea turtles after chain mats were implemented would have been 125 turtles per year (CV = 0.15, 95% CI: 88-163) [equivalent to 22 adults], 95 of which were loggerheads [equivalent to 16 adults]. Interaction rates of hard-shelled turtles were correlated with sea surface temperature, depth, and use of a chain mat. Results from this recent analysis suggest that chain mats and fishing effort reductions have contributed to the decline in estimated loggerhead sea turtle interactions with scallop dredge gear after 2006 (Murray 2011).

An estimate of the number of loggerheads taken annually in U.S. Mid-Atlantic gillnet fisheries has also recently been published (Murray 2009a, b). From 1995-2006, the annual bycatch of loggerheads in U.S. Mid-Atlantic gillnet gear was estimated to average 350 turtles (CV=0.20, 95% CI over the 12-year period: 234 to 504). Bycatch rates were correlated with latitude, sea surface temperature, and mesh size. The highest predicted bycatch rates occurred in warm waters of the southern Mid-Atlantic in large-mesh gillnets (Murray 2009a).

The U.S. tuna and swordfish longline fisheries that are managed under the Highly Migratory Species (HMS) FMP are estimated to capture 1,905 loggerheads (no more than 339 mortalities) for each 3-year period starting in 2007 (NMFS 2004a). NMFS has mandated gear changes for the HMS fishery to reduce sea turtle bycatch and the likelihood of death from those incidental takes that would still occur (Garrison and Stokes 2010). In 2010, there were 40 observed interactions between loggerhead sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2011a, 2011b). All of the loggerheads were released alive, with the vast majority released with all gear removed. While 2010 total estimates are not yet available, in 2009, 242.9 (95% CI: 167.9-351.2) loggerhead sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP based on the observed takes (Garrison and Stokes 2010). The 2009 estimate is considerably lower than those in 2006 and 2007 and is consistent with historical averages since 2001 (Garrison and Stokes 2010). This fishery represents just one of several longline fisheries operating in the Atlantic Ocean. Lewison *et al.* (2004) estimated that 150,000-200,000 loggerheads were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries as well as others).

Documented takes also occur in other fishery gear types and by non-fishery mortality sources (*e.g.*, hopper dredges, power plants, vessel collisions), but quantitative estimates are unavailable.

The most recent Recovery Plan for loggerhead sea turtles as well as the 2009 Status Review Report identifies global climate change as a threat to loggerhead sea turtles. However, trying to assess the likely effects of climate change on loggerhead sea turtles is extremely difficult given the uncertainty in all climate change models and the difficulty in determining the likely rate of temperature increases and the scope and scale of any accompanying habitat effects. Additionally, no significant climate change-related impacts to loggerhead sea turtle populations have been observed to date. Over the long-term, climate change related impacts are expected to influence biological trajectories on a century scale (Parmesan and Yohe 2003). As noted in the 2009 Status Review (Conant *et al.* 2009), impacts from global climate change induced by human activities are likely to become more apparent in future years (Intergovernmental Panel on Climate Change (IPCC) 2007). Climate change related increasing temperatures, sea level rise, changes in ocean productivity, and increased frequency of storm events may affect loggerhead sea turtles.

Increasing temperatures are expected to result in rising sea levels (Titus and Narayanan 1995 in Conant *et al.* 2009), which could result in increased erosion rates along nesting beaches. Sea level rise could result in the inundation of nesting sites and decrease available nesting habitat (Daniels *et al.* 1993; Fish *et al.* 2005; Baker *et al.* 2006). The BRT noted that the loss of habitat as a result of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis *et al.* 2006; Baker *et al.* 2006; both in Conant *et al.* 2009). Along developed coastlines, and especially in areas where erosion control structures have been constructed to limit shoreline movement, rising sea levels may cause severe effects on nesting females and their eggs as nesting females may deposit eggs seaward of the erosion control structures potentially subjecting them to repeated tidal inundation. However, if global temperatures increase and there is a range shift northwards, beaches not currently used for nesting may become available for loggerhead sea turtles, which may offset some loss of accessibility to beaches in the southern portions of the range.

Climate change has the potential to result in changes at nesting beaches that may affect loggerhead sex ratios. Loggerhead sea turtles exhibit temperature-dependent sex determination. Rapidly increasing global temperatures may result in warmer incubation temperatures and highly female-biased sex ratios (*e.g.*, Glen and Mrosovsky 2004; Hawkes *et al.* 2009); however, to the extent that nesting can occur at beaches further north where sand temperatures are not as warm, these effects may be partially offset. The BRT specifically identified climate change as a threat to loggerhead sea turtles in the neritic/oceanic zone where climate change may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution. In the threats matrix analysis, climate change was considered for oceanic juveniles and adults and eggs/hatchlings. The report states that for oceanic juveniles and adults, "although the effect of trophic level change from…climate change…is unknown it is believed to be very low." For eggs/hatchlings the report states that total mortality from anthropogenic causes, including sea level rise resulting from climate change, is believed to be low relative to the entire life stage. However, only limited data are available on past trends related to climate effects on loggerhead sea turtles; current scientific methods are not able to reliably predict the future magnitude of climate change, associated impacts, whether and to what extent some impacts will offset others, or the adaptive capacity of this species.

However, Van Houtan and Halley (2011) recently developed climate based models to investigate loggerhead nesting (considering juvenile recruitment and breeding remigration) in the North Pacific and Northwest Atlantic. These models found that climate conditions/oceanographic influences explain loggerhead nesting variability, with climate models alone explaining an average 60% (range 18%-88%) of the observed nesting changes over the past several decades. In terms of future nesting projections, modeled climate data show a future positive trend for Florida nesting, with increases through 2040 as a result of the Atlantic Multidecadal Oscillation signal.

While there is a reasonable degree of certainty that certain climate change related effects will be experienced globally (*e.g.*, rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects to sea turtles resulting from climate change are not predictable or quantifiable at this time (Hawkes *et al.* 2009). However, given this uncertainty and the likely rate of change associated with climate impacts (i.e., the century scale), it is unlikely that climate related impacts will have a significant effect on the status of loggerhead sea turtles over the temporal scale of the proposed action (*i.e.*, through 2018).

Summary of Status for Loggerhead Sea Turtles

Loggerheads are a long-lived species and reach sexual maturity relatively late at around 32-35 years in the Northwest Atlantic (NMFS and USFWS 2008). The species continues to be affected by many factors occurring on nesting beaches and in the water. These include poaching, habitat loss, and nesting predation that affects eggs, hatchlings, and nesting females on land, as well as fishery interactions, vessel interactions, marine pollution, and non-fishery (*e.g*., dredging) operations affecting all sexes and age classes in the water (NRC 1990; NMFS and USFWS 2007a, 2008). As a result, loggerheads still face many of the original threats that were the cause of their listing under the ESA.

As mentioned previously, a final revised recovery plan for loggerhead sea turtles in the Northwest Atlantic was recently published by NMFS and FWS in December 2008. The revised recovery plan is significant in that it identifies five unique recovery units, which comprise the population of loggerheads in the Northwest Atlantic, and describes specific recovery criteria for each recovery unit. The recovery plan noted a decline in annual nest counts for three of the five recovery units for loggerheads in the Northwest Atlantic, including the PFRU, which is the largest (in terms of number of nests laid) in the Atlantic Ocean. The nesting trends for the other two recovery units could not be determined due to an absence of long term data.

NMFS convened a new Loggerhead Turtle Expert Working Group (TEWG) to review all available information on Atlantic loggerheads in order to evaluate the status of this species in the Atlantic. A final report from the Loggerhead TEWG was published in July 2009. In this report, the TEWG indicated that it could not determine whether the decreasing annual numbers of nests among the Northwest Atlantic loggerhead subpopulations were due to stochastic processes resulting in fewer nests, a decreasing average reproductive output of adult females, decreasing numbers of adult females, or a combination of these factors. Many factors are responsible for past or present loggerhead mortality that could impact current nest numbers; however, no single mortality factor stands out as a likely primary factor. It is likely that several factors compound to create the current decline, including incidental capture (in fisheries, power plant intakes, and dredging operations), lower adult female survival rates, increases in the proportion of first-time nesters, continued directed harvest, and increases in mortality due to disease. Regardless, the

TEWG stated that “it is clear that the current levels of hatchling output will result in depressed recruitment to subsequent life stages over the coming decades” (TEWG 2009). However, the report does not provide information on the rate or amount of expected decrease in recruitment but goes on to state that the ability to assess the current status of loggerhead subpopulations is limited due to a lack of fundamental life history information and specific census and mortality data.

While several documents reported the decline in nesting numbers in the NWA DPS (NMFS and USFWS 2008, TEWG 2009), when nest counts through 2010 are analyzed, the nesting trends from 1989-2010 are not significantly different than zero for all recovery units within the NWA DPS for which there are enough data to analyze (76 FR 58868, September 22, 2011). The SEFSC (2009) estimated the number of adult females in the NWA DPS at 30,000, and if a 1:1 adult sex ratio is assumed, the result is 60,000 adults in this DPS. Based on the reviews of nesting data, as well as information on population abundance and trends, NMFS and USFWS determined in the September 2011 listing rule that the NWA DPS should be listed as threatened. They found that an endangered status for the NWA DPS was not warranted given the large size of the nesting population, the overall nesting population remains widespread, the trend for the nesting population appears to be stabilizing, and substantial conservation efforts are underway to address threats.

4.2.8 Kemp’s Ridley Sea Turtles

Distribution and Life History
The Kemp’s ridley is one of the least abundant of the world’s sea turtle species. In contrast to loggerhead, leatherback, and green sea turtles, which are found in multiple oceans of the world, Kemp’s ridleys typically occur only in the Gulf of Mexico and the northwestern Atlantic Ocean (NMFS *et al.* 2011).

Kemp’s ridleys mature at 10-17 years (Caillouet *et al.* 1995; Schmid and Witzell 1997; Snover *et al.* 2007; NMFS and USFWS 2007c). Nesting occurs from April through July each year with hatchlings emerging after 45-58 days (NMFS *et al.* 2011). Females lay an average of 2.5 clutches within a season (TEWG 1998, 2000) and the mean remigration interval for adult females is 2 years (Marquez *et al.* 1982; TEWG 1998, 2000).

Once they leave the nesting beach, hatchlings presumably enter the Gulf of Mexico where they feed on available S*argassum* and associated infauna or other epipelagic species (NMFS *et al.* 2011). The presence of juvenile turtles along both the U.S. Atlantic and Gulf of Mexico coasts, where they are recruited to the coastal benthic environment, indicates that post-hatchlings are distributed in both the Gulf of Mexico and Atlantic Ocean (TEWG 2000).

The location and size classes of dead turtles recovered by the STSSN suggests that benthic immature developmental areas occur along the U.S. coast and that these areas may change given resource quality and quantity (TEWG 2000). Developmental habitats are defined by several characteristics, including coastal areas sheltered from high winds and waves such as embayments and estuaries, and nearshore temperate waters shallower than 50 m (NMFS and USFWS 2007c). The suitability of these habitats depends on resource availability, with optimal environments providing rich sources of crabs and other invertebrates. Kemp's ridleys consume a variety of crab species, including *Callinectes*, *Ovalipes*, *Libinia*, and *Cancer* species. Mollusks, shrimp, and fish are consumed less frequently (Bjorndal 1997). A wide variety of substrates have been documented to provide good foraging habitat, including seagrass beds, oyster reefs, sandy and mud bottoms, and rock outcroppings (NMFS and USFWS 2007c).

Foraging areas documented along the U.S. Atlantic coast include Charleston Harbor, Pamlico Sound (Epperly *et al.* 1995c), Chesapeake Bay (Musick and Limpus 1997), Delaware Bay (Stetzar 2002), and Long Island Sound (Morreale and Standora 1993; Morreale *et al.* 2005). For instance, in the Chesapeake Bay, Kemp's ridleys frequently forage in submerged aquatic grass beds for crabs (Musick and Limpus 1997). Upon leaving Chesapeake Bay in autumn, juvenile Kemp's ridleys migrate down the coast, passing Cape Hatteras in December and January (Musick and Limpus 1997). These larger juveniles are joined by juveniles of the same size from North Carolina sounds and smaller juveniles from New York and New England to form one of the densest concentrations of Kemp's ridleys outside of the Gulf of Mexico (Epperly *et al.* 1995a, 1995b; Musick and Limpus 1997).

Adult Kemp's ridleys are found in the coastal regions of the Gulf of Mexico and southeastern United States, but are typically rare in the northeastern U.S. waters of the Atlantic (TEWG 2000). Adults are primarily found in nearshore waters of 37 m or less that are rich in crabs and have a sandy or muddy bottom (NMFS and USFWS 2007c).

Population Dynamics and Status

The majority of Kemp's ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; NMFS and USFWS 2007c; NMFS *et al.* 2011). There is a limited amount of scattered nesting to the north and south of the primary nesting beach (NMFS and USFWS 2007c). Nesting often occurs in synchronized emergences termed *arribadas*. The number of recorded nests reached an estimated low of 702 nests in 1985, corresponding to fewer than 300 adult females nesting in that season (TEWG 2000; NMFS and USFWS 2007c; NMFS *et al.* 2011). Conservation efforts by Mexican and U.S. agencies have aided this species by eliminating egg harvest, protecting eggs and hatchlings, and reducing at-sea mortality through fishing regulations (TEWG 2000). Since the mid-1980s, the number of nests observed at Rancho Nuevo and nearby beaches has increased 14-16% per year (Heppell *et al.* 2005), allowing cautious optimism that the population is on its way to recovery. An estimated 5,500 females nested in the State of Tamaulipas over a 3-day period in May 2007 and over 4,000 of those nested at Rancho Nuevo (NMFS and USFWS 2007c). In 2008, 17,882 nests were documented on Mexican nesting beaches (NMFS 2011). There is limited nesting in the United States, most of which is located in South Texas. While six nests were documented in 1996, a record 195 nests were found in 2008 (NMFS 2011).

Threats

Kemp's ridleys face many of the same natural threats as loggerheads, including destruction of nesting habitat from storm events, predators, and oceanographic-related events such as cold-stunning. Although cold-stunning can occur throughout the range of the species, it may be a greater risk for sea turtles that utilize the more northern habitats of Cape Cod Bay and Long

Island Sound. In the last five years (2006-2010), the number of cold-stunned turtles on Cape Cod beaches averaged 115 Kemp's ridleys, 7 loggerheads, and 7 greens (NMFS unpublished data). The numbers ranged from a low in 2007 of 27 Kemp's ridleys, 5 loggerheads, and 5 greens to a high in 2010 of 213 Kemp's ridleys, 4 loggerheads, and 14 greens. Annual cold stun events vary in magnitude; the extent of episodic major cold stun events may be associated with numbers of turtles utilizing Northeast U.S. waters in a given year, oceanographic conditions, and/or the occurrence of storm events in the late fall. Although many cold-stunned turtles can survive if they are found early enough, these events represent a significant source of natural mortality for Kemp's ridleys.

Like other sea turtle species, the severe decline in the Kemp's ridley population appears to have been heavily influenced by a combination of exploitation of eggs and impacts from fishery interactions. From the 1940s through the early 1960s, nests from Ranch Nuevo were heavily exploited, but beach protection in 1967 helped to curtail this activity (NMFS *et al.* 2011). Following World War II, there was a substantial increase in the number of trawl vessels, particularly shrimp trawlers, in the Gulf of Mexico where adult Kemp's ridley sea turtles occur. Information from fisheries observers helped to demonstrate the high number of turtles taken in these shrimp trawls (USFWS and NMFS 1992). Subsequently, NMFS has worked with the industry to reduce sea turtle takes in shrimp trawls and other trawl fisheries, including the development and use of turtle excluder devices (TEDs). As described above, there is lengthy regulatory history with regard to the use of TEDs in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries (NMFS 2002a; Epperly 2003; Lewison *et al.* 2003). The 2002 Biological Opinion on shrimp trawling in the southeastern United States concluded that 155,503 Kemp's ridley sea turtles would be taken annually in the fishery with 4,208 of the takes resulting in mortality (NMFS 2002a).

Although modifications to shrimp trawls have helped to reduce mortality of Kemp's ridleys, a recent assessment found that the Southeast/Gulf of Mexico shrimp trawl fishery remained responsible for the vast majority of U.S. fishery interactions (up to 98%) and mortalities (more than 80%). Finkbeiner *et al.* (2011) compiled cumulative sea turtle bycatch information in U.S. fisheries from 1990 through 2007, before and after implementation of bycatch mitigation measures. Information was obtained from peer reviewed publications and NMFS documents (e.g., Biological Opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, occurred annually (since implementation of bycatch mitigation measures). Kemp's ridleys interacted with fisheries most frequently, with the highest level of mean annual mortality (2,700), followed by loggerheads (1,400), greens (300), and leatherbacks (40). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations.

This species is also affected by other sources of anthropogenic impact (fishery and non-fishery related), similar to those discussed above. Three Kemp's ridley captures in Mid-Atlantic trawl fisheries were documented by NMFS observers between 1994 and 2008 (Warden and Bisack 2010), and eight Kemp's ridleys were documented by NMFS observers in mid-Atlantic sink gillnet fisheries between 1995 and 2006 (Murray 2009a). Additionally, in the spring of 2000, a total of five Kemp's ridley carcasses were recovered from the same North Carolina beaches where 275 loggerhead carcasses were found. The cause of death for most of the turtles recovered was unknown, but the mass mortality event was suspected by NMFS to have been from a large-mesh gillnet fishery for monkfish and dogfish operating offshore in the preceding weeks (67 FR 71895, December 3, 2002). The five Kemp's ridley carcasses that were found are likely to have been only a minimum count of the number of Kemp's ridleys that were killed or seriously injured as a result of the fishery interaction, since it is unlikely that all of the carcasses washed ashore. The NMFS Northeast Fisheries Science Center also documented 14 Kemp's ridleys entangled in or impinged on Virginia pound net leaders from 2002-2005. Note that bycatch estimates for Kemp's ridleys in various fishing gear types (*e.g.*, trawl, gillnet, dredge) are not available at this time, largely due to the low number of observed interactions precluding a robust estimate. Kemp's ridley interactions in non-fisheries have also been observed; for example, the Oyster Creek Nuclear Generating Station in Barnegat Bay, New Jersey, recorded a total of 27 Kemp's ridleys (15 of which were found alive) impinged or captured on their intake screens from 1992-2006 (NMFS 2006).

The recovery plan for Kemp's ridley sea turtles (NMFS *et al.* 2011) identifies climate change as a threat; however, as with the other species discussed above, no significant climate change-related impacts to Kemp's ridley sea turtles have been observed to date. Atmospheric warming could cause habitat alteration which may change food resources such as crabs and other invertebrates. It may increase hurricane activity, leading to an increase in debris in nearshore and offshore waters, which may result in an increase in entanglement, ingestion, or drowning. In addition, increased hurricane activity may cause damage to nesting beaches or inundate nests with sea water. Atmospheric warming may change convergence zones, currents and other oceanographic features that are relevant to Kemp's ridleys, as well as change rain regimes and levels of nearshore runoff.

Considering that the Kemp's ridley has temperature-dependent sex determination (Wibbels 2003) and the vast majority of the nesting range is restricted to the State of Tamaulipas, Mexico, global warming could potentially shift population sex ratios towards females and thus change the reproductive ecology of this species. A female bias is presumed to increase egg production (assuming that the availability of males does not become a limiting factor) (Coyne and Landry 2007) and increase the rate of recovery; however, it is unknown at what point the percentage of males may become insufficient to facilitate maximum fertilization rates in a population. If males become a limiting factor in the reproductive ecology of the Kemp's ridley, then reproductive output in the population could decrease (Coyne 2000). Low numbers of males could also result in the loss of genetic diversity within a population; however, there is currently no evidence that this is a problem in the Kemp's ridley population (NMFS *et al.* 2011). Models (Davenport 1997, Hulin and Guillon 2007, Hawkes *et al.* 2007, all referenced in NMFS *et al.* 2011) predict very long-term reductions in fertility in sea turtles due to climate change, but due to the relatively long life cycle of sea turtles, reductions may not be seen until 30 to 50 years in the future.

Another potential impact from global climate change is sea level rise, which may result in increased beach erosion at nesting sites. Beach erosion may be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents. In the case of the Kemp's ridley where most of the critical nesting beaches are undeveloped, beaches may shift landward and still be available for nesting. The Padre Island National Seashore (PAIS) shoreline is accreting, unlike much of the Texas coast, and with nesting increasing and the sand temperatures slightly cooler than at Rancho Nuevo, PAIS could become an increasingly important source of males for the population.

As with the other sea turtle species discussed in this section, while there is a reasonable degree of certainty that certain climate change related effects will be experienced globally (*e.g.*, rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects of climate change on this species are not predictable or quantifiable at this time (Hawkes *et al.* 2009). However, given the likely rate of change associated with climate impacts (*i.e.*, the century scale), it is unlikely that climate change will have a significant effect on the status of Kemp's ridley sea turtles over the temporal scale of the proposed action (*i.e.*, through 2018).

Summary of Status for Kemp's Ridley Sea Turtles

The majority of Kemp's ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; NMFS and USFWS 2007c; NMFS *et al.* 2011). The number of nesting females in the Kemp's ridley population declined dramatically from the late 1940s through the mid-1980s, with an estimated 40,000 nesting females in a single *arribada* in 1947 and fewer than 300 nesting females in the entire 1985 nesting season (TEWG 2000; NMFS *et al.* 2011). However, the total annual number of nests at Rancho Nuevo gradually began to increase in the 1990s (NMFS and USFWS 2007c). Based on the number of nests laid in 2006 and the remigration interval for Kemp's ridley sea turtles (1.8-2 years), there were an estimated 7,000-8,000 adult female Kemp's ridley sea turtles in 2006 (NMFS and USFWS 2007c). The number of adult males in the population is unknown, but sex ratios of hatchlings and immature Kemp's ridleys suggest that the population is female-biased, suggesting that the number of adult males is less than the number of adult females (NMFS and USFWS 2007c). While there is cautious optimism for recovery, events such as the Deepwater Horizon oil release, and stranding events associated increased skimmer trawl use and poor TED compliance in the northern Gulf of Mexico may dampen recent population growth.

As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like dredging, pollution, and habitat destruction account for an unknown level of other mortality. Based on their 5-year status review of the species, NMFS and USFWS (2007c) determined that Kemp's ridley sea turtles should not be reclassified as threatened under the ESA. A revised bi-national recovery plan was published for public comment in 2010, and in September 2011, NMFS, USFWS, and the Services and the Secretary of Environment and Natural Resources, Mexico (SEMARNAT) released the second revision to the Kemp's ridley recovery plan.

4.2.9 Green Sea Turtles

Green sea turtles are distributed circumglobally, and can be found in the Pacific, Indian, and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 1991, 2007d; Seminoff 2004). In 1978, the Atlantic population of the green sea turtle was listed as threatened under the

ESA, except for the breeding populations in Florida and on the Pacific coast of Mexico, which were listed as endangered. As it is difficult to differentiate between breeding populations away from the nesting beaches, all green sea turtles in the water are considered endangered.

Pacific Ocean
Green sea turtles occur in the western, central, and eastern Pacific. Foraging areas are also found throughout the Pacific and along the southwestern U.S. coast (NMFS and USFWS 1998b). In the western Pacific, major nesting rookeries at four sites including Heron Island (Australia), Raine Island (Australia), Guam, and Japan were evaluated and determined to be increasing in abundance, with the exception of Guam which appears stable (NMFS and USFWS 2007d). In the central Pacific, nesting occurs on French Frigate Shoals, Hawaii, which has also been reported as increasing with a mean of 400 nesting females annually from 2002-2006 (NMFS and USFWS 2007d). The main nesting sites for the green sea turtle in the eastern Pacific are located in Michoacan, Mexico and in the Galapagos Islands, Ecuador (NMFS and USFWS 2007d). The number of nesting females per year exceeds 1,000 females at each site (NMFS and USFWS 2007d). However, historically, greater than 20,000 females per year are believed to have nested in Michoacan alone (Cliffton *et al.* 1982; NMFS and USFWS 2007d). The Pacific Mexico green turtle nesting population (also called the black turtle) is considered endangered.

Historically, green sea turtles were used in many areas of the Pacific for food. They were also commercially exploited, which, coupled with habitat degradation, led to their decline in the Pacific (NMFS and USFWS 1998b). Green sea turtles in the Pacific continue to be affected by poaching, habitat loss or degradation, fishing gear interactions, and fibropapillomatosis, which is a viral disease that causes tumors in affected turtles (NMFS and USFWS 1998b; NMFS 2004b).

Indian Ocean
There are numerous nesting sites for green sea turtles in the Indian Ocean. One of the largest nesting sites for green sea turtles worldwide occurs on the beaches of Oman where an estimated 20,000 green sea turtles nest annually (Hirth 1997; Ferreira *et al.* 2003). Based on a review of the 32 Index Sites used to monitor green sea turtle nesting worldwide, Seminoff (2004) concluded that declines in green sea turtle nesting were evident for many of the Indian Ocean Index Sites. While several of these had not demonstrated further declines in the more recent past, only the Comoros Island Index Site in the western Indian Ocean showed evidence of increased nesting (Seminoff 2004).

Mediterranean Sea
There are four nesting concentrations of green sea turtles in the Mediterranean from which data are available – Turkey, Cyprus, Israel, and Syria. Currently, approximately 300-400 females nest each year, about two-thirds of which nest in Turkey and one-third in Cyprus. Although green sea turtles are depleted from historic levels in the Mediterranean Sea (Kasparek *et al.* 2001), nesting data gathered since the early 1990s in Turkey, Cyprus, and Israel show no apparent trend in any direction. However, a declining trend is apparent along the coast of Palestine/Israel, where 300-350 nests were deposited each year in the 1950s (Sella 1982) compared to a mean of 6 nests per year from 1993-2004 (Kuller 1999; Y. Levy, Israeli Sea Turtle Rescue Center, unpublished data). A recent discovery of green sea turtle nesting in Syria

adds roughly 100 nests per year to green sea turtle nesting activity in the Mediterranean (Rees *et al.* 2005). That such a major nesting concentration could have gone unnoticed until recently (the Syria coast was surveyed in 1991, but nesting activity was attributed to loggerheads) bodes well for the ongoing speculation that the unsurveyed coast of Libya may also host substantial nesting.

Atlantic Ocean -Distribution and Life History
As has occurred in other oceans of its range, green sea turtles were once the target of directed fisheries in the United States and throughout the Caribbean. In 1890, over one million pounds of green sea turtles were taken in a directed fishery in the Gulf of Mexico (Doughty 1984). However, declines in the turtle fishery throughout the Gulf of Mexico were evident by 1902 (Doughty 1984).

In the western Atlantic, large juvenile and adult green sea turtles are largely herbivorous, occurring in habitats containing benthic algae and seagrasses from Massachusetts to Argentina, including the Gulf of Mexico and Caribbean (Wynne and Schwartz 1999). Green sea turtles occur seasonally in Mid-Atlantic and Northeast waters such as Chesapeake Bay and Long Island Sound (Musick and Limpus 1997; Morreale and Standora 1998; Morreale *et al.* 2005), which serve as foraging and developmental habitats.

Some of the principal feeding areas in the western Atlantic Ocean include the upper west coast of Florida, the Florida Keys, and the northwestern coast of the Yucatán Peninsula. Additional important foraging areas in the western Atlantic include the Mosquito and Indian River Lagoon systems and nearshore wormrock reefs between Sebastian and Ft. Pierce Inlets in Florida, Florida Bay, the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean coast of Panama, and scattered areas along Colombia and Brazil (Hirth 1971). The waters surrounding the island of Culebra, Puerto Rico, and its outlying keys are designated critical habitat for the green sea turtle.

Age at maturity for green sea turtles is estimated to be 20-50 years (Balazs 1982; Frazer and Ehrhart 1985; Seminoff 2004). As is the case with the other sea turtle species described above, adult females may nest multiple times in a season (average 3 nests/season with approximately 100 eggs/nest) and typically do not nest in successive years (NMFS and USFWS 1991; Hirth 1997).

Population Dynamics and Status
Like other sea turtle species, nest count information for green sea turtles provides information on the relative abundance of nesting, and the contribution of each nesting group to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually. The 5-year status review for the species identified eight geographic areas considered to be primary sites for threatened green sea turtle nesting in the Atlantic/Caribbean, and reviewed the trend in nest count data for each (NMFS and USFWS 2007d). These include: (1) Yucatán Peninsula, Mexico, (2) Tortuguero, Costa Rica, (3) Aves Island, Venezuela, (4) Galibi Reserve, Suriname, (5) Isla Trindade, Brazil, (6) Ascension Island, United Kingdom, (7) Bioko Island, Equatorial Guinea, and (8) Bijagos Achipelago, Guinea-Bissau (NMFS and USFWS 2007d). Nesting at all of these sites is considered to be stable or

increasing with the exception of Bioko Island, which may be declining. However, the lack of sufficient data precludes a meaningful trend assessment for this site (NMFS and USFWS 2007d).

Seminoff (2004) reviewed green sea turtle nesting data for eight sites in the western, eastern, and central Atlantic, including all of the above threatened nesting sites with the exception that nesting in Florida was reviewed in place of Isla Trindade, Brazil. He concluded that all sites in the central and western Atlantic showed increased nesting with the exception of nesting at Aves Island, Venezuela, while both sites in the eastern Atlantic demonstrated decreased nesting. These sites are not inclusive of all green sea turtle nesting in the Atlantic Ocean. However, other sites are not believed to support nesting levels high enough that would change the overall status of the species in the Atlantic (NMFS and USFWS 2007d).

By far, the most important nesting concentration for green sea turtles in the western Atlantic is in Tortuguero, Costa Rica (NMFS and USFWS 2007d). Nesting in the area has increased considerably since the 1970s and nest count data from 1999-2003 suggest nesting by 17,402-37,290 females per year (NMFS and USFWS 2007d). The number of females nesting per year on beaches in the Yucatán, at Aves Island, Galibi Reserve, and Isla Trindade number in the hundreds to low thousands, depending on the site (NMFS and USFWS 2007d).

The status of the endangered Florida breeding population was also evaluated in the 5-year review (NMFS and USFWS 2007d). The pattern of green sea turtle nesting shows biennial peaks in abundance, with a generally positive trend since establishment of the Florida index beach surveys in 1989. This trend is perhaps due to increased protective legislation throughout the Caribbean (Meylan *et al.* 1995), as well as protections in Florida and throughout the United States (NMFS and USFWS 2007d).

The statewide Florida surveys (2000-2006) have shown that a mean of approximately 5,600 nests are laid annually in Florida, with a low of 581 in 2001 to a high of 9,644 in 2005 (NMFS and USFWS 2007d). Most nesting occurs along the east coast of Florida, but occasional nesting has been documented along the Gulf coast of Florida, at Southwest Florida beaches, as well as the beaches in the Florida Panhandle (Meylan *et al.* 1995). More recently, green sea turtle nesting occurred on Bald Head Island, North Carolina (just east of the mouth of the Cape Fear River), Onslow Island, and Cape Hatteras National Seashore. One green sea turtle nested on a beach in Delaware in 2011, although its occurrence was considered very rare.

Threats

Green sea turtles face many of the same natural threats as loggerhead and Kemp's ridley sea turtles. In addition, green sea turtles appear to be particularly susceptible to fibropapillomatosis, an epizootic disease producing lobe-shaped tumors on the soft portion of a turtle's body. Juveniles appear to be most affected in that they have the highest incidence of disease and the most extensive lesions, whereas lesions in nesting adults are rare. Also, green sea turtles frequenting nearshore waters, areas adjacent to large human populations, and areas with low water turnover, such as lagoons, have a higher incidence of the disease than individuals in deeper, more remote waters. The occurrence of fibropapilloma tumors may result in impaired foraging, breathing, or swimming ability, leading potentially to death (George 1997).

As with the other sea turtle species, incidental fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches. Witherington *et al.* (2009) observes that because green sea turtles spend a shorter time in oceanic waters and as older juveniles occur on shallow seagrass pastures (where benthic trawling is unlikely), they avoid high mortalities in pelagic longline and benthic trawl fisheries. Although the relatively low number of observed green sea turtle captures makes it difficult to estimate bycatch rates and annual take levels, green sea turtles have been observed captured in the pelagic driftnet, pelagic longline, southeast shrimp trawl, and mid-Atlantic trawl and gillnet fisheries. Murray (2009a) also lists five observed captures of green turtle in Mid-Atlantic sink gillnet gear between 1995 and 2006.

Finkbeiner *et al.* (2011) compiled cumulative sea turtle bycatch information in U.S. fisheries from 1990 through 2007, before and after implementation of bycatch mitigation measures. Information was obtained from peer reviewed publications and NMFS documents (e.g., Biological Opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, occurred annually (since implementation of bycatch mitigation measures). Kemp's ridleys interacted with fisheries most frequently, with the highest level of mean annual mortality (2,700), followed by loggerheads (1,400), greens (300), and leatherbacks (40). The Southeast/Gulf of Mexico shrimp trawl fishery was responsible for the vast majority of U.S. interactions (up to 98%) and mortalities (more than 80%). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations.

Other activities like channel dredging, marine debris, pollution, vessel strikes, power plant impingement, and habitat destruction account for an unquantifiable level of other mortality. Stranding reports indicate that between 200-400 green sea turtles strand annually along the eastern U.S. coast from a variety of causes most of which are unknown (STSSN database).

The five year status review for green sea turtles (NMFS and USFWS 2007d) notes that global climate change is affecting green sea turtles and is likely to continue to be a threat. There is an increasing female bias in the sex ratio of green turtle hatchlings. While this is partly attributable to imperfect egg hatchery practices, global climate change is also implicated as a likely cause as warmer sand temperatures at nesting beaches are likely to result in the production of more female embryos. At least one nesting site, Ascension Island, has had an increase in mean sand temperature in recent years (Hays *et al.* 2003 in NMFS and USFWS 2007d). Climate change may also impact nesting beaches through sea level rise which may reduce the availability of nesting habitat and increase the risk of nest inundation. Loss of appropriate nesting habitat may also be accelerated by a combination of other environmental and oceanographic changes, such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion. Oceanic changes related to rising water temperatures could result in changes in the abundance and distribution of the primary food sources of green sea turtles, which in turn could result in changes in behavior and distribution of this species. Seagrass habitats may suffer from decreased productivity and/or increased stress due to sea level rise, as well as salinity and temperature changes (Short and Neckles 1999; Duarte 2002).

As noted above, the increasing female bias in green sea turtle hatchlings is thought to be at least partially linked to increases in temperatures at nesting beaches. However, due to a lack of scientific data, the specific future effects of climate change on green sea turtles species are not predictable or quantifiable to any degree at this time (Hawkes *et al.* 2009). For example, information is not available to predict the extent and rate to which sand temperatures at the nesting beaches used by green sea turtles may increase in the short-term future and the extent to which green sea turtles may be able to cope with this change by selecting cooler areas of the beach or shifting their nesting distribution to other beaches at which increases in sand temperature may not be experienced.

Summary of Status of Green Sea Turtles

A review of 32 Index Sites[3] distributed globally revealed a 48-67% decline in the number of mature females nesting annually over the last three generations[4] (Seminoff 2004). An evaluation of green sea turtle nesting sites was also conducted as part of the 5-year status review of the species (NMFS and USFWS 2007d). Of the 23 threatened nesting groups assessed in that report for which nesting abundance trends could be determined, ten were considered to be increasing, nine were considered stable, and four were considered to be decreasing (NMFS and USFWS 2007d). Nesting groups were considered to be doing relatively well (the number of sites with increasing nesting were greater than the number of sites with decreasing nesting) in the Pacific, western Atlantic, and central Atlantic (NMFS and USFWS 2007d). However, nesting populations were determined to be doing relatively poorly in Southeast Asia, eastern Indian Ocean, and perhaps the Mediterranean. Overall, based on mean annual reproductive effort, the report estimated that 108,761 to 150,521 females nest each year among the 46 threatened and endangered nesting sites included in the evaluation (NMFS and USFWS 2007d). However, given the late age to maturity for green sea turtles, caution is urged regarding the status for any of the nesting groups since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d).

Seminoff (2004) and NMFS and USFWS (2007d) made comparable conclusions with regard to nesting for four nesting sites in the western Atlantic that indicate sea turtle abundance is increasing in the Atlantic Ocean. Each also concluded that nesting at Tortuguero, Costa Rica represented the most important nesting area for green sea turtles in the western Atlantic and that nesting had increased markedly since the 1970s (Seminoff 2004; NMFS and USFWS 2007d).

However, the 5-year review also noted that the Tortuguero nesting stock continued to be affected by ongoing directed take at their primary foraging area in Nicaragua (NMFS and USFWS 2007d). The endangered breeding population in Florida appears to be increasing based upon index nesting data from 1989-2010 (NMFS 2011).

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[3] The 32 Index Sites include all of the major known nesting areas as well as many of the lesser nesting areas for which quantitative data are available.

[4] Generation times ranged from 35.5 years to 49.5 years for the assessment depending on the Index Beach site

As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like hopper dredging, pollution, and habitat destruction account for an unknown level of other mortality. Based on its 5-year status review of the species, NMFS and USFWS (2007d) determined that the listing classification for green sea turtles should not be changed. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPSs should be identified (NMFS and USFWS 2007d).

4.2.10 Leatherback Sea Turtles

Leatherback sea turtles are widely distributed throughout the oceans of the world, including the Atlantic, Pacific, and Indian Oceans, and the Mediterranean Sea (Ernst and Barbour 1972). Leatherbacks are the largest living turtles and range farther than any other sea turtle species. Their large size and tolerance of relatively low water temperatures allows them to occur in boreal waters such as those off Labrador and in the Barents Sea (NMFS and USFWS 1995).

In 1980, the leatherback population was estimated at approximately 115,000 adult females globally (Pritchard 1982). By 1995, this global population of adult females was estimated to have declined to 34,500 (Spotila *et al.* 1996). The most recent population size estimate for the North Atlantic alone is a range of 34,000-94,000 adult leatherbacks (TEWG 2007). Thus, there is substantial uncertainty with respect to global population estimates of leatherback sea turtles.

Pacific Ocean

Leatherback nesting has been declining at all major Pacific basin nesting beaches for the last two decades (Spotila *et al.* 1996, 2000; NMFS and USFWS 1998, 2007b; Sarti *et al.* 2000). In the western Pacific, major nesting beaches occur in Papua New Guinea, Indonesia, Solomon Islands, and Vanuatu, with an approximate 2,700-4,500 total breeding females, estimated from nest counts (Dutton *et al.* 2007). While there appears to be overall long term population decline, the Indonesian nesting aggregation at Jamursba-Medi is currently stable (since 1999), although there is evidence to suggest a significant and continued decline in leatherback nesting in Papua New Guinea and Solomon Islands over the past 30 years (NMFS 2011). Leatherback sea turtles disappeared from India before 1930, have been virtually extinct in Sri Lanka since 1994, and appear to be approaching extinction in Malaysia (Spotila *et al.* 2000). In Fiji, Thailand, and Australia, leatherback sea turtles have only been known to nest in low densities and scattered sites.

The largest, extant leatherback nesting group in the Indo-Pacific lies on the North Vogelkop coast of West Papua, Indonesia, with 3,000-5,000 nests reported annually in the 1990s (Suárez *et al.* 2000). However, in 1999, local villagers started reporting dramatic declines in sea turtles near their villages (Suárez 1999). Declines in nesting groups have been reported throughout the western Pacific region where observers reported that nesting groups were well below abundance levels that were observed several decades ago (*e.g.*, Suárez 1999). Recent studies indicate a continual and significant long term nesting decline of 5.9% per year, at primary western Pacific beaches, since 1984 (Tapilatu *et al. 2013)*.

Leatherback sea turtles in the western Pacific are threatened by poaching of eggs, killing of nesting females, human encroachment on nesting beaches, incidental capture in fishing gear, beach erosion, and egg predation by animals.

In the eastern Pacific Ocean, major leatherback nesting beaches are located in Mexico and Costa Rica, where nest numbers have been declining. According to reports from the late 1970s and early 1980s, beaches located on the Mexican Pacific coasts of Michoacán, Guerrero, and Oaxaca sustained a large portion, perhaps 50%, of all global nesting by leatherbacks (Sarti *et al.* 1996). A dramatic decline has been seen on nesting beaches in Pacific Mexico, where aerial survey data was used to estimate that tens of thousands of leatherback nests were laid on the beaches in the 1980s (Pritchard 1982), but a total of only 120 nests on the four primary index beaches (combined) were counted in the 2003-2004 season (Sarti Martinez *et al.* 2007). Since the early 1980s, the Mexican Pacific population of adult female leatherback turtles has declined to slightly more than 200 during 1998-1999 and 1999-2000 (Sarti *et al.* 2000). Spotila *et al.* (2000) reported the decline of the leatherback nesting at Playa Grande, Costa Rica, which had been the fourth largest nesting group in the world and the most important nesting beach in the Pacific. Between 1988 and 1999, the nesting group declined from 1,367 to 117 female leatherback sea turtles. Based on their models, Spotila *et al.* (2000) estimated that the group could fall to less than 50 females by 2003-2004. Another, more recent, analysis of the Costa Rican nesting beaches indicates a decline in nesting during 15 years of monitoring (1989-2004) with approximately 1,504 females nesting in 1988-1989 to an average of 188 females nesting in 2000-2001 and 2003-2004 (NMFS and USFWS 2007b), indicating that the reductions in nesting females were not as extreme as the reductions predicted by Spotila *et al.* (2000).

On September 26, 2007, NMFS received a petition to revise the critical habitat designation for leatherback sea turtles to include waters along the U.S. West Coast. On December 28, 2007, NMFS published a positive 90-day finding on the petition and convened a critical habitat review team. On January 26, 2012, NMFS published a final rule to revise the critical habitat designation to include three particular areas of marine habitat. The designation includes approximately 16,910 square miles along the California coast from Point Arena to Point Arguello east of the 3,000 meter depth contour, and 25,004 square miles from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 meter depth contour. The areas comprise approximately 41,914 square miles of marine habitat and include waters from the ocean surface down to a maximum depth of 262 feet. The designated critical habitat areas contain the physical or biological feature essential to the conservation of the species that may require special management conservation or protection. In particular, the team identified one Primary Constituent Element: the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae, of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks.

Leatherbacks in the eastern Pacific face a number of threats to their survival. For example, commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru; purse seine fisheries for tuna in the eastern tropical Pacific Ocean; and California/Oregon drift gillnet fisheries are known to capture, injure, or kill leatherbacks in the eastern Pacific Ocean. Given the declines in leatherback nesting in the Pacific, some researchers have concluded that the leatherback is on the verge of extinction in the Pacific Ocean (*e.g.*, Spotila *et al.* 1996, 2000).

Indian Ocean
Leatherbacks nest in several areas around the Indian Ocean. These sites include Tongaland, South Africa (Pritchard 2002) and the Andaman and Nicobar Islands (Andrews *et al.* 2002). Intensive survey and tagging work in 2001 provided new information on the level of nesting in the Andaman and Nicobar Islands (Andrews *et al.* 2002). Based on the survey and tagging work, it was estimated that 400-500 female leatherbacks nest annually on Great Nicobar Island (Andrews *et al.* 2002). The number of nesting females using the Andaman and Nicobar Islands combined was estimated around 1,000 (Andrews and Shanker 2002). Some nesting also occurs along the coast of Sri Lanka, although in much smaller numbers than in the past (Pritchard 2002).

Mediterranean Sea
Casale *et al.* (2003) reviewed the distribution of leatherback sea turtles in the Mediterranean. Among the 411 individual records of leatherback sightings in the Mediterranean, there were no nesting records. Nesting in the Mediterranean is believed to be extremely rare if it occurs at all. Leatherbacks found in Mediterranean waters originate from the Atlantic Ocean (P. Dutton, NMFS, unpublished data).

Atlantic Ocean - Distribution and Life History
Evidence from tag returns and strandings in the western Atlantic suggests that adult leatherback sea turtles engage in routine migrations between northern temperate and tropical waters (NMFS and USFWS 1992). Leatherbacks are frequently thought of as a pelagic species that feed on jellyfish (*e.g., Stomolophus, Chryaora*, and *Aurelia* species) and tunicates (*e.g.*, salps, pyrosomas) (Rebel 1974; Davenport and Balazs 1991). However, leatherbacks are also known to use coastal waters of the U.S. continental shelf (James *et al.* 2005a; Eckert *et al.* 2006; Murphy *et al.* 2006), as well as the European continental shelf on a seasonal basis (Witt *et al.* 2007).

Tagging and satellite telemetry data indicate that leatherbacks from the western North Atlantic nesting beaches use the entire North Atlantic Ocean (TEWG 2007). For example, leatherbacks tagged at nesting beaches in Costa Rica have been found in Texas, Florida, South Carolina, Delaware, and New York (STSSN database). Leatherback sea turtles tagged in Puerto Rico, Trinidad, and the Virgin Islands have also been subsequently found on U.S. beaches of southern, Mid-Atlantic, and northern states (STSSN database). Leatherbacks from the South Atlantic nesting assemblages (West Africa, South Africa, and Brazil) have not been re-sighted in the western North Atlantic (TEWG 2007).

The CETAP aerial survey of the outer Continental Shelf from Cape Hatteras, North Carolina to Cape Sable, Nova Scotia conducted between 1978 and 1982 showed leatherbacks to be present throughout the area with the most numerous sightings made from the Gulf of Maine south to Long Island. Leatherbacks were sighted in water depths ranging from 1 to 4,151 m, but 84.4%

of sightings were in waters less than 180 m (Shoop and Kenney 1992). Leatherbacks were sighted in waters within a sea surface temperature range similar to that observed for loggerheads; from 7°-27.2°C (Shoop and Kenney 1992). However, leatherbacks appear to have a greater tolerance for colder waters in comparison to loggerhead sea turtles since more leatherbacks were found at the lower temperatures (Shoop and Kenney 1992). Studies of satellite tagged leatherbacks suggest that they spend 10%-41% of their time at the surface, depending on the phase of their migratory cycle (James *et al.* 2005b). The greatest amount of surface time (up to 41%) was recorded when leatherbacks occurred in continental shelf and slope waters north of 38°N (James *et al.* 2005b).

In 1979, the waters adjacent to Sandy Point, St. Croix, U.S. Virgin Islands were designated as critical habitat for the leatherback sea turtle. On February 2, 2010, NMFS received a petition to revise the critical habitat designation for leatherback sea turtles to include waters adjacent to a major nesting beach in Puerto Rico. NMFS published a 90-day finding on the petition on July 16, 2010, which found that the petition did not present substantial scientific information indicating that the petitioned revision was warranted. The original petitioners submitted a second petition on November 2, 2010 to revise the critical habitat designation to again include waters adjacent to a major nesting beach in Puerto Rico, including additional information on the usage of the waters. NMFS determined on May 5, 2011, that a revision to critical habitat off Puerto Rico may be warranted, and an analysis is underway. Note that on August 4, 2011, FWS issued a determination that revision to critical habitat along Puerto Rico should be made and will be addressed during the future planned status review.

Leatherbacks are a long lived species (>30 years). They were originally believed to mature at a younger age than loggerhead sea turtles, with a previous estimated age at sexual maturity of about 13-14 years for females with 9 years reported as a likely minimum (Zug and Parham 1996) and 19 years as a likely maximum (NMFS SEFSC 2001). However, new sophisticated analyses suggest that leatherbacks in the Northwest Atlantic may reach maturity at 24.5-29 years of age (Avens *et al.* 2009). In the United States and Caribbean, female leatherbacks nest from March through July. In the Atlantic, most nesting females average between 150-160 cm curved carapace length (CCL), although smaller (<145 cm CCL) and larger nesters are observed (Stewart *et al.* 2007, TEWG 2007). They nest frequently (up to seven nests per year) during a nesting season and nest about every 2-3 years. They produce 100 eggs or more in each clutch and can produce 700 eggs or more per nesting season (Schultz 1975). However, a significant portion (up to approximately 30%) of the eggs can be infertile. Therefore, the actual proportion of eggs that can result in hatchlings is less than the total number of eggs produced per season. As is the case with other sea turtle species, leatherback hatchlings enter the water soon after hatching. Based on a review of all sightings of leatherback sea turtles of <145 cm CCL, Eckert (1999) found that leatherback juveniles remain in waters warmer than 26°C until they exceed 100 cm CCL.

Population Dynamics and Status

As described earlier, sea turtle nesting survey data is important in that it provides information on the relative abundance of nesting, and the contribution of each population/subpopulation to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually, and as an indicator of the trend in the number of nesting females in the nesting group. The 5-year review for leatherback sea turtles (NMFS and USFWS 2007b) compiled the most recent information on mean number of leatherback nests per year for each of the seven leatherback populations or groups of populations that were identified by the Leatherback TEWG as occurring within the Atlantic. These are: Florida, North Caribbean, Western Caribbean, Southern Caribbean, West Africa, South Africa, and Brazil (TEWG 2007).

In the United States, the Florida Statewide Nesting Beach Survey program has documented an increase in leatherback nesting numbers from 98 nests in 1988 to between 800 and 900 nests in the early 2000s (NMFS and USFWS 2007b). Stewart *et al.* (2011) evaluated nest counts from 68 Florida beaches over 30 years (1979-2008) and found that nesting increased at all beaches with trends ranging from 3.1%-16.3% per year, with an overall increase of 10.2% per year. An analysis of Florida's index nesting beach sites from 1989-2006 shows a substantial increase in leatherback nesting in Florida during this time, with an annual growth rate of approximately 1.17 (TEWG 2007). The TEWG reports an increasing or stable nesting trend for all of the seven populations or groups of populations with the exception of the Western Caribbean and West Africa. The leatherback rookery along the northern coast of South America in French Guiana and Suriname supports the majority of leatherback nesting in the western Atlantic (TEWG 2007), and represents more than half of total nesting by leatherback sea turtles worldwide (Hilterman and Goverse 2004). Nest numbers in Suriname have shown an increase and the long-term trend for the Suriname and French Guiana nesting group seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests for Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). The TEWG (2007) report indicates that using nest numbers from 1967-2005, a positive population growth rate was found over the 39-year period for French Guinea and Suriname, with a 95% probability that the population was growing. Given the magnitude of leatherback nesting in this area compared to other nest sites, negative impacts in leatherback sea turtles in this area could have profound impacts on the entire species.

The CETAP aerial survey conducted from 1978-1982 estimated the summer leatherback population for the northeastern United States at approximately 300-600 animals (from near Nova Scotia, Canada to Cape Hatteras, North Carolina) (Shoop and Kenney 1992). However, the estimate was based on turtles visible at the surface and does not include those that were below the surface out of view. Therefore, it likely underestimated the leatherback population for the northeastern United States at the time of the survey. Estimates of leatherback abundance of 1,052 turtles (C.V. = 0.38) and 1,174 turtles (C.V. = 0.52) were obtained from surveys conducted from Virginia to the Gulf of St. Lawrence in 1995 and 1998, respectively (Palka 2000). However, since these estimates were also based on sightings of leatherbacks at the surface, the author considered the estimates to be negatively biased and the true abundance of leatherbacks may be 4.27 times higher (Palka 2000).

Threats

The 5-year status review (NMFS and USFWS 2007b) and TEWG (2007) report provide summaries of natural as well as anthropogenic threats to leatherback sea turtles. Of the Atlantic sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear, trap/pot gear in particular. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), their diving and foraging behavior, their distributional overlap with the gear, their possible attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, and perhaps to the lightsticks used to attract target species in longline fisheries. Leatherbacks entangled in fishing gear generally have a reduced ability to feed, dive, surface to breathe, or perform any other behavior essential to survival (Balazs 1985). In addition to drowning from forced submergence, they may be more susceptible to boat strikes if forced to remain at the surface, and entangling lines can constrict blood flow resulting in tissue necrosis. The long-term impacts of entanglement on leatherback health remain unclear. Innis *et al.* (2010) conducted a health evaluation of leatherback sea turtles during direct capture (n=12) and disentanglement (n=7). They found no significant difference in many of the measured health parameters between entangled and directly captured turtles. However, blood parameters, including but not limited to sodium, chloride, and blood urea nitrogen, for entangled turtles showed several key differences that were most likely due to reduced foraging and associated seawater ingestion, as well as a general stress response.

Finkbeiner *et al.* (2011) compiled cumulative sea turtle bycatch information in U.S. fisheries from 1990 through 2007, before and after implementation of bycatch mitigation measures. Information was obtained from peer reviewed publications and NMFS documents (e.g., Biological Opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, occurred annually (since implementation of bycatch mitigation measures). Kemp's ridleys interacted with fisheries most frequently, with the highest level of mean annual mortality (2,700), followed by loggerheads (1,400), greens (300), and leatherbacks (40). The Southeast/Gulf of Mexico shrimp trawl fishery was responsible for the vast majority of U.S. interactions (up to 98%) and mortalities (more than 80%). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations.

Leatherbacks have been documented interacting with longline, trap/pot, trawl, and gillnet fishing gear. For instance, an estimated 6,363 leatherback sea turtles were documented as caught by the U.S. Atlantic tuna and swordfish longline fisheries between 1992-1999 (NMFS SEFSC 2001). Currently, the U.S. tuna and swordfish longline fisheries managed under the HMS FMP are estimated to capture 1,764 leatherbacks (no more than 252 mortalities) for each 3-year period starting in 2007 (NMFS 2004a). In 2010, there were 26 observed interactions between leatherback sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2011a, 2011b). All leatherbacks were released alive, with all gear removed for the majority of captures. While 2010 total estimates are not yet available, in 2009, 285.8 (95% CI: 209.6-389.7) leatherback sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP based on the observed takes (Garrison and Stokes 2010). The 2009 estimate continues a downward trend since 2007 and remains well below the average prior to implementation of gear regulations (Garrison and Stokes 2010). Since the U.S. fleet accounts for only 5%-8% of the longline hooks fished in the Atlantic Ocean, adding up the under-represented observed takes of the other 23 countries actively fishing in the area would likely result in annual take estimates of thousands of leatherbacks over different life stages (NMFS SEFSC 2001). Lewison *et al.* (2004) estimated that 30,000-60,000 leatherbacks were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries, as well as others).

Leatherbacks are susceptible to entanglement in the lines associated with trap/pot gear used in several fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine (Dwyer *et al.* 2002). Additional leatherbacks stranded wrapped in line of unknown origin or with evidence of a past entanglement (Dwyer *et al.* 2002). More recently, from 2002 to 2010, NMFS received 137 reports of sea turtles entangled in vertical lines from Maine to Virginia, with 128 events confirmed (verified by photo documentation or response by a trained responder; NMFS 2008a). Of the 128 confirmed events during this period, 117 events involved leatherbacks. NMFS identified the gear type and fishery for 72 of the 117 confirmed events, which included lobster (42[5]), whelk/conch (15), black sea bass (10), crab (2), and research pot gear (1). A review of leatherback mortality documented by the STSSN in Massachusetts suggests that vessel strikes and entanglement in fixed gear (primarily lobster pots and whelk pots) are the principal sources of this mortality (Dwyer *et al.* 2002).

Leatherback interactions with the U.S. South Atlantic and Gulf of Mexico shrimp fisheries are also known to occur (NMFS 2002). Leatherbacks are likely to encounter shrimp trawls working in the coastal waters off the U.S. Atlantic coast (from Cape Canaveral, Florida through North Carolina) as they make their annual spring migration north. For many years, TEDs that were required for use in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries were less effective for leatherbacks as compared to the smaller, hard-shelled turtle species, because the TED openings were too small to allow leatherbacks to escape. To address this problem, NMFS issued a final rule on February 21, 2003, to amend the TED regulations (68 FR 8456, February 21, 2003). Modifications to the design of TEDs are now required in order to exclude leatherbacks as well as large benthic immature and sexually mature loggerhead and green sea turtles. Given those modifications, Epperly *et al.* (2002) anticipated an average of 80 leatherback mortalities a year in shrimp gear interactions, dropping to an estimate of 26 leatherback mortalities in 2009 due to effort reduction in the Southeast shrimp fishery (Memo from Dr. B. Ponwith, SEFSC, to Dr. R. Crabtree, SERO, January 5, 2011).

Other trawl fisheries are also known to interact with leatherback sea turtles although on a much smaller scale. In October 2001, for example, a NMFS fisheries observer documented the take of a leatherback in a bottom otter trawl fishing for *Loligo* squid off of Delaware. TEDs are not currently required in this fishery. In November 2007, fisheries observers reported the capture of a leatherback sea turtle in bottom otter trawl gear fishing for summer flounder.

Gillnet fisheries operating in the waters of the Mid-Atlantic states are also known to capture, injure, and/or kill leatherbacks when these fisheries and leatherbacks co-occur. Data collected by the NEFSC Fisheries Observer Program from 1994-1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54%-92%. In North Carolina, six additional leatherbacks were reported captured in gillnet sets in the spring (NMFS SEFSC 2001). In addition to these, in September 1995, two dead

[5] One case involved both lobster and whelk/conch gear.

leatherbacks were removed from an 11-inch (28.2-cm) monofilament shark gillnet set in the nearshore waters off of Cape Hatteras (STSSN unpublished data reported in NMFS SEFSC 2001). Lastly, Murray (2009a) reports five observed leatherback captures in Mid-Atlantic sink gillnet fisheries between 1994 and 2008.

Fishing gear interactions can occur throughout the range of leatherbacks. Entanglements occur in Canadian waters where Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland/Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line, and crab pot line. Leatherbacks are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo *et al.* 1994; Graff 1995). Gillnets are one of the suspected causes for the decline in the leatherback sea turtle population in French Guiana (Chevalier *et al.* 1999), and gillnets targeting green and hawksbill sea turtles in the waters of coastal Nicaragua also incidentally catch leatherback sea turtles (Lagueux 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alio-M. 2000). An estimated 1,000 mature female leatherback sea turtles are caught annually in fishing nets off of Trinidad and Tobago with mortality estimated to be between 50%-95% (Eckert and Lien 1999). Many of the sea turtles do not die as a result of drowning, but rather because the fishermen cut them out of their nets (NMFS SEFSC 2001).

Leatherbacks may be more susceptible to marine debris ingestion than other sea turtle species due to the tendency of floating debris to concentrate in convergence zones that juveniles and adults use for feeding (Shoop and Kenney 1992; Lutcavage *et al.* 1997). Investigations of the necropsy results of leatherback sea turtles revealed that a substantial percentage (34% of the 408 leatherback necropsies' recorded between 1885 and 2007) reported plastic within the turtles' stomach contents, and in some cases (8.7% of those cases in which plastic was reported), blockage of the gut was found in a manner that may have caused the mortality (Mrosovsky *et al.* 2009). An increase in reports of plastic ingestion was evident in leatherback necropsies conducted after the late 1960s (Mrosovsky *et al.* 2009). Along the coast of Peru, intestinal contents of 19 of 140 (13%) leatherback carcasses were found to contain plastic bags and film (Fritts 1982). The presence of plastic debris in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items (*e.g.*, jellyfish) and plastic debris (Mrosovsky 1981). Balazs (1985) speculated that plastic objects may resemble food items by their shape, color, size, or even movements as they drift about, and induce a feeding response in leatherbacks.

Global climate change has been identified as a factor that may affect leatherback habitat and biology (NMFS and USFWS 2007b); however, no significant climate change related impacts to leatherback sea turtle populations have been observed to date. Over the long term, climate change related impacts will likely influence biological trajectories in the future on a century scale (Parmesan and Yohe 2003). Changes in marine systems associated with rising water temperatures, changes in ice cover, salinity, oxygen levels and circulation including shifts in ranges and changes in algal, plankton, and fish abundance could affect leatherback prey distribution and abundance. Climate change is expected to expand foraging habitats into higher latitude waters and some concern has been noted that increasing temperatures may increase the female:male sex ratio of hatchlings on some beaches (Morosovsky *et al.* 1984 and Hawkes *et al.* 2007 in NMFS and USFWS 2007b). However, due to the tendency of leatherbacks to have individual nest placement preferences and deposit some clutches in the cooler tide zone of beaches, the effects of long-term climate on sex ratios may be mitigated (Kamel and Mrosovsky 2004 in NMFS and USFWS 2007b). Additional potential effects of climate change on leatherbacks include range expansion and changes in migration routes as increasing ocean temperatures shift range-limiting isotherms north (Robinson *et al.* 2008). Leatherbacks have expanded their range in the Atlantic north by 330 km in the last 17 years as warming has caused the northerly migration of the 15°C sea surface temperature (SST) isotherm, the lower limit of thermal tolerance for leatherbacks (McMahon and Hays 2006). Leatherbacks are speculated to be the best able to cope with climate change of all the sea turtle species due to their wide geographic distribution and relatively weak beach fidelity. Leatherback sea turtles may be most affected by any changes in the distribution of their primary jellyfish prey, which may affect leatherback distribution and foraging behavior (NMFS and USFWS 2007b). Jellyfish populations may increase due to ocean warming and other factors (Brodeur *et al.* 1999; Attrill *et al.* 2007; Richardson *et al.* 2009). However, any increase in jellyfish populations may or may not impact leatherbacks as there is no evidence that any leatherback populations are currently food-limited.

As discussed for loggerheads, increasing temperatures are expected to result in rising sea levels (Titus and Narayanan 1995 in Conant *et al.* 2009), which could result in increased erosion rates along nesting beaches. Sea level rise could result in the inundation of nesting sites and decrease available nesting habitat (Fish *et al.* 2005). This effect would potentially be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents. While there is a reasonable degree of certainty that climate change related effects will be experienced globally (*e.g.*, rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects of climate change on this species are not predictable or quantifiable at this time (Hawkes *et al.* 2009). However, given the likely rate of change associated with climate impacts (*i.e.*, the century scale), it is unlikely that climate related impacts will have a significant effect on the status of leatherback sea turtles in the short-term future.

Summary of Status for Leatherback Sea Turtles

In the Pacific Ocean, the abundance of leatherback sea turtles on nesting beaches has declined dramatically over the past 10 to 20 years. Nesting groups throughout the eastern and western Pacific Ocean have been reduced to a fraction of their former abundance by the combined effects of human activities that have reduced the number of nesting females and reduced the reproductive success of females that manage to nest (for example, egg poaching) (NMFS and USFWS 2007b). No reliable long term trend data for the Indian Ocean populations are currently available. While leatherbacks are known to occur in the Mediterranean Sea, nesting in this region is not known to occur (NMFS and USFWS 2007b).

Nest counts in many areas of the Atlantic Ocean show increasing trends, including for beaches in Suriname and French Guiana which support the majority of leatherback nesting (NMFS and USFWS 2007b). The species as a whole continues to face numerous threats in nesting and marine habitats. As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like pollution and habitat destruction account for an unknown level of other mortality. The long term recovery potential of this species may be further threatened by observed low genetic diversity, even in the largest nesting groups like French Guiana and Suriname (NMFS and USFWS 2007b).

Based on its 5-year status review of the species, NMFS and USFWS (2007b) determined that endangered leatherback sea turtles should not be delisted or reclassified. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPSs should be identified (NMFS and USFWS 2007b).

4.2.11 Status of Atlantic sturgeon

The section below describes the Atlantic sturgeon listing, provides life history information that is relevant to all DPSs of Atlantic sturgeon and then provides information specific to the status of each DPS of Atlantic sturgeon. Below, we also provide a description of which Atlantic sturgeon DPSs likely occur in the action area and provide information on the use of the action area by Atlantic sturgeon.

The Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) is a subspecies of sturgeon distributed along the eastern coast of North America from Hamilton Inlet, Labrador, Canada to Cape Canaveral, Florida, USA (Scott and Scott, 1988; ASSRT, 2007; T. Savoy, CT DEP, pers. comm.). NMFS has delineated U.S. populations of Atlantic sturgeon into five DPSs[6] (77 FR 5880 and 77 FR 5914). These are: the Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs (see Figure 3). The results of genetic studies suggest that natal origin influences the distribution of Atlantic sturgeon in the marine environment (Wirgin and King, 2011). However, genetic data as well as tracking and tagging data demonstrate sturgeon from each DPS and Canada occur throughout the full range of the subspecies. Therefore, sturgeon originating from any of the five DPSs can be affected by threats in the marine, estuarine and riverine environment that occur far from natal spawning rivers.

On February 6, 2012, we published notice in the *Federal Register* that we were listing the New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs as "endangered," and the Gulf of Maine DPS as "threatened" (77 FR 5880 and 77 FR 5914). The effective date of the listings was April 6, 2012. The DPSs do not include Atlantic sturgeon that are spawned in Canadian rivers. Therefore, Canadian spawned fish are not included in the listings.

As described below, individuals originating from the five listed DPSs may occur in the action area. Information general to all Atlantic sturgeon as well as information specific to each of the relevant DPSs, is provided below.

Figure 2. Map Depicting the Boundaries of the five Atlantic sturgeon DPSs

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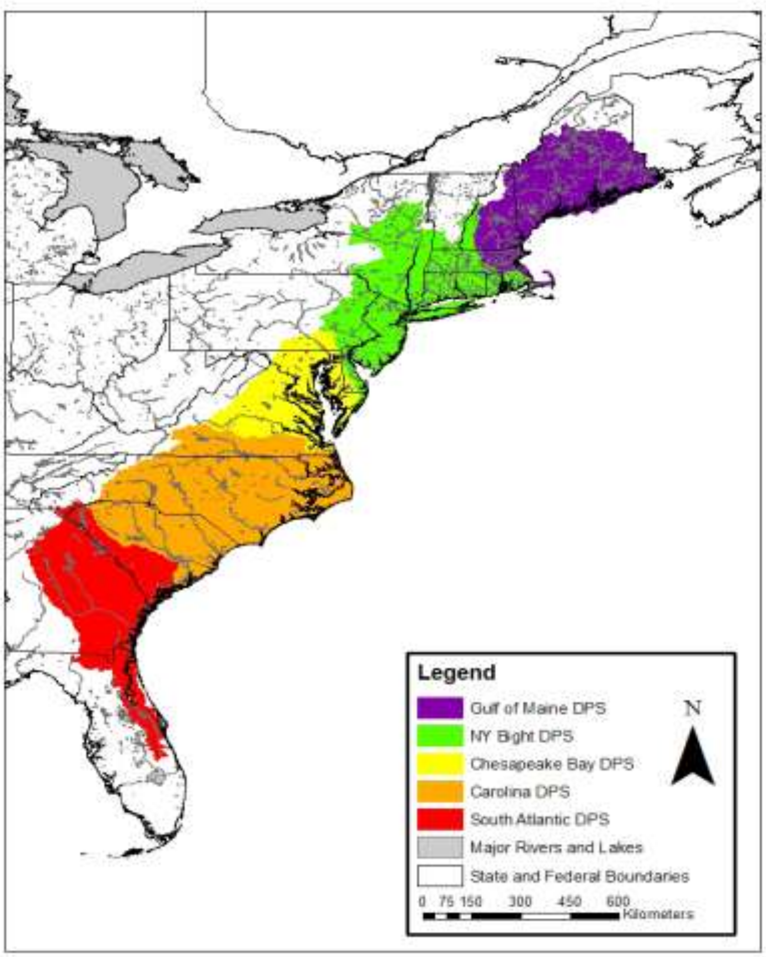
[6] To be considered for listing under the ESA, a group of organisms must constitute a "species." A "species" is defined in section 3 of the ESA to include "any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature."



4.2.10.1 Atlantic sturgeon life history

Atlantic sturgeon are long lived (approximately 60 years), late maturing, estuarine dependent, anadromous[7] fish (Bigelow and Schroeder, 1953; Vladykov and Greeley 1963; Mangin, 1964; Pikitch *et al.*, 2005; Dadswell, 2006; ASSRT, 2007).

The life history of Atlantic sturgeon can be divided up into five general categories as described in the table below (adapted from ASSRT 2007).

Age Class	**Size**	**Description**
Egg		Fertilized or unfertilized

[7] Anadromous refers to a fish that is born in freshwater, spends most of its life in the sea, and returns to freshwater to spawn (NEFSC FAQs, available at http://www.nefsc.noaa.gov/faq/fishfaq1a.html, modified June 16, 2011).

Age Class	Size	Description
Larvae		Negative photo-taxic, nourished by yolk sac
Young of Year (YOY)	**0.3 grams <41 cm TL**	Fish that are > 3 months and < one year; capable of capturing and consuming live food
Sub-adults	**>41 cm and <150 cm TL**	Fish that are at least age 1 and are not sexually mature
Adults	**>150 cm TL**	Sexually mature fish

Table 4. Descriptions of Atlantic sturgeon life history stages.

They are a relatively large fish, even amongst sturgeon species (Pikitch *et al.*, 2005). Atlantic sturgeon are bottom feeders that suck food into a ventrally-located protruding mouth (Bigelow and Schroeder, 1953). Four barbels in front of the mouth assist the sturgeon in locating prey (Bigelow and Schroeder, 1953). Diets of adult and migrant subadult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as sand lance (Bigelow and Schroeder, 1953; ASSRT, 2007; Guilbard *et al.*, 2007; Savoy, 2007). While in the river, Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (Bigelow and Schroeder, 1953; ASSRT, 2007; Guilbard *et al.*, 2007).

Rate of maturation is affected by water temperature and gender. In general: (1) Atlantic sturgeon that originate from southern systems grow faster and mature sooner than Atlantic sturgeon that originate from more northern systems; (2) males grow faster than females; (3) fully mature females attain a larger size (i.e. length) than fully mature males; and (4) the length of Atlantic sturgeon caught since the mid-late 20th century have typically been less than 3 meters (m) (Smith *et al.*, 1982; Smith *et al.*, 1984; Smith, 1985; Scott and Scott, 1988; Young *et al.*, 1998; Collins *et al.*, 2000; Caron *et al.*, 2002; Dadswell, 2006; ASSRT, 2007; Kahnle *et al.*, 2007; DFO, 2011). The largest recorded Atlantic sturgeon was a female captured in 1924 that measured approximately 4.26 m (Vladykov and Greeley, 1963). Dadswell (2006) reported seeing seven fish of comparable size in the St. John River estuary from 1973 to 1995. Observations of large-sized sturgeon are particularly important given that egg production is correlated with age and body size (Smith *et al.*, 1982; Van Eenennaam *et al.*, 1996; Van Eenennaam and Doroshov, 1998; Dadswell, 2006). However, while females are prolific with egg production ranging from 400,000 to 4 million eggs per spawning year, females spawn at intervals of 2-5 years (Vladykov and Greeley, 1963; Smith *et al.*, 1982; Van Eenennaam *et al.*, 1996; Van Eenennaam and

Doroshov, 1998; Stevenson and Secor, 1999; Dadswell, 2006). Given spawning periodicity and a female's relatively late age to maturity, the age at which 50 percent of the maximum lifetime egg production is achieved is estimated to be 29 years (Boreman, 1997). Males exhibit spawning periodicity of 1-5 years (Smith, 1985; Collins *et al.*, 2000; Caron *et al.*, 2002). While long-lived, Atlantic sturgeon are exposed to a multitude of threats prior to achieving maturation and have a limited number of spawning opportunities once mature.

Water temperature plays a primary role in triggering the timing of spawning migrations (ASMFC, 2009). Spawning migrations generally occur during February-March in southern systems, April-May in Mid-Atlantic systems, and May-July in Canadian systems (Murawski and Pacheco, 1977; Smith, 1985; Bain, 1997; Smith and Clugston, 1997; Caron *et al.*, 2002). Male sturgeon begin upstream spawning migrations when waters reach approximately 6° C (43° F) (Smith *et al.*, 1982; Dovel and Berggren, 1983; Smith, 1985; ASMFC, 2009), and remain on the spawning grounds throughout the spawning season (Bain, 1997). Females begin spawning migrations when temperatures are closer to 12° C to 13° C (54° to 55° F) (Dovel and Berggren, 1983; Smith, 1985; Collins *et al.*, 2000), make rapid spawning migrations upstream, and quickly depart following spawning (Bain, 1997).

The spawning areas in most U.S. rivers have not been well defined. However, the habitat characteristics of spawning areas have been identified based on historical accounts of where fisheries occurred, tracking and tagging studies of spawning sturgeon, and physiological needs of early life stages. Spawning is believed to occur in flowing water between the salt front of estuaries and the fall line of large rivers, when and where optimal flows are 46-76 cm/s and depths are 3-27 m (Borodin, 1925; Dees, 1961; Leland, 1968; Scott and Crossman, 1973; Crance, 1987; Shirey *et al.* 1999; Bain *et al.*, 2000; Collins *et al.*, 2000; Caron *et al.* 2002; Hatin *et al.* 2002; ASMFC, 2009). Sturgeon eggs are deposited on hard bottom substrate such as cobble, coarse sand, and bedrock (Dees, 1961; Scott and Crossman, 1973; Gilbert, 1989; Smith and Clugston, 1997; Bain *et al.* 2000; Collins *et al.*, 2000; Caron *et al.*, 2002; Hatin *et al.*, 2002; Mohler, 2003; ASMFC, 2009), and become adhesive shortly after fertilization (Murawski and Pacheco, 1977; Van den Avyle, 1983; Mohler, 2003). Incubation time for the eggs increases as water temperature decreases (Mohler, 2003). At temperatures of 20° and 18° C, hatching occurs approximately 94 and 140 hours, respectively, after egg deposition (ASSRT, 2007).

Larval Atlantic sturgeon (i.e. less than 4 weeks old, with total lengths (TL) less than 30 mm; Van Eenennaam *et al.* 1996) are assumed to undertake a demersal existence and inhabit the same riverine or estuarine areas where they were spawned (Smith *et al.*, 1980; Bain *et al.*, 2000; Kynard and Horgan, 2002; ASMFC, 2009). Studies suggest that age-0 (i.e., young-of-year), age-1, and age-2 Atlantic sturgeon occur in low salinity waters of the natal estuary (Haley, 1999; Hatin *et al.*, 2007; McCord *et al.*, 2007; Munro *et al.*, 2007) while older fish are more salt tolerant and occur in higher salinity waters as well as low salinity waters (Collins *et al.*, 2000). Atlantic sturgeon remain in the natal estuary for months to years before emigrating to open ocean as subadults (Holland and Yelverton, 1973; Dovel and Berggen, 1983; Waldman *et al.*, 1996; Dadswell, 2006; ASSRT, 2007).

After emigration from the natal estuary, subadults and adults travel within the marine environment, typically in waters less than 50 m in depth, using coastal bays, sounds, and ocean waters (Vladykov and Greeley, 1963; Murawski and Pacheco, 1977; Dovel and Berggren, 1983; Smith, 1985; Collins and Smith, 1997; Welsh *et al.*, 2002; Savoy and Pacileo, 2003; Stein *et al.*, 2004; USFWS, 2004; Laney *et al.*, 2007; Dunton *et al.*, 2010; Erickson *et al.*, 2011; Wirgin and King, 2011). Tracking and tagging studies reveal seasonal movements of Atlantic sturgeon along the coast. Satellite-tagged adult sturgeon from the Hudson River concentrated in the southern part of the Mid-Atlantic Bight at depths greater than 20 m during winter and spring, and in the northern portion of the Mid-Atlantic Bight at depths less than 20 m in summer and fall (Erickson *et al.*, 2011). Shirey (Delaware Department of Fish and Wildlife, unpublished data reviewed in ASMFC, 2009) found a similar movement pattern for juvenile Atlantic sturgeon based on recaptures of fish originally tagged in the Delaware River. After leaving the Delaware River estuary during the fall, juvenile Atlantic sturgeon were recaptured by commercial fishermen in nearshore waters along the Atlantic coast as far south as Cape Hatteras, North Carolina from November through early March. In the spring, a portion of the tagged fish re-entered the Delaware River estuary. However, many fish continued a northerly coastal migration through the Mid-Atlantic as well as into southern New England waters where they were recovered throughout the summer months. Movements as far north as Maine were documented. A southerly coastal migration was apparent from tag returns reported in the fall. The majority of these tag returns were reported from relatively shallow near shore fisheries with few fish reported from waters in excess of 25 m (C. Shirey, Delaware Department of Fish and Wildlife, unpublished data reviewed in ASMFC, 2009). Areas where migratory Atlantic sturgeon commonly aggregate include the Bay of Fundy (e.g., Minas and Cumberland Basins), Massachusetts Bay, Connecticut River estuary, Long Island Sound, New York Bight, Delaware Bay, Chesapeake Bay, and waters off of North Carolina from the Virginia/North Carolina border to Cape Hatteras at depths up to 24 m (Dovel and Berggren, 1983; Dadswell *et al.*, 1984; Johnson *et al.*, 1997; Rochard *et al.*, 1997; Kynard *et al.*, 2000; Eyler *et al.*, 2004; Stein *et al.*, 2004; Wehrell, 2005; Dadswell, 2006; ASSRT, 2007; Laney *et al.*, 2007). These sites may be used as foraging sites and/or thermal refuge.

Determination of DPS Composition in the Action Area

As explained above, the range of all five DPSs overlaps and extends from Canada through Cape Canaveral, Florida. The action area is known to be used by Atlantic sturgeon originating from all five DPSs. We have considered the best available information to determine from which DPSs individuals in the action area are likely to have originated. Genetic analysis has been completed on 173 samples obtained through NMFS NEFOP program. These fish have been captured in commercial fishing gear from Maine to North Carolina. Because this sampling overlaps with the action area, we consider it to be the best available information from which to determine the DPS composition in the action area. Based on the mixed-stock analysis resulting from the genetic assignments of the NEFOP samples, we have determined that Atlantic sturgeon in the action area likely originate from the five DPSs at the following frequencies: Canada = 1%
NYB 51%; South Atlantic 22%; Chesapeake Bay 13%; Gulf of Maine 11%; and Carolina 2%. One percent of Atlantic sturgeon in the action area may originate from the St. John's River in Canada; these fish are not included in the 2012 ESA listing. The genetic assignments have a plus/minus 5% confidence interval; however, for purposes of section 7 consultation we have selected the reported values above, which approximate the mid-point of the range, as a

reasonable indication of the likely genetic makeup of Atlantic sturgeon in the action area. These assignments and the data from which they are derived are described in detail in Damon-Randall *et al.* (2013).

4.2.10.2 Distribution and Abundance

Atlantic sturgeon underwent significant range-wide declines from historical abundance levels due to overfishing in the mid to late 19th century when a caviar market was established (Scott and Crossman, 1973; Taub, 1990; Kennebec River Resource Management Plan, 1993; Smith and Clugston, 1997; Dadswell, 2006; ASSRT, 2007). Abundance of spawning-aged females prior to this period of exploitation was predicted to be greater than 100,000 for the Delaware, and at least 10,000 females for other spawning stocks (Secor and Waldman, 1999; Secor, 2002). Historical records suggest that Atlantic sturgeon spawned in at least 35 rivers prior to this period. Currently, only 16 U.S. rivers are known to support spawning based on available evidence (i.e., presence of young-of-year or gravid Atlantic sturgeon documented within the past 15 years) (ASSRT, 2007). While there may be other rivers supporting spawning for which definitive evidence has not been obtained (e.g., in the Penobscot and York Rivers), the number of rivers supporting spawning of Atlantic sturgeon are approximately half of what they were historically. In addition, only four rivers (Kennebec, Hudson, Delaware, James) are known to currently support spawning from Maine through Virginia where historical records support there used to be fifteen spawning rivers (ASSRT, 2007). While spawning may also be occurring in other rivers (e.g., the Androscoggin River in Maine), we do not yet have confirmation of spawning in other Northeast rivers. Thus, there are substantial gaps in the range between Atlantic sturgeon spawning rivers amongst northern and mid-Atlantic states which could make recolonization of extirpated populations more difficult.

There are no current, published population abundance estimates for any of the currently known spawning stocks. Therefore, there are no published abundance estimates for any of the five DPSs of Atlantic sturgeon. An annual mean estimate of 863 mature adults (596 males and 267 females) was calculated for the Hudson River based on fishery-dependent data collected from 1985-1995 (Kahnle *et al.*, 2007). An estimate of 343 spawning adults per year is available for the Altamaha River, GA, based on fishery-independent data collected in 2004 and 2005 (Schueller and Peterson, 2006). Using the data collected from the Hudson River and Altamaha River to estimate the total number of Atlantic sturgeon in either subpopulation is not possible, since mature Atlantic sturgeon may not spawn every year (Vladykov and Greeley, 1963; Smith, 1985; Van Eenennaam *et al.*, 1996; Stevenson and Secor, 1999; Collins *et al.* 2000; Caron *et al.*, 2002), the age structure of these populations is not well understood, and stage to stage survival is unknown. In other words, the information that would allow us to take an estimate of annual spawning adults and expand that estimate to an estimate of the total number of individuals (e.g., yearlings, subadults, and adults) in a population is lacking. The ASSRT presumed that the Hudson and Altamaha rivers had the most robust of the remaining U.S. Atlantic sturgeon spawning populations and concluded that the other U.S. spawning populations were likely less than 300 spawning adults per year (ASSRT, 2007).

Kahnle *et al.* (2007) estimated the number of total mature adults per year in the Hudson River using data from surveys in the 1980s to mid-1990s and based on mean harvest by sex divided by sex specific exploitation rate. While this data is over 20 years old, it is currently the best available data on the abundance of Hudson River origin Atlantic sturgeon. The sex ratio of spawners is estimated to be approximately 70% males and 30% females. As noted above, Kahnle *et al.* (2007) estimated a mean annual number of mature adults at 596 males and 267 females. It is important to note that the authors of this paper have stated that this is an estimate of the annual mean number of Hudson River mature adults during the 1985-1995 period, not an estimate of the number of spawners per year.

4.2.10.3 Threats faced by Atlantic sturgeon throughout their range

Atlantic sturgeon are susceptible to over exploitation given their life history characteristics (e.g., late maturity, dependence on a wide-variety of habitats). Similar to other sturgeon species (Vladykov and Greeley, 1963; Pikitch *et al.*, 2005), Atlantic sturgeon experienced range-wide declines from historical abundance levels due to overfishing (for caviar and meat) and impacts to habitat in the 19^{th} and 20^{th} centuries (Taub, 1990; Smith and Clugston, 1997; Secor and Waldman, 1999).

Based on the best available information, NMFS has concluded that unintended catch of Atlantic sturgeon in fisheries, vessel strikes, poor water quality, water availability, dams, lack of regulatory mechanisms for protecting the fish, and dredging are the most significant threats to Atlantic sturgeon (77 FR 5880 and 77 FR 5914; February 6, 2012). While all of the threats are not necessarily present in the same area at the same time, given that Atlantic sturgeon subadults and adults use ocean waters from the Labrador, Canada to Cape Canaveral, FL, as well as estuaries of large rivers along the U.S. East Coast, activities affecting these water bodies are likely to impact more than one Atlantic sturgeon DPS. In addition, given that Atlantic sturgeon depend on a variety of habitats, every life stage is likely affected by one or more of the identified threats.

An ASMFC interstate fishery management plan for sturgeon (Sturgeon FMP) was developed and implemented in 1990 (Taub, 1990). In 1998, the remaining Atlantic sturgeon fisheries in U.S. state waters were closed per Amendment 1 to the Sturgeon FMP. Complementary regulations were implemented by NMFS in 1999 that prohibit fishing for, harvesting, possessing or retaining Atlantic sturgeon or its parts in or from the Exclusive Economic Zone in the course of a commercial fishing activity.

Commercial fisheries for Atlantic sturgeon still exist in Canadian waters (DFO, 2011). Sturgeon belonging to one or more of the DPSs may be harvested in the Canadian fisheries. In particular, the Bay of Fundy fishery in the Saint John estuary may capture sturgeon of U.S. origin given that sturgeon from the Gulf of Maine and the New York Bight DPSs have been incidentally captured in other Bay of Fundy fisheries (DFO, 2010; Wirgin and King, 2011). Because Atlantic sturgeon are listed under Appendix II of the Convention on International Trade in Endangered Species (CITES), the U.S. and Canada are currently working on a conservation strategy to address the potential for captures of U.S. fish in Canadian directed Atlantic sturgeon fisheries and of Canadian fish incidentally in U.S. commercial fisheries. At this time, there are no estimates of the number of individuals from any of the DPSs that are captured or killed in Canadian fisheries each year.

Based on geographic distribution, most U.S. Atlantic sturgeon that are intercepted in Canadian fisheries are likely to originate from the Gulf of Maine DPS, with a smaller percentage from the New York Bight DPS.

Fisheries bycatch in U.S. waters is the primary threat faced by all 5 DPSs. At this time, we have an estimate of the number of Atlantic sturgeon captured and killed in sink gillnet and otter trawl fisheries authorized by Federal FMPs (NMFS NEFSC 2011b) in the Northeast Region but do not have a similar estimate for Southeast fisheries. We also do not have an estimate of the number of Atlantic sturgeon captured or killed in state fisheries. At this time, we are not able to quantify the effects of other significant threats (e.g., vessel strikes, poor water quality, water availability, dams, and dredging) in terms of habitat impacts or loss of individuals. While we have some information on the number of mortalities that have occurred in the past in association with certain activities (e.g., mortalities in the Delaware and James rivers that are thought to be due to vessel strikes), we are not able to use those numbers to extrapolate effects throughout one or more DPS. This is because of (1) the small number of data points and, (2) lack of information on the percent of incidences that the observed mortalities represent.

As noted above, the NEFSC prepared an estimate of the number of encounters of Atlantic sturgeon in fisheries authorized by Northeast FMPs (NEFSC 2011b). The analysis prepared by the NEFSC estimates that from 2006 through 2010 there were 2,250 to 3,862 encounters per year in observed gillnet and trawl fisheries, with an average of 3,118 encounters. Mortality rates in gillnet gear are approximately 20%. Mortality rates in otter trawl gear are believed to be lower at approximately 5%.

4.2.12 Gulf of Maine DPS of Atlantic sturgeon

The Gulf of Maine DPS includes the following: all anadromous Atlantic sturgeons that are spawned in the watersheds from the Maine/Canadian border and, extending southward, all watersheds draining into the Gulf of Maine as far south as Chatham, MA. Within this range, Atlantic sturgeon historically spawned in the Androscoggin, Kennebec, Merrimack, Penobscot, and Sheepscot Rivers (ASSRT, 2007). Spawning still occurs in the Kennebec River, and it is possible that it still occurs in the Penobscot River as well. Spawning in the Androscoggin River was just recently confirmed by the Maine Department of Marine Resources when they captured a larval Atlantic sturgeon during the 2011 spawning season below the Brunswick Dam; however, the extent of spawning in this river is unknown. There is no evidence of recent spawning in the remaining rivers. In the 1800s, construction of the Essex Dam on the Merrimack River at river kilometer (rkm) 49 blocked access to 58 percent of Atlantic sturgeon habitat in the river (Oakley, 2003; ASSRT, 2007). However, the accessible portions of the Merrimack seem to be suitable habitat for Atlantic sturgeon spawning and rearing (i.e., nursery habitat) (Keiffer and Kynard, 1993). Therefore, the availability of spawning habitat does not appear to be the reason for the lack of observed spawning in the Merrimack River. Studies are on-going to determine whether Atlantic sturgeon are spawning in these rivers. Atlantic sturgeons that are spawned elsewhere continue to use habitats within all of these rivers as part of their overall marine range (ASSRT, 2007). The movement of subadult and adult sturgeon between rivers, including to and from the Kennebec River and the Penobscot River, demonstrates that coastal and marine migrations are key elements of Atlantic sturgeon life history for the Gulf of Maine DPS as well as likely throughout the entire range (ASSRT, 2007; Fernandes, *et al.*, 2010).

Bigelow and Schroeder (1953) surmised that Atlantic sturgeon likely spawned in Gulf of Maine Rivers in May-July. More recent captures of Atlantic sturgeon in spawning condition within the Kennebec River suggest that spawning more likely occurs in June-July (Squiers *et al.*, 1981; ASMFC, 1998; NMFS and USFWS, 1998). Evidence for the timing and location of Atlantic sturgeon spawning in the Kennebec River includes: (1) the capture of five adult male Atlantic sturgeon in spawning condition (i.e., expressing milt) in July 1994 below the (former) Edwards Dam; (2) capture of 31 adult Atlantic sturgeon from June 15,1980, through July 26,1980, in a small commercial fishery directed at Atlantic sturgeon from the South Gardiner area (above Merrymeeting Bay) that included at least 4 ripe males and 1 ripe female captured on July 26,1980; and, (3) capture of nine adults during a gillnet survey conducted from 1977-1981, the majority of which were captured in July in the area from Merrymeeting Bay and upriver as far as Gardiner, ME (NMFS and USFWS, 1998; ASMFC 2007). The low salinity values for waters above Merrymeeting Bay are consistent with values found in other rivers where successful Atlantic sturgeon spawning is known to occur.

Several threats play a role in shaping the current status of Gulf of Maine DPS Atlantic sturgeon. Historical records provide evidence of commercial fisheries for Atlantic sturgeon in the Kennebec and Androscoggin Rivers dating back to the 17th century (Squiers *et al.*, 1979). In 1849, 160 tons of sturgeon was caught in the Kennebec River by local fishermen (Squiers *et al.*, 1979). Following the 1880's, the sturgeon fishery was almost non-existent due to a collapse of the sturgeon stocks. All directed Atlantic sturgeon fishing as well as retention of Atlantic sturgeon by catch has been prohibited since 1998. Nevertheless, mortalities associated with bycatch in fisheries occurring in state and federal waters still occurs. In the marine range, Gulf of Maine DPS Atlantic sturgeon are incidentally captured in federal and state managed fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (Stein *et al.*, 2004; ASMFC 2007). As explained above, we have estimates of the number of subadults and adults that are killed as a result of bycatch in fisheries authorized under Northeast FMPs. At this time, we are not able to quantify the impacts from other threats or estimate the number of individuals killed as a result of other anthropogenic threats. Habitat disturbance and direct mortality from anthropogenic sources are the primary concerns.

Riverine habitat may be impacted by dredging and other in-water activities, disturbing spawning habitat and also altering the benthic forage base. Many rivers in the Gulf of Maine region have navigation channels that are maintained by dredging. Dredging outside of Federal channels and in-water construction occurs throughout the Gulf of Maine region. While some dredging projects operate with observers present to document fish mortalities, many do not. To date we have not received any reports of Atlantic sturgeon killed during dredging projects in the Gulf of Maine region; however, as noted above, not all projects are monitored for interactions with fish. At this time, we do not have any information to quantify the number of Atlantic sturgeon killed or disturbed during dredging or in-water construction projects are also not able to quantify any effects to habitat.

Connectivity is disrupted by the presence of dams on several rivers in the Gulf of Maine region, including the Penobscot and Merrimack Rivers. While there are also dams on the Kennebec, Androscoggin and Saco Rivers, these dams are near the site of natural falls and likely represent the maximum upstream extent of sturgeon occurrence even if the dams were not present. Because no Atlantic sturgeon are known to occur upstream of any hydroelectric projects in the Gulf of Maine region, passage over hydroelectric dams or through hydroelectric turbines is not a source of injury or mortality in this area. While not expected to be killed or injured during passage at a dam, the extent that Atlantic sturgeon are affected by the existence of dams and their operations in the Gulf of Maine region is currently unknown. The documentation of an Atlantic sturgeon larvae downstream of the Brunswick Dam in the Androscoggin River suggests that Atlantic sturgeon spawning may be occurring in the vicinity of at least that project and therefore, may be affected by project operations. The range of Atlantic sturgeon in the Penobscot River is limited by the presence of the Veazie and Great Works Dams. Together these dams prevent Atlantic sturgeon from accessing approximately 29 km of habitat, including the presumed historical spawning habitat located downstream of Milford Falls, the site of the Milford Dam. While removal of the Veazie and Great Works Dams is anticipated to occur in the near future, the presence of these dams is currently preventing access to significant habitats within the Penobscot River. While Atlantic sturgeon are known to occur in the Penobscot River, it is unknown if spawning is currently occurring or whether the presence of the Veazie and Great Works Dams affects the likelihood of spawning occurring in this river. The Essex Dam on the Merrimack River blocks access to approximately 58% of historically accessible habitat in this river. Atlantic sturgeon occur in the Merrimack River but spawning has not been documented. Like the Penobscot, it is unknown how the Essex Dam affects the likelihood of spawning occurring in this river.

Gulf of Maine DPS Atlantic sturgeon may also be affected by degraded water quality. In general, water quality has improved in the Gulf of Maine over the past decades (Lichter *et al.* 2006; EPA, 2008). Many rivers in Maine, including the Androscoggin River, were heavily polluted in the past from industrial discharges from pulp and paper mills. While water quality has improved and most discharges are limited through regulations, many pollutants persist in the benthic environment. This can be particularly problematic if pollutants are present on spawning and nursery grounds as developing eggs and larvae are particularly susceptible to exposure to contaminants.

There are no empirical abundance estimates for the Gulf of Maine DPS. The Atlantic sturgeon SRT (2007) presumed that the Gulf of Maine DPS was comprised of less than 300 spawning adults per year, based on abundance estimates for the Hudson and Altamaha River riverine populations of Atlantic sturgeon. Surveys of the Kennebec River over two time periods, 1977-1981 and 1998-2000, resulted in the capture of nine adult Atlantic sturgeon (Squiers, 2004). However, since the surveys were primarily directed at capture of shortnose sturgeon, the capture gear used may not have been selective for the larger-sized, adult Atlantic sturgeon; several hundred subadult Atlantic sturgeon were caught in the Kennebec River during these studies.

Summary of the Gulf of Maine DPS

Spawning for the Gulf of Maine DPS is known to occur in the Kennebec and recent evidence suggests it may also be occurring in the Androscoggin. Spawning may be occurring in other rivers, such as the Sheepscot or Penobscot, but has not been confirmed. There are indications of increasing abundance of Atlantic sturgeon belonging to the Gulf of Maine DPS. Atlantic sturgeon continue to be present in the Kennebec River; in addition, they are captured in directed research projects in the Penobscot River, and are observed in rivers where they were unknown to occur or had not been observed to occur for many years (e.g., the Saco, Presumpscot, and Charles rivers). These observations suggest that abundance of the Gulf of Maine DPS of Atlantic sturgeon is sufficient such that recolonization to rivers historically suitable for spawning may be occurring. However, despite some positive signs, there is not enough information to establish a trend for this DPS.

Some of the impacts from the threats that contributed to the decline of the Gulf of Maine DPS have been removed (e.g., directed fishing), or reduced as a result of improvements in water quality and removal of dams (e.g., the Edwards Dam on the Kennebec River in 1999). There are strict regulations on the use of fishing gear in Maine state waters that incidentally catch sturgeon. In addition, there have been reductions in fishing effort in state and federal waters, which most likely would result in a reduction in bycatch mortality of Atlantic sturgeon. A significant amount of fishing in the Gulf of Maine is conducted using trawl gear, which is known to have a much lower mortality rate for Atlantic sturgeon caught in the gear compared to sink gillnet gear (ASMFC, 2007). Atlantic sturgeon from the GOM DPS are not commonly taken as bycatch in areas south of Chatham, MA, with only 8 percent (e.g., 7 of the 84 fish) of interactions observed in the Mid Atlantic/Carolina region being assigned to the Gulf of Maine DPS (Wirgin and King, 2011). Tagging results also indicate that Gulf of Maine DPS fish tend to remain within the waters of the Gulf of Maine and only occasionally venture to points south. However, data on Atlantic sturgeon incidentally caught in trawls and intertidal fish weirs fished in the Minas Basin area of the Bay of Fundy(Canada) indicate that approximately 35 percent originated from the Gulf of Maine DPS (Wirgin *et al.*, in draft).

As noted previously, studies have shown that in order to rebuild, Atlantic sturgeon can only sustain low levels of bycatch and other anthropogenic mortality (Boreman, 1997; ASMFC, 2007; Kahnle *et al.*, 2007; Brown and Murphy, 2010). NMFS has determined that the Gulf of Maine DPS is at risk of becoming endangered in the foreseeable future throughout all of its range (i.e., is a threatened species) based on the following: (1) significant declines in population sizes and the protracted period during which sturgeon populations have been depressed; (2) the limited amount of current spawning; and, (3) the impacts and threats that have and will continue to affect recovery.

4.2.13 New York Bight DPS of Atlantic sturgeon

The New York Bight DPS includes the following: all anadromous Atlantic sturgeon spawned in the watersheds that drain into coastal waters from Chatham, MA to the Delaware-Maryland border on Fenwick Island. Within this range, Atlantic sturgeon historically spawned in the Connecticut, Delaware, Hudson, and Taunton Rivers (Murawski and Pacheco, 1977; Secor, 2002; ASSRT, 2007). Spawning still occurs in the Delaware and Hudson Rivers, but there is no recent evidence (within the last 15 years) of spawning in the Connecticut and Taunton Rivers (ASSRT, 2007). Atlantic sturgeon that are spawned elsewhere continue to use habitats within the

Connecticut and Taunton Rivers as part of their overall marine range (ASSRT, 2007; Savoy, 2007; Wirgin and King, 2011).

The abundance of the Hudson River Atlantic sturgeon riverine population prior to the onset of expanded exploitation in the 1800s is unknown but has been conservatively estimated at 10,000 adult females (Secor, 2002). Current abundance is likely at least one order of magnitude smaller than historical levels (Secor, 2002; ASSRT, 2007; Kahnle *et al.*, 2007). As described above, an estimate of the mean annual number of mature adults (863 total; 596 males and 267 females) was calculated for the Hudson River riverine population based on fishery-dependent data collected from 1985-1995 (Kahnle *et al.*, 2007). Kahnle *et al.* (1998; 2007) also showed that the level of fishing mortality from the Hudson River Atlantic sturgeon fishery during the period of 1985-1995 exceeded the estimated sustainable level of fishing mortality for the riverine population and may have led to reduced recruitment. No data on abundance of juveniles are available prior to the 1970s; however, two estimates of immature Atlantic sturgeon have been calculated for the Hudson River population, one for the 1976 year class and one for the 1994 year class. Dovel and Berggren (1983) marked immature fish from 1976-1978. Estimates for the 1976 year class at age were approximately 25,000 individuals. Dovel and Berggren estimated that in 1976 there were approximately 100,000 juvenile (non-migrant) Atlantic sturgeon from approximately 6 year classes, excluding young of year.

In October of 1994, the NYDEC stocked 4,929 marked age-0 Atlantic sturgeon, provided by a USFWS hatchery, into the Hudson Estuary at Newburgh Bay. These fish were reared from Hudson River brood stock. In 1995, Cornell University sampling crews collected 15 stocked and 14 wild age-1 Atlantic sturgeon (Peterson *et al*. 2000). A Petersen mark-recapture population estimate from these data suggests that there were 9,529 (95% CI = 1,916 – 10,473) age-0 Atlantic sturgeon in the estuary in 1994. Since 4,929 were stocked, 4,600 fish were of wild origin, assuming equal survival for both hatchery and wild fish and that stocking mortality for hatchery fish was zero.

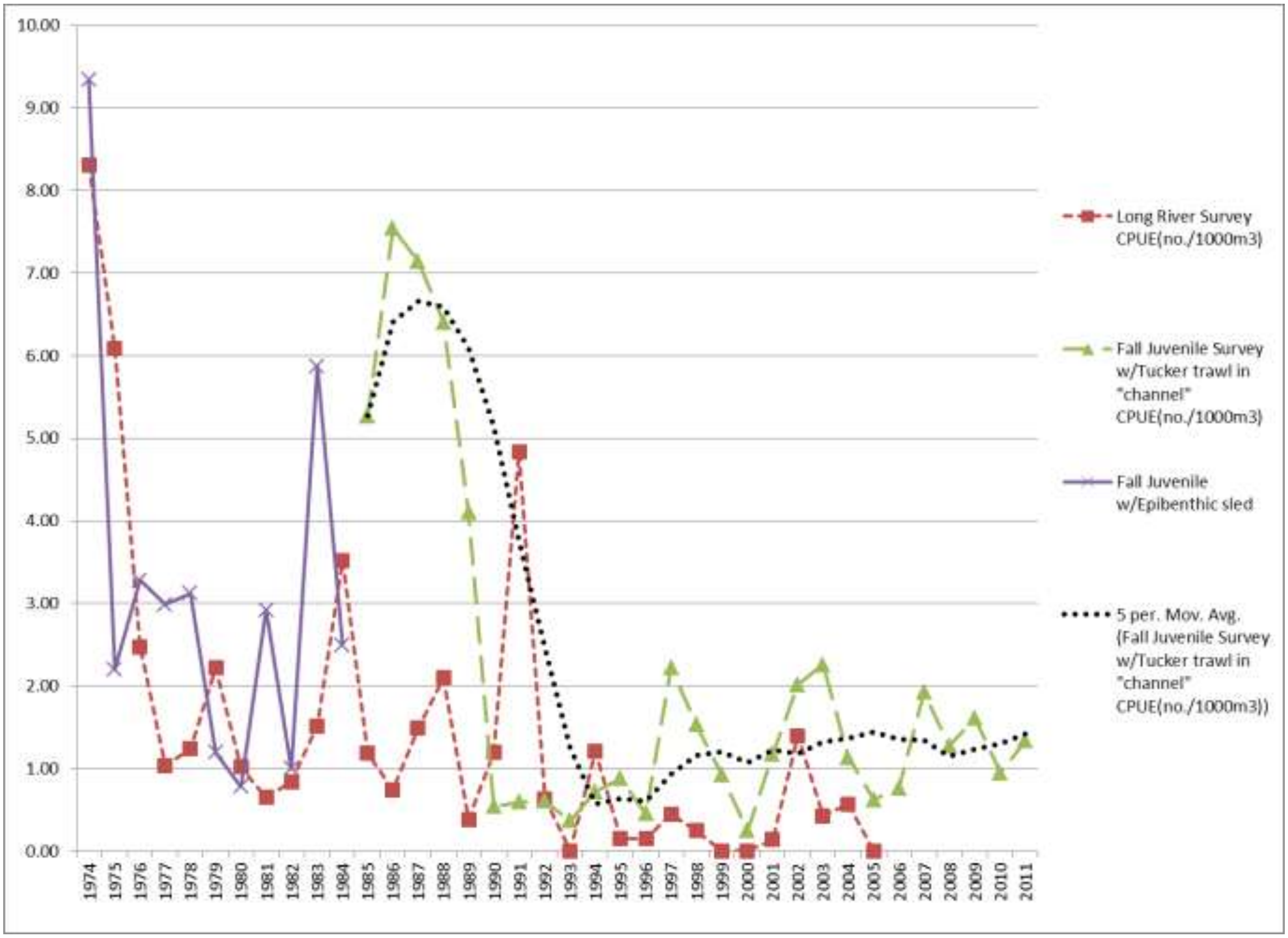
Information on trends for Atlantic sturgeon in the Hudson River are available from a number of long term surveys. From July to November during 1982-1990 and 1993, the NYSDEC sampled the abundance of juvenile fish in Haverstraw Bay and the Tappan Zee Bay. The CPUE of immature Atlantic sturgeon was 0.269 in 1982 and declined to zero by 1990. This study has not been carried out since this time.

The Long River Survey (LRS) samples ichthyoplankton river-wide from the George Washington Bridge (rkm 19) to Troy (rkm 246) using a stratified random design (CONED 1997). These data, which are collected from May-July, provide an annual index of juvenile Atlantic sturgeon in the Hudson River estuary since 1974. The Fall Juvenile Survey (FJS), conducted from July – October by the utilities, calculates an annual index of the number of fish captured per haul. Between 1974 and 1984, the shoals in the entire river (rkm 19-246) were sampled by epibenthic sled; in 1985 the gear was changed to a three-meter beam trawl. While neither of these studies were designed to catch sturgeon, given their consistent implementation over time they provide indications of trends in abundance, particularly over long time series. When examining CPUE, these studies suggest a sharp decline in the number of young Atlantic sturgeon in the early

1990s. While the amount of interannual variability makes it difficult to detect short term trends, a five year running average of CPUE from the FJS indicates a slowly increasing trend since about 1996. Interestingly, that is when the in-river fishery for Atlantic sturgeon closed. While that fishery was not targeting juveniles, a reduction in the number of adult mortalities would be expected to result in increased recruitment and increases in the number of young Atlantic sturgeon in the river. There also could have been bycatch of juveniles that would have suffered some mortality.

In 2000, the NYSDEC created a sturgeon juvenile survey program to supplement the utilities' survey; however, funds were cut in 2000, and the USFWS was contracted in 2003 to continue the program. In 2003 – 2005, 579 juveniles were collected (N = 122, 208, and 289, respectively) (Sweka et al. 2006). Pectoral spine analysis showed they ranged from 1 – 8 years of age, with the majority being ages 2 – 6. There has not been enough data collected to use this information to detect a trend, but at least during the 2003-2005 period, the number of juveniles collected increased each year which could be indicative of an increasing trend for juveniles.

As evidenced by estimates of juvenile abundance, the Atlantic sturgeon population in the Hudson River has declined over time. Peterson et al. (2000) found that the abundance of age-1 Atlantic sturgeon in the Hudson River declined 80% from 1977 to 1995. Similarly, longterm indices of juvenile abundance (the Hudson River Long River and Fall Shoals surveys) demonstrate a longterm declining trend in juvenile abundance. The figure below (Figure 7) illustrates the CPUE of Atlantic sturgeon in the two longterm surveys of the Hudson River. Please note that the Fall Shoals survey switched gear types in 1985. We do not have the CPUE data for the Long River Survey for 2006-2011.



CPUE for the Fall Juvenile Survey for the most recent five year period (2007-2011) is approximately 27% of the CPUE from 1985-1990, but is more than two times higher than the CPUE from 1991-1996 which may be suggestive of an increasing trend in juvenile abundance. Given the high variability between years, it is difficult to use this data to assess short term trends, however, when looking at a five-year moving average, the index appears to be increasing from lows in the early 1990s, but is still much lower than the 1970s and 1980s.

There is no abundance estimate for the Delaware River population of Atlantic sturgeon. Harvest records from the 1800's indicate that this was historically a large population with an estimated 180,000 adult females prior to 1890 (Secor and Waldman, 1999; Secor, 2002). Sampling in 2009 to target young-of- the year (YOY) Atlantic sturgeon in the Delaware River (i.e., natal sturgeon) resulted in the capture of 34 YOY, ranging in size from 178 to 349 mm TL (Fisher, 2009) and the collection of 32 YOY Atlantic sturgeon in a separate study (Brundage and O'Herron in Calvo *et al.*, 2010). Genetics information collected from 33 of the 2009 year class YOY indicates that at least 3 females successfully contributed to the 2009 year class (Fisher, 2011). Therefore, while the capture of YOY in 2009 provides evidence that successful spawning is still occurring in the

Delaware River, the relatively low numbers suggest the existing riverine population is limited in size.

Several threats play a role in shaping the current status and trends observed in the Delaware River and Estuary. In-river threats include habitat disturbance from dredging, and impacts from historical pollution and impaired water quality. A dredged navigation channel extends from Trenton seaward through the tidal river (Brundage and O'Herron, 2009), and the river receives significant shipping traffic. Vessel strikes have been identified as a threat in the Delaware River; however, at this time we do not have information to quantify this threat or its impact to the population or the New York Bight DPS. Similar to the Hudson River, there is currently not enough information to determine a trend for the Delaware River population.

Summary of the New York Bight DPS

Atlantic sturgeon originating from the New York Bight DPS spawn in the Hudson and Delaware rivers. While genetic testing can differentiate between individuals originating from the Hudson or Delaware river the available information suggests that the straying rate is high between these rivers. There are no indications of increasing abundance for the New York Bight DPS (ASSRT, 2009; 2010). Some of the impact from the threats that contributed to the decline of the New York Bight DPS have been removed (e.g., directed fishing) or reduced as a result of improvements in water quality since passage of the Clean Water Act (CWA). In addition, there have been reductions in fishing effort in state and federal waters, which may result in a reduction in bycatch mortality of Atlantic sturgeon. Nevertheless, areas with persistent, degraded water quality, habitat impacts from dredging, continued bycatch in state and federally-managed fisheries, and vessel strikes remain significant threats to the New York Bight DPS.

In the marine range, New York Bight DPS Atlantic sturgeon are incidentally captured in federal and state managed fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (Stein *et al.*, 2004; ASMFC 2007). As explained above, currently available estimates indicate that at least 4% of adults may be killed as a result of bycatch in fisheries authorized under Northeast FMPs. Based on mixed stock analysis results presented by Wirgin and King (2011), over 40 percent of the Atlantic sturgeon bycatch interactions in the Mid Atlantic Bight region were sturgeon from the New York Bight DPS. Individual-based assignment and mixed stock analysis of samples collected from sturgeon captured in Canadian fisheries in the Bay of Fundy indicated that approximately 1-2% were from the New York Bight DPS. At this time, we are not able to quantify the impacts from other threats or estimate the number of individuals killed as a result of other anthropogenic threats.

Riverine habitat may be impacted by dredging and other in-water activities, disturbing spawning habitat and also altering the benthic forage base. Both the Hudson and Delaware rivers have navigation channels that are maintained by dredging. Dredging is also used to maintain channels in the nearshore marine environment. Dredging outside of Federal channels and in-water construction occurs throughout the New York Bight region. While some dredging projects operate with observers present to document fish mortalities many do not. We have reports of one Atlantic sturgeon entrained during hopper dredging operations in Ambrose Channel, New Jersey. At this time, we do not have any information to quantify the number of Atlantic sturgeon killed or disturbed during dredging or in-water construction projects, we are also not able to quantify any effects to habitat.

In the Hudson and Delaware Rivers, dams do not block access to historical habitat. The Holyoke Dam on the Connecticut River blocks further upstream passage; however, the extent that Atlantic sturgeon would historically have used habitat upstream of Holyoke is unknown. Connectivity may be disrupted by the presence of dams on several smaller rivers in the New York Bight region. Because no Atlantic sturgeon occur upstream of any hydroelectric projects in the New York Bight region, passage over hydroelectric dams or through hydroelectric turbines is not a source of injury or mortality in this area. The extent that Atlantic sturgeon are affected by operations of dams in the New York Bight region is currently unknown.

New York Bight DPS Atlantic sturgeon may also be affected by degraded water quality. In general, water quality has improved in the Hudson and Delaware over the past decades (Lichter *et al.* 2006; EPA, 2008). Both the Hudson and Delaware rivers, as well as other rivers in the New York Bight region, were heavily polluted in the past from industrial and sanitary sewer discharges. While water quality has improved and most discharges are limited through regulations, many pollutants persist in the benthic environment. This can be particularly problematic if pollutants are present on spawning and nursery grounds as developing eggs and larvae are particularly susceptible to exposure to contaminants.

Vessel strikes occur in the Delaware River. Twenty-nine mortalities believed to be the result of vessel strikes were documented in the Delaware River from 2004 to 2008, and at least 13 of these fish were large adults. Given the time of year in which the fish were observed (predominantly May through July, with two in August), it is likely that many of the adults were migrating through the river to the spawning grounds. Because we do not know the percent of total vessel strikes that the observed mortalities represent, we are not able to quantify the number of individuals likely killed as a result of vessel strikes in the New York Bight DPS.

Studies have shown that to rebuild, Atlantic sturgeon can only sustain low levels of anthropogenic mortality (Boreman, 1997; ASMFC, 2007; Kahnle *et al.*, 2007; Brown and Murphy, 2010). There are no empirical abundance estimates of the number of Atlantic sturgeon in the New York Bight DPS. As described in the final listing rule, NMFS has determined that the New York Bight DPS is currently at risk of extinction due to: (1) precipitous declines in population sizes and the protracted period in which sturgeon populations have been depressed; (2) the limited amount of current spawning; and (3) the impacts and threats that have and will continue to affect population recovery.

4.2.14 Chesapeake Bay DPS of Atlantic sturgeon

The Chesapeake Bay DPS includes the following: all anadromous Atlantic sturgeons that are spawned in the watersheds that drain into the Chesapeake Bay and into coastal waters from the Delaware-Maryland border on Fenwick Island to Cape Henry, VA. Within this range, Atlantic sturgeon historically spawned in the Susquehanna, Potomac, James, York, Rappahannock, and Nottoway Rivers (ASSRT, 2007). Based on the review by Oakley (2003), 100 percent of Atlantic sturgeon habitat is currently accessible in these rivers since most of the barriers to passage (i.e. dams) are located upriver of where spawning is expected to have historically occurred (ASSRT, 2007). Spawning still occurs in the James River, and the presence of juvenile and adult sturgeon in the York River suggests that spawning may occur there as well (Musick *et al.*, 1994; ASSRT, 2007; Greene, 2009). However, conclusive evidence of current spawning is only available for the James River. Atlantic sturgeon that are spawned elsewhere are known to use the Chesapeake Bay for other life functions, such as foraging and as juvenile nursery habitat prior to entering the marine system as subadults (Vladykov and Greeley, 1963; ASSRT, 2007; Wirgin *et al.*, 2007; Grunwald *et al.*, 2008).

Age to maturity for Chesapeake Bay DPS Atlantic sturgeon is unknown. However, Atlantic sturgeon riverine populations exhibit clinal variation with faster growth and earlier age to maturity for those that originate from southern waters, and slower growth and later age to maturity for those that originate from northern waters (75 FR 61872; October 6, 2010). Age at maturity is 5 to 19 years for Atlantic sturgeon originating from South Carolina rivers (Smith *et al.*, 1982) and 11 to 21 years for Atlantic sturgeon originating from the Hudson River (Young *et al.*, 1998). Therefore, age at maturity for Atlantic sturgeon of the Chesapeake Bay DPS likely falls within these values.

Several threats play a role in shaping the current status of Chesapeake Bay DPS Atlantic sturgeon. Historical records provide evidence of the large-scale commercial exploitation of Atlantic sturgeon from the James River and Chesapeake Bay in the 19th century (Hildebrand and Schroeder, 1928; Vladykov and Greeley, 1963; ASMFC, 1998; Secor, 2002; Bushnoe *et al.*, 2005; ASSRT, 2007) as well as subsistence fishing and attempts at commercial fisheries as early as the 17th century (Secor, 2002; Bushnoe *et al.*, 2005; ASSRT, 2007; Balazik *et al.*, 2010). Habitat disturbance caused by in-river work such as dredging for navigational purposes is thought to have reduced available spawning habitat in the James River (Holton and Walsh, 1995; Bushnoe *et al.*, 2005; ASSRT, 2007). At this time, we do not have information to quantify this loss of spawning habitat.

Decreased water quality also threatens Atlantic sturgeon of the Chesapeake Bay DPS, especially since the Chesapeake Bay system is vulnerable to the effects of nutrient enrichment due to a relatively low tidal exchange and flushing rate, large surface to volume ratio, and strong stratification during the spring and summer months (Pyzik *et al.*, 2004; ASMFC, 1998; ASSRT, 2007; EPA, 2008). These conditions contribute to reductions in dissolved oxygen levels throughout the Bay. The availability of nursery habitat, in particular, may be limited given the recurrent hypoxia (low dissolved oxygen) conditions within the Bay (Niklitschek and Secor, 2005; 2010). At this time we do not have sufficient information to quantify the extent that degraded water quality effects habitat or individuals in the James River or throughout the Chesapeake Bay.

Vessel strikes have been observed in the James River (ASSRT, 2007). Eleven Atlantic sturgeon were reported to have been struck by vessels from 2005 through 2007. Several of these were mature individuals. Because we do not know the percent of total vessel strikes that the observed mortalities represent, we are not able to quantify the number of individuals likely killed as a result of vessel strikes in the New York Bight DPS.

In the marine and coastal range of the Chesapeake Bay DPS from Canada to Florida, fisheries bycatch in federally and state managed fisheries poses a threat to the DPS, reducing survivorship of subadults and adults and potentially causing an overall reduction in the spawning population (Stein *et al.*, 2004; ASMFC, 2007; ASSRT, 2007).

Summary of the Chesapeake Bay DPS

Spawning for the Chesapeake Bay DPS is known to occur in only the James River. Spawning may be occurring in other rivers, such as the York, but has not been confirmed. There are anecdotal reports of increased sightings and captures of Atlantic sturgeon in the James River. However, this information has not been comprehensive enough to develop a population estimate for the James River or to provide sufficient evidence to confirm increased abundance. Some of the impact from the threats that facilitated the decline of the Chesapeake Bay DPS have been removed (e.g., directed fishing) or reduced as a result of improvements in water quality since passage of the Clean Water Act (CWA). We do not currently have enough information about any life stage to establish a trend for this DPS.

Areas with persistent, degraded water quality, habitat impacts from dredging, continued bycatch in U.S. state and federally-managed fisheries, Canadian fisheries and vessel strikes remain significant threats to the Chesapeake Bay DPS of Atlantic sturgeon. Studies have shown that Atlantic sturgeon can only sustain low levels of bycatch mortality (Boreman, 1997; ASMFC, 2007; Kahnle *et al.*, 2007). The Chesapeake Bay DPS is currently at risk of extinction given (1) precipitous declines in population sizes and the protracted period in which sturgeon populations have been depressed; (2) the limited amount of current spawning; and, (3) the impacts and threats that have and will continue to affect the potential for population recovery.

4.2.15 Carolina DPS of Atlantic sturgeon

The Carolina DPS includes all Atlantic sturgeon that spawn or are spawned in the watersheds (including all rivers and tributaries) from Albemarle Sound southward along the southern Virginia, North Carolina, and South Carolina coastal areas to Charleston Harbor. The marine range of Atlantic sturgeon from the Carolina DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida. Sturgeon are commonly captured 40 miles offshore (D. Fox, DSU, pers. comm.). Records providing fishery bycatch data by depth show the vast majority of Atlantic sturgeon bycatch via gillnets is observed in waters less than 50 meters deep (Stein et al. 2004, ASMFC 2007), but Atlantic sturgeon are recorded as bycatch out to 500 fathoms.

Rivers known to have current spawning populations within the range of the Carolina DPS include the Roanoke, Tar-Pamlico, Cape Fear, Waccamaw, and Pee Dee Rivers. We determined spawning was occurring if young-of-the-year (YOY) were observed, or mature adults were present, in freshwater portions of a system (Table 5). However, in some rivers, spawning by Atlantic sturgeon may not be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development. There may also be spawning populations in the Neuse, Santee and Cooper Rivers, though it is uncertain. Historically, both the Sampit and Ashley Rivers were documented to have spawning populations at one time. However, the spawning population in the Sampit River is believed to be extirpated and the current status of the spawning population in the Ashley River is unknown. Both rivers may be used as nursery habitat by young Atlantic sturgeon originating from other spawning populations. This represents our current knowledge of the river systems utilized by the Carolina DPS for specific life functions, such as spawning, nursery habitat, and foraging. However, fish from the Carolina DPS likely use other river systems than those listed here for their specific life functions.

River/Estuary	**Spawning Population**	**Data**
Roanoke River, VA/NC; Albemarle Sound, NC	Yes	collection of 15 YOY (1997-1998); single YOY (2005)
Tar-Pamlico River, NC; Pamlico Sound	Yes	one YOY (2005)
Neuse River, NC; Pamlico Sound	Unknown	
Cape Fear River, NC	Yes	upstream migration of adults in the fall, carcass of a ripe female upstream in mid-September (2006)
Waccamaw River, SC; Winyah Bay	Yes	age-1, potentially YOY (1980s)
Pee Dee River, SC; Winyah Bay	Yes	running ripe male in Great Pee Dee River (2003)
Sampit, SC; Winyah Bay	Extirpated	
Santee River, SC	Unknown	
Cooper River, SC	Unknown	
Ashley River, SC	Unknown	

Table 5. Major rivers, tributaries, and sounds within the range of the Carolina DPS and currently available data on the presence of an Atlantic sturgeon spawning population in each system.

The riverine spawning habitat of the Carolina DPS occurs within the Mid-Atlantic Coastal Plain ecoregion (TNC 2002a), which includes bottomland hardwood forests, swamps, and some of the world's most active coastal dunes, sounds, and estuaries. Natural fires, floods, and storms are so dominant in this region that the landscape changes very quickly. Rivers routinely change their courses and emerge from their banks. The primary threats to biological diversity in the Mid-Atlantic Coastal Plain, as listed by TNC are: global climate change and rising sea level; altered surface hydrology and landform alteration (e.g., flood-control and hydroelectric dams, inter-basin transfers of water, drainage ditches, breached levees, artificial levees, dredged inlets and river channels, beach renourishment, and spoil deposition banks and piles); a regionally receding water table, probably resulting from both over-use and inadequate recharge; fire suppression; land fragmentation, mainly by highway development; land-use conversion (e.g., from forests to timber plantations, farms, golf courses, housing developments, and resorts); the invasion of exotic plants and animals; air and water pollution, mainly from agricultural activities including concentrated animal feed operations; and over-harvesting and poaching of species. Many of the Carolina DPS' spawning rivers, located in the Mid-Coastal Plain, originate in areas of marl. Waters draining calcareous, impervious surface materials such as marl are: (1) likely to be alkaline; (2) dominated by surface run-off; (3) have little groundwater connection; and, (4) are seasonally ephemeral.

Historical landings data indicate that between 7,000 and 10,500 adult female Atlantic sturgeon were present in North Carolina prior to 1890 (Armstrong and Hightower 2002, Secor 2002). Secor (2002) estimates that 8,000 adult females were present in South Carolina during that same time-frame. Reductions from the commercial fishery and ongoing threats have drastically reduced the numbers of Atlantic sturgeon within the Carolina DPS. Currently, the Atlantic sturgeon spawning population in at least one river system within the Carolina DPS has been extirpated, with a potential extirpation in an additional system. The abundances of the remaining river populations within the DPS, each estimated to have fewer than 300 spawning adults, is estimated to be less than 3 percent of what they were historically (ASSRT 2007).

Threats

The Carolina DPS was listed as endangered under the ESA as a result of a combination of habitat curtailment and modification, overutilization (i.e, being taken as bycatch) in commercial fisheries, and the inadequacy of regulatory mechanisms in ameliorating these impacts and threats.

The modification and curtailment of Atlantic sturgeon habitat resulting from dams, dredging, and degraded water quality is contributing to the status of the Carolina DPS. Dams have curtailed Atlantic sturgeon spawning and juvenile developmental habitat by blocking over 60 percent of the historical sturgeon habitat upstream of the dams in the Cape Fear and Santee-Cooper River systems. Water quality (velocity, temperature, and dissolved oxygen (DO)) downstream of these dams, as well as on the Roanoke River, has been reduced, which modifies and curtails the extent of spawning and nursery habitat for the Carolina DPS. Dredging in spawning and nursery grounds modifies the quality of the habitat and is further curtailing the extent of available habitat in the Cape Fear and Cooper Rivers, where Atlantic sturgeon habitat has already been modified and curtailed by the presence of dams. Reductions in water quality from terrestrial activities have modified habitat utilized by the Carolina DPS. In the Pamlico and Neuse systems, nutrient-loading and seasonal anoxia are occurring, associated in part with concentrated animal feeding operations (CAFOs). Heavy industrial development and CAFOs have degraded water quality in the Cape Fear River. Water quality in the Waccamaw and Pee Dee rivers have been affected by industrialization and riverine sediment samples contain high levels of various toxins, including dioxins. Additional stressors arising from water allocation and climate change threaten to exacerbate water quality problems that are already present throughout the range of the Carolina DPS. Twenty interbasin water transfers in existence prior to 1993, averaging 66.5 million gallons per day (mgd), were authorized at their maximum levels without being subjected to an evaluation for certification by North Carolina Department of Environmental and Natural Resources or other resource agencies. Since the 1993 legislation requiring certificates for transfers, almost 170 mgd of interbasin water withdrawals have been authorized, with an additional 60 mgd pending certification. The removal of large amounts of water from the system will alter flows, temperature, and DO. Existing water allocation issues will likely be compounded by population growth and potentially climate change. Climate change is also predicted to elevate water temperatures and exacerbate nutrient-loading, pollution inputs, and lower DO, all of which are current stressors to the Carolina DPS.

Overutilization of Atlantic sturgeon from directed fishing caused initial severe declines in Atlantic sturgeon populations in the Southeast, from which they have never rebounded. Further, continued overutilization of Atlantic sturgeon as bycatch in commercial fisheries is an ongoing impact to the Carolina DPS. Little data exists on bycatch in the Southeast and high levels of bycatch underreporting are suspected. Further, a total population abundance for the DPS is not available, and it is therefore not possible to calculate the percentage of the DPS subject to bycatch mortality based on the available bycatch mortality rates for individual fisheries. However, fisheries known to incidentally catch Atlantic sturgeon occur throughout the marine range of the species and in some riverine waters as well. Because Atlantic sturgeon mix extensively in marine waters and may access multiple river systems, they are subject to being caught in multiple fisheries throughout their range. In addition, stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (e.g., exposure to toxins and low DO). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even post-capture mortality.

As a wide-ranging anadromous species, Carolina DPS Atlantic sturgeon are subject to numerous Federal (U.S. and Canadian), state and provincial, and inter-jurisdictional laws, regulations, and agency activities. While these mechanisms have addressed impacts to Atlantic sturgeon through directed fisheries, there are currently no mechanisms in place to address the significant risk posed to Atlantic sturgeon from commercial bycatch. Though statutory and regulatory mechanisms exist that authorize reducing the impact of dams on riverine and anadromous species, such as Atlantic sturgeon, and their habitat, these mechanisms have proven inadequate for preventing dams from blocking access to habitat upstream and degrading habitat downstream. Further, water quality continues to be a problem in the Carolina DPS, even with existing controls on some pollution sources. Current regulatory regimes are not necessarily effective in controlling water allocation issues (e.g., no restrictions on interbasin water transfers in South Carolina, the lack of ability to regulate non-point source pollution, etc.)

The recovery of Atlantic sturgeon along the Atlantic Coast, especially in areas where habitat is limited and water quality is severely degraded, will require improvements in the following areas: (1) elimination of barriers to spawning habitat either through dam removal, breaching, or installation of successful fish passage facilities; (2) operation of water control structures to provide appropriate flows, especially during spawning season; (3) imposition of dredging restrictions including seasonal moratoriums and avoidance of spawning/nursery habitat; and, (4) mitigation of water quality parameters that are restricting sturgeon use of a river (i.e., DO). Additional data regarding sturgeon use of riverine and estuarine environments is needed.

The concept of a viable population able to adapt to changing environmental conditions is critical to Atlantic sturgeon, and the low population numbers of every river population in the Carolina DPS put them in danger of extinction throughout their range; none of the populations are large or stable enough to provide with any level of certainty for continued existence of Atlantic sturgeon in this part of its range. Although the largest impact that caused the precipitous decline of the species has been curtailed (directed fishing), the population sizes within the Carolina DPS have remained relatively constant at greatly reduced levels (approximately 3 percent of historical population sizes) for 100 years. Small numbers of individuals resulting from drastic reductions in populations, such as occurred with Atlantic sturgeon due to the commercial fishery, can remove the buffer against natural demographic and environmental variability provided by large populations (Berry, 1971; Shaffer, 1981; Soulé, 1980). Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon, and they continue to face a variety of other threats that contribute to their risk of extinction. While a long life-span also allows multiple opportunities to contribute to future generations, it also results increases the timeframe over which exposure to the multitude of threats facing the Carolina DPS can occur.

The viability of the Carolina DPS depends on having multiple self-sustaining riverine spawning populations and maintaining suitable habitat to support the various life functions (spawning, feeding, growth) of Atlantic sturgeon populations. Because a DPS is a group of populations, the stability, viability, and persistence of individual populations affects the persistence and viability of the larger DPS. The loss of any population within a DPS will result in: (1) a long-term gap in the range of the DPS that is unlikely to be recolonized; (2) loss of reproducing individuals; (3) loss of genetic biodiversity; (4) potential loss of unique haplotypes; (5) potential loss of adaptive traits; and (6) reduction in total number. The loss of a population will negatively impact the persistence and viability of the DPS as a whole, as fewer than two individuals per generation spawn outside their natal rivers (Secor and Waldman 1999). The persistence of individual populations, and in turn the DPS, depends on successful spawning and rearing within the freshwater habitat, the immigration into marine habitats to grow, and then the return of adults to natal rivers to spawn.

Summary of the Status of the Carolina DPS of Atlantic Sturgeon

In summary, the Carolina DPS is estimated to number less than 3 percent of its historic population size. There are estimated to be less than 300 spawning adults per year (total of both sexes) in each of the major river systems occupied by the DPS in which spawning still occurs, whose freshwater range occurs in the watersheds (including all rivers and tributaries) from Albemarle Sound southward along the southern Virginia, North Carolina, and South Carolina coastal areas to Charleston Harbor. Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon. While a long life-span also allows multiple opportunities to contribute to future generations, this is hampered within the Carolina DPS by habitat alteration and bycatch. This DPS was severely depleted by past directed commercial fishing, and faces ongoing impacts and threats from habitat alteration or inaccessibility, bycatch, and the inadequacy of existing regulatory mechanisms to address and reduce habitat alterations and bycatch that have prevented river populations from rebounding and will prevent their recovery.

The presence of dams has resulted in the loss of over 60 percent of the historical sturgeon habitat on the Cape Fear River and in the Santee-Cooper system. Dams are contributing to the status of the Carolina DPS by curtailing the extent of available spawning habitat and further modifying the remaining habitat downstream by affecting water quality parameters (such as depth, temperature, velocity, and DO) that are important to sturgeon. Dredging is also contributing to the status of the Carolina DPS by modifying Atlantic sturgeon spawning and nursery habitat. Habitat modifications through reductions in water quality are contributing to the status of the Carolina DPS due to nutrient-loading, seasonal anoxia, and contaminated sediments. Interbasin water transfers and climate change threaten to exacerbate existing water quality issues. Bycatch is also a current threat to the Carolina DPS that is contributing to its status. Fisheries known to incidentally catch Atlantic sturgeon occur throughout the marine range of the species and in some riverine waters as well. Because Atlantic sturgeon mix extensively in marine waters and may utilize multiple river systems for nursery and foraging habitat in addition to their natal spawning river, they are subject to being caught in multiple fisheries throughout their range. In addition to direct mortality, stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (e.g., exposure to toxins). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even post-capture mortality. While many of the threats to the Carolina DPS have been ameliorated or reduced due to the existing regulatory mechanisms, such as the moratorium on directed fisheries for Atlantic sturgeon, bycatch is currently not being addressed through existing mechanisms. Further, access to habitat and water quality continues to be a problem even with NMFS' authority under the Federal Power Act to recommend fish passsage and existing controls on some pollution sources. The inadequacy of regulatory mechanisms to control bycatch and habitat alterations is contributing to the status of the Carolina DPS.

4.2.16 South Atlantic DPS of Atlantic sturgeon

The South Atlantic DPS includes all Atlantic sturgeon that spawn or are spawned in the watersheds (including all rivers and tributaries) of the Ashepoo, Combahee, and Edisto Rivers (ACE) Basin southward along the South Carolina, Georgia, and Florida coastal areas to the St. Johns River, Florida. The marine range of Atlantic sturgeon from the South Atlantic DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida.

Rivers known to have current spawning populations within the range of the South Atlantic DPS include the Combahee, Edisto, Savannah, Ogeechee, Altamaha, and Satilla Rivers. We determined spawning was occurring if young-of-the-year (YOY) were observed, or mature adults were present, in freshwater portions of a system (Table 6). However, in some rivers, spawning by Atlantic sturgeon may not be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development. Historically, both the Broad-Coosawatchie and St. Marys Rivers were documented to have spawning populations at one time; there is also evidence that spawning may have occurred in the St. Johns River or one of its tributaries. However, the spawning population in the St. Marys River, as well as any historical spawning population present in the St. Johns, is believed to be extirpated, and the status of the spawning population in the Broad-Coosawatchie is unknown. Both the St. Marys and St. Johns Rivers are used as nursery habitat by young Atlantic sturgeon originating from other spawning populations. The use of the Broad-Coosawatchie by sturgeon from other spawning populations is unknown at this time. The presence of historical and current spawning populations in the Ashepoo River has not been documented; however, this river may currently be used for nursery habitat by young Atlantic sturgeon originating from other spawning populations. This represents our current knowledge of the river systems utilized by the South Atlantic DPS for specific life functions, such as spawning, nursery habitat, and foraging. However, fish from the South Atlantic DPS likely use other river systems than those listed here for their specific life functions.

River/Estuary	**Spawning Population**	**Data**
ACE (Ashepoo, Combahee, and Edisto Rivers) Basin, SC; St. Helena Sound	Yes	1,331 YOY (1994-2001); gravid female and running ripe male in the Edisto (1997); 39 spawning adults (1998)
Broad-Coosawhatchie Rivers, SC; Port Royal Sound	Unknown	
Savannah River, SC/GA	Yes	22 YOY (1999-2006); running ripe male (1997)
Ogeechee River, GA	Yes	age-1 captures, but high inter-annual variability (1991-1998); 17 YOY (2003); 9 YOY (2004)
Altamaha River, GA	Yes	74 captured/308 estimated spawning adults (2004); 139 captured/378 estimated spawning adults (2005)
Satilla River, GA	Yes	4 YOY and spawning adults (1995-1996)
St. Marys River, GA/FL	Extirpated	
St. Johns River, FL	Extirpated	

Table 6. Major rivers, tributaries, and sounds within the range of the South Atlantic DPS and currently available data on the presence of an Atlantic sturgeon spawning population in each system.

The riverine spawning habitat of the South Atlantic DPS occurs within the South Atlantic Coastal Plain ecoregion (TNC 2002b), which includes fall-line sandhills, rolling longleaf pine uplands, wet pine flatwoods, isolated depression wetlands, small streams, large river systems, and estuaries. Other ecological systems in the ecoregion include maritime forests on barrier islands, pitcher plant seepage bogs and Altamaha grit (sandstone) outcrops. Other ecological systems in the ecoregion include maritime forests on barrier islands, pitcher plant seepage bogs and Altamaha grit (sandstone) outcrops. The primary threats to biological diversity in the South Atlantic Coastal Plain listed by TNC are intensive silvicultural practices, including conversion of natural forests to highly managed pine monocultures and the clear-cutting of bottomland hardwood forests. Changes in water quality and quantity, caused by hydrologic alterations (impoundments, groundwater withdrawal, and ditching), and point and nonpoint pollution, are threatening the aquatic systems. Development is a growing threat, especially in coastal areas. Agricultural conversion, fire regime alteration, and the introduction of nonnative species are additional threats to the ecoregion's diversity. The South Atlantic DPS' spawning rivers, located in the South Atlantic Coastal Plain, are primarily of two types: brownwater (with headwaters north of the Fall Line, silt-laden) and blackwater (with headwaters in the coastal plain, stained by tannic acids).

Secor (2002) estimates that 8,000 adult females were present in South Carolina prior to 1890. Prior to the collapse of the fishery in the late 1800s, the sturgeon fishery was the third largest fishery in Georgia. Secor (2002) estimated from U.S. Fish Commission landing reports that approximately 11,000 spawning females were likely present in the state prior to 1890. Reductions from the commercial fishery and ongoing threats have drastically reduced the numbers of Atlantic sturgeon within the South Atlantic DPS. Currently, the Atlantic sturgeon spawning population in at least two river systems within the South Atlantic DPS has been extirpated. The Altamaha River population of Atlantic sturgeon, with an estimated 343 adults spawning annually, is believed to be the largest population in the Southeast, yet is estimated to be only a small percent of its historical population size. The abundances of the remaining river populations within the DPS, each estimated to have fewer than 300 spawning adults, is estimated to be a small percent of what they were historically (ASSRT 2007).

Threats

The South Atlantic DPS was listed as endangered under the ESA as a result of a combination of habitat curtailment and modification, overutilization (i.e, being taken as bycatch) in commercial fisheries, and the inadequacy of regulatory mechanisms in ameliorating these impacts and threats.

The modification and curtailment of Atlantic sturgeon habitat resulting from dredging and degraded water quality is contributing to the status of the South Atlantic DPS. Dredging is a present threat to the South Atlantic DPS and is contributing to their status by modifying the quality and availability of Atlantic sturgeon habitat. Maintenance dredging is currently modifying Atlantic sturgeon nursery habitat in the Savannah River and modeling indicates that the proposed deepening of the navigation channel will result in reduced DO and upriver movement of the salt wedge, curtailing spawning habitat. Dredging is also modifying nursery and foraging habitat in the St. Johns Rivers. Reductions in water quality from terrestrial activities have modified habitat utilized by the South Atlantic DPS. Low DO is modifying sturgeon habitat in the Savannah due to dredging, and non-point source inputs are causing low DO in the Ogeechee River and in the St. Marys River, which completely eliminates juvenile nursery habitat in summer. Low DO has also been observed in the St. Johns River in the summer. Sturgeon are more sensitive to low DO and the negative (metabolic, growth, and feeding) effects caused by low DO increase when water temperatures are concurrently high, as they are within the range of the South Atlantic DPS. Additional stressors arising from water allocation and climate change threaten to exacerbate water quality problems that are already present throughout the range of the South Atlantic DPS. Large withdrawals of over 240 million gallons per day mgd of water occur in the Savannah River for power generation and municipal uses. However, users withdrawing less than 100,000 gallons per day (gpd) are not required to get permits, so actual water withdrawals from the Savannah and other rivers within the range of the South Atlantic DPS are likely much higher. The removal of large amounts of water from the system will alter flows, temperature, and DO. Water shortages and "water wars" are already occurring in the rivers occupied by the South Atlantic DPS and will likely be compounded in the future by population growth and potentially by climate change. Climate change is also predicted to elevate water temperatures and exacerbate nutrient-loading, pollution inputs, and lower DO, all of which are current stressors to the South Atlantic DPS.

Overutilization of Atlantic sturgeon from directed fishing caused initial severe declines in Atlantic sturgeon populations in the Southeast, from which they have never rebounded. Further, continued overutilization of Atlantic sturgeon as bycatch in commercial fisheries is an ongoing impact to the South Atlantic DPS. The loss of large subadults and adults as a result of bycatch impacts Atlantic sturgeon populations because they are a long-lived species, have an older age at maturity, have lower maximum fecundity values, and a large percentage of egg production occurs later in life. Little data exists on bycatch in the Southeast and high levels of bycatch underreporting are suspected. Further, a total population abundance for the DPS is not available, and it is therefore not possible to calculate the percentage of the DPS subject to bycatch mortality based on the available bycatch mortality rates for individual fisheries. However, fisheries known to incidentally catch Atlantic sturgeon occur throughout the marine range of the species and in some riverine waters as well. Because Atlantic sturgeon mix extensively in marine waters and may access multiple river systems, they are subject to being caught in multiple fisheries throughout their range. In addition, stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (e.g., exposure to toxins and low DO). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even post-capture mortality.

As a wide-ranging anadromous species, Atlantic sturgeon are subject to numerous Federal (U.S. and Canadian), state and provincial, and inter-jurisdictional laws, regulations, and agency activities. While these mechanisms have addressed impacts to Atlantic sturgeon through directed fisheries, there are currently no mechanisms in place to address the significant risk posed to Atlantic sturgeon from commercial bycatch. Though statutory and regulatory mechanisms exist that authorize reducing the impact of dams on riverine and anadromous species, such as Atlantic sturgeon, and their habitat, these mechanisms have proven inadequate for preventing dams from blocking access to habitat upstream and degrading habitat downstream. Further, water quality continues to be a problem in the South Atlantic DPS, even with existing controls on some pollution sources. Current regulatory regimes are not necessarily effective in controlling water allocation issues (e.g., no permit requirements for water withdrawals under 100,000 gpd in Georgia, no restrictions on interbasin water transfers in South Carolina, the lack of ability to regulate non-point source pollution.)

The recovery of Atlantic sturgeon along the Atlantic Coast, especially in areas where habitat is limited and water quality is severely degraded, will require improvements in the following areas: (1) elimination of barriers to spawning habitat either through dam removal, breaching, or installation of successful fish passage facilities; (2) operation of water control structures to provide appropriate flows, especially during spawning season; (3) imposition of dredging restrictions including seasonal moratoriums and avoidance of spawning/nursery habitat; and, (4) mitigation of water quality parameters that are restricting sturgeon use of a river (i.e., DO). Additional data regarding sturgeon use of riverine and estuarine environments is needed.

A viable population able to adapt to changing environmental conditions is critical to Atlantic sturgeon, and the low population numbers of every river population in the South Atlantic DPS put them in danger of extinction throughout their range; none of the populations are large or stable enough to provide with any level of certainty for continued existence of Atlantic sturgeon in this part of its range. Although the largest impact that caused the precipitous decline of the species has been curtailed (directed fishing), the population sizes within the South Atlantic DPS have remained relatively constant at greatly reduced levels (a small percent of historical population sizes in the Altamaha River and the remainder of the DPS) for 100 years. Small numbers of individuals resulting from drastic reductions in populations, such as occurred with Atlantic sturgeon due to the commercial fishery, can remove the buffer against natural demographic and environmental variability provided by large populations (Berry, 1971; Shaffer, 1981; Soulé, 1980). Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon, and they continue to face a variety of other threats that contribute to their risk of extinction. While a long life-span also allows multiple opportunities to contribute to future generations, it also results increases the timeframe over which exposure to the multitude of threats facing the South Atlantic DPS can occur.

Summary of the Status of the South Atlantic DPS of Atlantic Sturgeon

The South Atlantic DPS is estimated to number fewer than 6 percent of its historical population size, with all river populations except the Altamaha estimated to be a small percent of historical abundance. There are an estimated 343 spawning adults per year in the Altamaha and less than 300 spawning adults per year (total of both sexes) in each of the other major river systems occupied by the DPS in which spawning still occurs, whose freshwater range occurs in the watersheds (including all rivers and tributaries) of the ACE Basin southward along the South Carolina, Georgia, and Florida coastal areas to the St. Johns River, Florida. Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon. Their late age at maturity provides more opportunities for individuals to be removed from the population before reproducing. While a long life-span also allows multiple opportunities to contribute to future generations, this is hampered within the South Atlantic DPS by habitat alteration, bycatch, and from the inadequacy of existing regulatory mechanisms to address and reduce habitat alterations and bycatch.

Dredging is contributing to the status of the South Atlantic DPS by modifying spawning, nursery, and foraging habitat. Habitat modifications through reductions in water quality are also contributing to the status of the South Atlantic DPS through reductions in DO, particularly during times of high water temperatures, which increase the detrimental effects on Atlantic sturgeon habitat. Interbasin water transfers and climate change threaten to exacerbate existing water quality issues. Bycatch is also a current impact to the South Atlantic DPS that is contributing to its status. Fisheries known to incidentally catch Atlantic sturgeon occur throughout the marine range of the species and in some riverine waters as well. Because Atlantic sturgeon mix extensively in marine waters and may utilize multiple river systems for nursery and foraging habitat in addition to their natal spawning river, they are subject to being caught in multiple fisheries throughout their range. In addition to direct mortality, stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (e.g., exposure to toxins). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even post-capture mortality. While many of the threats to the South Atlantic DPS have been ameliorated or reduced due to the existing regulatory mechanisms, such as the moratorium on directed fisheries for Atlantic sturgeon, bycatch is currently not being addressed through existing mechanisms. Further, access to habitat and water quality continues to be a problem even with NMFS' authority under the Federal Power Act to recommend fish passage and existing controls on some pollution sources. There is a lack of regulation for some large water withdrawals, which threatens sturgeon habitat. Current regulatory regimes do not require a permit for water withdrawals under 100,000 gpd in Georgia and there are no restrictions on interbasin water transfers in South Carolina. Data required to evaluate water allocation issues are either very weak, in terms of determining the precise amounts of water currently being used, or non-existent, in terms of our knowledge of water supplies available for use under historical hydrologic conditions in the region. Existing water allocation issues will likely be compounded by population growth, drought, and potentially climate change. The inadequacy of regulatory mechanisms to control bycatch and habitat alterations is contributing to the status of the South Atlantic DPS.

5.0 CLIMATE CHANGE

The discussion below presents background information on predicted global climate change and information on past and predicted future effects of global climate change throughout the range of the listed species considered here. Additionally, we present the available information on predicted effects of climate change in the action area and how listed whales, sea turtles and sturgeon may be affected by those predicted environmental changes over the life of the proposed action (i.e., five years). Climate change is relevant to the Status of the Species, Environmental Baseline and Cumulative Effects sections of this Opinion; rather than include partial discussion in several sections of this Opinion, we are synthesizing this information into one discussion. Consideration of effects of the proposed action in light of predicted changes in environmental conditions due to anticipated climate change are included in the Effects of the Action section below (section 7.0 below).

5.1 Background Information on Global climate change

The global mean temperature has risen 0.76ºC (1.36°F) over the last 150 years, and the linear trend over the last 50 years is nearly twice that for the last 100 years (Intergovernmental Panel on Climate Change (IPCC) 2007a) and precipitation has increased nationally by 5%-10%, mostly due to an increase in heavy downpours (NAST 2000). There is a high confidence, based on substantial new evidence, that observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. Ocean acidification resulting from massive amounts of carbon dioxide and other pollutants released into the air can have major adverse impacts on the calcium balance in the oceans. Changes to the marine ecosystem due to climate change include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007b); these trends are most apparent over the past few decades. Information on future impacts of climate change in the action area is discussed below.

Climate model projections exhibit a wide range of plausible scenarios for both temperature and precipitation over the next century. Both of the principal climate models used by the National Assessment Synthesis Team (NAST) project warming in the southeast by the 2090s, but at different rates (NAST 2000): the Canadian model scenario shows the southeast U.S. experiencing a high degree of warming, which translates into lower soil moisture as higher temperatures increase evaporation; the Hadley model scenario projects less warming and a significant increase in precipitation (about 20%). The scenarios examined, which assume no major interventions to reduce continued growth of world greenhouse gases (GHG), indicate that temperatures in the U.S. will rise by about 3^{o}-5^{o}C (5^{o}-9^{o}F) on average in the next 100 years which is more than the projected global increase (NAST 2000). A warming of about 0.2^{o}C (0.4°F) per decade is projected for the next two decades over a range of emission scenarios (IPCC 2007). This temperature increase will very likely be associated with more extreme precipitation and faster evaporation of water, leading to greater frequency of both very wet and very dry conditions. Climate warming has resulted in increased precipitation, river discharge, and glacial and sea-ice melting (Greene *et al.* 2008).

The past three decades have witnessed major changes in ocean circulation patterns in the Arctic, and these were accompanied by climate associated changes as well (Greene *et al.* 2008). Shifts in atmospheric conditions have altered Arctic Ocean circulation patterns and the export of freshwater to the North Atlantic (Greene *et al.* 2008, IPCC 2006). With respect specifically to the North Atlantic Oscillation (NAO), changes in salinity and temperature are thought to be the result of changes in the earth's atmosphere caused by anthropogenic forces (IPCC 2006). The NAO impacts climate variability throughout the northern hemisphere (IPCC 2006). Data from the 1960s through the present show that the NAO index has increased from minimum values in the 1960s to strongly positive index values in the 1990s and somewhat declined since (IPCC 2006). This warming extends over 1000m (0.62 miles) deep and is deeper than anywhere in the world oceans and is particularly evident under the Gulf Stream/ North Atlantic Current system (IPCC 2006). On a global scale, large discharges of freshwater into the North Atlantic subarctic seas can lead to intense stratification of the upper water column and a disruption of North Atlantic Deepwater (NADW) formation (Greene *et al.* 2008, IPCC 2006). There is evidence that the NADW has already freshened significantly (IPCC 2006). This in turn can lead to a slowing down of the global ocean thermohaline (large-scale circulation in the ocean that transforms low-density upper ocean waters to higher density intermediate and deep waters and returns those waters back to the upper ocean), which can have climatic ramifications for the whole earth system (Greene *et al.* 2008).

While predictions are available regarding potential effects of climate change globally, it is more difficult to assess the potential effects of climate change over the next few decades on coastal and marine resources on smaller geographic scales, especially as climate variability is a dominant factor in shaping coastal and marine systems. The effects of future change will vary greatly in diverse coastal regions for the U.S. Warming is very likely to continue in the U.S.

over the next 25 to 50 years regardless of reduction in GHGs, due to emissions that have already occurred (NAST 2000). It is very likely that the magnitude and frequency of ecosystem changes will continue to increase in the next 25 to 50 years, and it is possible that the rate of change will accelerate. Climate change can cause or exacerbate direct stress on ecosystems through high temperatures, a reduction in water availability, and altered frequency of extreme events and severe storms. Water temperatures in streams and rivers are likely to increase as the climate warms and are very likely to have both direct and indirect effects on aquatic ecosystems. Changes in temperature will be most evident during low flow periods when they are of greatest concern (NAST 2000). In some marine and freshwater systems, shifts in geographic ranges and changes in algal, plankton, and fish abundance are associated with high confidence with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels and circulation (IPCC 2007).

A warmer and drier climate is expected to result in reductions in stream flows and increases in water temperatures. Expected consequences could be a decrease in the amount of dissolved oxygen in surface waters and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing rate (Murdoch *et al.* 2000). Because many rivers are already under a great deal of stress due to excessive water withdrawal or land development, and this stress may be exacerbated by changes in climate, anticipating and planning adaptive strategies may be critical (Hulme 2005). A warmer-wetter climate could ameliorate poor water quality conditions in places where human-caused concentrations of nutrients and pollutants other than heat currently degrade water quality (Murdoch *et al.* 2000). Increases in water temperature and changes in seasonal patterns of runoff will very likely disturb fish habitat and affect recreational uses of lakes, streams, and wetlands. Surface water resources in the southeast are intensively managed with dams and channels and almost all are affected by human activities; in some systems water quality is either below recommended levels or nearly so. A global analysis of the potential effects of climate change on river basins indicates that due to changes in discharge and water stress, the area of large river basins in need of reactive or proactive management interventions in response to climate change will be much higher for basins impacted by dams than for basins with free-flowing rivers (Palmer *et al.* 2008). Human-induced disturbances also influence coastal and marine systems, often reducing the ability of the systems to adapt so that systems that might ordinarily be capable of responding to variability and change are less able to do so. Because stresses on water quality are associated with many activities, the impacts of the existing stresses are likely to be exacerbated by climate change. Within 50 years, river basins that are impacted by dams or by extensive development may experience greater changes in discharge and water stress than unimpacted, free-flowing rivers (Palmer *et al.* 2008).

While debated, researchers anticipate: 1) the frequency and intensity of droughts and floods will change across the nation; 2) a warming of about $0.2^{o}C$ (0.4°F) per decade; and 3) a rise in sea level (NAST 2000). A warmer and drier climate will reduce stream flows and increase water temperature resulting in a decrease of DO and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing. Sea level is expected to continue rising: during the 20th century global sea level has increased 15 to 20 cm (6-8 inches).

5.2 Species Specific Information on Anticipated Effects of Predicted Climate Change

5.2.1 Right, Humpback, Sei, Fin and Sperm Whales

The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats and potential shifts in the distribution and abundance of prey species. Of the main factors affecting distribution of cetaceans, water temperature appears to be the main influence on geographic ranges of cetacean species (Macleod 2009). As such, depending on habitat preferences, changes in water temperature due to climate change may affect the distribution of certain species of cetacean. For instance, fin and humpback whales are distributed in all water temperatures zones, therefore, it is unlikely that their range will be directly affected by an increase in water temperatures (MacLeod 2009). However, North Atlantic right whales, which currently have a range of sub-polar to sub-tropical, may respond to an increase in water temperature by shifting their range northward, with both the northern and southern limits moving poleward.

In regards to marine mammal prey species, there are many potential direct and indirect effects that global climate change may have on prey abundance and distribution, which in turn, poses potential behavioral and physiological effects to marine mammals, including listed whales. For example, Greene *et al.* (2003) described the potential oceanographic processes linking climate variability to the reproduction of North Atlantic right whales. Climate-driven changes in ocean circulation have had a significant impact on the plankton ecology of the Gulf of Maine, including effects on *Calanus finmarchicus*, a primary prey resource for right whales. More information is needed in order to determine the potential impacts global climate change will have on the timing and extent of population movements, abundance, recruitment, distribution and species composition of prey (Learmonth *et al.* 2006). Changes in climate patterns, ocean currents, storm frequency, rainfall, salinity, melting ice, and an increase in river inputs/runoff (nutrients and pollutants) will all directly affect the distribution, abundance and migration of prey species (Waluda *et al.* 2001; Tynan & DeMaster 1997; Learmonth *et al.* 2006). These changes will likely have several indirect effects on marine mammals, which may include changes in distribution including displacement from ideal habitats, decline in fitness of individuals, population size due to the potential loss of foraging opportunities, abundance, migration, community structure, susceptibility to disease and contaminants, and reproductive success (Macleod 2009). Global climate change may also result in changes to the range and abundance of competitors and predators which will also indirectly affect marine mammals (Learmonth *et al.* 2006). A decline in the reproductive fitness as a result of global climate change could have profound effects on the abundance and distribution of large whales in the Atlantic.

5.2.2 Loggerhead Sea Turtles

The most recent Recovery Plan for loggerhead sea turtles as well as the 2009 Status Review Report identifies global climate change as a threat to loggerhead sea turtles. However, trying to assess the likely effects of climate change on loggerhead sea turtles is extremely difficult given the uncertainty in all climate change models and the difficulty in determining the likely rate of temperature increases and the scope and scale of any accompanying habitat effects.

Additionally, no significant climate change-related impacts to loggerhead sea turtle populations have been observed to date. Over the long-term, climate change related impacts are expected to influence biological trajectories on a century scale (Parmesan and Yohe 2003). As noted in the 2009 Status Review (Conant *et al.* 2009), impacts from global climate change induced by human activities are likely to become more apparent in future years (IPCC 2007). Climate change related increasing temperatures, sea level rise, changes in ocean productivity, and increased frequency of storm events may affect loggerhead sea turtles.

Increasing temperatures are expected to result in increased polar melting and changes in precipitation which may lead to rising sea levels (Titus and Narayanan 1995 in Conant *et al.* 2009), which could result in increased erosion rates along nesting beaches. Sea level rise could result in the inundation of nesting sites and decrease available nesting habitat (Daniels *et al.* 1993; Fish *et al.* 2005; Baker *et al.* 2006). The BRT noted that the loss of habitat as a result of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis *et al.* 2006; Baker *et al.* 2006; both in Conant *et al.* 2009). Along developed coastlines, and especially in areas where erosion control structures have been constructed to limit shoreline movement, rising sea levels may cause severe effects on nesting females and their eggs as nesting females may deposit eggs seaward of the erosion control structures potentially subjecting them to repeated tidal inundation. However, if global temperatures increase and there is a range shift northwards, beaches not currently used for nesting may become available for loggerhead sea turtles, which may offset some loss of accessibility to beaches in the southern portions of the range.

Climate change has the potential to result in changes at nesting beaches that may affect loggerhead sex ratios. Loggerhead sea turtles exhibit temperature-dependent sex determination. Rapidly increasing global temperatures may result in warmer incubation temperatures and highly female-biased sex ratios (*e.g.*, Glen and Mrosovsky 2004; Hawkes *et al.* 2009); however, to the extent that nesting can occur at beaches further north where sand temperatures are not as warm, these effects may be partially offset. The BRT specifically identified climate change as a threat to loggerhead sea turtles in the neritic/oceanic zone where climate change may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution. In the threats matrix analysis, climate change was considered for oceanic juveniles and adults and eggs/hatchlings. The report states that for oceanic juveniles and adults, "although the effect of trophic level change from…climate change…is unknown it is believed to be very low." For eggs/hatchlings the report states that total mortality from anthropogenic causes, including sea level rise resulting from climate change, is believed to be low relative to the entire life stage. However, only limited data are available on past trends related to climate effects on loggerhead sea turtles; current scientific methods are not able to reliably predict the future magnitude of climate change, associated impacts, whether and to what extent some impacts will offset others, or the adaptive capacity of this species.

However, Van Houtan and Halley (2011) recently developed climate based models to investigate loggerhead nesting (considering juvenile recruitment and breeding remigration) in the North

Pacific and Northwest Atlantic. These models found that climate conditions/oceanographic influences explain loggerhead nesting variability, with climate models alone explaining an average 60% (range 18%-88%) of the observed nesting changes over the past several decades. In terms of future nesting projections, modeled climate data show a future positive trend for Florida nesting, with increases through 2040 as a result of the Atlantic Multidecadal Oscillation signal.

5.2.3 Kemp's Ridley Sea Turtles

The recovery plan for Kemp's ridley sea turtles (NMFS *et al.* 2011) identifies climate change as a threat; however, as with the other species discussed above, no significant climate change-related impacts to Kemp's ridley sea turtles have been observed to date. Atmospheric warming could cause habitat alteration which may change food resources such as crabs and other invertebrates. It may increase hurricane activity, leading to an increase in debris in nearshore and offshore waters, which may result in an increase in entanglement, ingestion, or drowning. In addition, increased hurricane activity may cause damage to nesting beaches or inundate nests with sea water. Atmospheric warming may change convergence zones, currents and other oceanographic features that are relevant to Kemp's ridleys, as well as change rain regimes and levels of nearshore runoff.

Considering that the Kemp's ridley has temperature-dependent sex determination (Wibbels 2003) and the vast majority of the nesting range is restricted to the State of Tamaulipas, Mexico, global warming could potentially shift population sex ratios towards females and thus change the reproductive ecology of this species. A female bias is presumed to increase egg production (assuming that the availability of males does not become a limiting factor) (Coyne and Landry 2007) and increase the rate of recovery; however, it is unknown at what point the percentage of males may become insufficient to facilitate maximum fertilization rates in a population. If males become a limiting factor in the reproductive ecology of the Kemp's ridley, then reproductive output in the population could decrease (Coyne 2000). Low numbers of males could also result in the loss of genetic diversity within a population; however, there is currently no evidence that this is a problem in the Kemp's ridley population (NMFS *et al.* 2011). Models (Davenport 1997, Hulin and Guillon 2007, Hawkes *et al.* 2007, all referenced in NMFS *et al.* 2011) predict very long-term reductions in fertility in sea turtles due to climate change, but due to the relatively long life cycle of sea turtles, reductions may not be seen until 30 to 50 years in the future.

Another potential impact from global climate change is sea level rise, which may result in increased beach erosion at nesting sites. Beach erosion may be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents. In the case of the Kemp's ridley where most of the critical nesting beaches are undeveloped, beaches may shift landward and still be available for nesting. The Padre Island National Seashore (PAIS) shoreline is accreting, unlike much of the Texas coast, and with nesting increasing and the sand temperatures slightly cooler than at Rancho Nuevo, PAIS could become an increasingly important source of males for the population.

5.2.4 Green Sea Turtles

The five year status review for green sea turtles (NMFS and USFWS 2007d) notes that global climate change is affecting green sea turtles and is likely to continue to be a threat. There is an increasing female bias in the sex ratio of green turtle hatchlings. While this is partly attributable to imperfect egg hatchery practices, global climate change is also implicated as a likely cause. This is because warmer sand temperatures at nesting beaches are likely to result in the production of more female embryos. At least one nesting site, Ascension Island, has had an increase in mean sand temperature in recent years (Hays *et al.* 2003 in NMFS and USFWS 2007d). Climate change may also affect nesting beaches through sea level rise, which may reduce the availability of nesting habitat and increase the risk of nest inundation. Loss of appropriate nesting habitat may also be accelerated by a combination of other environmental and oceanographic changes, such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion. Oceanic changes related to rising water temperatures could result in changes in the abundance and distribution of the primary food sources of green sea turtles, which in turn could result in changes in behavior and distribution of this species. Seagrass habitats may suffer from decreased productivity and/or increased stress due to sea level rise, as well as salinity and temperature changes (Short and Neckles 1999; Duarte 2002).

As noted above, the increasing female bias in green sea turtle hatchlings is thought to be at least partially linked to increases in temperatures at nesting beaches. However, at this time, we do not know how much of this bias is due to hatchery practice and how much is due to increased sand temperature. Because we do not have information to predict the extent and rate to which sand temperatures at the nesting beaches used by green sea turtles may increase in the short-term future, we cannot predict the extent of any future bias. Also, we do not know to what extent to which green sea turtles may be able to cope with this change by selecting cooler areas of the beach or shifting their nesting distribution to other beaches at which increases in sand temperature may not be experienced.

5.2.5 Leatherback sea turtles

Global climate change has been identified as a factor that may affect leatherback habitat and biology (NMFS and USFWS 2007b); however, no significant climate change related impacts to leatherback sea turtle populations have been observed to date. Over the long term, climate change related impacts will likely influence biological trajectories in the future on a century scale (Parmesan and Yohe 2003). Changes in marine systems associated with rising water temperatures, changes in ice cover, salinity, oxygen levels and circulation including shifts in ranges and changes in algal, plankton, and fish abundance could affect leatherback prey distribution and abundance. Climate change is expected to expand foraging habitats into higher latitude waters and some concern has been noted that increasing temperatures may increase the female:male sex ratio of hatchlings on some beaches (Morosovsky *et al.* 1984 and Hawkes *et al.* 2007 in NMFS and USFWS 2007b). However, due to the tendency of leatherbacks to have individual nest placement preferences and deposit some clutches in the cooler tide zone of beaches, the effects of long-term climate on sex ratios may be mitigated (Kamel and Mrosovsky 2004 in NMFS and USFWS 2007b).

Additional potential effects of climate change on leatherbacks include range expansion and changes in migration routes as increasing ocean temperatures shift range-limiting isotherms north (Robinson *et al.* 2008). Leatherbacks have expanded their range in the Atlantic north by 330 km in the last 17 years as warming has caused the northerly migration of the 15°C sea surface temperature (SST) isotherm, the lower limit of thermal tolerance for leatherbacks (McMahon and Hays 2006). Leatherbacks are speculated to be the best able to cope with climate change of all the sea turtle species due to their wide geographic distribution and relatively weak beach fidelity. Leatherback sea turtles may be most affected by any changes in the distribution of their primary jellyfish prey, which may affect leatherback distribution and foraging behavior (NMFS and USFWS 2007b). Jellyfish populations may increase due to ocean warming and other factors (Brodeur *et al.* 1999; Attrill *et al.* 2007; Richardson *et al.* 2009). However, any increase in jellyfish populations may or may not impact leatherbacks as there is no evidence that any leatherback populations are currently food-limited.

Increasing temperatures are expected to result in increased polar melting and changes in precipitation which may lead to rising sea levels (Titus and Narayanan 1995 in Conant *et al.* 2009), which could result in increased erosion rates along nesting beaches. Sea level rise could result in the inundation of nesting sites and decrease available nesting habitat (Fish *et al.* 2005). This effect would potentially be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents. While there is a reasonable degree of certainty that climate change related effects will be experienced globally (*e.g.*, rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects of climate change on this species are not quantifiable at this time (Hawkes *et al.* 2009).

5.2.6 Atlantic sturgeon

Global climate change may affect all DPSs of Atlantic sturgeon in the future; however, effects of increased water temperature and decreased water availability are most likely to effect the South Atlantic and Carolina DPSs. Rising sea level may result in the salt wedge moving upstream in affected rivers. Atlantic sturgeon spawning occurs in fresh water reaches of rivers because early life stages have little to no tolerance for salinity. Similarly, juvenile Atlantic sturgeon have limited tolerance to salinity and remain in waters with little to no salinity. If the salt wedge moves further upstream, Atlantic sturgeon spawning and rearing habitat could be restricted. In river systems with dams or natural falls that are impassable by sturgeon, the extent that spawning or rearing may be shifted upstream to compensate for the shift in the movement of the saltwedge would be limited. While there is an indication that an increase in sea level rise would result in a shift in the location of the salt wedge, at this time there are no predictions on the timing or extent of any shifts that may occur; thus, it is not possible to predict any future loss in spawning or rearing habitat. However, in all river systems, spawning occurs miles upstream of the saltwedge. It is unlikely that shifts in the location of the saltwedge would eliminate freshwater spawning or rearing habitat. If habitat was severely restricted, productivity or survivability may decrease.

The increased rainfall predicted by some models in some areas may increase runoff and scour spawning areas and flooding events could cause temporary water quality issues. Rising temperatures predicted for all of the U.S. could exacerbate existing water quality problems with DO and temperature. While this occurs primarily in rivers in the southeast U.S. and the Chesapeake Bay, it may start to occur more commonly in the northern rivers. Atlantic sturgeon prefer water temperatures up to approximately 28°C (82.4°F); these temperatures are experienced naturally in some areas of rivers during the summer months. If river temperatures rise and temperatures above 28°C are experienced in larger areas, sturgeon may be excluded from some habitats.

Increased droughts (and water withdrawal for human use) predicted by some models in some areas may cause loss of habitat including loss of access to spawning habitat. Drought conditions in the spring may also expose eggs and larvae in rearing habitats. If a river becomes too shallow or flows become intermittent, all Atlantic sturgeon life stages, including adults, may become susceptible to strandings or habitat restriction. Low flow and drought conditions are also expected to cause additional water quality issues. Any of the conditions associated with climate change are likely to disrupt river ecology causing shifts in community structure and the type and abundance of prey. Additionally, cues for spawning migration and spawning could occur earlier in the season causing a mismatch in prey that are currently available to developing sturgeon in rearing habitat.

5.3 Effects of climate Change to Listed Species in the Action Area

As there is significant uncertainty in the rate and timing of change as well as the effect of any changes that may be experienced in the action area due to climate change, it is difficult to predict the impact of these changes on listed species; however, we have considered the available information to consider likely impacts to these species in the action area.

5.3.1 Right, Humpback, Fin, Sei and Sperm Whales

As described above, the impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of seawater due to melting ice and increased rainfall, sea level rise, the loss of polar habitats, and potential shifts in the distribution and abundance of prey species. These impacts, in turn, are likely to affect the distribution of species of whales. As described in section 4.0, listed species of whales may be found throughout the action area. Within this portion of the action area, the most likely effect to whales from climate change would be if warming temperatures led to changes in the seasonal distribution of whales. This may mean that ranges and seasonal migratory patterns are altered to coincide with changes in prey distribution on foraging grounds located outside of the action area, which may result in an increase or decrease of listed species of whales in the action area. As humpback and fin whales are distributed in all water temperature zones, it is unlikely that their range will be directly affected by an increase in water temperature; however, for right whales, increases in water temperature may result in a northward shift of their range. This may result in an unfavorable effect on the North Atlantic right whale due to an increase in the length of migrations (Macleod 2009) or a favorable effect by allowing them to expand their range. However, over the life of the action (through 2018) it is unlikely that this possible shift in range will be observed due the extremely small increase in water temperature predicted to occur during this period (i.e., less than 1^{o}C); if any shift does occur, it is likely to be minimal and thus, it seems unlikely that this small increase in temperature will cause a significant effect to right whales or a significant modification to the number of whales likely to be present in the action area through 2018.

5.3.2 Sea Turtles

Sea turtles are most likely to be affected by climate change due to increasing sand temperatures at nesting beaches which in turn would result in increased female: male sex ratio among hatchlings, sea level rise which could result in a reduction in available nesting beach habitat, increased risk of nest inundation, changes in the abundance and distribution of forage species which could result in changes in the foraging behavior and distribution of sea turtle species, and changes in water temperature which could possibly lead to a northward shift in their range.

Over the five-year time period considered in this Opinion, any increase in sea surface temperatures attributable to global climate change is expected to be very small. It is unlikely to be enough of a change to contribute to shifts in the range or distribution of sea turtles. Theoretically, we expect that as waters in the action area warm, more sea turtles could be present or sea turtles could be present for longer periods of time. However, if temperature affected the distribution of sea turtle forage in a way that decreased forage in the action area, sea turtles may be less likely to occur in the action area.

It has been speculated that the nesting range of some sea turtle species may shift northward. Nesting in the mid-Atlantic generally is extremely rare and no nesting has been documented at any beach in the action area. In 2010, one green sea turtle came up on the beach in Sea Isle City, New Jersey; however, it did not lay any eggs. In August 2011, a loggerhead came up on the beach in Stone Harbor, New Jersey but did not lay any eggs. On August 18, 2011, a green sea turtle laid one nest at Cape Henlopen Beach in Lewes Delaware near the entrance to Delaware Bay. The nest contained 190 eggs and was transported indoors to an incubation facility on October 7. A total of twelve eggs hatched, with eight hatchlings surviving. In December, seven of the hatchlings were released in Cape Hatteras, North Carolina. It is important to consider that in order for nesting to be successful in the mid-Atlantic, fall and winter temperatures need to be warm enough to support the successful rearing of eggs and sea temperatures must be warm enough for hatchlings not to die when they enter the water. Predicted increases in water temperatures over the next five years are not great enough to allow successful rearing of sea turtle eggs in the action area or the survival of hatchlings that enter the water outside of the summer months. Predicted increases in water temperatures between now and 2018 are not great enough to allow successful rearing of sea turtle eggs in the action area. Therefore, it is unlikely that over the time period considered here, that there would be an increase in nesting activity in the action area or that hatchlings would be present in the action area.

5.3.3 Atlantic sturgeon

Over time, the most likely effect to Atlantic sturgeon would be if sea level rise was great enough to consistently shift the salt wedge far enough north which would restrict the range of juvenile sturgeon and may affect the development of these life stages. Upstream shifts in spawning or rearing habitat in some rivers are limited by the existence of a dam or other impassable barrier (natural falls). The available habitat for juvenile sturgeon could decrease over time; however, even if the saltwedge shifted several miles upstream, it seems unlikely that the decrease in available habitat would have a significant effect on juvenile sturgeon because there would still be many miles of available low salinity habitat between the salt wedge and these barriers.

In the action area, it is possible that changing seasonal temperature regimes could result in changes in the timing of seasonal migrations through the area as sturgeon move to spawning and overwintering grounds. There could be shifts in the timing of spawning; presumably, if water temperatures warm earlier in the spring, and water temperature is a primary spawning cue, spawning migrations and spawning events could occur earlier in the year. However, because spawning is not triggered solely by water temperature, but also by day length (which would not be affected by climate change) and river flow (which could be affected by climate change), it is not possible to predict how any change in water temperature or river flow alone will affect the seasonal movements of sturgeon through the action area.

Any forage species that are temperature dependent may also shift in distribution as water temperatures warm. However, because we do not know the adaptive capacity of these individuals or how much of a change in temperature would be necessary to cause a shift in distribution, it is not possible to predict how these changes may affect foraging sturgeon. If sturgeon distribution shifted along with prey distribution, it is likely that there would be minimal, if any, impact on the availability of food. Similarly, if sturgeon shifted to areas where different forage was available and sturgeon were able to obtain sufficient nutrition from that new source of forage, any effect would be minimal. The greatest potential for effect to forage resources would be if sturgeon shifted to an area or time where insufficient forage was available; however, the likelihood of this happening seems low because sturgeon feed on a wide variety of species and in a wide variety of habitats.

Limited information on the thermal tolerances of Atlantic sturgeon is available. Atlantic sturgeon have been observed in water temperatures above 30°C in the south (see Damon-Randall *et al.* 2010); in the wild, shortnose sturgeon are typically found in waters less than 28°C. In the laboratory, juvenile Atlantic sturgeon showed negative behavioral and bioenergetics responses (related to food consumption and metabolism) after prolonged exposure to temperatures greater than 28°C (82.4°F) (Niklitschek 2001). Tolerance to temperatures is thought to increase with age and body size (Ziegweid *et al.* 2008 and Jenkins *et al.* 1993), however, no information on the lethal thermal maximum or stressful temperatures for subadult or adult Atlantic sturgeon is available. Shortnose sturgeon, have been documented in the lab to experience mortality at temperatures of 33.7°C (92.66°F) or greater and are thought to experience stress at temperatures above 28°C. For purposes of considering thermal tolerances, we consider Atlantic sturgeon to be a reasonable surrogate for shortnose sturgeon given similar geographic distribution and known biological similarities.

Normal surface water temperatures in the action area can be as high as 28°C at some times and in some areas during the summer months; temperatures in deeper waters and near the bottom are cooler. A predicted increase in water temperature of 3-4°C within 100 years is expected to result in temperatures approaching the preferred temperature of shortnose and Atlantic sturgeon (28°C) on more days and/or in larger areas. However, over the next five years, any increase in water temperatures is expected to be very small. While over time warming water temperatures could result in shifts in the distribution of sturgeon out of certain areas during the warmer months, we do not expect these type of large scale changes in the next five years.

As described above, over the long term, global climate change may affect Atlantic sturgeon by affecting the location of the salt wedge in rivers, distribution of prey, water temperature and water quality. However, there is significant uncertainty, due to a lack of scientific data, on the degree to which these effects may be experienced and the degree to which shortnose or Atlantic sturgeon will be able to successfully adapt to any such changes. Any activities occurring within and outside the action area that contribute to global climate change are also expected to affect shortnose and Atlantic sturgeon in the action area. While we can make some predictions on the likely effects of climate change on these species, without modeling and additional scientific data these predictions remain speculative. Additionally, these predictions do not take into account the adaptive capacity of these species which may allow them to deal with change better than predicted.

6.0 ENVIRONMENTAL BASELINE

Environmental baselines for biological opinions include the past and present impacts of all state, federal or private actions and other human activities in the action area, the anticipated impacts of all proposed federal projects in the action area that have already undergone formal or early Section 7 consultation, and the impact of state or private actions that are contemporaneous with the consultation in process (50 CFR 402.02). The environmental baseline for this Opinion includes the effects of several activities that may affect the survival and recovery of the listed species in the action area.

6.1 Federal Actions that have Undergone Section 7 Consultation

NMFS has undertaken several ESA Section 7 consultations to address the effects of various federal actions on threatened and endangered species in the action area. Each of those consultations sought to develop ways of reducing the probability of adverse impacts of the action on listed species.

6.1.1 Authorization of Fisheries through Fishery Management Plans

NMFS authorizes the operation of several fisheries in the action area under the authority of the Magnuson-Stevens Fishery Conservation Act and through Fishery Management Plans and their implementing regulations. Commercial and recreational fisheries in the action area employ gear that is known to harass, injure, and/or kill sea turtles, whales and Atlantic sturgeon. In the Northeast Region (Maine through Virginia), formal ESA section 7 consultations have been conducted on the American lobster, Atlantic bluefish, Atlantic mackerel/squid/ butterfish, Atlantic sea scallop, monkfish, northeast multispecies, red crab, spiny dogfish, summer flounder/scup/black sea bass, and tilefish fisheries. These consultations have considered effects to loggerhead, green, Kemp's ridley and leatherback sea turtles as well as ESA-listed whales. We have completed Biological Opinions on the operations of these fisheries. In each of these Opinions, we concluded that the ongoing action was likely to adversely affect but was not likely to jeopardize the continued existence of any sea turtle or whale species. Each of these Opinions included an incidental take statement (ITS) exempting a certain amount of lethal and/or non-lethal take of sea turtles resulting from interactions with the fishery. These ITSs are summarized in the table below. Further, in each Opinion, we concluded that the potential for interactions (*i.e.*, vessel strikes) between sea turtles and fishing vessels was extremely low and similarly that any effects to sea turtle prey and/or habitat would be insignificant and discountable. We have also determined that the Atlantic herring and surf clam/ocean quahog fisheries do not adversely affect any species of listed sea turtles or whales.

In addition to these consultations, NMFS has conducted a formal consultation on the pelagic longline component of the Atlantic highly migratory species FMP. Portions of this fishery occur within the action area. In a June 1, 2004 Opinion, NMFS concluded that the ongoing action was likely to adversely affect but was not likely to jeopardize the continued existence of loggerhead, Kemp's ridley or green sea turtles but was likely to jeopardize the continued existence of leatherback sea turtles. This Opinion included a Reasonable and Prudent Alternative that when implemented would modify operations of the fishery in a way that would remove jeopardy. This fishery is currently operated in a manner that is consistent with the RPA. The RPA included an ITS which is reflected in the table below. Unless specifically noted, all numbers denote an annual number of captures that may be lethal or non-lethal.

Table 6. Information on Fisheries Opinions conducted by NMFS NERO and SERO for federally managed fisheries that operate in the action area

FMP	**Date of Most Recent Opinion**	**Loggerhead**	**Kemp's ridley**	**Green**	**Leatherback**
American lobster	August 3, 2012	1	0	0	5
Atlantic bluefish	October 29, 2010	82 (34 lethal)	4	5	4
Monkfish	October 29, 2010	173 (70 lethal)	4	5	4
Multispecies	October 29, 2010	46 in trawls (21 lethal)	4	5	4
Skate	October 29, 2010	39 (17 lethal)	4	5	4
Spiny dogfish	October 29, 2010	2	4	5	4
Mackerel/squid/butterfish	October 29, 2010	62 (25 lethal)	2	2	2
Summer flounder/scup/black sea bass	October 29, 2010	205 (85 lethal)	4	5	6
Shark fisheries as managed under the Consolidated HMS FMP	May 20, 2008	679 (349 lethal) every 3 years	2 (1 lethal) every 3 years	2 (1 lethal) every 3 years	74 (47 lethal) every 3 years

Atlantic sea scallop	July 12, 2012	2013 and beyond: 301 (115 lethal)	3	2	2
Coastal migratory pelagic	August 13, 2007	33 every 3 years	4 every 3 years	14 every 3 years	2 every 3 years
Pelagic longline under the HMS FMP (per the RPA)	June 1, 2004	1,905 (339 lethal) every 3 years	*105 (18 lethal) every 3 years	*105 (18 lethal) every 3 years	1764 (252 lethal) every 3 years

**combination of 105 (18 lethal) Kemp's ridley, green, hawksbill, or Olive ridley*

Sturgeon originating from all five DPSs are captured and killed in fisheries operated in the action area. We have reinitiated consultations that consider fisheries actions that may affect Atlantic sturgeon. To date, consultations considering effects to Atlantic sturgeon from Federally authorized fisheries in the action area have been completed for the scallop FMP and the smooth dogfish fishery. As noted in the Status of the Species section above, the NEFSC prepared a bycatch estimate for Atlantic sturgeon captured in sink gillnet and otter trawl fisheries operated from Maine through Virginia. This estimate indicates that, based on data from 2006-2010, annually, an average of 3,118 Atlantic sturgeon are captured in these fisheries with 1,569 in sink gillnet and 1,548 in otter trawls. The mortality rate in sink gillnets is estimated at approximately 20% and the mortality rate in otter trawls is estimated at 5%. Based on this estimate, a total of 391 Atlantic sturgeon are estimated to be killed annually in these fisheries that are prosecuted in the action area. We are currently in the process of determining the effects of this annual loss to each of the DPSs.

6.1.2 Vessel Activity and Military Operations

Potential sources of adverse effects to sea turtles from Federal vessel operations in the action area include operations of the U.S. Navy (USN), U.S. Coast Guard (USCG), Environmental Protection Agency (EPA), Army Corps of Engineers (ACOE), and NOAA to name a few. NMFS has previously conducted formal consultations with the USN, USCG, and NOAA on their vessel-based operations. NMFS has also conducted section 7 consultations with the Minerals Management Service (MMS), Federal Energy Regulatory Commission (FERC), and Maritime Administration (MARAD) on vessel traffic related to energy projects in the Northeast Region and has implemented conservation measures. Through the section 7 process, where applicable, NMFS has and will continue to establish conservation measures for all these agency vessel operations to avoid or minimize adverse effects to listed species. We are currently in the process of determining if any of these activities may affect Atlantic sturgeon and if any existing section 7 consultations on these actions need to be reinitiated. To date, ocean going vessels and military activities have not been identified as significant threats to Atlantic sturgeon. However, the possibility exists for interactions between vessels and Atlantic sturgeon in the marine environment. Because of a lack of information on the effects of these activities on Atlantic sturgeon, the discussion below focuses on sea turtles.

Although consultations on individual USN and USCG activities have been completed, only one formal consultation on overall military activities in all of the Atlantic has been completed at this time. In June 2009, NMFS prepared an Opinion on USN activities in each of their four training range complexes along the U.S. Atlantic coast—Northeast, Virginia Capes, Cherry Point, and Jacksonville (NMFS 2009c). In addition, the following Opinions for the USN (NMFS 1996, 1997, 2008c, 2009e) and USCG (NMFS 1995, 1998b) contain details on the scope of vessel operations for these agencies and the conservation measures that are being implemented as standard operating procedures. In the U.S. Atlantic, the operation of USCG boats and cutters is estimated to take no more than one individual sea turtle, of any species, per year (NMFS 1995).

Military activities such as ordnance detonation also affect listed species of sea turtles. A section 7 consultation was conducted in 1997 for USN aerial bombing training in the ocean off the southeast U.S. coast, involving drops of live ordnance (500 and 1,000-lb bombs). The resulting Opinion for this consultation determined that the activity was likely to adversely affect sea turtles but would not jeopardize their continued existence. In the ITS included within the Opinion, these training activities were estimated to have the potential to injure or kill, annually, 84 loggerheads, 12 leatherbacks, and 12 greens or Kemp's ridleys, in combination (NMFS 1997).

NMFS has also conducted more recent section 7 consultations on USN explosive ordnance disposal, mine warfare, sonar testing (e.g., AFAST, SURTASS LFA), and other major training exercises (e.g., bombing, Naval gunfire, combat search and rescue, anti-submarine warfare, and torpedo and missile exercises) in the Atlantic Ocean. These consultations have determined that the proposed USN activities may adversely affect but would not jeopardize the continued existence of ESA-listed sea turtles (NMFS 2008b, 2009c). NMFS estimated that five loggerhead and six Kemp's ridley sea turtles are likely to be harmed as a result of training activities in the Virginia Capes Range Complex from June 2009 to June 2010, and that nearly 1,500 sea turtles, including 10 leatherbacks, are likely to experience harassment (NMFS 2009c).

6.2 Other Activities

6.2.1 Maritime Industry

Private and commercial vessels, including fishing vessels, operating in the action area of this consultation also have the potential to interact with sea turtles and Atlantic sturgeon. The effects of fishing vessels, recreational vessels, or other types of commercial vessels on ESA-listed species may involve disturbance or injury/mortality due to collisions or entanglement in anchor lines. It is important to note that minor vessel collisions may not kill an animal directly, but may weaken or otherwise affect it so it is more likely to become vulnerable to effects such as entanglement. Listed species may also be affected by fuel oil spills resulting from vessel accidents. Fuel oil spills could affect animals through the food chain. However, these spills typically involve small amounts of material that are unlikely to adversely affect listed species. Larger oil spills may result from severe accidents, although these events would be rare and involve small areas. No direct adverse effects on listed sea turtles or Atlantic sturgeon resulting from fishing vessel fuel spills have been documented.

6.2.2 Pollution

Anthropogenic sources of marine pollution, while difficult to attribute to a specific Federal, state, local, or private action, may affect sea turtles and Atlantic sturgeon in the action area. Sources of pollutants in the action area include atmospheric loading of pollutants such as PCBs; storm water runoff from coastal towns, cities, and villages; runoff into rivers emptying into bays; groundwater discharges; sewage treatment plant effluents; and oil spills. The pathological effects of oil spills on sea turtles have been documented in several laboratory studies (Vargo *et al.* 1986).

Nutrient loading from land-based sources, such as coastal communities and agricultural operations, is known to stimulate plankton blooms in closed or semi-closed estuarine systems. The effect to larger embayments is unknown. Contaminants could degrade habitat if pollution and other factors reduce the food available to marine animals.

6.2.3 Coastal development
Beachfront development, lighting, and beach erosion control all are ongoing activities along the Mid- and South Atlantic coastlines of the U.S. These activities potentially reduce or degrade sea turtle nesting habitats or interfere with hatchling movement to sea. Nocturnal human activities along nesting beaches may also discourage sea turtles from nesting sites. The extent to which these activities reduce sea turtle nesting and hatchling production is unknown. However, more and more coastal counties are adopting stringent protective measures to protect hatchling sea turtles from the disorienting effects of beach lighting. Coastal development may also impact Atlantic sturgeon if it disturbs or degrades foraging habitats or otherwise affects the ability of sturgeon to use coastal habitats.

6.3 Reducing Threats to ESA-listed Species

6.3.1 Whales

Atlantic Large Whale Take Reduction Plan
The ALWTRP reduces the risk of serious injury to or mortality of large whales due to incidental entanglement in U.S. commercial trap/pot and gillnet fishing gear. The ALWTRP focuses on the critically endangered North Atlantic right whale, but is also intended to reduce entanglement of endangered humpback and fin whales. The plan is required by the MMPA and has been developed by NMFS. The ALWTRP covers the U.S. Atlantic EEZ from Maine through Florida (26°46.5‘N). The requirements are year-round in the Northeast, and seasonal in the Mid and South Atlantic.

The plan has been developed in collaboration with the Atlantic Large Whale Take Reduction Team (ALWTRT), which consists of fishing industry representatives, environmentalists, state and federal officials, and other interested parties. The ALWTRP is an evolving plan that changes as NMFS and the ALWTRT learn more about why whales become entangled and how fishing practices might be modified to reduce the risk of entanglement. Regulatory actions are directed at reducing serious entanglement injuries and mortalities of right, humpback, and fin whales from fixed gear fisheries (*i.e.*, trap/pot and gillnet fisheries). The non-regulatory component of the ALWTRP is composed of four principal parts: (1) gear research and development, (2) disentanglement, (3) the Sighting Advisory System (SAS), and (4) education/outreach**.** These components will be discussed in more detail below. The first ALWTRP went into effect in 1997.

Regulatory Measures to Reduce the Threat of Entanglement on Whales

The regulatory component of the ALWTRP includes a combination of broad fishing gear modifications and time-area restrictions supplemented by progressive gear research to reduce the chance that entanglements will occur, or that whales will be seriously injured or die as a result of an entanglement. The long-term goal, established by the 1994 Amendments to the MMPA, is to reduce entanglement related serious injuries and mortalities of right, humpback and fin whales to insignificant levels approaching zero within five years of its implementation. Despite these measures, entanglements, some of which resulted in serious injuries or mortalities, continued to occur. Data on whale distribution, gear distribution and configuration, and all gear observed on or taken off whales was examined. The ALWTRP is an evolving plan, and revisions are made to the regulations as new information and technology becomes available. Because serious injury and mortality of right, humpback, and fin whales have continued to occur due to gear entanglements, new and revised regulatory measures have been issued since the original plan was developed.

The ALWTRT initially concluded that all parts of gillnet and trap/pot gear can and have caused entanglements. Initial measures in the ALWTRP addressed both parts of the gear, and since then, the ALWTRT has identified the need to further reduce risk posed by both vertical and horizontal portions of gear. Research and testing has been ongoing to identify risk reduction measures that are feasible. The regulations focus on reducing the risk associated with horizontal (ground line) lines.

The ALWTRP measures vary by designated area that roughly approximate the federal Lobster Management Areas (FLMAs) designated in the Federal lobster regulations. The major requirements of the ALWTRP are:

- No buoy line floating at the surface.
- No wet storage of gear (all gear must be hauled out of the water at least once every 30 days).
- Surface buoys and buoy line need to be marked to identify the vessel or fishery.
- All buoys, flotation devices, and/or weights must be attached to the buoy line with a weak link. This measure is designed so that if a large whale does become entangled, it could extert enough force to break the weak link and break free of the gear, reducing the risk of injury or mortality.
- All groundline must be made of sinking line (year-round in the Northeast; seasonal in the Mid- and South Atlantic).

In addition to gear modification requirements, the ALWTRP prohibits all trap/pot and gillnet fishing in the Great South Channel from April 1 to June 30. Cape Cod Bay is also closed to gillnet fishing from January 1 to May 15. These time periods coincide with the presence of right whales in these areas.

In addition to the regulatory measures recently implemented to reduce the risk of entanglement in horizontal/ground lines, NMFS, in collaboration with the ALWTRT, has developed a strategy to further reduce risk associated with vertical lines.

It is anticipated that the final regulations implementing the vertical line strategy will prioritize risk reduction in areas where there is the greatest co-occurrence of vertical lines and large whales. There are two ways to achieve a reduced risk: (1) maintain the same number of active lines but decrease the risk from each one (not currently feasible), or (2) reduce the number of lines in the water column.

Whale distribution data are being used to help prioritize areas for implementation of future vertical line action(s). These data are overlaid with the vertical line distribution data to look at the combined densities by area. A model has been developed and was constructed to allow gear configurations to be manipulated and determine what relative co-occurrence reductions (as a proxy for risk) can be achieved by gear configuration changes and/or effort reductions by area. This co-occurrence analysis is an integral component of the vertical line strategy that will further minimize the risk of large whale entanglement and associated serious injury and death. The actions and timeframe for the implementation of the vertical line strategy are as follows:

- Vertical line model development for all areas to gather as much information as possible regarding the distribution and density of vertical line fishing gear. Status: completed;

- Compile and analyze whale distribution and density data in a manner to overlay with vertical line density data. Status: completed;

- Development of vertical line and whale distribution co-occurrence overlays. Status: completed;

- Develop an ALWTRP monitoring plan designed to track implementation of vertical line strategy, including risk reduction. Status: completed, with annual interim reports beginning in July 2012.

- Analyze and develop potential management measures. Time frame: ongoing;

- Develop and publish proposed rule to implement risk reduction from vertical lines. Time frame: by Mid- 2013;

- Develop and publish final rule to implement risk reduction from vertical lines. Time frame: by Mid- 2014;

- Implement final rule to implement risk reduction from vertical lines. Time frame: by early 2015;

Non-regulatory Components of the ALWTRP
Gear Research and Development

Gear research and development is a critical component of the ALWTRP, with the aim of finding new ways of reducing the number and severity of protected species-gear interactions while still allowing for fishing activities. At the outset, the gear research and development program followed two approaches: (a) reducing the number of lines in the water while still allowing fishing, and (b) devising lines that are weak enough to allow whales to break free and at the same time strong enough to allow continued fishing. Development of gear modifications are ongoing and are primarily used to minimize risk of large whale entanglement. The ALWTRT has now moved into the next phase with the focus and priority being research to reduce risk associated with vertical lines. This aspect of the ALWTRP is important because it incorporates the knowledge and encourages the participation of industry in the development and testing of modified and experimental gear. Currently, NMFS is refining a co-occurrence risk model that allows us to examine the density of whales and vertical lines in time and space to identify those areas and times that pose the greatest vertical line risk. These areas would be prioritized for management. The current schedule would result in a proposed rule for additional vertical line risk reduction to be published in 2013.

The NMFS, in consultation with the ALWTRT, has developed a monitoring plan for the ALWTRP. While the number of serious injuries and mortalities caused by entanglements is higher than our goal, it is still a relatively small number, which makes monitoring difficult. Specifically, we want to know if the most recent management measures, which became fully effective April 2009, have resulted in a reduction in entanglement related serious injuries and mortalities of right, humpback and fin whales. Because these are relatively rare events and the data obtained from each event is sparse, this is a difficult question to answer. The NEFSC has identified proposed metrics that will be used to monitor progress. They project that five years of data would be required before a change may be able to be detected. Therefore, data from 2010 to 2014 may be required to answer this question. The analysis of that data would not be able to occur until 2016 due to the availability of the five years of data after new regulations have been in place.

Large Whale Disentanglement Program

Entanglement of marine mammals in fishing gear and/or marine debris is a significant problem throughout the world's oceans. NMFS created and manages a Whale Disentanglement Network, purchasing equipment to be located at strategic spots along the Atlantic coastline, supporting training for fishermen and biologists, purchasing telemetry equipment, etc. This has resulted in an expanded capacity for disentanglement along the Atlantic seaboard including offshore areas. Along the U.S. eastern seaboard, reports of entangled humpback whales and North Atlantic right whales, and to a lesser extent fin whales and sei whales, have been received. In 1984 the Provincetown Center for Coastal Studies (PCCS) in partnership with NMFS developed a technique for disentangling free-swimming large whales from life threatening entanglements. Over the next decade, PCCS and NMFS continued working on the development of the technique to safely disentangle both anchored and free swimming large whales. In 1995 NMFS issued a permit to PCCS to disentangle large whales. Additionally, NMFS and PCCS have established a large whale disentanglement program, also referred to as the Atlantic Large Whale Disentanglement Network (ALWDN), based on successful disentanglement efforts by many researchers and partners. Memorandums of Agreement were also issued between NMFS and other federal government agencies to increase the resources available to respond to reports of entangled large whales anywhere along the U.S. eastern seaboard. NMFS has established agreements with many coastal states to collaboratively monitor and respond to entangled whales. As a result of the success of the disentanglement network, NMFS believes whales that may otherwise have succumbed to complications from entangling gear have been freed and have survived.

Sighting Advisory System (SAS)
Although the Sighting Advisory System (SAS) was developed primarily as a method of locating right whales and alerting mariners to right whale sighting locations in a real time manner, the SAS also addresses entanglement threats. Fishermen can obtain SAS sighting reports and make necessary adjustments in operations to decrease the potential for interactions with right whales. Some of these sighting efforts have resulted in successful disentanglement of right whales. The SAS is discussed further in section 4.4.7.5.

Educational Outreach
Education and outreach activities are considered some of the primary tools needed to reduce the threats to all protected species from human activities, including fishing activities. Outreach efforts for fishermen under the ALWTRP are fostering a more cooperative relationship between all parties interested in the conservation of threatened and endangered species. Type of outreach/education include website updates, attendance at industry meetings and outreach events, publications in industry trade journals, training for observer program and Coast Guard and state/federel enforcement agents.

Ship Strike Reduction Program
The Ship Strike Reduction Program is currently focused on protecting the North Atlantic right whale, but the operational measures are expected to reduce the incidence of ship strike on other large whales to some degree. The program consists of five basic elements and includes both regulatory and non-regulatory components: 1) operational measures for the shipping industry, including speed restrictions and routing measures, 2) section 7 consultations with federal agencies that maintain vessel fleets, 3) education and outreach programs, 4) a bilateral conservation agreement with Canada, and 5) continuation of ongoing measures to reduce ship strikes of right whales (*e.g.*, SAS, ongoing research into the factors that contribute to ship strikes, and research to identify new technologies that can help mariners and whales avoid each other).

Regulatory Measures to Reduce Vessel Strikes to Large Whales

Restricting Vessel Approach to Right Whales
In one recovery action aimed at reducing vessel-related impacts, including disturbance, NMFS published a proposed rule in August 1996 restricting vessel approach to right whales (61 FR 41116, August 7, 1996) to a distance of 500 yards. The Recovery Plan for the North Atlantic right whale identified anthropogenic disturbance as one of many factors that had some potential to impede right whale recovery (NMFS 2005a). Following public comment, NMFS published an interim final rule in February 1997 codifying the regulations. With certain exceptions, the

rule prohibits both boats and aircraft from approaching any right whale closer than 500 yards. Exceptions for closer approach are provided for the following situations, when: (a) compliance would create an imminent and serious threat to a person, vessel, or aircraft; (b) a vessel is restricted in its ability to maneuver around the 500-yard perimeter of a whale; (c) a vessel is investigating or involved in the rescue of an entangled or injured right whale; or (d) the vessel or aircraft is participating in a permitted activity, such as a research project. If a vessel operator finds that he or she has unknowingly approached closer than 500 yards, the rule requires that a course be steered away from the whale at slow, safe speed. In addition, all aircraft, except those involved in whale watching activities, are exempted from these approach regulations. This rule is expected to reduce the potential for vessel collisions and other adverse vessel-related effects in the environmental baseline.

Mandatory Ship Reporting System (MSR)
In April 1998, the USCG submitted, on behalf of the US, a proposal to the International Maritime Organization (IMO) requesting approval of a mandatory ship reporting system (MSR) in two areas off the east coast of the U.S., the right whale feeding grounds in the Northeast, and the right whale calving grounds in the Southeast. The USCG worked closely with NMFS and other agencies on technical aspects of the proposal. The package was submitted to the IMO's Subcommittee on Safety and Navigation for consideration. It was then submitted to the Marine Safety Committee at IMO and approved in December 1998. The USCG and NOAA play important roles in helping to operate the MSR system, which was implemented on July 1, 1999. Ships entering the northeast and southeast MSR boundaries are required to report the vessel identity, date, time, course, speed, destination, and other relevant information. In return, the vessel receives an automated reply with the most recent right whale sightings or management areas and information on precautionary measures to take while in the vicinity of right whales.

Vessel Speed Restrictions
A key component of NOAA's right whale ship strike reduction program is the implementation of speed restrictions for vessels transiting the US Atlantic in areas and seasons where right whales predictably occur in high concentrations. The Northeast Implementation Team (NEIT)-funded report "Recommended Measures to Reduce Ship Strikes of North Atlantic Right Whales" found that seasonal speed and routing measures could be an effective means of reducing the risk of ship strike along the U.S. East Coast. Based on these recommendations, NMFS published an Advance Notice of Proposed Rulemaking (ANPR) in June 2004 (69 FR 30857; June 1, 2004), and subsequently published a proposed rule on June 26, 2006 (71 FR 36299; June 26, 2006). NMFS published regulations on October 10, 2008 to implement a 10-knot speed restriction for all vessels 19.8 meters (65 feet) or longer in Seasonal Management Areas (SMAs) along the East Coast of the U.S. Atlantic seaboard at certain times of the year (73 FR 60173; October 10, 2008).

SMAs are supplemented by Dynamic Management Areas (DMAs) that are implemented for 15 day periods in areas in which right whales are sighted outside of SMA boundaries. When NOAA aerial surveys or other reliable sources report aggregations of three or more right whales in a density that indicates the whales are likely to persist in the area, NOAA calculates a buffer zone around the aggregation and announces the boundaries of the zone to mariners via various mariner communication outlets, including NOAA Weather Radio, USCG Broadcast Notice to

Mariners, MSR return messages, email distribution lists, and the Right Whale Sighting Advisory System (SAS). NOAA requests mariners route around these zones or transit through them at 10 knots or less. Compliance with these zones is voluntary.

The rule will expire five years from the date of effectiveness. NOAA is currently analyzing data on compliance with, and effectiveness of the rule since its implementation to determine the next steps as its expiration in December 2013 approaches.

Sighting Advisory System (SAS)
The right whale Sighting Advisory System (SAS) was initiated in early 1997 as a partnership among several federal and state agencies and other organizations to conduct aerial and ship board surveys to locate right whales and to alert mariners to right whale sighting locations in a near real time manner. The SAS surveys and opportunistic sightings reports document the presence of right whales and are provided to mariners via fax, email, NAVTEX, Broadcast Notice to Mariners, NOAA Weather Radio, several websites, and the Traffic Controllers at the Cape Cod Canal. Fishermen and other vessel operators can obtain SAS sighting reports, and make necessary adjustments in operations to decrease the potential for interactions with right whales. The SAS has also served as the only form of active entanglement monitoring in the Cape Cod Bay and Great South Channel feeding areas. Some of these sighting efforts have resulted in successful disentanglement of right whales. SAS flights have also contributed sightings of dead floating animals that can occasionally be retrieved to increase our knowledge of the biology of the species and effects of human impacts.

In 2009, with the implementation of the new ship strike regulations and the DMA program, the SAS alerts were modified to provide current SMA and DMA information to mariners on a weekly basis in an effort to maximize compliance with all active right whale protection zones.

Marine Mammal Health and Stranding Response Program (MMHSRP)
NMFS was designated the lead agency to coordinate the MMHSRP which was formalized by the 1992 Amendments to the MMPA. The program consists of the following components, all of which contribute important information on endangered large whales through stranding response and data collection:

- All coastal states established volunteer stranding networks and are authorized through Letters of Authority from NMFS regional offices to respond to marine mammal strandings.

- Biomonitoring helps assess the health and contaminant loads of marine mammals, but also to assist in determining anthropogenic impacts on marine mammals, marine food chains and marine ecosystem health.

- The Analytical Quality Assurance (AQA) was designed to ensure accuracy, precision, level or detection, and intercomparability of data in the chemical analyses of marine mammal tissue samples.

- NMFS established a Working Group on Marine Mammal Unusual Mortality Events to provide criteria to determine when a UME is occurring and how to direct responses to such events. The group meets annually to discuss many issues including recent mortality events involving endangered species both in the United States and abroad.

- The National Marine Mammal Tissue Bank provides protocols and techniques for the long-term storage of tissues from marine mammals for retrospective contaminant analyses. Additionally, a serum bank and long-term storage of histopathology tissue are being developed.

Magnuson-Stevens Fishery Conservation and Management Act
There are numerous regulations mandated by the Magnuson-Stevens Fishery Conservation and Management Act that may benefit ESA-listed species. Many fisheries are subject to different time and area closures. These area closures can be seasonal or year-round. Closure areas may benefit ESA-listed species due to elimination of active gear in areas where sea turtle and cetaceans are present. However, if closures shift effort to areas or seasons with a comparable or higher density of marine mammals or sea turtles, then risk of interaction could actually increase. Fishing effort reduction (*i.e.*, landing/possession limits or trap allocations) measures may also benefit ESA-listed species by limiting the amount of time that gear is present in the species environment. Additionally, gear restrictions and modifications required for fishing regulations may also decrease the risk of entanglement with endangered species. For a complete listing of fishery regulations in the action area visit: http://www.nero.noaa.gov/nero/regs/info.html.

6.3.2 Sea Turtles

Numerous efforts are ongoing to reduce threats to listed sea turtles. Below, we detail efforts that are ongoing within the action area. The majority of these activities are related to regulations that have been implemented to reduce the potential for incidental mortality of sea turtles from commercial fisheries. These include sea turtle release gear requirements for Atlantic HMS; TED requirements for Southeast shrimp trawl fishery and the southern part of the summer flounder trawl fishery; mesh size restrictions in the North Carolina gillnet fishery and Virginia's gillnet and pound net fisheries; modified leader requirements in the Virginia Chesapeake Bay pound net fishery; area closures in the North Carolina gillnet fishery; and gear modifications in the Atlantic sea scallop dredge fishery. In addition to regulations, outreach programs have been established and data on sea turtle interactions and strandings are collected. The summaries below discuss all of these measures in more detail.

Final Rules for Large-Mesh Gillnets
In March 2002, NMFS published new restrictions for the use of gillnets with larger than 8-inch (20.3 cm) stretched mesh, in Federal waters (3-200 nautical miles) off of North Carolina and Virginia. These restrictions were published in an interim final rule under the authority of the ESA (67 FR 13098) and were implemented to reduce the impact of the monkfish and other large-mesh gillnet fisheries on ESA-listed sea turtles in areas where sea turtles are known to concentrate. Following review of public comments submitted on the interim final rule, NMFS published a final rule on December 3, 2002, that established the restrictions on an annual basis. As a result, gillnets with larger than 8-inch (20.3 cm) stretched mesh are not allowed in Federal

waters (3-200 nautical miles) in the areas described as follows: (1) North of the North Carolina/South Carolina border at the coast to Oregon Inlet at all times; (2) north of Oregon Inlet to Currituck Beach Light, NC from March 16 through January 14; (3) north of Currituck Beach Light, NC, to Wachapreague Inlet, VA, from April 1 through January 14; and (4) north of Wachapreague Inlet, VA, to Chincoteague, VA, from April 16 through January 14. On April 26, 2006, NMFS published a final rule (71 FR 24776) that included modifications to the large-mesh gillnet restrictions. The new final rule revised the gillnet restrictions to apply to stretched mesh that is ≥7 inches (17.9 cm). Federal waters north of Chincoteague, VA, remain unaffected by the large-mesh gillnet restrictions. These measures are in addition to Harbor Porpoise Take Reduction Plan measures that prohibit the use of large-mesh gillnets in southern Mid-Atlantic waters (territorial and Federal waters from Delaware through North Carolina out to 72°30'W longitude) from February 15 through March 15, annually. The measures are also in addition to comparable North Carolina and Virginia regulations for large-mesh gillnet fisheries in their respective state waters that were enacted in 2005.

NMFS has also issued a rule addressing capture of sea turtles in gillnet gear fished in the southern flounder fishery in Pamlico Sound. NMFS issued a final rule (67 FR 56931), effective September 3, 2002, that closed the waters of Pamlico Sound, NC, to fishing with gillnets with larger than 4 ¼-inch (10.8 cm) stretched mesh from September 1 through December 15 each year to protect migrating sea turtles. The closed area includes all inshore waters of Pamlico Sound south of 35°46.3'N latitude, north of 35°00'N latitude, and east of 76°30'W longitude.

TED requirements for the summer flounder fishery

As mentioned above, significant measures have been developed to reduce the incidental take of sea turtles in summer flounder trawls and trawls that meet the definition of a summer flounder trawl (which would include fisheries for other species like scup and black sea bass) by requiring TEDs in trawl nets fished in trawls used in the area of greatest turtle bycatch off the North Carolina and part of the Virginia coast from North Carolina/South Carolina border to Cape Charles, Virginia. The TED requirements for the summer flounder trawl fishery do not, however, require the use of larger TEDs that are required to be used in the U.S. Southeast shrimp trawl fisheries.

HMS Sea Turtle Protection Measures

NMFS completed the most recent biological opinion on the FMP for the Atlantic HMS fisheries for tuna and swordfish on June 1, 2004, and concluded that the pelagic longline component of the fishery was likely to jeopardize the continued existence of leatherback sea turtles. An RPA was provided to avoid jeopardy to leatherback sea turtles as a result of the operation of this component of the fishery. The RPA was also expected to benefit loggerhead sea turtles by reducing the likelihood of mortality resulting from interactions with the gear. Regulatory components of the RPA have been implemented through rulemaking. Since 2004, bycatch estimates for both loggerheads and leatherbacks in pelagic longline gear have been well below the average prior to implementation of gear regulations under the RPA (Garrison *et al.* 2009).

Use of a Chain-Mat Modified Scallop Dredge in the Mid-Atlantic

In response to the observed capture of sea turtles in scallop dredge gear, including serious injuries and sea turtle mortality as a result of capture, NMFS proposed a modification to scallop dredge gear (70 FR 30660, May 27, 2005). The rule was finalized as proposed (71 FR 50361, August 25, 2006) and required federally permitted scallop vessels fishing with dredge gear to modify their gear by adding an arrangement of horizontal and vertical chains (hereafter referred to as a "chain mat") between the sweep and the cutting bar when fishing in Mid-Atlantic waters south of 41°9'N from the shoreline to the outer boundary of the EEZ during the period of May 1-November 30 each year. The requirement was subsequently modified by emergency rule on November 15, 2006 (71 FR 66466), and by a final rule published on April 8, 2008 (73 FR 18984). On May 5, 2009, NMFS proposed additional minor modifications to the regulations on how chain mats are configured (74 FR 20667). In general, the chain mat gear modification is expected to reduce the severity of some sea turtle interactions with scallop dredge gear. However, this modification is not expected to reduce the overall number of sea turtle interactions with scallop dredge gear.

Sea Turtle Handling and Resuscitation Techniques

NMFS has developed and published as a final rule in the *Federal Register* (66 FR 67495, December 31, 2001) sea turtle handling and resuscitation techniques for sea turtles that are incidentally caught during scientific research or fishing activities. Persons participating in fishing activities or scientific research are required to handle and resuscitate (as necessary) sea turtles as prescribed in the final rule. These measures help to prevent mortality of hard-shelled turtles caught in fishing or scientific research gear.

Sea Turtle Entanglements and Rehabilitation

A final rule (70 FR 42508) published on July 25, 2005, allows any agent or employee of NMFS, the USFWS, the U.S. Coast Guard, or any other Federal land or water management agency, or any agent or employee of a state agency responsible for fish and wildlife, when acting in the course of his or her official duties, to take endangered sea turtles encountered in the marine environment if such taking is necessary to aid a sick, injured, or entangled endangered sea turtle, or dispose of a dead endangered sea turtle, or salvage a dead endangered sea turtle that may be useful for scientific or educational purposes. NMFS already affords the same protection to sea turtles listed as threatened under the ESA (50 CFR 223.206(b)).

Education and Outreach Activities

Education and outreach activities do not directly reduce the threats to ESA-listed sea turtles. However, education and outreach are a means of better informing the public of steps that can be taken to reduce impacts to sea turtles (*i.e.*, reducing light pollution in the vicinity of nesting beaches) and increasing communication between affected user groups (*e.g.*, the fishing community). For the HMS fishery, NMFS has been active in public outreach to educate fishermen regarding sea turtle handling and resuscitation techniques. For example, NMFS has conducted workshops with longline fishermen to discuss bycatch issues including protected species, and to educate them regarding handling and release guidelines. NMFS intends to continue these outreach efforts in an attempt to increase the survival of protected species through education on proper release techniques.

Sea Turtle Stranding and Salvage Network (STSSN)
As is the case with education and outreach, the STSSN does not directly reduce the threats to sea turtles. However, the extensive network of STSSN participants along the Atlantic and Gulf of Mexico coasts not only collects data on dead sea turtles, but also rescues and rehabilitates live stranded turtles. Data collected by the STSSN are used to monitor stranding levels and identify areas where unusual or elevated mortality is occurring. These data are also used to monitor incidence of disease, study toxicology and contaminants, and conduct genetic studies to determine population structure. All of the states that participate in the STSSN tag live turtles when encountered (either via the stranding network through incidental takes or in-water studies). Tagging studies help provide an understanding of sea turtle movements, longevity, and reproductive patterns, all of which contribute to our ability to reach recovery goals for the species.

6.3.3 Reducing Threats to Atlantic sturgeon

Several conservation actions aimed at reducing threats to Atlantic sturgeon are currently ongoing. In the near future, NMFS will be convening a recovery team and will be drafting a recovery plan which will outline recovery goals and criteria and steps necessary to recover all Atlantic sturgeon DPSs. Numerous research activities are underway, involving NMFS and other Federal, State and academic partners, to obtain more information on the distribution and abundance of Atlantic sturgeon throughout their range, including in the action area, and to develop population estimates for each DPS. Efforts are also underway to better understand threats faced by the DPSs and ways to minimize these threats, including bycatch and water quality. Fishing gear research is underway to design fishing gear that minimizes interactions with Atlantic sturgeon while maximizing retention of targeted fish species. Several states are in the process of preparing ESA Section 10 Habitat Conservation Plans aimed at minimizing the effects of state fisheries on Atlantic sturgeon.

7.0 EFFECTS OF THE ACTION
In order to assess the potential effects of BOEM's issuance of renewable energy leases in the WEAs and approval of SAPs and the carrying out of site assessment activities by lessees, we assessed the likelihood that listed species or designated critical habitat, if present in the action area would be affected by the proposed actions considered in this consultation. We have considered the scenarios outlined in BOEM's EA and BA which considers 100% coverage of the WEAs by leases, surveys sufficient to cover the lease areas, and the construction of up to 4 met towers in the RI/MA WEA, 5 met towers in the MA WEA and the installation of up to 8 met buoys in the RI/MA WEA and 10 met buoys in the MA WEA. Any activities that exceed this amount would be considered to be outside the scope of this consultation. Additionally, any proposal to conduct activities that is not wholly consistent with the activities described herein or is not wholly consistent with the PDCs outlined above, would not be considered to be eligible for coverage under this consultation and would require a separate section 7 consultation.

The proposed action involves several stages of activity. The sections below will outline potential effects from the following sources: (1) pre-construction surveys; (2) installation of the met tower foundations and construction of the met tower; (3) installation and operation of met buoys; (4) operation of the met towers, and; (5) decommissioning. In addition to these categories of effects, BOEM provided information in the BA on non-routine and accidental events. These events include oil spills, vessel collisions with a met tower and destructive natural events. Effects of these non-routine and accidental events are also discussed below. Potential effects of the proposed action can be broadly categorized into the following categories: (1) acoustic effects, (2) effects to benthic habitat, (3) effects of an increase in vessel traffic, (4) effects of met tower and met buoy operation, (5) effects of non-routine and accidental events, and (6) effects of decommissioning. The effects analysis will be organized along these topics.

As explained above, BOEM's proposed action would not authorize the construction or operation of any electricity generating facility (i.e., wind energy facility) or transmission cables with the potential to export electricity; thus, this consultation does not consider the effects of any future potential construction or operation of a wind facility or associated transmission equipment. It is also important to note that the Project Design Criteria noted above (see section 3.6) will be required by BOEM as conditions of their approval of any action considered under this consultation; therefore, they are considered part of the proposed action.

7.1 Acoustic Effects

Sources of noise associated with the proposed action include pile driving, construction and maintenance vessel transits and the geotechnical and geophysical survey equipment.

Frequency (i.e., number of cycles per unit of time, with hertz (Hz) as the unit of measurement) and amplitude (loudness, measured in decibels, or dB) are the measures typically used to describe sound. An acoustic field from any source consists of a propagating pressure wave, generated from particle motions in the medium that causes compression and rarefaction. This sound wave consists of both pressure and particle motion components that propagate from the source. Sound in water follows the same physical principles as sound in air. The major difference is that due to the density of water, sound in water travels about 4.5 times faster than in air (approx. 4900 feet/s vs. 1100 feet/s), and attenuates much less rapidly than in air. As a result of the greater speed, the wavelength of a particular sound frequency is about 4.5 times longer in water than in air (Rogers and Cox 1988; Bass and Clarke 2003).

The level of a sound in water can be expressed in several different ways, but always in terms of dB relative to 1 micro-Pascal (µPa). Decibels are a log scale; each 10 dB increase is a ten-fold increase in sound pressure. Accordingly, a 10 dB increase is a 10x increase in sound pressure, and a 20 dB increase is a 100x increase in sound pressure.

The following are commonly used measures of sound:

- Peak sound pressure level (SPL): the maximum sound pressure level (highest level of sound) in a signal measured in dB re 1 µPa.

- Sound exposure level (SEL): the integral of the squared sound pressure over the duration of the pulse (e.g., a full pile driving strike.) SEL is the integration over time of the square of the acoustic pressure in the signal and is thus an indication of the total acoustic energy received by an organism from a particular source (such as pile strikes).

Measured in dB re 1μPa^2-s.

- Single Strike SEL: the amount of energy in one strike of a pile.

- Cumulative SEL (cSEL or SEL_{cum}): the energy accumulated over multiple strikes. cSEL indicates the full energy to which an animal is exposed during any kind of signal. The rapidity with which the cSEL accumulates depends on the level of the single strike SEL. The actual level of accumulated energy (cSEL) is the logarithmic sum of the total number of single strike SELs. Thus, cSEL (dB) = Single-strike SEL + $10\log_{10}(N)$; where N is the number of strikes.

- Root Mean Square (RMS): the average level of a sound signal over a specific period of time.

7.1.1 Background Information on Acoustics and Marine Mammals and Sea Turtles

When anthropogenic disturbances elicit responses from sea turtles and marine mammals, it is not always clear whether they are responding to visual stimuli, the physical presence of humans or man-made structures, or acoustic stimuli. However, because sound travels well underwater, it is reasonable to assume that, in many conditions, marine organisms would be able to detect sounds from anthropogenic activities before receiving visual stimuli. As such, exploring the acoustic effects of the proposed project provides a reasonable and conservative estimate of the magnitude of disturbance caused by the general presence of a manmade, industrial structure in the marine environment, as well as effects of sound on marine mammal and sea turtle behavior.

Marine organisms rely on sound to communicate with conspecifics and derive information about their environment. There is growing concern about the effect of increasing ocean noise levels due to anthropogenic sources on marine organisms, particularly marine mammals. Effects of noise exposure on marine organisms can be characterized by the following range of physical and behavioral responses (Richardson et al. 1995):

1. Behavioral reactions – Range from brief startle responses, to changes or interruptions in feeding, diving, or respiratory patterns, to cessation of vocalizations, to temporary or permanent displacement from habitat.
2. Masking – Reduction in ability to detect communication or other relevant sound signals due to elevated levels of background noise.
3. Temporary threshold shift (TTS) – Temporary, fully recoverable reduction in hearing sensitivity caused by exposure to sound.
4. Permanent threshold shift (PTS) – Permanent, irreversible reduction in hearing sensitivity due to damage or injury to ear structures caused by prolonged exposure to sound or temporary exposure to very intense sound.
5. Non-auditory physiological effects – Effects of sound exposure on tissues in non-auditory systems either through direct exposure or as a consequence of changes in behavior, e.g., resonance of respiratory cavities or growth of gas bubbles in body fluids.

Several components of the proposed action will produce sound that may affect listed sea turtles and whales. NMFS is in the process of developing a comprehensive acoustic policy that will provide guidance on managing sources of anthropogenic sound based on each species' sensitivity to different frequency ranges and intensities of sound. The available information on the hearing capabilities of cetaceans and the mechanisms they use for receiving and interpreting sounds remains limited due to the difficulties associated with conducting field studies on these animals. However, current thresholds for determining impacts to marine mammals typically center around root-mean-square (RMS) received levels of 180 dB re 1μPa for potential injury, 160 dB re 1μPa for behavioral disturbance/harassment from a non-continuous noise source, and 120 dB re 1μPa for behavioral disturbance/harassment from a continuous noise source. These thresholds are based on a limited number of experimental studies on captive odontocetes, a limited number of controlled field studies on wild marine mammals, observations of marine mammal behavior in the wild, and inferences from studies of hearing in terrestrial mammals. In addition, marine mammal responses to sound can be highly variable, depending on the individual hearing sensitivity of the animal, the behavioral or motivational state at the time of exposure, past exposure to the noise which may have caused habituation or sensitization, demographic factors, habitat characteristics, environmental factors that affect sound transmission, and non-acoustic characteristics of the sound source, such as whether it is stationary or moving (NRC 2003). Nonetheless, the threshold levels referred to above are considered conservative based on the best available scientific information at this time and will be used in the analysis of effects for this consultation.

Right, Humpback, Fin, Sei and Sperm Whale Hearing

In order for whales to be adversely affected by construction noise, they must be able to perceive the noises produced by the activities. If a species cannot hear a sound, or hears it poorly, then the sound is unlikely to have a significant effect (Ketten 1998). Baleen whale hearing has not been studied directly, and there are no specific data on sensitivity, frequency or intensity discrimination, or localization (Richardson et al. 1995) for these whales. Thus, predictions about probable impact on baleen whales are based on assumptions about their hearing rather than actual studies of their hearing (Richardson et al. 1995; Ketten 1998).

Ketten (1998) summarized that the vocalizations of most animals are tightly linked to their peak hearing sensitivity. Hence, it is generally assumed that baleen whales hear in the same range as their typical vocalizations, even though there are no direct data from hearing tests on any baleen whale. Most baleen whale sounds are concentrated at frequencies less than 1 kHz (Richardson et al. 1995), although humpback whales can produce songs up to 8 kHz (Payne and Payne 1985). Based on indirect evidence, at least some baleen whales are quite sensitive to frequencies below 1 kHz but can hear sounds up to a considerably higher but unknown frequency. Most of the manmade sounds that elicited reactions by baleen whales were at frequencies below 1 kHz (Richardson et al. 1995). Some or all baleen whales may hear infrasounds, sounds at frequencies well below those detectable by humans. Functional models indicate that the functional hearing of baleen whales extends to 20 Hz, with an upper range of 30 Hz. Even if the range of sensitive hearing does not extend below 20-50 Hz, whales may hear strong infrasounds at considerably lower frequencies. Based on work with other marine mammals, if hearing sensitivity is good at 50 Hz, strong infrasounds at 5 Hz might be detected (Richardson et al. 1995). Fin whales are predicted to hear at frequencies as low as 10-15 Hz. The right whale uses tonal signals in the frequency range from roughly 20 to 1000 Hz, with broadband source levels ranging from 137 to

162 dB (RMS) re 1 μPa at 1 m (Parks & Tyack 2005). One of the more common sounds made by right whales is the "up call," a frequency-modulated upsweep in the 50–200 Hz range (Mellinger 2004). The following table summarizes the range of sounds produced by right, humpback, fin, sei and sperm whales (from Au et al. 2000):

Table 1. Summary of known right, humpback, fin, sei and sperm whale vocalizations

Species	Signal type	Frequency Limits (Hz)	Dominant Frequencies (Hz)	Source Level (dB re 1μPa RMS)	References
North Atlantic Right	Moans Tonal Gunshots	< 400 20-1000	-- 100-2500 50-2000	-- 137-162 174-192	Watkins and Schevill (1972) Parks and Tyack (2005) Parks et al. (2005)
Humpback	Grunts Pulses Songs	25-1900 25-89 30-8000	25-1900 25-80 120-4000	-- 176 144-174	Thompson, Cummings, and Ha (1986) Thompson, Cummings, and Ha (1986) Payne and Payne (1985)
Fin	FM moans Tonal Songs	14-118 34-150 17-25	20 34-150 17-25	160-186 186	Watkins (1981), Edds (1988), Cummings and Thompson (1994) Edds (1988) Watkins (1981)
Sei	FM Sweeps	1500-3500	-	-	T. Thompson et al. 1979; Knowlton et al. 1991
Sperm	Clicks	0.1-30 kHz 5-20 kHz	2-4 kHz 10-16 kHz	160-180	Backus & Shevill 1996; Levenson 1974; Watkins 1980; Ridgeway & Carter 2001

Most species also have the ability to hear beyond their region of best sensitivity. This broader range of hearing probably is most likely related to their need to detect other important environmental phenomena, such as the locations of predators or prey. Among marine mammal species, considerable variation exists in hearing sensitivity and absolute hearing range (Richardson et al. 1995; Ketten 1998). However, from what is known of right, humpback, sei, sperm and fin whale hearing and the source levels and dominant frequencies of the site assessment and characterization activities, it is expected that if these whales are present in the area where the underwater noise occurs they would be capable of perceiving those noises. The baleen whales have hearing ranges that are likely to have peak sensitivities with low frequencies (below 1 kHz) while the sperm whale is characterized as a mid-frequency cetacean (above 1 kHz) that overlap with frequencies of site assessment and site characterization sounds. We assume that sources with frequencies above 200 kHz are not perceived by these species.

Criteria for Assessing Effects to Listed Whales

NOAA is developing comprehensive guidance on sound characteristics likely to cause injury and behavioral disruption in the context of the MMPA, ESA and other statutes. Until formal guidance is available, we will use conservative thresholds of received sound pressure levels from broad band sounds that may cause behavioral disturbance and injury. These conservative thresholds are applied in MMPA permits and ESA Section 7 consultations, including this Opinion, for marine mammals to evaluate the potential for sound effects. The criterion levels specified below are specific to the levels of harassment permitted under the MMPA.

Criterion	Criterion Definition	Threshold
Level A	PTS (injury) conservatively based on TTS	190 dB_{rms} for pinnipeds 180 dB_{rms} for cetaceans
Level B	Behavioral disruption for impulsive noise (e.g., impact pile driving)	160 dB_{rms}
Level B	Behavioral disruption for non-pulse noise (e.g., vibratory pile driving, drilling)	120* dB_{rms}

All decibels referenced to 1 micro Pascal (re: 1μPa). Note all thresholds are based off root mean square (rms) levels.
* The 120 dB threshold may be slightly adjusted if background noise levels are at or above this level.

As such, we would consider there to be a potential for injury to any whales exposed to pile driving or G&G survey noise greater than 180 dB re 1uPa RMS and would consider there to be a potential for behavioral disruption to any whales exposed to pile driving or geophysical survey noise (impulsive sources) greater than 160 dB re 1uPa RMS or geotechnical drilling (continuous noise) greater than 120 dB re 1uPa RMS. Under the 1994 Amendments to the MMPA, harassment is statutorily defined as, any act of pursuit, torment, or annoyance which--

(Level A Harassment) has the potential to injure a marine mammal or marine mammal stock in the wild; or,

(Level B Harassment) has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering but which does not have the potential to injure a marine mammal or marine mammal stock in the wild.

Sea Turtle Hearing

The hearing capabilities of sea turtles are poorly known. Few experimental data exist, and since sea turtles do not vocalize, inferences cannot be made from their vocalizations as is the case with baleen whales. Direct hearing measurements have been made in only a few species. The limited information available suggests that the auditory capabilities of sea turtles are centered in the low frequency range (<1 kHz) (Ridgway *et al.* 1969; Lenhardt *et al.* 1983; Bartol *et al.* 1999, Lenhardt 1994, O'Hara and Wilcox 1990). An early experiment measured cochlear potential in three Pacific green turtles and suggested a best hearing sensitivity in air of 300–500 Hz and an effective hearing range of 60–1,000 Hz (Ridgway et al. 1969). Sea turtle underwater hearing is believed to be about 10 dB less sensitive than their in-air hearing (Lenhardt 1994). Lenhardt et al. (1996) used a behavioral "acoustic startle response" to measure the underwater hearing sensitivity of a juvenile Kemp's ridley and a juvenile loggerhead turtle to a 430-Hz tone. Their results suggest that those species have a hearing sensitivity at a frequency similar to those of the green turtles studied by Ridgway et al. (1969). Lenhardt (1994) was also able to induce startle responses in loggerhead turtles to low frequency (20–80 Hz) sounds projected into their tank. He suggested that sea turtles have a range of best hearing from 100–800 Hz, an upper limit of about 2,000 Hz, and serviceable hearing abilities below 80 Hz. More recently, the hearing abilities of loggerhead sea turtles were measured using auditory evoked potentials in 35 juvenile animals caught in tributaries of Chesapeake Bay (Bartol et al. 1999). Those experiments suggest that the effective hearing range of the loggerhead sea turtle is 250–750 Hz and that its most sensitive hearing is at 250 Hz. In general, however, these experiments indicate that sea turtles generally hear best at low frequencies and that the upper frequency limit of their hearing is likely about 1 kHz.

Ridgway *et al*. (1969) studied the auditory evoked potentials of three green sea turtles (in air and through mechanical stimulation of the ear) and concluded that their maximum sensitivity occurred from 300 to 400 Hz with rapid declines for tones at lower and higher frequencies. They reported an upper limit for cochlear potentials without injury of 2000 Hz and a practical limit of about 1000 Hz. This is similar to estimates for loggerhead sea turtles, which had most sensitive hearing between 250 and 1000 Hz, with rapid decline above 1000 Hz (Bartol *et al*. 1999). These hearing sensitivities are similar to the hearing sensitivities reported for two terrestrial species: pond turtles (*Pseudemys scripta*) and wood turtles (*Chrysemys inscuplta*). Pond turtles are reported to have best hearing responsiveness between 200 and 700 Hz, with slow declines below 100 Hz and rapid declines above 700 Hz and almost no sensitivity above 3000 Hz (Wever and Vernon 1956). Wood turtles have sensitivities up to about 500 Hz, followed by a rapid decline above 1000 Hz and almost no responses beyond 3000 or 4000 Hz (Peterson 1966). We assume that these sensitivities to sound apply to all of the sea turtles in the action area (i.e., the green, hawksbill, Kemp's ridley, leatherback and loggerhead sea turtles).

A study on the effects of airguns on sea turtle behavior also suggests that sea turtles are most likely to respond to low-frequency sounds. McCauley *et al*. (2000) reported that green and loggerhead sea turtles will avoid air-gun arrays at 2 km and at 1 km with received levels of 166 dB re 1 Pa and 175 dB re 1 uPa, respectively. The sea turtles responded consistently: above a level of approximately 166 dB re 1 uPa RMS the turtles noticeably increased their swimming activity compared to non-airgun operation periods. Above 175 dB re 1 Pa mean squared pressure their behavior became more erratic possibly indicating the turtles were in an agitated state.

Criteria for Assessing Effects to Sea Turtles
Currently there are no established thresholds for injury or behavioral disturbance/harassment for sea turtles. As noted above, the hearing capabilities of sea turtles are poorly known and there is little available information on the effects of noise on sea turtles; however, McCauley *et al.* (2000) noted that decibel levels of 166 dB re 1 µPa RMS were required before any behavioral reaction (e.g., increased swimming speed) was observed, and decibel levels above 175 dB re 1 µPa RMS elicited avoidance behavior of sea turtles. Based on this and the best available information, NMFS believes any sea turtles exposed to underwater noise greater than 166 dB re 1uPa RMS may experience behavioral disturbance. While there is some information suggesting the noise levels that might result in injury to sea turtles from exposure to underwater explosives, no such information is available for pile driving or low frequency geophysical survey equipment, such as the boomer. However, all available information indicates that injury is not expected upon exposure to impulsive noises less than 180 dB re 1uPa RMS. Therefore, we do not anticipate any injury to sea turtles exposed to impulsive noise, such as the boomer and pile driving, that is less than 180 dB re 1uPa RMS.

7.1.2 Basic Background on Fish Bioacoustics

7.1.2.1 Summary of Available Information on Underwater Noise and Sturgeon

Sturgeon have swim bladders, but they are not located very close to the ear; thus, they are assumed to detect primarily particle motion rather than pressure. Sturgeon rely primarily on particle motion to detect sounds (Lovell *et al.* 2005). While there are no data both in terms of hearing sensitivity and structure of the auditory system for shortnose or Atlantic sturgeon, there are data for the closely related lake sturgeon (Lovell *et al.* 2005; Meyer *et al.* 2010), which for the purpose of considering acoustic impacts can be considered as a surrogate for Atlantic sturgeon. The available data suggest that lake sturgeon can hear sounds from below 100 Hz to 800 Hz (Lovell *et al.* 2005; Meyer *et al.* 2010). However, since these two studies examined responses of the ear and did not examine whether fish would behaviorally respond to sounds detected by the ear, it is hard to determine thresholds for hearing (that is, the lowest sound levels that an animal can hear at a particular frequency) using information from these studies.

The swim bladder of sturgeon is relatively small compared to other species (Beregi *et al.* 2001). While there are no data that correlate effects of noise on fishes and swim bladder size, the potential for damage to body tissues from rapid expansion of the swim bladder likely is reduced in a fish where the structure occupies less of the body cavity, and, thus, is in contact with less body tissue. Although there are no experimental data that enable one to predict the potential effects of sound on sturgeon, the physiological effects of impulsive noises, such as pile driving, on sturgeon may actually be less than on other species due to the small size of their swim bladder.

Sound is an important source of environmental information for most vertebrates (e.g., Fay and Popper, 2000). Fish are thought to use sound to learn about their general environment, the presence of predators and prey, and, for some species, for acoustic communication. As a consequence, sound is important for fish survival, and anything that impedes the ability of fish to detect a biologically relevant sound could affect individual fish.

Richardson *et al.* (1995) defined different zones around a sound source that could result in different types of effects on fish. There are a variety of different potential effects from any sound, with a decreasing range of effects at greater distances from the source. Thus, very close to the source, effects may range from mortality to behavioral changes. Somewhat further from the source mortality is no longer an issue, and effects range from physiological to behavioral. As one gets even further, the potential for effects declines. The actual nature of effects, and the distance from the source at which they could be experienced will vary and depend on a large number of factors, such as fish hearing sensitivity, source level, how the sounds propagate away from the source and the resultant sound level at the fish, whether the fish stays in the vicinity of the source, the motivation level of the fish, etc.

Underwater sound pressure waves can injure or kill fish (Reyff 2003, Abbott and Bing-Sawyer 2002, Caltrans 2001, Longmuir and Lively 2001, Stotz and Colby 2001). Fish with swim bladders, including shortnose and Atlantic sturgeon are particularly sensitive to underwater impulsive sounds with a sharp sound pressure peak occurring in a short interval of time (Caltrans 2001). As the pressure wave passes through a fish, the swim bladder is rapidly squeezed due to the high pressure, and then rapidly expanded as the under pressure component of the wave passes through the fish. The pneumatic pounding on tissues contacting the swim bladder may rupture capillaries in the internal organs as indicated by observed blood in the abdominal cavity, and maceration of the kidney tissues (Caltrans 2001).

There are limited data from other projects to demonstrate the circumstances under which immediate mortality occurs: mortality appears to occur when fish are close (within a few feet to 30 feet) to driving of relatively large diameter piles. Studies conducted by California Department of Transportation (Caltrans, 2001) showed some mortality for several different species of wild fish exposed to driving of steel pipe piles 8 feet in diameter, whereas Ruggerone *et al.* (2008) found no mortality to caged yearling coho salmon (*Oncorhynchus kisutch*) placed as close as 2 feet from a 1.5 foot diameter pile and exposed to over 1,600 strikes. As noted above, species are thought to have different tolerances to noise and may exhibit different responses to the same noise source.

Physiological effects that could potentially result in mortality may also occur upon sound exposure as could minor physiological effects that would have no effect on fish survival. Potential physiological effects are highly diverse, and range from very small ruptures of capillaries in fins (which are not likely to have any effect on survival) to severe hemorrhaging of major organ systems such as the liver, kidney, or brain (Stephenson *et al.*, 2010). Other potential effects include rupture of the swim bladder (the bubble of air in the abdominal cavity of most fish species that is involved in maintenance of buoyancy). See Halvorsen *et al.* 2011 for a review of potential injuries from pile driving.

Effects on body tissues may result from barotrauma or result from rapid oscillations of air bubbles. Barotrauma occurs when there is a rapid change in pressure that directly affects the body gasses. Gas in the swim bladder, blood, and tissue of fish can experience a change in state, expand and contract during rapid pressure changes, which can lead to tissue damage and organ failure (Stephenson *et al.* 2010).

Related to this are changes that result from very rapid and substantial excursions (oscillations) of the walls of air-filled chambers, such as the swim bladder, striking near-by structures. Under normal circumstances the walls of the swim bladder do not move very far during changes in depth or when impinged upon by normal sounds. However, very intense sounds, and particularly those with very sharp onsets (also called "rise time") will cause the swim bladder walls to move much greater distances and thereby strike near-by tissues such as the kidney or liver. Rapid and frequent striking (as during one or more sound exposures) can result in bruising, and ultimately in damage, to the nearby tissues.

There is some evidence to suggest that very intense signals may not necessarily have substantial physiological effects and that the extent of effect will vary depending on a number of factors including sound level, rise time of the signal, duration of the signal, signal intensity, etc. For example, investigations on the effects of very high intensity sonar showed no damage to ears and other tissues of several different fish species (Kane *et al.* 2010). Some studies involving exposure of fish to sounds from seismic air guns, signal sources that have very sharp onset times, as found in pile driving, also did not result in any tissue damage (Popper *et al.* 2007; Song *et al.* 2008). However, the extent that results from one study are comparable to another is difficult to determine due to difference in species, individuals, and experimental design. Recent studies of the effects of pile driving sounds on fish showed that there is a clear relationship between onset of physiological effects and single strike and cumulative sound exposure level, and that the initial effects are very small and would not harm an animal (and from which there is rapid and complete recovery), whereas the most intense signals (e.g., >210 dB cumulative SEL) may result in tissue damage that could have long-term mortal effects (Halvorsen *et al.* 2011; Casper *et al.* 2011, in prep.)

7.1.2.2 Criteria for Assessing the Potential for Physiological Effects

The Fisheries Hydroacoustic Working Group (FHWG) was formed in 2004 and consists of biologists from NMFS, USFWS, FHWA, and the California, Washington and Oregon DOTs, supported by national experts on sound propagation activities that affect fish and wildlife species of concern. In June 2008, the agencies signed an MOA documenting criteria for assessing physiological effects of pile driving on fish. The criteria were developed for the acoustic levels at which physiological effects to fish could be expected. It should be noted, that these are onset of physiological effects (Stadler and Woodbury, 2009), and not levels at which fish are necessarily mortally damaged. These criteria were developed to apply to all species, including listed green sturgeon, which are biologically similar to Atlantic sturgeon and for these purposes can be considered a surrogate. The interim criteria are:

- Peak SPL: 206 decibels relative to 1 micro-Pascal (dB re 1 µPa).
- cSEL: 187 decibels relative to 1 micro-Pascal-squared second (dB re 1µPa^2-s) for fishes above 2 grams (0.07 ounces).
- cSEL: 183 dB re 1µPa^2-s for fishes below 2 grams (0.07 ounces).

NMFS has relied on these criteria in determining the potential for physiological effects in ESA Section 7 consultations related to pile driving conducted on the US West Coast. At this time, they represent the best available information on the thresholds at which physiological effects to sturgeon are likely to occur. It is important to note that physiological effects may range from minor injuries from which individuals are anticipated to completely recover with no impact to fitness to significant injuries that will lead to death. The severity of injury is related to the distance from the pile being installed and the duration of exposure. The closer to the source and the greater the duration of the exposure, the higher likelihood of significant injury.

A recent peer-reviewed study from the Transportation Research Board (TRB) of the National Research Council of the National Academies of Science describes a carefully controlled experimental study of the effects of pile driving sounds on fish (Halvorsen *et al.* 2011). This investigation documented effects of pile driving sounds (recorded by actual pile driving operations) under simulated free-field acoustic conditions where fish could be exposed to signals that were precisely controlled in terms of number of strikes, strike intensity, and other parameters. The study used Chinook salmon and determined that onset of physiological effects that have the potential of reduced fitness, and thus a potential effect on survival, started at above 210 dB re 1 µPa^2·s cSEL. Smaller injuries, such as ruptured capillaries near the fins, which the authors noted were not expected to impact fitness, occurred at lower noise levels. The peak noise level that resulted in physiological effects was about the same as the FHWG criteria.

Based on the available information, for the purposes of this Opinion, we consider the potential for physiological effects upon exposure to pile driving noise of 206dB re 1 µPa peak and 187 dB re 1 µPa^2-s cSEL. Use of the 183 dB re 1 µPa^2-s cSEL threshold, is not appropriate for this consultation because all Atlantic sturgeon in the action area will be larger than 2 grams. As explained here, physiological effects could range from minor injuries that a fish is expected to completely recover from with no impairment to survival to major injuries that increase the potential for mortality, or result in death.

7.1.2.3 Available Information for Assessing Behavioral Effects

Results of empirical studies of hearing of fishes, amphibians, birds, and mammals (including humans), in general, show that behavioral responses vary substantially, even within a single species, depending on a wide range of factors, such as the motivation of an animal at a particular time, the nature of other activities that the animal is engaged in when it detects a new stimulus, the hearing capabilities of an animal or species, and numerous other factors (Brumm and Slabbekoorn 2005). Thus, it may be difficult to assign a single criterion above which behavioral responses to noise would occur.

In order to be detected, a sound must be above the "background" level. Additionally, results from some studies suggest that sound may need to be biologically relevant to an individual to elicit a behavioral response. For example, in an experiment on responses of American shad to sounds produced by their predators (dolphins), it was found that if the predator sound is detectable, but not very loud, the shad will not respond (Plachta and Popper 2003). But, if the sound level is raised an additional 8 or 10 dB, the fish will turn and move away from the sound source. Finally, if the sound is made even louder, as if a predator were nearby, the American shad go into a frenzied series of motions that probably helps them avoid being caught. It was speculated by the researchers that the lowest sound levels were those recognized by the American shad as being from very distant predators, and thus, not worth a response. At somewhat higher levels, the shad recognized that the predator was closer and then started to swim away. Finally, the loudest sound was thought to indicate a very near-by predator, eliciting maximum response to avoid predation. Similarly, results from Doksaeter *et al.* (2009) suggest that fish will only respond to sounds that are of biological relevance to them. This study showed no responses by free-swimming herring (*Clupea* spp.) when exposed to sonars produced by naval vessels; but, sounds at the same received level produced by major predators of the herring (killer whales) elicited strong flight responses. Sound levels at the fishes from the sonar in this experiment were from 197 dB to 209 dB (rms) re 1 µPa at 1,000 to 2,000Hz.

For purposes of assessing behavioral effects of pile driving at several West Coast projects, NMFS has employed a 150dB re 1 µPa RMS SPL criterion at several sites including the San Francisco-Oakland Bay Bridge and the Columbia River Crossings. For the purposes of this consultation we will use 150 dB re 1 µPa RMS as a conservative indicator of the noise level at which there is the potential for behavioral effects. That is not to say that exposure to noise levels of 150 dB re 1 µPa RMS will always result in behavioral modifications or that any behavioral modifications will rise to the level of "take" (i.e., harm or harassment) but that there is the potential, upon exposure to noise at this level, to experience some behavioral response. Behavioral responses could range from a temporary startle to avoidance of an ensonified area.

As hearing generalists, sturgeon rely primarily on particle motion to detect sounds (Lovell *et al.* 2005), which does not propagate as far from the sound source as does pressure. However, a clear threshold for particle motion was not provided in the Lovell study. In addition, flanking of the sounds through the substrate may result in higher levels of particle motion at greater distances than would be expected from the non-flanking sounds. Unfortunately, data on particle motion from pile driving is not available at this time, and we are forced to rely on sound pressure level criteria. Although we agree that more research is needed, the studies noted above support the 150 dB re 1 µPa RMS criterion as an indication for when behavioral effects could be expected. We are not aware of any studies that have considered the behavior of shortnose or Atlantic sturgeon in response to pile driving noise. However, given the available information from studies on other fish species, we consider 150 dB re 1 µPa RMS to be a reasonable estimate of the noise level at which exposure may result in behavioral modifications.

Unfortunately, there is not an extensive body of literature on effects of anthropogenic sounds on fish behavior, and even fewer studies on effects of pile driving, and many of these were conducted under conditions that make the interpretation of the results uncertain. The most information is available for seismic airguns; the air gun sound spectrum is reasonably similar to that of pile driving. The results of the studies, summarized below, suggest that there is a potential for underwater sound of certain levels and frequencies to affect behavior of fish, but that it varies with fish species and the existing hydroacoustic environment. In addition, behavioral response may change over time as fish individuals habituate to the presence of the sound. Behavioral responses to other noise sources, such as noise associated with vessel traffic, and the results of noise deterrent studies, are also summarized below.

Mueller-Blenke *et al.* (2010), attempted to evaluate response of Atlantic cod (*Gadus morhua*) and Dover sole (*Solea solea*) held in large pens to playbacks of pile driving sounds recorded during construction of Danish wind farms. The investigators reported that a few representatives of both species exhibited some movement response, reported as increased swimming speed or freezing to the pile-driving stimulus at peak sound pressure levels ranging from 144 to 156 dB re 1 µPa for sole and 140 to 161 dB re 1 µPa for cod. These results must be interpreted cautiously as fish position was not able to be determined more frequently than once every 80 seconds.

Feist (1991) examined the responses of juvenile pink (*Oncorhyncus gorbuscha*) and chum (*O. keta*) salmon behavior during pile driving operations. Feist had observers watching fish schools in less than 1.5 m water depth and within 2 m of the shore over the course of a pile driving operation. The report gave limited information on the types of piles being installed and did not give pile size. Feist did report that there were changes in distribution of schools at up to 300 m from the pile driving operation, but that of the 973 schools observed, only one showed any overt startle or escape reaction to the onset of a pile strike. There was no statistical difference in the number of schools in the area on days with and without pile driving, although other behaviors changed somewhat.

Any analysis of the Feist data is complicated by a lack of data on pile type, size and source sound level. Without this data, it is very difficult to use the Feist data to help understand how fish would respond to pile driving and whether such sounds could result in avoidance or other behaviors. It is interesting to note that the size of the stocks of salmon never changed, but appeared to be transient, suggesting that normal fish behavior of moving through the study area was taking place no differently during pile driving operations than in quiet periods. This may suggest that the fish observed during the study were not avoiding pile driving operations.

Andersson *et al.* (2007) presents information on the response of sticklebacks (*Gasterosteus aculeatus*), a hearing generalist, to pure tones and broadband sounds from wind farm operations. Sticklebacks responded by freezing in place and exhibiting startle responses at SPLs of 120 dB (re: 1µPa) and less. Purser and Radford (2011) examined the response of three-spined sticklebacks to short and long duration white noise. This exposure resulted in increased startle responses and reduced foraging efficiency, although they did not reduce the total number of prey ingested. Foraging was less efficient due to attacks on non-food items and missed attacks on food items. The SPL of the white noise was reported to be similar (at frequencies between 100 and 1000 Hz) to the noise environment in a shoreline area with recreational speedboat activity. While this does not allow a comparison to the 150 dB re 1 µPa RMS guideline, it does demonstrate that significant noise-induced effects on behavior are possible, and that behaviors other than avoidance can occur.

Several of the studies (Andersson *et al.* 2007, Purser and Radford 2011, Wysocki *et al.* 2007) support our use of the 150 dB re 1 µPa RMS as a threshold for examining the potential for behavioral responses. We will use 150 dB re 1 µPa RMS as a guideline for assessing when behavioral responses to pile driving noise may be expected. The effect of any anticipated response on individuals will be considered in the effects analysis below.

7.1.3 Effects of Noise Exposure

The acoustic effects analysis will:

- characterize the various sources of noise attributed to the proposed action;
- determine which species are likely to be exposed to each type of noise;
- characterize the range of expected or possible responses of sea turtles and marine mammals exposed to the noise; and,
- determine the significance of those effects to individuals and populations.

7.1.3.1 Geophysical and Geotechnical Surveys

<u>7.1.3.1.1 Geophysical Surveys</u>

It is anticipated that all lessees will conduct a high resolution geophysical survey. The survey would investigate the shallow subsurface for geohazards and sediment conditions, as well as to identify potential benthic biological communities (or habitats) and archaeological resources. In general, the survey ship travels at less than 4.5knots (8.3 km/hour), and the source is activated every 7-8 seconds (or about every 12.5 m). All involved ships are designed to reduce self-noise, as the higher frequencies used in high-resolution work are easily masked by the vessel noise if special attention is not paid to keeping the ships quiet. The sound levels at the source (i.e., the survey vessel) will depend on the type of equipment used for the survey. As outlined above several types of equipment will be used including fathometers, sub-bottom profilers (chirp or boomer) as well as side scan and multibeam sonar. Noise levels at the source are expected to range from 220 – 201 dB re 1uPa peak (see Table 2).

In the BA, BOEM provided estimates of the distance from the source to the 180 dB RMS radius and the 160 dB RMS radius for the different survey instruments. The source levels and operating frequencies are noted in Table 1 above and the distances to the 180 dB and 160 dB isopleth radii are noted in Table 2. As demonstrated below, the boomer is the only source that can be heard by sea turtles and Atlantic sturgeon. Because sea turtles and Atlantic sturgeon can not perceive the sound emitted by the side scan sonar, chirp or depth sounder, these species will not experience any effects due to exposure to increased underwater noise associated with these surveys. Because sei and sperm whales do not occur in the NY and NJ WEAs they will only be exposed to effects of underwater noise in the MA and MA/RI WEAs.

Table 2. Estimated Ranges to 180dB and 160dB

Source	Pulse Length	Broadband Source Level (dB re 1 µPa at 1 m)	Operating Frequencies	180-dB Radius (m)	160-dB Radius (m)	Within Hearing Range		
						Whales	Sea Turtles	Atlantic sturgeon
Boomer	180µs	212	200 Hz – 16kHz	38-45	1,054-2,138	**Yes**	**Yes**	**Yes**
Side-scan sonar	20 ms	226	100 kHz	128-192	500-655	**Yes**	No	No
			400 kHz			No	No	No
Chirp sub-bottom Profiler	64 ms	222	3.5 kHz	32-42	359-971	**Yes**	No	No
			12 kHz			**Yes**	No	No
			200kHz			No	No	No
Multi-beam depth sounder	225 µs	213	240kHz	27	147-156	No	No	No

It should be noted that while the modeling scenarios are based on sites offshore of the BOEM's Mid and South Atlantic Planning Areas, the modeling scenarios included similar bottom sediments, and depth ranges as found in the North Atlantic Planning Area. The sound velocity profiles are expected to be inclusive of what would be expected in the Project Area. See Appendix D in the Atlantic OCS Proposed Geological and Geophysical Activities Mid-Atlantic and South Atlantic Planning Areas Draft Programmatic Environmental Impact Statement for a full explanation of the threshold radii modeling (USDOI, BOEM 2012a). The distance ranges captured in the table above are expected to represent the maximum distances to attenuation of 160 dB RMS and can be considered as a "worst case" representation.

Effects to Whales

The HRG survey exclusion zones are based on preventing any whales from experiencing Level A harassment (i.e., noise that is loud enough to cause injury) from a non-continuous noise source as defined for the purposes of the MMPA. Because the exclusion zones will be continuously maintained and no surveys will occur if whales are near enough to experience noise above 180 dB re 1uPa RMS, we do not anticipate that any whales will be injured.

The source level (i.e., within 1 meter) for the equipment that can be heard by whales ranges from 212-226 dB re 1uPa; the sound attenuates with distance so noise levels are greatest closest to the source and diminish the further from the source. As noted above, injury can result to whales upon exposure to impulsive noises, such as the geophysical survey equipment, above 180 dB re

1uPa RMS. According to the information provided by BOEM, noise levels greater than 180 dB re 1uPa RMS will be experienced within 32-192 meters of the source, depending on the type of equipment being operated. BOEM is requiring that all lessees maintain a 200-meter exclusion zone during the survey and that this exclusion zone be monitored for at least 60 minutes prior to ramp up of the survey equipment. The equipment will not be started until the exclusion zone is free of whales for at least 60 minutes. As whales typically surface at least once every 60 minutes, it is reasonable to expect that monitoring the exclusion zone for at least 60 minutes will allow the PSO to detect any whales that may be submerged in the exclusion zone. Once the equipment is turned on, should a whale be detected within 200 meters of the survey vessel, all operations will be halted or delayed until the exclusion zone is clear of whales for at least 60 minutes. Based on this, it is extremely unlikely that a whale will be present within 200 m of the source while the geophysical survey equipment is operating. This is because the exclusion zone will be monitored throughout operations and the survey will stop if a whale is detected within 200 m of the source. Therefore, because the survey will not operate if there is a whale in the area where it could be exposed to noise levels greater than 180 dB re 1uPa RMS, it is extremely unlikely that any whales will be injured.

The distance to Level B harassment levels (160 dB) ranges from 359-2,138 meters, depending on the type of equipment being used. Taking into account the standard operating conditions that will be implemented, effects on whale behavior are generally expected to be temporary disruption of normal behaviors (foraging, migrating, resting) while making movements to avoid the area around the HRG survey, and changes in vocalizations due to masking caused by the additional background noise. As whales are mobile species, they have the ability to move away from the sound should disturbance occur. It is expected that areas avoided by whales during noise producing activity would be available and used by whales after the survey had left the area. Once an area has been surveyed, it is not likely that it will be surveyed again, therefore reducing the likelihood of repeated HRG-related impacts within the Project Area. Thus, the exposure to noise that equates to Level B harassment is expected to be temporary.

Aside from the case of mass strandings of beaked whales in response to acoustic activities, no scientific studies have conclusively demonstrated a link between exposure to sound and adverse effects on a marine mammal population (NRC 2005). Any animals that are exposed to underwater noise associated with the surveys (i.e., noise between 160 and 180 dB re 1uPa RMS) may display behavioral reactions to the sounds by temporarily ceasing resting, migration, and foraging activities and moving away from the sound source. As explained above, the area ensonified at any given time (approximately 3-14 square kilometres) will be only a small portion of the action area, so there is sufficient habitat available for whales that swim away from the noise source. In addition, after the survey vessel had left the area, any animal temporarily displaced for the duration of the surveys would likely return to the area without additional impairment of migrating, feeding, resting, or other behaviors. Major shifts in habitat use or distribution or foraging success are not expected. Based on what we know about their responses upon exposure to such sound sources in other instances, we expect that long-term adverse effects on individuals are unlikely, and as such would be unlikely to reduce the overall reproductive success, feeding, or migration of any individual animal. Therefore, while the proposed survey activities may result in temporary harassment of right, humpback, fin, sperm and sei whales, they are not likely to result in death or injury of any individuals.

Feeding behavior is not likely to be significantly impacted, as whales appear to be less likely to exhibit behavioral reactions or avoidance responses while engaged in feeding activities (Richardson et al. 1995). Whale prey species are mobile, and are broadly distributed throughout the action area; therefore, whales that may be temporarily displaced during survey activities are expected to be able to resume foraging once they have moved away from areas with disturbing levels of underwater noise.

Masking

Masking is a natural phenomenon which marine mammals must cope with even in the absence of man-made noise (Richardson et al. 1995). Since the sound produced by the surveys would be intermittent and transient in nature, masking would not be a continuous phenomenon, but would occur for only a few seconds at a time in a particular area. Marine mammals demonstrate strategies for reducing the effects of masking, including changing the source level of calls, increasing the frequency or duration of calls, and changing the timing of calls (NRC 2003). Although these strategies are not necessarily without energetic costs, the consequences of temporary and localized increases in background noise level are impossible to determine from the available data (Richardson et al. 1995; NRC 2005). However, one relevant factor in attempting to consider the effect of elevated noise levels on marine mammal populations is the size of the area affected versus the habitat available. The endangered whale species likely to be affected by the survey noise (right, humpback, fin, sei and sperm whales) are widely dispersed. As such, only a very small percentage of the population is likely to be within the radius of masking at any given time. Richardson et al. (1995) concludes broadly that, although further data are needed, localized or temporary increases in masking probably cause few problems for marine mammals, with the possible exception of populations highly concentrated in an ensonified area. Although a high proportion of the known right whale population can be concentrated in Cape Cod Bay or the Great South Channel at one time, these areas are beyond the predicted zone of ensonification from the surveys in the MA/RI and MA WEA. Given the location of the surveys, no increase in noise is expected in Cape Cod Bay, the closest habitat area where a significant portion of the population may be at any one time. As such, although some number of right, humpback, fin, sei and sperm whales are likely to be subject to occasional masking as a result of survey activity, temporary shifts in calling behavior to reduce the effects of masking, on the scale of no more than a few minutes, are not likely to result in failure of an animal to feed successfully, breed successfully, or complete its life history.

Acoustically Induced Stress

Stress can be defined in different ways, but in general, a stress response demonstrates a perturbation to homeostasis (NRC 2003), or a physiological change that increases an animal's ability to cope with challenges. Typical manifestations of stress include changes in heart rate, blood pressure, or gastrointestinal activity. Stress can also involve activation of the pituitary-adrenal axis, which stimulates the release of more adrenal corticoid hormones. Stress-induced changes in the secretion of pituitary hormones have been implicated in failed reproduction

(Moberg 1987, Rivest and Rivier 1995) and altered metabolism (Elasser *et al*. 2000), immune competence (Blecha 2000) and behavior.

Generally, stress is a normal, adaptive response, and the body returns to homeostasis with minimal biotic cost to the animal. However, stress can turn to "distress" or become pathological if the perturbation is frequent, outside of the normal physiological response range, or persistent (NRC 2003). In addition, an animal that is already in a compromised state may not have sufficient reserves to satisfy the biotic cost of a stress response, and then must divert resources away from other functions. In these cases, stress can inhibit critical functions such as growth (in a young animal), reproduction, or immune responses.

There are very few studies on the effects of stress on marine mammals, and even fewer on noise-induced stress in particular. One controlled laboratory experiment on captive bottlenose dolphins showed cardiac responses to acoustic playbacks, but no changes in the blood chemistry parameters measured (Miksis et al. 2001 in NRC 2003). Beluga whales exposed to playbacks of drillrig noise (30 minutes at 134-153 dB re 1μPa) exhibited no short term behavioral responses and no changes in catecholamine levels or other blood parameters (Thomas et al. 1990 in NRC 2003). However, techniques to identify the most reliable indicators of stress in natural marine mammal populations have not yet been fully developed, and as such it is difficult to draw conclusions about potential noise-induced stress from the limited number of studies conducted.

There have been some studies on terrestrial mammals, including humans, that may provide additional insight on the potential for noise exposure to cause stress. Marine mammals are likely to exhibit some of the same stress symptoms as terrestrial mammals. For example, the stress caused by pursuit and capture activates similar physiological responses in terrestrial mammals (Harlow et al. 1992 in NRC 2003) and cetaceans (St. Aubin and Geraci 1992 in NRC 2003). Jansen (1998) reported on the relationship between acoustic exposures and physiological responses that are indicative of stress responses in humans (for example, elevated respiration and increased heart rates). Jones and Broadbent (1998) reported on reductions in human performance when faced with acute, repetitive exposures to acoustic disturbance. Trimper *et al*. (1998) reported on the physiological stress responses of osprey to low-level aircraft noise while Krausman *et al*. (2004) reported on the auditory and physiological stress responses of endangered Sonoran pronghorn to military overflights.

These studies on stress in terrestrial mammals lead us to believe that this type of stress is likely to result from chronic acoustic exposure. Because we do not expect any chronic acoustic exposure to any individuals from the proposed surveys, we do not anticipate this type of stress response from the survey activities.

It is difficult to predict the number of whales that may be exposed to potentially disturbing levels of noise associated with the geophysical surveys. BOEM reports maximum sightings per unit effort (SPUE) levels within the MA/RI and MA WEAs for right, humpback, fin, and sei whales in the BA (number of animals per 1 km surveyed, reported from the Right Whale Consortium sighting database) as follows: right whales 0.1; humpback 0.2; fin 0.135; sei, 0.1. SPUE levels for sperm whales are also reported (0.335); however, this SPUE is considered to be unreasonably high given the estimated number of sperm whales in the North Atlantic as a whole. Sei and Sperm whales do not occur in the NJ and NY WEAs. Reported sightings data indicates that densities of right, fin and humpback whale species are higher in the MA/RI and MA WEAs than in the NJ and NY WEAs; thus, using the MA/RI and MA WEAs to predict density throughout the project area is likely to result in an overestimate of whale density and the number of animals that may be exposed. This method is also likely to overestimate the number of animals exposed because it uses the maximum SPUEs, regardless of season, to predict exposure. Because it is likely that not all surveys will occur during the time of highest whale density, this causes this method to overestimate the number of animals that may be exposed. Further, it is possible that a single individual may be exposed to disturbing levels of noise multiple times; in that case, the actual number of individuals exposed to increased underwater noise would be less than what is predicted here. It is also important to note that this number of exposures will occur over a five-year period and that there is no danger of injury, death or hearing impairment from the exposure to these noise levels.

In calculating the area density of these species from these linear density data, we use 1.85km as the strip width (W). This strip width is based on the distance of visibility used in the NARWC data. However, those surveys used a strip transect instead of a line transect methodology. Therefore, in order to obtain a strip width, one must divide the visibility or transect value in half. Since the visibility value used in the NARWC data was 3.7km, it gives a strip width of 1.85km. Based on this, the area density (D, whales/km^2) of whale species in the WEAs can be obtained by the following formula:

D=SPUE/2W

Using these calculated area densities and the physical area where noise between 180 dB and 160 dB will be experienced for the different types of geophysical surveys (see Table above), we can calculate the number of whales that may be exposed to these noise levels. The table below illustrates these calculations. As can be seen, the number of right, humpback, fin or sei whales likely to be exposed to disturbing levels of noise at any one time during these surveys is very low (calculated to be less than one). Applying the density estimates to the entire area of all four WEAs, we have estimated the number of whales that may experience behavioral disturbance over the five-year period as: 147 right whales, 288 humpback whales, 192 fin whales and 147 sei whales. We calculated these estimates by applying the density estimate to the entirety of the four WEAs. For example, for right whales the density is 0.027 (D=0.1/2(1.85km). This density estimate is then multiplied by the size of the four WEAs (because we assume 100% coverage by geophysical surveys) to generate a total exposure estimate of 147 right whales (N= (0.027 right whales)(5,439 square kilometers)). As explained above, while we anticipate that whales will have their normal behaviors disrupted, all behavioral effects will be minor and temporary with no injury or mortality.

Species	SPUE	Area Density	Number exposed to noise <180dB >160 dB	Number exposed to <180dB >160 dB from side	Number exposed to <180dB >160 dB from	Total Animals exposed to<180dB >160 dB,

			from chirp (0.404 km2)	scan sonar (1.35 km2)	boomer (14.36 km2)	all surveys (5,439km2)
Right	0.1	0.027	0.01	0.04	0.39	147
Humpback	0.2	0.054	0.02	0.07	0.78	294
Fin	0.135	0.036	0.01	0.05	0.52	196
Sei	0.1	0.027	0.01	0.04	0.39	147

As explained above, sperm whales are not expected in the NY and NJ WEAs because of the shallow depths in that area. Sperm whales are occasional visitors to the MA and MA/RI WEAs. Sightings in these areas are recorded in the Right Whale Consortium database. Small numbers of sperm whales are also reported in the MA WEA in a report prepared by the Massachusetts Clean Energy Center (Kraus et al. 2013, *In Review*). For this report, the distribution and abundance of marine mammals and sea turtles in the Massachusetts wind energy area was assessed with aerial survey and acoustic methods. The New England Aquarium and Provincetown Center for Coastal Studies conducted aerial surveys over the Massachusetts WEA for whales and sea turtles twice a month from October 9, 2011 to September 17, 2012. The Cornell University's Bioacoustics Team placed Marine Autonomous Recording Units (MARU's) at 6 locations within the MA WEA on November 10, 2011, which recorded all large whale sounds continuously (with a single day swap-over break) through October 3, 2012. For all whale and sea turtle species, except loggerhead and leatherback sea turtles, sightings were too few to calculate density estimates. A total of six sperm whales were sighted during this survey. Based on the best available information, we expect fewer sperm whales in the action area than any of the other ESA listed whale species. Therefore, while we cannot generate a quantitative estimate of the number of sperm whales likely to be exposed to potentially disturbing levels of noise, we expect that number to be less than the other whale species. Like the other whale species, all behavioral effects will be minor and temporary with no injury or mortality.

Effects to Sea Turtles

As noted above, the only geophysical survey equipment that operates in a range that can be heard by sea turtles is the boomer. The source level (i.e., within 1 meter) for the boomer is 212 dB re 1uPa; the sound attenuates with distance so noise levels are greatest closest to the source and diminish the further from the source. As noted above, we do not expect injury to sea turtles upon exposure to impulsive noises, such as the boomer, less than 180 dB re 1uPa RMS. According to the information provided by BOEM, noise levels greater than 180 dB re 1uPa RMS will only be experienced within 45 meters of the source. BOEM is requiring that all lessees maintain a 200-meter exclusion zone during the survey and that this exclusion zone be monitored for at least 60 minutes prior to ramp up of the survey equipment. The equipment will not be started until the exclusion zone is free of sea turtles for at least 60 minutes. The normal duration of sea turtle dives ranges from 5-40 minutes depending on species, with a maximum duration of 45-66 minutes depending on species (Spotila 2004). As sea turtles typically surface at least once every 60 minutes, it is reasonable to expect that monitoring the exclusion zone for at least 60 minutes will allow the endangered species monitor to detect any sea turtles that may be submerged in the exclusion zone. Once the equipment is turned on, should a sea turtle be detected within 200 meters of the survey vessel, all operations will be halted or delayed until the exclusion zone is clear of turtles for at least 60 minutes. Based on this, it is extremely unlikely that sea turtle will be present within 45 m of the source while the boomer is operating. This is because the exclusion zone will be monitored throughout operations and the survey will stop if a sea turtle is detected within 200 m of the source. Additionally, given the noise levels produced by the survey equipment and given the expected behavioral response of avoiding noise levels greater than 166 dB, it is extremely unlikely that any sea turtles would swim towards the survey vessel. Any sea turtles within 45 meters of the equipment at the beginning of the survey are expected to move away during ramp-up and not be injured. Therefore, because the survey will not operate if there is a sea turtle within the area where it could be exposed to noise levels greater than 180 dB re 1uPa RMS, it is extremely unlikely that any sea turtles will be injured.

As explained above, the best available information indicates that sea turtles will respond behaviorally to impulsive noises greater than 166 dB re 1uPa by actively avoiding the area where noise is greater than 166 dB. As the survey vessel travels along the transects it is expected that any sea turtles in the area that are close enough to perceive the sound will swim away from it. The noise emitted from the boomer dissipates as it gets further from the source, it attenuates to 180 dB re 1uPa at a distance of 45 meters and attenuates to 160 dB re 1uPa at a distance of 1,054-2,138 meters from the survey equipment. Each pulse of the boomer lasts for less than one second and the equipment will be pulsed approximately every 12 meters. Each time the boomer is activated, an area with a radius of 1.054-2.138 km will have noise levels greater than 160 dB re 1uPa RMS. At any given time during the survey when a boomer is used, an approximately 3.49-14.36 square kilometer area will have noise levels between 160 and 180 dB. We expect that sea turtles will avoid this noisy area and move away. Because the vessel will be traveling, it will only be in a particular area for a very short time (seconds to minutes); thus, no one area will experience disturbing levels of noise for more than a few minutes and the area in which sea turtles would be disturbed would be constantly changing. Thus, we do not anticipate that sea turtles would be excluded from any one area for more than a few minutes. Sea turtles whose behavior is disrupted would be expected to resume their behavior after the disturbance has stopped. Sea turtles that avoided ensonified areas would return to those areas after the survey vessel left the area. Effects to sea turtle from this avoidance behavior are expected to be limited to temporary disruption of normal behaviors and increased energy expenditure to move away from the noisy area.

The surveys would likely use the full daylight hours available, approximately 10 hours per day. However, the time that any particular area would experience elevated, detectable sound levels would be significantly shorter as the vessel would be ensonifying a limited area along each transect. Available information indicates that sea turtle forage items may be present in the action area, therefore if sea turtles were present and feeding or resting in an area where HRG surveys were passing through, it is expected that they could find alternative forage and resting locations nearby. Sea turtles are not expected to be excluded from large areas due to the temporary nature of HRG activities. The avoidance of ensonified areas will be temporary and localized. It is not expected that any impacts would result in injury or overall behavioral impairment to any individual. Major shifts in habitat use, interruption of foraging or major displacement of migration pathways, are not expected.

Because only a relatively small, transient area will be impacted at any one time and sea turtle density in the action area is generally low, it is difficult to calculate the number of sea turtles that may experience disturbing levels of sound. Few researchers have reported on the density of sea turtles in Northeastern waters. However, this information is available from one source (Shoop and Kenney 1992). Shoop and Kenney (1992) used information from the University of Rhode Island's Cetacean and Turtle Assessment Program (CETAP[8]) as well as other available sightings information to estimate seasonal abundances of loggerhead and leatherback sea turtles in northeastern waters. The authors calculated overall ranges of abundance estimates for the summer of 7,000-10,000 loggerheads and 300-600 leatherbacks present in the study area from Nova Scotia to Cape Hatteras. Using the available sightings data (2,841 loggerheads, 128 leatherbacks and 491 unidentified sea turtles), the authors calculated density estimates for loggerhead and leatherback sea turtles (reported as number of turtles per square kilometer). These calculations resulted in density estimates of 0.00164 – 0.510 loggerheads per square kilometer and 0.00209 – 0.0216 leatherbacks per square kilometer. It is important to note, however, that this estimate assumes that sea turtles are evenly distributed throughout the waters off the northeast, even though Shoop and Kenney report several concentration areas where loggerhead or leatherback abundance is much higher than in other areas. Further, the data do not include any sightings from Massachusetts and only considered the presence of leatherback and loggerhead sea turtles. The Shoop and Kenney data, despite considering only the presence of loggerhead and leatherback sea turtles, likely overestimates the number of sea turtles present in the action area. This is due to the assumption that sea turtle abundance will be even throughout the Nova Scotia to Cape Hatteras study area, which is an invalid assumption.

Kraus et al. (2013 *In Review*), presents SPUE-based density estimates for loggerhead and leatherback sea turtles in the MA WEA. During this survey effort (described above), sightings were recorded of 93 leatherbacks, 76 loggerheads and 6 Kemp's ridleys and 9 unidentified sea turtles. The number of Kemp's ridley observations was too small to calculate a density estimate. The majority of sea turtle sightings were in August and September. While the reported density estimates for loggerheads (summer 0.072/km^2 and fall 0.037/km^2) are within the range reported by Shoop and Kenney from the CETAP data (0.00164-0.510), the density estimates reported by Kraus et al. for leatherbacks are higher than those reported by Shoop and Kenney (summer 0.033/km^2 and fall 0.037/km^2 compared to 0.00209-0.0216).

Using the maximum reported density estimates (0.510/km2 for loggerheads and 0.037/km2 for leatherbacks), and the area where noise levels greater than 160 dB re 1uPa will be experienced during surveys using the boomer (3.49-14.36 square kilometer), we can estimate the number of loggerhead and leatherback sea turtles that may experience disturbing levels of noise. These calculations lead to an estimate of up to 7 loggerheads and 1 leatherback sea turtle are likely to be exposed to potentially disturbing levels of noise each time the boomer is operated. If we assume that the entirety of the NY, NJ, RI/MA and MA WEA are surveyed, an area that totals approximately 5,439 square kilometers, we would expect that up to 201 leatherbacks and up to

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[8] The CETAP survey consisted of three years of aerial and shipboard surveys conducted between 1978 and 1982 and provided the first comprehensive assessment of the sea turtle population between Nova Scotia, Canada and Cape Hatteras, North Carolina.

2,774 loggerheads may be exposed to potentially disturbing levels of noise associated with the boomer surveys over a five year period. As explained above, we expect this exposure to result in minor and temporary behavioral disturbance with no injury or mortality. No density estimates are available for Kemp's ridley or green sea turtles; however, we expect fewer sea turtles of these species than leatherbacks in the action area. This assumption is supported by the sightings data reported by Kraus et al. (2013 *In Review*) of no green sea turtles and only 6 Kemp's ridleys (compared to 93 leatherbacks and 76 loggerheads). Therefore, each time the boomer is operated, no more than 1 Kemp's ridley and 1 green sea turtles are likely to experience potentially disturbing levels of noise. In total, we expect no more than 201 Kemp's ridleys and 201 green sea turtles to be exposed to potentially disturbing levels of noise from the boomers. We consider this a worst case estimate because it assumes that sea turtle density will be at the maximum reported level throughout the action area, which is unlikely to occur, it uses the maximum distances modeled by BOEM for noise attenuation, and it assumes that all surveys will occur at a time of year when sea turtles are present (June – November) and that sea turtles will be present at every location that the boomer is operated. However, despite these assumptions, this is the best available estimate of the number of sea turtles that may be exposed to disturbing levels of noise from the boomer.

As explained above, the effects of exposure to noise from the boomer will be minor and temporary and limited to temporary disruption of normal behaviors as sea turtles move away from the sound source and avoid the temporarily ensonified area. Sea turtles in the areas where surveys will take place are likely to be migrating, foraging and resting. If these behaviors are disrupted, we expect that they will quickly resume once the survey vessel has left the area. No sea turtles will be displaced from a particular area for more than a few minutes. While the movements of individual sea turtles while be affected by the sound associated with the survey, these effects will be temporary and localized and sea turtles are not expected to be excluded from the action area. Major shifts in habitat use or distribution or foraging success are not expected. As changes to individuals movements are expected to be minor and short-term, there are not likely to be any population-level effects.

Effects to Sturgeon

Sturgeon are only expected to be able to perceive the noise associated with the boomer. All other survey equipment operates at frequency higher than sturgeon can hear, therefore we do not expect any effects to sturgeon exposed to increased underwater noise from the other higher frequency survey equipment. As noted above, the available information on effects of fish to exposure of sound is extremely limited. There are no known studies examining the effects of boomers on fish generally, or sturgeon specifically. However, based on the available information summarized above, we expect fish to react to noise that is disturbing by moving away from the sound source and avoiding further exposure. Injury and mortality is only known to occur when fish are very close to the noise source and the noise is very loud and typically associated with pressure changes (i.e., impulsive pile driving or blasting).

All surveys will use a ramp up procedure for the boomer; that is, it will not be used at full energy right away. This gives any fish in the immediate area a "warning" and an opportunity to leave the area before the full energy of the boomer is used. Given the location of the areas to be surveyed, we do not expect sturgeon to occur in dense aggregations in the survey areas. We expect any sturgeon that are in the survey areas to be individuals migrating through that may forage opportunistically. The available information suggests that for pile driving, peak noise levels need to be at least 212 dB re 1uPa before physiological impacts are likely. In order to be exposed to peak energy of 212 dB re 1uPa from the boomer, a sturgeon would need to be within 1 meter of the source. This is extremely unlikely to occur.

The likelihood of exposure to potentially injurious levels of noise is further reduced by the use of the ramp up and the expectation that sturgeon will react to the increase in noise by leaving the area. Available information suggests that noise above 150 dB re 1uPa RMS may trigger a behavioral response in fish. In the worst case, we expect that sturgeon would completely avoid the area ensonified by the boomer. However, because the area where increased underwater noise will be experienced is transient and increased underwater noise will only be experienced in a particular area for only seconds, we expect any effects to behavior to be minor and limited to a temporary disruption of normal behaviors, temporary avoidance of the ensonified area and additional energy expenditure spent while swimming away from the noisy area. If foraging or migrations are disrupted, we expect that they will quickly resume once the survey vessel has left the area. No sturgeon will be displaced from a particular area for more than a few minutes. While the movements of individual Atlantic sturgeon while be affected by the sound associated with the survey, these effects will be temporary and localized and these fish are not expected to be excluded from the action area and there will be only a minimal impact on foraging, migrating or resting sturgeon. Major shifts in habitat use or distribution or foraging success are not expected. There are no available density estimates that would allow us to calculate the number of Atlantic sturgeon that could potentially be exposed to disturbing levels of sound. As explained above, we expect this exposure to result in minor and temporary behavioral disturbance with no injury or mortality.

7.1.3.1.2 Geotechnical Surveys – Drilling

As explained above, geotechnical drilling will take place in each lease block. Limited information is available on underwater noise from underwater construction drilling operations. Richardson *et al.,* (1990) reported that shallow water measurements (19.6 to 22.9 feet [6 to 7 meters] deep) taken in the vicinity of a drill rig on an ice pad produced approximately 125 dB re 1 μPa at 130 meters, and 86 dB re 1 μPa at 480 meters. Hall *et al.,*'s (1991, as cited in Nedwell and Howell 2004) measurements of drilling from a concrete caisson showed little difference in levels of frequencies above 30 Hz between drilling and background noise. Drill ships and semi-submersible drill rigs have been reported to have a source level from 145 (Gales 1982) to 191 dB re 1 μPa at 1 meter (Greene 1987), but are unlikely to be used during windfarm development. It is anticipated that the majority of the work will be accomplished by CPT which does not require deep borehole drilling. However, should CPT be found to be an inappropriate technique given the conditions encountered, borehole drilling may be required. Previous estimates submitted to BOEM have source sound levels not exceeding 145 dB re 1 μPa at a frequency of 120Hz (USDOI, BOEM, OREP 2012), which are similar to those from historical drilling studies cited previously. Previous submissions to BOEM also indicated that boring sound should attenuate to below 120 dB re 1 μPa by the 492 foot (150 meter) isopleth.

As noted above, a 200 meter exclusion zone around the geotechnical survey vessel will be maintained such that no drilling will occur should a whale or sea turtle occur within 200 meters of the survey vessel. As no geotechnical drilling will occur if whales or sea turtles will occur are within 200 meters, and we expect noise to attenuate to below 120 dB within 150 meters, no whales will be exposed to sound levels at which harassment (i.e., 120 dB re 1uPa for a continuous noise source such as drilling) could occur. Sea turtles are only thought to be able to hear noises above 126 dB re 1uPa. A sea turtle would need to be within 150 meters of the drill to hear the noise. Because drilling will not occur if sea turtles are that close to the drill, we do not expect any sea turtles to be exposed to drilling noise.

Given the noise level at the source (145 dB re 1uPa) is below the level that we expect may result in behavioral responses by sturgeon, we expect all effects to sturgeon from exposure to drilling noise to be insignificant and discountable.

7.1.3.2 Pile Driving

Sound levels associated with the driving of piles have been modeled and presented by BOEM. BOEM has estimated that up to four met towers could be constructed in the RI/MA WEA and up to five additional towers in the MA WEA. Any additional construction of met towers would be considered to be outside the scope of this programmatic consultation. Estimates of pile driving noise associated with the installation of met towers are varied. The majority of estimates indicate that, depending on the size of the pile being driven, underwater sound levels at the source could range from 185 dB re 1uPa to 200 dB re 1uPa with noise levels dissipating to below 180dB re 1uPa RMS by a distance of 500-1,000 meters from the source and to below 160 dB re 1uPa RMS within 3.4-7.2 km.

BOEM will require that an exclusion zone be maintained at the distance of the 180 dB isopleth. This is designed to insure that no whales or sea turtles will be exposed to noise levels that could result in injury. It is expected to be impractical to maintain an exclusion zone of the size necessary to insure that no whales or sea turtles were present in the area where noise levels between 160 and 180 dB could be experienced. If it is not possible, then we anticipate that whales and sea turtles may be exposed to disturbing levels of noise (i.e., between 160 and 180dB).

It is important to note that pile driving will only occur for 3-8 hours for each pile to be installed and that each met tower will involve the driving of only 1-3 piles. Therefore, we anticipate only 12 – 96 hours of non-continuous pile driving in the RI/MA WEA and only 15 – 120 hours of non-continuous pile driving in the MA WEA. Piles will only be installed from May – October. During these months, right whales are not likely to occur in the RI/MA or MA WEA. Humpback, fin, sei, and sperm whales may be present as well as Atlantic sturgeon and sea turtles.

Effects to Whales

As noted above, pile driving will not occur during the time of year when right whales may be present in the MA/RI or MA WEAs (November – April). Therefore, no right whales will be exposed to any increased underwater noise associated with pile installation.

It is expected that disturbance/harassment (Level B) levels of sound (i.e. 160 dB re 1 μPa) would occur within 4 miles (7 kilometers), and Level A harassment (180 dB re 1 μPa) would occur within 1,000 m (3,281 feet) of the activity. BOEM will require an exclusion zone of 1,000 m to be monitored from the sound source and an additional observation vessel circling the sound source at 500 m from the source. Therefore, BOEM anticipates that no whales will be exposed to sound level greater than 180dB as pile driving would not occur should a whale enter within 1,000 m (3,281 feet) of the active source. Therefore, no whales are expected to be exposed to sound levels that would cause injury (i.e. above 180 dB re 1μPA). Should future field-verified acoustic data indicate the 180 dB isopleth is greater than 1,000 m (3,281 feet), then future mitigation measures in lease stipulations would be modified to reflect the new data. In the case where more than one monopole is being installed per meteorological tower (e.g. tripod structure), then field verifications could modify the mitigation measures for the installation of additional monopoles (i.e., the exclusion zone would be expanded or retracted based on the in-water monitoring data).

Large whales present within the RI/MA and MA WEAs and the surrounding waters are expected to be transiting between summer feeding grounds in the north, and winter calving grounds in the south, however there are also observations of large whales feeding within the vicinity of the RI/MA and MA WEAs. Because of the large size of the area where noise levels will be greater than 160 dB Re 1uPa, BOEM does not anticipate that lessees will be able to effectively monitor and maintain an exclusion zone such that pile driving could be delayed or stopped if whales were present in the area where noise is louder than 160 dB. Therefore, we anticipate that some humpback, fin, sei and sperm whales could be exposed to underwater noise between 160 and 180 dB.

Any animals that are exposed to pile driving noise between 160 and 180 dB re 1uPa RMS may display behavioral reactions to the sounds by temporarily ceasing resting, migration, and foraging activities and moving away from the sound source. As explained above, the area ensonified during pile driving will range from approximately 36 to 163 square kilometers. The increase in underwater noise will be temporary (3-8 hours per day for no more than 3 consecutive days). Once pile driving stops, any animal temporarily displaced for the duration of construction activity would likely return to the area without additional impairment of migrating, feeding, resting, or other behaviors. Major shifts in habitat use or distribution or foraging success are not expected. Based on what we know about their responses upon exposure to such sound sources in other instances, we expect that long-term adverse effects on individuals are unlikely, and as such would be unlikely to reduce the overall reproductive success, feeding, or migration of any individual animal. Therefore, while the proposed pile driving may result in temporary disruption of normal behaviors of humpback, fin, sperm and sei whales, they are not likely to result in death or injury of any individuals.

Masking

Since the sound produced by the pile driving would only last for 3 to 8 hours for no more than 3 days for each of nine piles, masking would not be a continuous phenomenon, but would occur for only a few hours at a time in a particular area. Background information on masking is presented above. Although some number of humpback fin, sperm and sei whales are likely to be subject to occasional masking as a result of pile installation, temporary shifts in calling behavior to reduce the effects of masking, on the scale of no more than 3 to 8 hours for no more than 3 consecutive days on no more than 9 occasions over five years, are not likely to result in failure of an animal to feed successfully, breed successfully, or complete its life history.

Acoustically Induced Stress

Background information on acoustically induced stress is presented above. As noted above, available studies on stress in terrestrial mammals lead us to believe that this type of stress is likely to result from chronic acoustic exposure. Because we do not expect any chronic acoustic exposure to any individuals from the proposed pile driving, we do not anticipate this type of stress response from the survey activities.

It is difficult to predict the number of whales that may be exposed to potentially disturbing levels of noise associated with the pile driving; however, using the SPUE information explained above, we can calculate an estimate of the numbers humpback, fin, sperm and sei whales that may be exposed to noise between 160 and 180 dB re 1uPa RMS during pile driving. For the same reasons discussed above in the geophysical survey analysis, these calculations are likely to overestimate the number of individuals that may be exposed. This number of exposures will occur over a five year period and that there is no danger of injury, death or hearing impairment from the exposure to these noise levels.

Using the calculation method discussed above and considering that an area of 36.32 – 162.86 square kilometers may have noise levels between 160 and 180 dB re 1uPa RMS for the duration of each pile installation, the calculated number of whales that may experience behavioral disturbance during the installation of any one pile is: 7-33 humpback whales; 5-22 fin whales and 3-16 sei whales. We expect fewer sperm whales to be present in the action area. Therefore, we expect less than 3-16 sperm whales to be exposed to disturbing levels of noise during the installation of any one pile. As explained above, we expect this exposure to result in temporary disturbance of normal behaviors, avoidance of the temporarily ensonified area, and increased energy expenditure to avoid this area. These behaviors will be limited to the duration of the pile driving activity. We do not anticipate any injury, mortality or hearing loss.

Sea Turtles

An exclusion zone of approximately 1,000 meters, or the distance where noise will be attenuated to below 180 dB re 1uPa RMS, will be maintained for all pile driving. Pile driving will not begin until this area is clear of sea turtles for 60 minutes. As explained above, we do not anticipate injury to sea turtles exposed to impulsive sounds lower than 180 dB re 1uPa. Because of the exclusion zone, we do not expect any sea turtles to be exposed to noise levels that could result in injury. We expect sea turtles to respond behaviorally to underwater noise greater than 166 dB re 1uPa by actively avoiding those ensonified areas. As noted above, modeling results reported by BOEM indicate that sound levels could be higher than 160 dB within 3.4 – 7.2 km of the pile being driven. As such, any sea turtles occurring within that area would be exposed to potentially disturbing sound levels.

Using the density estimates referenced above and the area where noise levels will be less than 180 dB re 1uPa but greater than 160 dB re 1uPa during the pile driving activity (36.32 – 162.86 square kilometer), we can estimate the number of loggerhead (maximum reported density estimate of 0.510 from Shoop and Kenney based on CETAP data) and leatherback sea turtles (maximum reported density estimate of 0.037 from Kraus et al. 2013, *In Review*) that may experience disturbing levels of noise. Using this method, we estimate that up to 83 loggerheads (162.86 km2 * 0.510 loggerheads/km2) and up to 6 leatherback (162.86 km2 * 0.037 leatherbacks/km2) sea turtles are likely to be exposed to potentially disturbing levels of noise for each pile that is installed. Depending on the type of met tower installed (monopole or tripod) and the total number of met towers installed, there could be a total of 9-27 piles installed over the entirety of the RI/MA and MA WEAs. In total, we would expect that no more than 2,241 loggerheads and no more than 162 leatherback sea turtles may be exposed to potentially disturbing levels of noise. No density estimates are available for Kemp's ridley or green sea turtles; however, we expect fewer sea turtles of these species than leatherbacks in the MA/RI and MA WEAs. Therefore, for each pile that is installed, we expect no more than 6 Kemp's ridley or green sea turtles are likely to experience potentially disturbing levels of noise. In total, we expect no more than 162 Kemp's ridleys and 162 green sea turtles to be exposed to potentially disturbing levels of noise from pile installation. We consider this a worst case estimate because it assumes that sea turtle density will be at the maximum reported level throughout the action area, which is unlikely to occur, it uses the maximum distances modeled by BOEM for noise attenuation, and it assumes that sea turtles will be present at every location that a pile is installed. However, despite these assumptions, this is the best available estimate of the number of sea turtles that may be exposed to disturbing levels of noise from pile driving.

Sea turtles behaviorally disrupted would be expected to resume their behavior after the pile driving has stopped. As pile driving will occur for approximately 3-8 hours, sea turtles will be excluded from the area with disturbing levels of sound for this period for each of the one to three days it takes to install the met tower foundation. Available information indicates that sea turtle forage items are available throughout the action area; therefore, while sea turtles may move to other areas within the action area to forage during the times when pile driving is occurring, the ability of individual sea turtles to find suitable forage is not expected to be impacted. Likewise, if sea turtles were resting in a particular area they are expected to be able to find an alternate resting area within the action area. Additionally, if sea turtles are migrating through the action area, they may avoid the area with disturbing levels of sound and choose an alternate route through the action area. However, as at all times there will be areas where noise levels are not at disturbing levels, the ability of sea turtles to migrate through the action area will not be affected. As such, while the movements of individual sea turtles while be affected by the sound associated with the pile driving, these effects will be temporary and localized and sea turtles are expected to be excluded from the ensonified area for a short period of time (no more than 8 hours). Any disruption of normal behaviors will be temporary as well any increase in energy expenditure resulting from swimming away from the ensonified area. Major shifts in habitat use or distribution or foraging success are not expected. As changes to individuals movements are expected to be minor and short-term, and are therefore not likely to have population-level effects.

Atlantic sturgeon

Current guidelines for assessing the potential for physiological effects to Atlantic sturgeon center around exposure to peak noise of 206 dB re 1uPa and cSEL of 187 dB re 1uPa. Calculations provided by BOEM indicate that peak noise of 206 dB or greater is only expected within 1 meter of the pile and that a sturgeon would need to be within 117.4 meters of the pile being installed to be potential exposed to noise at a level of cSEL 187 dB re 1uPa. Given that BOEM will require a soft start, where pile driving energy will not be at its maximum when it begins, and that we expect sturgeon to avoid noise that is potentially disturbing (i.e., greater than 150 dB re 1uPa RMS, which will be experienced at least 3km from the source), it is extremely unlikely that any sturgeon would be close enough to the piles during pile driving activities to experience noise that could result in any physiological effects. Therefore, we do not anticipate the injury or mortality of any Atlantic sturgeon.

Sturgeon that are exposed to noise levels greater than 150 dB re 1uPa RMS may respond behaviorally. In the worst case, we expect that behavioral responses would result in sturgeon avoiding the ensonified area for the 3 to 8 hours that pile driving will occur. Sturgeon are expected to return to the area once pile driving ceases.

Because avoidance would be limited to the period when piles are being driven, no sturgeon will be displaced from a particular area for more than 8 hours on no more than three consecutive days. While the movements of individual Atlantic sturgeon while be affected by the sound associated with pile installation, and normal behaviors such as foraging, resting or migrating may be temporarily disrupted, these effects will be temporary. Major shifts in habitat use or distribution or foraging success are not expected. We do not anticipate any injury or mortality to any sturgeon from exposure to pile driving noise.

7.1.3.3 Vessel Noise

Exposure to individual vessel noise by ESA-listed marine mammals, sea turtles, and fish within the Project Area or in the surrounding waters would be transient and temporary as vessels passed through the area. ESA-listed marine mammal, sea turtle, and fish behavior and use of the habitat would be expected to return to normal following the passing of a vessel. Therefore, impacts from vessel noise would be short term and negligible. Support and vessel transits will occur regularly throughout the lease period. Vessels transmit noise through water and cumulatively are a significant contributor to increases in ambient noise levels in many areas. The dominant source of vessel noise from the proposed action is propeller cavitation, although other ancillary noises may be produced. The intensity of noise from service vessels is roughly related to ship size and speed. Large ships tend to be noisier than small ones, and ships underway with a full load (or towing or pushing a load) produce more noise than unladen vessels. Vessel traffic associated with the proposed action would produce levels of noise of 150 to 170 dB re 1 μPa-m at frequencies below 1,000 Hz. A tug pulling a barge generates 164 dB re 1 μPa-m when empty and 170 dB re 1 μPa-m loaded. A tug and barge underway at 18 km/h can generate broadband source levels of 171 dB re 1 μPa-m. A small crew boat produces 156 dB re 1 μPa-m at 90 Hz.

Vessel noises are within the range of frequencies that whales can detect. The noise produced by smaller crew support vessels is below the threshold of harassment from a non-continuous noise source (160 dB; while the vessel noise is continuous, whales will not be exposed continuously as the vessels will be transiting and only a small area will be ensonified at a given time). As such, any effects from noise associated with crew support vessels will be discountable. Project related vessel traffic traveling between the construction staging areas and the project site will consist of tugs and barges. As noted above, the source level for these vessels is approximately 164-171 dB re 1 μPa-m. However, operational noise sources are expected to diminish to below the 160 dB re 1uPa threshold within short distances. Based on the operating procedures which limit vessels from approaching within 100 meters of any whale and 500 meters of a right whale, it is extremely unlikely that any project vessel would come close enough to a whale in a manner that would result in exposure to harassing levels of noise. As such, no whales are expected to be exposed to injurious or harassing levels of sound. As no avoidance behaviors are anticipated, the distribution, abundance and behavior of whales in the action area is not likely to be affected by noise associated with construction or maintenance vessels and any effects will be insignificant or discountable.

As noted previously in relation to construction noise, sea turtles are thought to be far less sensitive to sound than marine mammals. Although vessel noises are within the limited range of frequencies they can detect, evidence suggests that sound levels of 110-126 dB re 1μPa are required before sea turtles can detect a sound (Ridgway 1969; Streeter, in press). McCauley (2000) noted that dB levels of 166 dB re 1μPa were required before any behavioral reaction was observed. As all operational noise sources are expected to diminish to below this threshold within very short distances, no sea turtles are expected to be exposed to injurious or harassing levels of sound. As no avoidance behaviors are anticipated, the distribution, abundance and behavior of sea turtles in the action area is not likely to be affected by noise associated with construction or maintenance vessels and any effects will be insignificant or discountable.

Sturgeon may exhibit behavioral responses to noise levels greater than 150 dB re 1uPa. These noise levels are only anticipated in very close proximity to the operating vessels. Because of this, we do not anticipate any avoidance behaviors or other impacts to sturgeon upon exposure to vessel noise. All effects are anticipated to insignificant and discountable.

7.2 Effects to Benthic Habitat

Activities that disturb the sea floor will also affect benthic communities, and can cause effects to listed species by reducing the numbers or altering the composition of the species upon which these species prey. Activities that may affect the sea floor and result in the loss of foraging resources for listed species include pile installation, geotechnical drilling, and installation of scour protection (scour mats and rock armoring). The proposed activities will result in both the temporary disturbance and permanent loss of benthic habitat. Effects to benthic resources and habitat will be restricted to the area within the project footprint where sediment disturbing activities will occur.

The geotechnical drilling will affect an extremely small area at each sampling location. While there will be some loss of benthic species, including potential forage items, at the site of the drill holes, the amount of habitat affected represents an extremely small percentage of the available foraging habitat in the lease blocks and in the mid-Atlantic. As such, any effects to listed species resulting from benthic disturbance during the geotechnical drilling will be insignificant and discountable.

BOEM has estimated that if the artificial seagrass mats are used, a total area of 3,700-7,800 square feet of bottom would be affected, depending on the number of piles supporting the met tower. If a rock armor system is used, BOEM estimates that 16,000 square feet of seabed could be affected. Using these estimates, and considering that up to 9 met towers could be installed, the installation of the piles and the scour protection will result in the permanent loss of no more than 3.3 acres of benthic habitat total (approximately 0.0004% of the total MA/RI and MA WEAs). Although these impacts would result in permanent loss of this benthic habitat, loss of this habitat is not likely to have a measurable adverse impact on any foraging activity or any other behavior of listed species. As such, any effects to listed species resulting from loss of benthic habitat resulting from the installation of piles and associated scour protection will be insignificant and discountable.

7.3 Vessel Traffic

Collision with vessels remains a source of anthropogenic mortality for both sea turtles and whales. The proposed project will lead to increased vessel traffic in the action area that would not exist but for the proposed action. This increase in vessel traffic will result in some increased risk of vessel strike of listed species. However, due to the limited information available regarding the incidence of ship strike and the factors contributing to ship strike events, it is difficult to determine how a particular number of vessel transits or a percentage increase in vessel traffic will translate into a number of likely ship strike events or percentage increase in collision risk. In spite of being one of the primary known sources of direct anthropogenic mortality to whales, and to a lesser degree, sea turtles, ship strikes remain relatively rare, stochastic events, and an increase in vessel traffic in the action area would not necessarily translate into an increase in ship strike events. As outlined in the Project Design Criteria above, several measures will be implemented to further reduce the likelihood of a project vessel interacting with a whale or sea turtle. These include mandatory adherence to any DMA associated speed restrictions, a requirement to have a dedicated lookout maintain vigilant watch for marine mammals and sea turtles during all transits, and mandatory adherence to vessel speed restrictions for all vessels greater than 65 feet in length during the November 1 – April 30 time period even in those areas of the WEA that do not overlap with the SMAs.

7.3.1 Whales

Vessel traffic will increase during survey activities; however, the increase in vessel activity will be limited to one or two slow moving vessels in each lease area. These vessels are expected to operate at speeds of no more than 4.5 knots during survey activities. The vessels will be required to maintain a distance of at least 500 meters from right whales, at least 100 meters from all other whales and at least 50 meters from dolphins and all sea turtles. Dedicated lookouts will be posted on all vessels and will communicate with the captain to ensure that all measures to avoid whales and sea turtles are taken.

The majority of whale interactions with vessels that have been reported as lethal are with vessels greater than 260 feet (80 meters). However, whale strikes can occur with any size vessel from large tankers to small recreational boats (Jensen and Silber, 2004). Vessels associated with the proposed action are not anticipated to be greater than 80 m, therefore reducing the potential for a lethal vessel-whale interaction. Strikes have been reported for vessels traveling between 2 and 51 knots (2 and 59 miles per hour [mph]), with most lethal or severe injuries occurring when vessels are traveling 14 knots (16 mph) or more (Jensen and Silber, 2004; Laist *et al.*, 2001; Vanderlaan and Taggart, 2006). Given the size and speed that the survey vessels will operate at combined with the expected operating conditions (daylight only), the required separation distances and the vigilant watch of dedicated lookouts who will be able to communicate with the captain regarding the presence of whales, the potential for vessel collisions is extremely low. Therefore, effects to whales from the survey vessels are discountable. While the towed gear has the potential to result in interaction with listed species, the speed of towing (less than 4.5 knots) minimizes the potential for entanglement during the survey, as whales would be able to avoid the slow moving gear and survey vessel. Therefore, we do not anticipate any whales will be entangled or otherwise contact the towed survey gear.

Activity associated with the installation and maintenance of met towers and met buoys will result in an increase in vessel traffic in the MA/RI and MA WEAs. All vessels associated with the proposed action that are 65 feet (19.8 meters) or longer are subject to NMFS vessel speed restriction. Under these regulations, which are in place until at least December 9, 2013, vessels will travel at no greater than 10 knots (<18.5 km/h)) in certain areas where right whales gather (SMAs). These regulations are in place to reduce the likelihood of death or serious injury to the endangered North Atlantic right whales that could result from a vessel collision. These regulations also benefit other marine mammals in the area by reducing the overall speed of transiting vessels. The restrictions extend out to 20 NM (37 kilometers) around major mid-Atlantic ports, (of which Rhode Island is included). With the exception of crew boats, which generally are smaller than 65 feet (19.8 meters), these restrictions would be applicable to most vessels associated with the proposed action. In addition to the SMA speed restrictions, all vessels associated with the proposed action would be required to check with NOAA's Sighting Advisory System and abide by dynamic management areas (DMAs) speed limits when they are in effect. It is important to note that that BOEM is requiring all project vessels 65 feet or longer to travel at speeds no greater than 10 knots between November 1 and July 31 within the WEAs, regardless of whether they are within a designated SMA. This restriction will remain in effect regardless of whether the existing ship strike reduction regulations expire in 2013.

Based on the measures in place, and the intermittent travel of vessels associated with the proposed action, the potential for a vessel strike is greatly reduced. The risk of a strike is further reduced by the required separation distances and the posting of a lookout to communicate with the captain regarding the presence of whales. Vessels will only travel between 0-4.5 knots while actually engaged in construction activities. At these speeds, vessel movements during construction are not likely to pose a vessel strike risk to whales. Based on the information presented here, we have determined that the potential for construction and maintenance related vessel collisions is extremely low. Therefore, effects to whales from these vessels are discountable.

7.3.2 Sea Turtles

Similar to marine mammals, sea turtles have been killed or injured due to collisions with vessels. Hatchlings and juveniles are more susceptible to vessel interactions than adults due to their limited swimming ability. The small size and darker coloration of hatchlings also makes them difficult to spot from transiting vessels. While adults and juveniles are larger in size and may be easier to spot when at the surface than hatchlings, they often spend time below the surface of the water, which makes them difficult to spot from a moving vessel. Due to the lack of nesting habitat present within the northeast, hatchlings are not likely to be present in the Project Area and its surrounding waters, therefore there would be no impacts to this life stage.

While adults and juveniles are more likely to be present within the Project Area and its surrounding waters, should HRG surveys occur between June and November, the slow speed of the survey vessels (typically about 4.5 knots) and the 45 m separation distance reduces the potential for interaction with vessels and the associated towed survey gear. At these speeds, sea turtles are expected to be able to avoid the vessels and gear. Hazel *et al.*, (2007) reported that green sea turtles ability to avoid an approaching vessel decreases significantly as the vessel speed increases. The amount of vessel traffic associated with meteorological tower/buoy construction, operation and decommissioning is expected to be low, occurring during a short duration and operating at slow speeds. Therefore, potential for vessel collisions is discountable.

7.3.3 Atlantic sturgeon

The factors relevant to determining the risk to Atlantic sturgeon from vessel strikes are currently unknown, but they may be related to size and speed of the vessels, navigational clearance (*i.e.*, depth of water and draft of the vessel) in the area where the vessel is operating, and the behavior of Atlantic sturgeon in the area (e.g., foraging, migrating, etc.). It is important to note that vessel strikes have only been identified as a significant concern in the upper Delaware and James rivers and current thinking suggests that there may be unique geographic features in these areas (e.g., potentially narrow migration corridors combined with shallow/narrow river channels) that increase the risk of interactions between vessels and Atlantic sturgeon. The risk of vessel strikes between Atlantic sturgeon and vessels operating in the action area is likely to be low given that the vessels are operating in the open ocean and there are no restrictions forcing Atlantic sturgeon into close proximity with the vessel as may be present in some rivers. We also expect Atlantic sturgeon in the action area to be at or near the bottom. Given the depths in the action area, interactions between surface vessels and fish at or near the bottom are extremely unlikely. Based on these factors, effects to Atlantic sturgeon from the increase in vessel traffic are likely to be discountable.

7.4 Operation of the Met Towers and Buoys

As noted in the Description of the Action, BOEM anticipates the installation of up to 4 met towers in the MA/RI WEA and 5 met towers in the MA WEA as well as the installation of up to 8 met buoys in the MA/RI WEA and 10 buoys in the MA WEA. Below, we consider the effects of the operation of the met towers and effects of deployment and operation of the met buoys.

7.4.1 Met Towers

As noted above, the met towers are designed to collect meteorological data for a period of four-five years. During this time, data will be collected and transmitted to onshore facilities. The operation of the meteorological data collection instrumentation will have no effect on listed species.

Per the USCG and the FAA, lighting will be required to operate on the towers at all times. Sea turtle hatchlings are known to be attracted to lights and adversely affected by artificial beach lighting, which disrupts proper orientation towards the sea. However, nesting does not occur in Rhode Island or Massachusetts, and hatchlings are not known to be present in the waters of the RI/MA or MA WEAs. If this lighting resulted in the attraction of sea turtles or marine mammals or their prey, no effects to sea turtles or marine mammals would occur as they are not likely to collide with the stationary met tower. As such, any effects of project lighting on sea turtles or whales will be discountable. We do not anticipate any effects to sturgeon from project lighting.

The presence of nine pile foundations in the WEAs and their associated scour control mats or rock armoring has the potential to shift the area immediately surrounding each met tower from soft sediment, open water habitat to a structure-oriented system. This may create localized changes, namely the establishment of “fouling communities” within the immediate area surrounding each met tower and an increased availability of shelter among the pile structure. The met tower foundations will represent a source of new substrate with vertical orientation in an area that has a limited amount of such habitat, and as such may attract finfish and benthic organisms, potentially affecting listed species by causing changes to prey distribution and/or abundance. While the aggregation of finfish around the piles will not attract sea turtles, some sea turtle species may be attracted to the met tower foundations for the fouling community and epifauna that may colonize the underwater structure as an additional food source for certain sea turtle species, especially loggerhead and Kemp’s ridley turtles. All four sea turtle species may be attracted to the underwater structure for shelter, especially loggerheads that have been reported to commonly occupy areas around oil platforms (NRC 1996) which also offer similar underwater vertical structure.

More specifically, loggerheads and Kemp’s ridleys could be attracted to the piles to feed on attached organisms since they feed on mollusks and crustaceans. Loggerheads are frequently observed around wrecks, underwater structures and reefs where they forage on a variety of mollusks and crustaceans (USFWS 2005). Leatherback turtles and green turtles however are less likely to be attracted to the met tower foundations for feeding since leatherbacks are strictly pelagic and feed from the water column primarily on jellyfish and green turtles are primarily herbivores feeding on seagrasses and algae. However, if either of these forage items occur in higher concentrations near the piles, these species of sea turtles could also be attracted to the piles. Despite possible localized changes in prey abundance and distribution, any changes are expected to be small due to the small number of met towers and the distance between them. Therefore, any effects to sea turtle foraging are expected to be minor and localized.

As explained above, right whales feed on copepods while humpback and fin whales feed on schooling fish. Sperm whales feed on squid and other fish. Sei whales feed on copepods, krill and fish. If the met tower foundations led to an increase in schooling fish around the piles, it is possible that individual whales could be attracted to the met tower foundations. However, the small number and low density of met tower foundations (i.e., 9 over the entirety of the MA/RI and MA WEA, an area of more than 800,000 acres) makes it extremely unlikely that the distribution of forage species in the action area would be altered in a way that would affect the distribution of any whales. As such, any effects to the distribution of forage species will be insignificant and discountable.

Sturgeon feed on benthic invertebrates and small benthic fish. It is possible that the distribution and abundance of these species could increase in the area immediately adjacent to the nine met towers. Despite possible localized changes in prey abundance and distribution, any changes are expected to be small due to the small number of met towers and the distance between them. Therefore, any effects to Atlantic sturgeon foraging are expected to be minor and localized.

Although the met tower foundations would create additional attachment sites for benthic organisms that require fixed (non-sand) substrates and additional structure that may attract certain finfish species, the additional amount of surface area being introduced (i.e., only nine met tower foundations over an 800,000 acre area) would be a minor addition to the hard substrate that is already present. Due to the small amount of additional surface area in relation to the total area of the proposed action and the spacing between met towers (at least 10 miles apart), the new additional structure is not expected to alter the species composition in the action area. While the increase in structure and localized alteration of species distribution in the action area around the met tower foundations may affect the localized movements of sea turtles in the action area and provide additional sheltering and foraging opportunities in the action area for these species, any effects will be beneficial or insignificant.

7.4.2 Deployment and Operation of Buoy and Monitoring System

As noted above, a met buoy is designed to collect meteorological data for a period of four-five years. During this time, data will be collected and transmitted to onshore facilities. The operation of the meteorological data collection instrumentation (i.e., LIDAR and ADCP) will have no effect on listed species.

As explained above, buoys are likely to be anchored to a clump weight anchor and attached to the anchor with heavy chain. NMFS has considered the potential for whales and/or sea turtles to interact with the buoy and to become entangled in the buoy or mooring system and has determined that this is extremely unlikely to occur for the reasons outlined below.

In order for an entanglement to occur, an animal must first encounter the gear. Since there will only be a total of no more than 18 buoys deployed in an over 1,000 square mile area, the likelihood of a whale or sea turtle encountering the gear is extremely low. The buoy will be attached to the anchor with chains. The use of heavy chain further reduces the risk of entanglement. In the event that an animal comes in contact with a chain, the risk of entanglement is unlikely because of the tension that the buoy will be under. There have been no documented incidences of whales or sea turtles entangled in navigational buoys which have a similar mooring configuration to these buoys. Based on the analysis herein, it is extremely unlikely that a whale or sea turtle will interact with the buoy and anchor system and become entangled. As such the effect of the deployment of any buoy and anchoring system on these species is discountable. We do not anticipate any effects to Atlantic sturgeon from the presence of the buoy or anchoring system.

7.5 Decommissioning

As required by BOEM, within a year from the expiration of the lease, met tower and buoy components would be retrieved and removed from the site. The use of explosives to remove foundations is not anticipated and is not considered part of the proposed action. Any proposal to use explosives would require separate section 7 consultation. The proposed removal method for all met towers considered here is cutting piles below the mud line. Removal activities are expected to have impacts similar to those discussed above in relation to construction activities, including temporary seafloor disturbance and turbidity. However, all impacts would be of less magnitude than those resulting from construction activities. As such, effects of decommissioning activities will be insignificant or discountable.

7.6 Unexpected Events

Vessel Collision with Met Tower or Damage Resulting from Natural Events
The extent of potential impacts that could result from a vessel collision with a met tower largely depends on the extent of damage to the tower, its foundation and the vessel. Some smaller vessels would merely strike a glancing blow and possibly suffer some hull damage but not sink. Other vessels may suffer enough damage to sink, causing a small release of fuel and debris. A larger vessel may cause a collapse of the tower. Similarly, a large storm could cause damage to the met tower and/or its foundation. Repair of a damaged or collapsed tower or its foundation would create short term and localized disturbances to the benthos, water column, and pelagic organisms similar to the construction and decommissioning of a single met tower, albeit in reverse order and combined in a single event. The effects of a vessel collision or destructive natural event are difficult to predict. However, effects to listed species from such an event are more likely to be attributable to the debris that enters the water and effects of any repair activities. As any effects are likely to be on a small scale and temporary, any effects, if adverse, will be insignificant.

Fuel Spill
A fuel spill could result from a diesel generator and would be an unintended, unpredictable event. Marine animals, including listed whales, sea turtles and sturgeon, are known to be negatively impacted by exposure to oil and other petroleum products. It is estimated that a buoy generator could contain 240 gallons of diesel fuel (FERN 2011). From 2000 to 2009, the average spill size for vessels similar to those anticipated to be used for these surveys, was 88.36 gallons (U.S. Department of Homeland Security, USCG, 2011) However, without an estimate of the amount of fuel released or the conditions surrounding the spill (weather, time of year, location, etc.) it is difficult to predict the likely effects on listed species. As the effects of a spill are likely to be localized and temporary, sea turtles and whales are not likely to be exposed to fuel and any effects would be discountable. Additionally, should a response be required by the US EPA or the USCG, there would be an opportunity for NMFS to conduct a consultation with the lead Federal agency on the spill response.

8.0 CUMULATIVE EFFECTS

Cumulative effects, as defined in 50 CFR 402.02, are those effects of future State or private activities, not involving Federal activities, which are reasonably certain to occur within the action area. Future Federal actions are not considered in the definition of "cumulative effects." It is important to note that the definition of "cumulative effects" in the section 7 regulations is not the same as the NEPA definition of cumulative effects.

Given the nature of the action area (i.e., nearshore and offshore areas off the coast of the U.S., mostly on the OCS), few activities that may affect listed species are likely to occur that do not require some Federal authorization or permitting. Therefore, Section 7 consultations with NMFS are anticipated to be necessary for the majority of future activities that could affect listed species in the action area.

The only portion of the action area that overlaps with state waters is the transit routes that may be used by project vessels. Actions carried out or regulated by the States within that portion of the action area that may affect listed species include the authorization of state fisheries. We are not aware of any local or private actions that are reasonably certain to occur in the action area that may affect listed species.

State Water Fisheries - Future recreational and commercial fishing activities in state waters may result in the capture, injury and mortality of listed species. Information on interactions with listed species for state fisheries operating in the action area is summarized in the Environmental Baseline section above, and it is not clear to what extent these future activities would affect listed species differently than the current state fishery activities described in the Status of the Species/Environmental Baseline section. However, this Opinion assumes effects in the future would be similar to those in the past and are, therefore, reflected in the anticipated trends described in the status of the species and environmental baseline sections of this Opinion.

Although not a traditional cumulative effects analysis in the context of Section 7 consultation, NMFS has evaluated the net additive effects of the full suite of anticipated activities that could occur under the terms of this programmatic consultation. Effects from the activities considered in this programmatic consultation are all temporary (e.g., increases in underwater noise associated with survey activities or pile driving and effects to the benthic environment resulting from geotechnical sampling, etc.). Therefore, we do not anticipate, as a result of the issuance of leases by BOEM or the carrying out of site assessment activities, any negative cumulative effects that will persist in the long-term leading to permanent effects to the environment that would affect listed species.

Any negative effects to listed species and their habitats as a result of the activities to be carried out under the terms of this programmatic consultation are anticipated to be temporary in duration and small in scope and, therefore, discountable and/or insignificant. Temporary, negative effects are only anticipated to occur during project construction or implementation and are only anticipated to occur over short durations on the order of minutes, hours or intermittently over a few days.

Predicting the spatial and temporal occurrences of activities to be carried out is difficult; however, using the leasing scenario established by BOEM, NMFS believes that the likelihood of multiple activities resulting in temporary negative effects that overlap spatially and temporally to the extent that the cumulative effects would result in an adverse effect is discountable. Thus, despite the potential for temporary negative effects, NMFS does not believe the cumulative effects of these activities will have any significant adverse effects to any species of listed whales or sea turtles.

The initial step prior to any activity in the WEAs is for an applicant to obtain a lease from BOEM. BOEM will provide us with notification of any proposed issuance of a lease that contains information on the location of the lease blocks. BOEM will also need to approve any lessees' SAP. BOEM will review each SAP and associated data collection plan to determine if it is consistent with the activities considered in this consultation. Prior to approval of the SAP, BOEM will provide us with written notification of its determination that the site assessment and data collection activities are wholly consistent with the activities and conditions outlined in this consultation and, if the activities are not wholly consistent, how the activities will be modified to be consistent. We will review this determination and provide BOEM written confirmation that we agree that the activities to be carried out are wholly consistent with the activities considered in this consultation. If the lessees plan is not wholly consistent with the activities considered in this consultation, the plan must be modified or BOEM must request a separate ESA Section 7 consultation to consider the activities to be carried out by the lessee. Submission of the notifications will allow us to monitor and track individual and cumulative effects of activities subject to this programmatic consultation. Furthermore, if at any time BOEM or NMFS obtain information that indicates that the proposed activities considered in this consultation are likely to result in impacts to listed species that were not considered herein, this consultation must be reinitiated. Thus, if information obtained through monitoring or other sources indicates that activities are resulting, individually or cumulatively, in adverse effects to listed species or critical habitat, this would represent new information and we would request re-initiation of this consultation.

9.0 INTEGRATION AND SYNTHESIS OF EFFECTS

We have determined that the proposed geophysical surveys in the MA/RI, MA, NY and NJ WEAs and pile driving associated with met tower installation in the MA/RI and MA WEAs are likely to result in the behavioral disturbance of listed whales, sea turtles and Atlantic sturgeon. However, because of the use of exclusion zones and other project design criteria, we do not anticipate the mortality or injury of any individual. All effects will be limited to minor and temporary behavioral disturbances that in the worst case will lead to avoidance of the ensonified areas while the geophysical surveys or pile driving is ongoing. While we are able to use available information to calculate estimates of the number of whales and sea turtles that may be exposed to underwater noise that may result in behavioral disturbance, these estimates are not critical to determining whether the proposed action is likely to jeopardize the continued existence of these species. That is because there is no danger of injury or death to any of these animals due to the expected noise exposure and because we expect any behavioral disturbance to be minor and temporary without any impact on fitness or survival. Therefore, the exact number of animals that will be affected is less important to our analysis than is the type of effects that those animals will experience. We have determined that all other effects to listed species, including benthic disturbance and increased vessel traffic, will be insignificant and discountable.

In the discussion below, we consider whether the effects of the proposed action reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of the listed species in the wild by reducing the reproduction, numbers, or distribution of any listed species. The purpose of this analysis is to determine whether the proposed action, in the context established by the status of the species, environmental baseline, and cumulative effects, would jeopardize the continued existence of any listed species. In the NMFS/USFWS Section 7 Handbook, for the purposes of determining jeopardy, survival is defined as, "the species' persistence as listed or as a recovery unit, beyond the conditions leading to its endangerment, with sufficient resilience to allow for the potential recovery from endangerment. Said in another way, survival is the condition in which a species continues to exist into the future while retaining the potential for recovery. This condition is characterized by a species with a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, which exists in an environment providing all requirements for completion of the species' entire life cycle, including reproduction, sustenance, and shelter." Recovery is defined as, "Improvement in the status of listed species to the point at which listing is no longer appropriate under the criteria set out in Section 4(a)(1) of the Act." Below, for the listed species that may be affected by the proposed action, NMFS summarizes the status of the species and considers whether the proposed action will result in reductions in reproduction, numbers or distribution of that species and then considers whether any reductions in reproduction, numbers or distribution resulting from the proposed action would reduce appreciably the likelihood of both the survival and recovery of that species, as those terms are defined for purposes of the federal Endangered Species Act.

9.1 North Atlantic right whales

As explained in the Opinion, we do not anticipate any injury or mortality of any right whales to result from the proposed action. Temporary, short-term behavioral effects during exposure to underwater noise between 160 and 180 dB re 1uPa such as cessation of feeding, resting, or other activities or temporary alterations in breathing, vocalizing, or diving rates are likely, although these effects are not likely to affect an individual's likelihood of survival or reproduction. We do not anticipate any impacts to the health, survival or reproductive success of any individual right whales. All other effects to right whales, including increased vessel traffic, installation of met buoys, operation of the met towers, geotechnical surveys, and impacts to benthic resources, will be insignificant and discountable.

The survival of any right whales will not be affected by the proposed action. As such, there will be no reduction in the numbers of North Atlantic right whales and no change in the status of this species or its trend.

Reproductive potential of right whales is not expected to be affected in any way. As all behavioral disruption will be minor and temporary and will not cause a delay or disruption of any essential behavior including reproduction there will be no reduction in individual fitness or any future reduction in numbers of individuals.

Any effects to distribution will be minor and temporary and limited to the temporary avoidance of ensonified areas. The proposed action is not likely to reduce distribution because the action will not impede right whales from accessing any known or regular seasonal concentration areas, including foraging or overwintering grounds in the action area or elsewhere.

Based on the information provided above, the behavioral disturbance of right whales due to exposure to noise between 160 and 180 dB re 1uPa (equivalent to MMPA level B harassment) during geophysical surveys in the MA/RI, MA, NY and NJ WEAs over a five-year period, will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of right whales; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of right whales in the action area (related to the temporary avoidance of temporarily ensonified areas) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the species will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery. Recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed action will affect the potential for the species to rebuild to a point where listing is no longer appropriate.

The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the species. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the species can be brought to the point at which it is no longer listed under the ESA. Based on the analysis presented herein, the proposed action, is not likely to appreciably reduce the survival and recovery of this species.

9.2 Humpback whales

As explained in the Opinion, we do not anticipate any injury or mortality of any humpback whales to result from the proposed action. Temporary, short-term behavioral effects during exposure to underwater noise between 160 and 180 dB re 1uPa such as cessation of feeding, resting, or other activities or temporary alterations in breathing, vocalizing, or diving rates are likely, although these effects are not likely to affect an individual's likelihood of survival or reproduction. We do not anticipate any impacts to the health, survival or reproductive success of any individual humpback whales. All other effects to humpback whales, including increased vessel traffic, installation of met buoys, operation of the met towers, geotechnical surveys, and impacts to benthic resources, will be insignificant and discountable.

The survival of any humpback whales will not be affected by the proposed action. As such, there will be no reduction in the numbers of humpback whales and no change in the status of this species or its trend.

Reproductive potential of humpback whales is not expected to be affected in any way. As all behavioral disruption will be minor and temporary and will not cause a delay or disruption of any essential behavior including reproduction there will be no reduction in individual fitness or any future reduction in numbers of individuals.

Any effects to distribution will be minor and temporary and limited to the temporary avoidance of ensonified areas. The proposed action is not likely to reduce distribution because the action will not impede humpback whales from accessing any known or regular seasonal concentration areas, including foraging or overwintering grounds in the action area or elsewhere.

Based on the information provided above, the behavioral disturbance of humpback due to exposure to noise between 160 and 180 dB re 1uPa (equivalent to MMPA level B harassment) during geophysical surveys in the MA/RI, MA, NY and NJ WEAs and pile driving activity in the MA/RI and MA WEAs over a five-year period, will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of humpback whales; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of humpback whales in the action area (related to the temporary avoidance of temporarily ensonified areas) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the species will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery. Recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed action will affect the potential for the species to rebuild to a point where listing is no longer appropriate.

The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the species. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the species can be brought to the point at which it is no longer listed under the ESA. Based on the analysis presented herein, the proposed action, is not likely to appreciably reduce the survival and recovery of this species.

9.3 Fin whales

As explained in the Opinion, we do not anticipate any injury or mortality of any fin whales to result from the proposed action. Temporary, short-term behavioral effects during exposure to underwater noise between 160 and 180 dB re 1uPa such as cessation of feeding, resting, or other activities or temporary alterations in breathing, vocalizing, or diving rates are likely, although these effects are not likely to affect an individual's likelihood of survival or reproduction. We do not anticipate any impacts to the health, survival or reproductive success of any individual fin whales. All other effects to fin whales, including increased vessel traffic, installation of met buoys, operation of the met towers, geotechnical surveys, and impacts to benthic resources, will be insignificant and discountable.

The survival of any fin whales will not be affected by the proposed action. As such, there will be no reduction in the numbers of fin whales and no change in the status of this species or its trend.

Reproductive potential of fin whales is not expected to be affected in any way. As all behavioral disruption will be minor and temporary and will not cause a delay or disruption of any essential behavior including reproduction there will be no reduction in individual fitness or any future reduction in numbers of individuals.

Any effects to distribution will be minor and temporary and limited to the temporary avoidance of ensonified areas. The proposed action is not likely to reduce distribution because the action will not impede fin whales from accessing any known or regular seasonal concentration areas, including foraging or overwintering grounds in the action area or elsewhere.

Based on the information provided above, the behavioral disturbance of fin whales due to exposure to noise between 160 and 180 dB re 1uPa (equivalent to MMPA level B harassment) during geophysical surveys in the MA/RI, MA, NY and NJ WEAs and pile driving activity in the MA/RI and MA WEAs over a five-year period, will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of fin whales; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of fin whales in the action area (related to the temporary avoidance of temporarily ensonified areas) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur.

As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the species will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery. Recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed action will affect the potential for the species to rebuild to a point where listing is no longer appropriate.

The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the species. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the species can be brought to the point at which it is no longer listed under the ESA. Based on the analysis presented herein, the proposed action, is not likely to appreciably reduce the survival and recovery of this species.

9.4 Sei whales

As explained in the Opinion, we do not anticipate any injury or mortality of any sei whales to result from the proposed action. Sei whales do not occur in the NY and NJ wind energy areas; therefore, they will only be exposed to effects of activities in the MA and MA/RI WEAs. Temporary, short-term behavioral effects during exposure to underwater noise between 160 and 180 dB re 1uPa such as cessation of feeding, resting, or other activities or temporary alterations in breathing, vocalizing, or diving rates are likely, although these effects are not likely to affect an individual's likelihood of survival or reproduction. We do not anticipate any impacts to the health, survival or reproductive success of any individual sei whales. All other effects to sei whales, including increased vessel traffic, installation of met buoys, operation of the met towers, geotechnical surveys, and impacts to benthic resources, will be insignificant and discountable.

The survival of any sei whales will not be affected by the proposed action. As such, there will be no reduction in the numbers of sei whales and no change in the status of this species or its trend.

Reproductive potential of sei whales is not expected to be affected in any way. As all behavioral disruption will be minor and temporary and will not cause a delay or disruption of any essential behavior including reproduction there will be no reduction in individual fitness or any future reduction in numbers of individuals.

Any effects to distribution will be minor and temporary and limited to the temporary avoidance of ensonified areas. The proposed action is not likely to reduce distribution because the action will not impede sei whales from accessing any seasonal concentration areas, including foraging or overwintering grounds in the action area or elsewhere.

Based on the information provided above, the behavioral disturbance of sei whales due to exposure to noise between 160 and 180 dB re 1uPa (equivalent to MMPA level B harassment) during geophysical surveys in the MA/RI and MA WEAs and pile driving activity in the MA/RI and MA WEAs over a five-year period, will not appreciably reduce the likelihood of survival of this species (*i.e.,* it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of sei whales; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of sei whales in the action area (related to the temporary avoidance of temporarily ensonified areas) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the species will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery. Recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed action will affect the potential for the species to rebuild to a point where listing is no longer appropriate.

The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the species. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the species can be brought to the point at which it is no longer listed under the ESA. Based on the analysis presented herein, the proposed action, is not likely to appreciably reduce the survival and recovery of this species.

9.5 Sperm whales

As explained in the Opinion, we do not anticipate any injury or mortality of any sperm whales to result from the proposed action. Sperm whales do not occur in the NY and NJ wind energy areas; therefore, they will only be exposed to effects of activities in the MA and MA/RI WEAs. Temporary, short-term behavioral effects during exposure to underwater noise between 160 and 180 dB re 1uPa such as cessation of feeding, resting, or other activities or temporary alterations in breathing, vocalizing, or diving rates are likely, although these effects are not likely to affect an individual's likelihood of survival or reproduction. We do not anticipate any impacts to the health, survival or reproductive success of any individual sperm whales. All other effects to sperm whales, including increased vessel traffic, installation of met buoys, operation of the met towers, geotechnical surveys, and impacts to benthic resources, will be insignificant and discountable.

The survival of any sperm whales will not be affected by the proposed action. As such, there will be no reduction in the numbers of sperm whales and no change in the status of this species or its trend.

Reproductive potential of sperm whales is not expected to be affected in any way. As all behavioral disruption will be minor and temporary and will not cause a delay or disruption of any essential behavior including reproduction there will be no reduction in individual fitness or any future reduction in numbers of individuals.

Any effects to distribution will be minor and temporary and limited to the temporary avoidance of ensonified areas. The proposed action is not likely to reduce distribution because the action will not impede sperm whales from accessing any seasonal concentration areas, including foraging or overwintering grounds in the action area or elsewhere.

Based on the information provided above, the behavioral disturbance of sperm whales due to exposure to noise between 160 and 180 dB re 1uPa (equivalent to MMPA level B harassment) during geophysical surveys in the MA/RI and MA WEAs and pile driving activity in the MA/RI and MA WEAs over a five-year period, will not appreciably reduce the likelihood of survival of this species (*i.e.,* it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of sperm whales; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of sperm whales in the action area (related to the temporary avoidance of temporarily ensonified areas) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the species will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery. Recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed action will affect the potential for the species to rebuild to a point where listing is no longer appropriate.

The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the species. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the species can be brought to the point at which it is no longer listed under the ESA. Based on the analysis presented herein, the proposed action, is not likely to appreciably reduce the survival and recovery of this species.

9.6 Northwest Atlantic DPS of Loggerhead Sea Turtles

As explained in the Opinion, we do not anticipate any injury or mortality of any loggerhead sea turtles to result from the proposed action. Temporary, short-term behavioral effects during exposure to underwater noise between 166 and 180 dB re 1uPa RMS are likely, although these effects are not likely to affect an individual's likelihood of survival or reproduction. We do not anticipate any impacts to the health, survival or reproductive success of any individual loggerhead sea turtles. All other effects to sea turtles, including increased vessel traffic, installation of met buoys, operation of the met towers, geotechnical surveys, and impacts to benthic resources, will be insignificant and discountable.

The survival of any sea turtles will not be affected by the proposed action. As such, there will be no reduction in the numbers of loggerhead sea turtles and no change in the status of this species or its trend.

Reproductive potential of sea turtles is not expected to be affected in any way. As all behavioral disruption will be minor and temporary and will not cause a delay or disruption of any essential behavior including reproduction there will be no reduction in individual fitness or any future reduction in numbers of individuals.

Any effects to distribution will be minor and temporary and limited to the temporary avoidance of ensonified areas. The proposed action is not likely to reduce distribution because the action will not impede sea turtles from accessing any known or regular seasonal concentration areas, including foraging or overwintering grounds in the action area or elsewhere.

Based on the information provided above, the behavioral disturbance of loggerhead sea turtles due to exposure to noise between 166 and 180 dB re 1uPa RMS during geophysical surveys in the MA/RI, MA, NY and NJ WEAs and pile driving activity in the MA/RI and MA WEAs over a five-year period, will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of loggerhead sea turtles; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of loggerhead sea turtles in the action area (related to the temporary avoidance of temporarily ensonified areas) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the species will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery. Recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed action will affect the potential for the species to rebuild to a point where listing is no longer appropriate.

The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the species. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the species can be brought to the point at which it is no longer listed under the ESA. Based on the analysis presented herein, the proposed action, is not likely to appreciably reduce the survival and recovery of this species.

9.6 Leatherback sea turtles

As explained in the Opinion, we do not anticipate any injury or mortality of any leatherback sea turtles to result from the proposed action. Temporary, short-term behavioral effects during exposure to underwater noise between 166 and 180 dB re 1uPa RMS are likely, although these effects are not likely to affect an individual's likelihood of survival or reproduction. We do not anticipate any impacts to the health, survival or reproductive success of any individual leatherback sea turtles. All other effects to sea turtles, including increased vessel traffic, installation of met buoys, operation of the met towers, geotechnical surveys, and impacts to benthic resources, will be insignificant and discountable.

The survival of any sea turtles will not be affected by the proposed action. As such, there will be no reduction in the numbers of leatherback sea turtles and no change in the status of this species or its trend.

Reproductive potential of sea turtles is not expected to be affected in any way. As all behavioral disruption will be minor and temporary and will not cause a delay or disruption of any essential behavior including reproduction there will be no reduction in individual fitness or any future reduction in numbers of individuals.

Any effects to distribution will be minor and temporary and limited to the temporary avoidance of ensonified areas. The proposed action is not likely to reduce distribution because the action will not impede sea turtles from accessing any seasonal concentration areas, including foraging or overwintering grounds in the action area or elsewhere.

Based on the information provided above, the behavioral disturbance of leatherback sea turtles due to exposure to noise between 166 and 180 dB re 1uPa RMS during geophysical surveys in the MA/RI, MA, NY and NJ WEAs and pile driving activity in the MA/RI and MA WEAs over a five-year period, will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of leatherback sea turtles; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of leatherback sea turtles in the action area (related to the temporary avoidance of temporarily ensonified areas) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the species will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery. Recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed action will affect the potential for the species to rebuild to a point where listing is no longer appropriate.

The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the species. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the species can be brought to the point at which it is no longer listed under the ESA. Based on the analysis presented herein, the proposed action, is not likely to appreciably reduce the survival and recovery of this species.

9.7 Kemp's ridley sea turtles

As explained in the Opinion, we do not anticipate any injury or mortality of any Kemp's ridley sea turtles to result from the proposed action. Temporary, short-term behavioral effects during exposure to underwater noise between 166 and 180 dB re 1uPa RMS are likely, although these effects are not likely to affect an individual's likelihood of survival or reproduction. We do not anticipate any impacts to the health, survival or reproductive success of any individual Kemp's ridley sea turtles. All other effects to sea turtles, including increased vessel traffic, installation of met buoys, operation of the met towers, geotechnical surveys, and impacts to benthic resources, will be insignificant and discountable.

The survival of any sea turtles will not be affected by the proposed action. As such, there will be no reduction in the numbers of Kemp's ridley sea turtles and no change in the status of this species or its trend.

Reproductive potential of sea turtles is not expected to be affected in any way. As all behavioral disruption will be minor and temporary and will not cause a delay or disruption of any essential behavior including reproduction there will be no reduction in individual fitness or any future reduction in numbers of individuals.

Any effects to distribution will be minor and temporary and limited to the temporary avoidance of ensonified areas. The proposed action is not likely to reduce distribution because the action will not impede sea turtles from accessing any seasonal concentration areas, including foraging or overwintering grounds in the action area or elsewhere.

Based on the information provided above, the behavioral disturbance of Kemp's ridley sea turtles due to exposure to noise between 166 and 180 dB re 1uPa RMS during geophysical surveys in the MA/RI, MA, NY and NJ WEAs and pile driving activity in the MA/RI and MA WEAs over a five-year period, will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of Kemp's ridley sea turtles; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of Kemp's ridley sea turtles in the action area (related to the temporary avoidance of temporarily ensonified areas) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the species will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery. Recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed action will affect the potential for the species to rebuild to a point where listing is no longer appropriate.

The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the species. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the species can be brought to the point at which it is no longer listed under the ESA. Based on the analysis presented herein, the proposed action, is not likely to appreciably reduce the survival and recovery of this species.

9.8 Green sea turtles

As explained in the Opinion, we do not anticipate any injury or mortality of any green sea turtles to result from the proposed action. Temporary, short-term behavioral effects during exposure to underwater noise between 166 and 180 dB re 1uPa RMS are likely, although these effects are not likely to affect an individual's likelihood of survival or reproduction. We do not anticipate any impacts to the health, survival or reproductive success of any individual green sea turtles. All other effects to sea turtles, including increased vessel traffic, installation of met buoys, operation of the met towers, geotechnical surveys, and impacts to benthic resources, will be insignificant and discountable.

The survival of any sea turtles will not be affected by the proposed action. As such, there will be no reduction in the numbers of green sea turtles and no change in the status of this species or its trend.

Reproductive potential of sea turtles is not expected to be affected in any way. As all behavioral disruption will be minor and temporary and will not cause a delay or disruption of any essential behavior including reproduction there will be no reduction in individual fitness or any future reduction in numbers of individuals.

Any effects to distribution will be minor and temporary and limited to the temporary avoidance of ensonified areas. The proposed action is not likely to reduce distribution because the action will not impede sea turtles from accessing any seasonal concentration areas, including foraging or overwintering grounds in the action area or elsewhere.

Based on the information provided above, the behavioral disturbance of green sea turtles due to exposure to noise between 166 and 180 dB re 1uPa RMS during geophysical surveys in the MA/RI, MA, NY and NJ WEAs and pile driving activity in the MA/RI and MA WEAs over a five-year period, will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of green sea turtles; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of green sea turtles in the action area (related to the temporary avoidance of temporarily ensonified areas) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the species will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery. Recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed action will affect the potential for the species to rebuild to a point where listing is no longer appropriate.

The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the species. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the species can be brought to the point at which it is no longer listed under the ESA. Based on the analysis presented herein, the proposed action, is not likely to appreciably reduce the survival and recovery of this species.

9.9 Atlantic sturgeon

Based on the mixed-stock analysis resulting from the genetic assignments of the NEFOP samples, we have determined that Atlantic sturgeon in the action area likely originate from the five DPSs at the following frequencies: NYB 49%; South Atlantic 20%; Chesapeake Bay 14%; Gulf of Maine 11%; and Carolina 4%. Two percent of Atlantic sturgeon in the action area may originate from the St. John's River in Canada; these fish are not included in the 2012 ESA listing. The genetic assignments have a plus/minus 5% confidence interval; however, for purposes of section 7 consultation we have selected the reported values above, which approximate the mid-point of the range, as a reasonable indication of the likely genetic makeup of Atlantic sturgeon in the action area. These assignments and the data from which they are derived are described in detail in Damon-Randall *et al.* (2013). We expect Atlantic sturgeon affected by the proposed action to originate from all five DPSs at the frequencies noted above.

As explained in the Opinion, we do not anticipate any injury or mortality of any Atlantic sturgeon to result from the proposed action. Temporary, short-term behavioral effects during exposure to underwater noise above 150 dB re 1uPa RMS are likely, although these effects are not likely to affect an individual's likelihood of survival or reproduction. We do not anticipate any impacts to the health, survival or reproductive success of any individual Atlantic sturgeon, from any DPS. All other effects to Atlantic sturgeon, including increased vessel traffic, installation of met buoys, operation of the met towers, geotechnical surveys, and impacts to benthic resources, will be insignificant and discountable.

The survival of any Atlantic sturgeon will not be affected by the proposed action. As such, there will be no reduction in the numbers of Atlantic sturgeon from any DPS and no change in the status or trend of any DPS.

Reproductive potential of Atlantic sturgeon is not expected to be affected in any way. As all behavioral disruption will be minor and temporary and will not cause a delay or disruption of any essential behavior including reproduction there will be no reduction in individual fitness or any future reduction in numbers of individuals.

Any effects to distribution will be minor and temporary and limited to the temporary avoidance of ensonified areas. The proposed action is not likely to reduce distribution because the action will not impede any Atlantic sturgeon from accessing any seasonal concentration areas, including foraging or overwintering grounds in the action area or elsewhere.

Based on the information provided above, the behavioral disturbance of Atlantic sturgeon due to exposure to noise above 150 dB re 1uPa RMS during geophysical surveys in the MA/RI, MA, NY and NJ WEAs and pile driving activity in the MA/RI and MA WEAs over a five-year period, will not appreciably reduce the likelihood of survival of any Atlantic sturgeon DPS (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of Atlantic sturgeon originating from any DPS; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of any DPS; (3) and, the action will have only a minor and temporary effect on the distribution of

Atlantic sturgeon in the action area (related to the temporary avoidance of temporarily ensonified areas) and no effect on the distribution of any DPS throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that any Atlantic sturgeon DPS will survive in the wild. Here, we consider the potential for the action to reduce the likelihood of recovery for any Atlantic sturgeon DPS. Recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed action will affect the potential for any Atlantic sturgeon DPS to rebuild to a point where listing is no longer appropriate.

The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of any Atlantic sturgeon DPS. There will not be a change in the status or trend of any Atlantic sturgeon DPS. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of any Atlantic sturgeon DPS. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for any Atlantic sturgeon DPS. The effects of the proposed action will also not reduce the likelihood that the status of any Atlantic sturgeon DPS can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that any Atlantic sturgeon DPS can be brought to the point at which it is no longer listed under the ESA. Based on the analysis presented herein, the proposed action, is not likely to appreciably reduce the survival and recovery of any Atlantic sturgeon DPS.

10.0 CONCLUSION

After reviewing the best available information on the status of endangered and threatened species under NMFS jurisdiction, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS's biological opinion that the proposed action may adversely affect but is not likely to jeopardize the continued existence of Kemp's ridley, green, or leatherback sea turtles; the NWA DPS of loggerhead sea turtles; North Atlantic right, humpback, fin, sei, or sperm whales, or the GOM, NYB, CB or SA DPSs of Atlantic sturgeon. Because no critical habitat is designated in the action area, none will be affected by the action.

11.0 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA prohibits the take of endangered species of fish and wildlife. "Fish and wildlife" is defined in the ESA "as any member of the animal kingdom, including without limitation any mammal, fish, bird (including any migratory, non-migratory, or endangered bird for which protection is also afforded by treaty or other international agreement), amphibian, reptile, mollusk, crustacean, arthropod or other invertebrate, and includes any part, product, egg, or offspring thereof, or the dead body or parts thereof." 16 U.S.C. 1532(8). "Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by NMFS to include any act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation that actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns including breeding, spawning, rearing, migrating, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. "Otherwise lawful activities" are those actions that meet all State and Federal legal requirements except for the prohibition against taking in ESA Section 9 (51 FR 19936, June 3, 1986), which would include any state endangered species laws or regulations. Section 9(g) makes it unlawful for any person "to attempt to commit, solicit another to commit, or cause to be committed, any offense defined [in the ESA.]" 16 U.S.C. 1538(g). A "person" is defined in part as any entity subject to the jurisdiction of the United States, including an individual, corporation, officer, employee, department or instrument of the Federal government (see 16 U.S.C. 1532(13)). Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

The measures described below are non-discretionary, and must be undertaken by BOEM and any lessees/applicants, for the exemption in section 7(o)(2) to apply. BOEM has a continuing duty to regulate the activity covered by this Incidental Take Statement. If BOEM (1) fails to assume and implement the terms and conditions consistent with its authority or (2) fails to require any lessee/applicant, to adhere to the terms and conditions of the Incidental Take Statement through enforceable terms that are added any leases or approvals consistent with its authority, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, BOEM must report the progress of the actions and their impact on the species to us as specified in the Incidental Take Statement [50 CFR §402.14(i)(3)] (See U.S. Fish and Wildlife Service and National Marine Fisheries Service's Joint Endangered Species Act Section 7 Consultation Handbook (1998) at 4-49).

We have concluded that the proposed actions are likely to result in take of North Atlantic right, humpback, fin, sei and sperm whales in the form of harassment, where habitat conditions (i.e., sound levels above the 160 dB threshold for pulsed noise used to determine harassment under the MMPA) will temporarily impair normal behavior patterns. This harassment will occur in the form of avoidance or displacement from preferred habitat and behavioral and/or metabolic compensations to deal with short-term masking or stress. While whales may experience temporary impairment of behavior patterns, no significant impairment resulting in injury (i.e., "harm") is likely due to: measures to ensure that no whales are exposed to sound levels that could result in injury, the ability of whales to easily move to areas beyond the impact zone that also provide suitable prey, and the limited exposure time to disturbing levels of sound.

While the Opinion includes an estimate of the number of whales that are likely to experience harassment, this Opinion does not include an incidental take authorization for right, humpback, fin, or sei whales at this time because the incidental take of these ESA-listed whales has not been authorized under section 101(a)(5) of the MMPA. Following the issuance of such authorizations, we may amend this Opinion to include an incidental take exemption for these species, as appropriate.

11.1 Anticipated Amount or Extent of Incidental Take

As established in the Biological Opinion, loggerhead, Kemp's ridley, green and leatherback sea turtles and Atlantic sturgeon from all five DPSs are likely to be exposed to increased underwater noise that will cause behavioral disruption. We have determined that the proposed action is likely to result in take of these species, in the form of harassment, where habitat conditions (i.e., increased underwater noise) will temporarily impair normal behavior patterns. This harassment will occur in the form of avoidance or displacement from ensonified areas and the temporary disruption of normal foraging, resting and migratory behaviors. Affected individuals will expend additional energy to swim away from ensonified areas. No mortality, injury or harm is anticipated. This is due to the level of noise that individuals will be exposed to as well as the temporary nature of this exposure and the extent of available habitat in the action area where noise levels will not be elevated and the ability of individuals to avoid noisy areas and move to areas without disturbing levels of underwater noise. Exposure of sea turtles to sound levels greater than 166 dB re 1uPa RMS will be considered harassment because that level of noise will disturb sea turtles and their normal behaviors (i.e., resting, foraging or migrating through the area) will be interrupted. Exposure of Atlantic sturgeon to sound levels greater than 150 dB re 1uPa RMS will be considered harassment because that level of noise will disturb Atlantic sturgeon and their normal behaviors (i.e., resting, foraging or migrating through the area) will be interrupted. Given the large size of the area where noise of this level will be experienced (approximately 162 square km during pile driving and 14 square km when the boomer is operated) there will be behavioral and/or metabolic (e.g., temporary increase in energy expenditure) costs associated with avoidance or displacement from the affected habitat.

Because there are no available estimates of Atlantic sturgeon density in the action area, we are not able to estimate the number of Atlantic sturgeon of any DPS that may be taken by harassment. Because both the distribution and numbers of Atlantic sturgeon in the action area during a geophysical survey or pile driving is likely to be highly variable and a function of the time of year, the behavior of individual fish, the distribution of prey and other environmental variables, and because incidental take is indirect and likely to occur from effects to habitat, the amount of take resulting from harassment is difficult, if not impossible, to estimate. In addition, because there are no known means to detect the presence of Atlantic sturgeon during geophysical surveys or pile driving activities, it would be extremely difficult, if not impossible, to monitor the behavior of all Atlantic sturgeon in the action area in a manner which would detect responses to geophysical surveys or pile driving, the likelihood of discovering take attributable to exposure to increased underwater noise is very limited. In such circumstances, NMFS uses a surrogate to estimate the extent of take. The surrogate must be rationally connected to the taking and provide a threshold of exempted take which, if exceeded, provides a basis for reinitiating consultation. For this proposed action, the spatial and temporal extent of the area where underwater noise is elevated above 150 dB re 1uPa RMS will serve as a surrogate for estimating the amount of incidental take from harassment as it allows NMFS to determine the area and time when sturgeon will be exposed to noise would result in behaviors consistent with the definition of harassment.

For sea turtles, we are able to use published density estimates to estimate the number of sea turtles that may be exposed to increased underwater noise that would cause harassment during geophysical surveys where a boomer is used and during pile driving. For loggerheads, the density estimates indicate that up to 7 loggerheads and 1 leatherback sea turtle are likely to be exposed to potentially disturbing levels of noise each time the boomer is operated. If we assume that the entirety of the NY, NJ, RI/MA and MA WEA are surveyed, an area that totals approximately 5,439 square kilometers, we would expect that a total of up to 201 leatherbacks and up to 2,774 loggerheads may be exposed to potentially disturbing levels of noise associated with the boomer surveys over a five year period. No density estimates are available for Kemp's ridley or green sea turtles; however, we expect fewer sea turtles of these species than leatherbacks in the action area. Therefore, each time the boomer is operated, no more than 1 Kemp's ridley and 1 green sea turtles are likely to experience potentially disturbing levels of noise. In total, we expect up to 201 Kemp's ridley and 201 green sea turtles to be exposed to potentially disturbing levels of noise from the boomers.

Our calculations based on reported density estimates indicate that up to 83 loggerheads and up to 6 leatherback sea turtles are likely to be exposed to potentially disturbing levels of noise for each pile that is installed. Depending on the type of met tower installed (monopole or tripod) and the total number of met towers installed, there could be a total of 9-27 piles installed over the entirety of the RI/MA and MA WEAs. In total, we would expect that no more than 2,774 loggerheads and no more than 162 leatherback sea turtles may be exposed to potentially disturbing levels of noise. No density estimates are available for Kemp's ridley or green sea turtles; however, we expect fewer sea turtles of these species than leatherbacks in the action area. Therefore, for each pile that is installed, we expect no more than 6 Kemp's ridley and 6 green sea turtles are likely to experience potentially disturbing levels of noise. In total, we expect no more than 162 Kemp's ridleys and 162 green sea turtles to be exposed to potentially disturbing levels of noise from pile driving.

As explained in the Opinion, these calculations are likely to result in overestimates of the number of individuals exposed. For the geophysical surveys, we consider this a worst case estimate because: (1) it assumes that sea turtle density will be at the maximum reported level throughout the action area, which is unlikely to occur; (2) it uses the maximum distances modeled by BOEM for noise attenuation; and, (3) it assumes that all surveys will occur at a time of year when sea turtles are present (June – November) and that sea turtles will be present at every location that the boomer is operated. For pile driving, we consider this a worst case estimate because: (1) it assumes that sea turtle density will be at the maximum reported level throughout the action area, which is unlikely to occur; (2) it uses the maximum distances modeled by BOEM for noise attenuation; and, (3) it assumes that sea turtles will be present at every location that a pile is installed.

Despite these assumptions, this is the best available estimate of the number of sea turtles that may be exposed to disturbing levels of noise from the boomer. Because both the distribution and numbers of sea turtles in the action area during a geophysical survey or pile driving is likely to be highly variable and a function of the time of year, the behavior of individual turtles, the distribution of prey and other environmental variables, and because incidental take is indirect and likely to occur from effects to habitat, the amount of take resulting from harassment is difficult, if not impossible, to estimate. In addition, because of the large size of ensonified area, we do not expect that BOEM or the lessees will be able to monitor the behavior of all sea turtles in the action area in a manner which would detect responses to geophysical surveys or pile driving; therefore, the likelihood of discovering take attributable to exposure to increased underwater noise is very limited. In such circumstances, NMFS uses a surrogate to estimate the extent of take. The surrogate must be rationally connected to the taking and provide a threshold of exempted take which, if exceeded, provides a basis for reinitiating consultation. For this proposed action, the spatial and temporal extent of the area where underwater noise is elevated above 166 dB re 1uPa RMS will serve as a surrogate for estimating the amount of incidental take from harassment as it allows NMFS to determine the area and time when loggerhead, leatherback, Kemp's ridley and green sea turtles will be exposed to noise would result in behaviors consistent with the definition of harassment. Based on the known distribution of sea turtles in the action area, we only anticipate harassment during pile driving and geophysical surveys that occur from May – November. In the accompanying biological opinion, we determined that this level of anticipated take is not likely to result in jeopardy to any of the affected species.

11.2 Reasonable and Prudent Measures

Reasonable and prudent measures are those measures necessary and appropriate to minimize and monitor incidental take of a listed species. These reasonable and prudent measures are in addition to the project design criteria proposed by BOEM that will be required for all lessees (see section 3.6, above) We believe the following reasonable and prudent measures are necessary and appropriate to minimize and monitor impacts of incidental take of sea turtles and Atlantic sturgeon. If additional RPMs are necessary and appropriate to minimize and monitor impacts of incidental take of listed whales, those RPMs will be added upon amendment of this ITS.

1. BOEM must provide NMFS with notice and opportunity to comment on the proposed issuance of leases, approval of SAPs, data collection plans, and decommissioning applications for facilities constructed under an approved SAP. This notification may occur through e-mail to NMFS staff that will be identified annually.
2. Field verification of modeled noise levels must be undertaken for electromechanical survey equipment operating below 200 kHz in each of the lease holds. This verification must take place prior to the equipment being used for any survey activities.
3. In order to monitor the acoustic effects of pile driving, acoustic monitoring of pile driving must be conducted to confirm the sound levels modeled by BOEM and reported in the BA.
4. BOEM must keep NMFS informed of all geophysical and geotechnical surveys and pile driving activity conducted by BOEM lessees in support of a SAP, COP, or GAP in the MA/RI, MA, NY and NJ WEAs.

11.3 Terms and conditions

In order to be exempt from prohibitions of section 9 of the ESA, BOEM must comply with the following terms and conditions, which implement the reasonable and prudent measures described above and which outline required minimization and monitoring requirements. These terms and conditions are non-discretionary.

1. To implement RPM #1, BOEM must provide NMFS (by e-mail to the NERO PRD Section 7 coordinator, or other contact provided annually by NMFS) with written notification of any proposed issuance of a lease. This must contain information on the location of the lease blocks and any proposed activities that will occur on the lease.
2. To implement RPM #1, BOEM must review each SAP, data collection plan, and/or decommissioning application for facilities in a BOEM-approved SAP to determine if it is wholly consistent with the activities considered in this consultation. At least 30 days prior to review of a survey plan, approval of a SAP or decommissioning application for facilities approved in a SAP BOEM will provide NMFS with written notification of its determination that the site assessment, data collection activities, or decommissioning application are wholly consistent with the activities and conditions outlined in this consultation. If BOEM has determined that the applicant's proposal is not consistent with the activities and conditions outlined in this consultation, BOEM must provide NMFS with a written explanation of how the plan will be modified. If the plan will not be modified, BOEM must request a separate section 7 consultation.
3. To implement RPM #2, sound source verification must be conducted prior to the commencement of surveys that involve the operation of the electromechanical survey equipment operating below 200 kHz in each of the lease holds in the four WEAs. Acoustic measurements must be sufficient to establish the following: source level (peak at 1 meter) and distance to the 180, 160 and 150 dB re 1uPa RMS isopleths. Results of this monitoring must be reported to NMFS as soon as practicable.
4. To implement RPM #3, acoustic monitoring must be conducted during the installation each meteorological tower requiring pile driving. Acoustic monitoring must be sufficient to determine the following: source level (peak at 1 meter) and distance to the 180, 160 and 150 dB re 1µPa RMS isopleths as well as 187 dB re 1µPa CSEL. Results of this monitoring must be reported to NMFS as soon as practicable after the completion of the pile driving activity.
5. To implement RPM #4, BOEM must provide NMFS with notice (email or telephone, to a contact provided annually by NMFS) no later than three days prior to scheduled geological and geophysical surveys and meteorological tower construction. BOEM must also provide notice when these activities are completed.
6. To implement RPM #4, prior to April 1 of each year, BOEM must submit a report to NMFS detailing the activities that occurred in the previous calendar year that were subject to this consultation and any impacts to listed species from those activities.

The reasonable and prudent measures, with their implementing terms and conditions, are designed to minimize and monitor the impact of incidental take that might otherwise result from the proposed action. Specifically, these RPMs and Terms and Conditions will ensure that no listed species are exposed to injurious levels of sound and will verify the modeling results provided by BOEM based on which NMFS has made conclusions regarding take. The RPMs and Terms and Conditions also serve to monitor and track individual and cumulative effects of activities subject to this programmatic consultation.

RPM #1 and Term and Condition #1 and #2 are necessary and appropriate because they will allow BOEM and NMFS to keep track of the activities that are being considered for coverage under this Opinion and ITS and will allow both agencies to track individual and cumulative effects of the activities considered here. This is only a minor change because it is not expected to result in any delay to the project or increased cost and will merely involve occasional communications and coordination between BOEM and NMFS staff.

RPM #2 and 3 and Term and Condition #3 and 4 are necessary and appropriate because they are designed to verify that the sound levels modeled by BOEM are valid and that the estimated areas where sound levels are expected to be greater than the threshold levels for effects to listed species are accurate. Any increases in cost or time are expected to be minor as these measurements will not be required for all survey activities or for the installation of all piles.

RPM #4 and Term and Condition #5 and 6 are necessary and appropriate because they will serve to ensure that we are aware of the dates and locations of all survey and pile driving activities. This will allow us to monitor the duration and seasonality of these activities as well as give us an opportunity to provide BOEM with any updated contact information for NMFS staff. This is only a minor change because it is not expected to result in any delay to the project and will merely involve an occasional telephone call or e-mail between BOEM and NMFS staff.

12.0 CONSERVATION RECOMMENDATIONS

In addition to section 7(a)(2), which requires agencies to ensure that proposed actions are not likely to jeopardize the continued existence of listed species, section 7(a)(1) of the ESA places a responsibility on all Federal agencies to utilize their authorities in furtherance of the purposes of the ESA by carrying out programs for the conservation of endangered and threatened species. Conservation Recommendations are discretionary activities designed to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. The following additional measures are recommended:

1. BOEM should use its authorities to support research on the effects of pile driving and geophysical survey noise on NMFS listed species.

13.0 REINITIATION OF CONSULTATION

This concludes formal consultation on the proposed action. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In the event that the amount or extent of incidental take is exceeded, Section 7 consultation must be reinitiated immediately.

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APPENDIX A

Incident Report: Protected Species Injury or Mortality

Photographs should be taken of all injured or dead animals.

Observer's full name:____
Reporter's full name:____

Species Identification:____

Activity ongoing at time of observation (e.g., transit, survey, pile driving, etc.):
____
____

Date animal observed:________________ Time animal observed: ________________________
Date animal collected:________________ Time animal collected:_________________________

Environmental conditions at time of observation (i.e., tidal stage, weather):
____
____

Water temperature (°C) at site and time of observation:_________________________
Describe location of animal and how it was documented (i.e., observer on boat):
____

Sturgeon Information:
Species ____________________________________

Fork length (or total length) ______________________ Weight _______________________

Condition of specimen/description of animal
____
____
____
____

Fish Decomposed: NO SLIGHTLY MODERATELY SEVERELY
Fish tagged: YES / NO *Please record all tag numbers.* Tag # _________________

Photograph taken: YES / NO
(please label *species, date, geographic site* and *vessel name* when transmitting photo)

Genetics Sample taken: YES / NO
Genetics sample transmitted to: _____________________ on ____/_____/20____

Sea Turtle Species Information: *(please designate cm/m or inches.)*
Species ______________________________ Weight (kg or lbs)____________________________

Sex (circle): Male Female Unknown How was sex determined? ____________________

Straight carapace length ________________ Straight carapace width ______________________

Curved carapace length ________________ Curved carapace width _______________________

Plastron length ________________________ Plastron width ______________________________

Tail length ____________________________ Head width ________________________________

Condition of specimen/description of animal____
____

Existing Flipper Tag Information
Left ______________________________ Right ___________________________________
PIT Tag # __________________________________
Miscellaneous:
Genetic biopsy taken: YES NO
Photos Taken: YES NO

Turtle Release Information:
Date ____________________________ Time ________________________________
Lat _____________________________ Long ________________________________
State ___________________________ County ______________________________

Remarks: (note if turtle was involved with tar or oil, gear or debris entanglement, wounds or mutilations, propeller damage, papillomas, old tag locations, etc.)
____
____

MARINE MAMMAL INFORMATION:
Species ____

Injuries Observed ____

Condition/Description of Animal ____
____

Other Remarks ____

Time Reported to NMFS Stranding Hotline: ____