NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
BIOLOGICAL OPINION

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

Activity: Deepwater Wind: Block Island Wind Farm and Transmission System
NER-2013-9995
GARFO-2013-00026

Conducted by: National Marine Fisheries Service
Northeast Regional Office

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Approved by: Harold C Mears

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1.0 INTRODUCTION

This constitutes NOAA's National Marine Fisheries Service's (NMFS) biological opinion, (Opinion) issued in accordance with section 7 of the Endangered Species Act of 1973 (ESA), as amended, on the effects of Deepwater Wind Block Island, LLC's, and Deepwater Wind Block Transmission, LLC's, proposals to construct and operate the Block Island Wind Farm (BIWF) and its associated Block Island Transmission System (BITS), respectively.[1] The U.S. Army Corps of Engineers (USACE) is the lead Federal agency for the proposed construction and operation of the BIWF and BITS. The UASCE will authorize the construction and operation of the BIWF and BITS via the issuance of individual permits under Section 10 of the Rivers and Harbors Act and Section 404 of the Clean Water Act to Deepwater Wind. Additionally, as Deepwater Wind has submitted to the Bureau of Ocean Energy Management (BOEM) a right-of-way (ROW) grant request for the installation of the portion of the BITS that traverses Federal waters, BOEM is also a Federal action agency for this consultation.[2]

This Opinion is based on information provided in the September 2012 and 2013 Deepwater Wind Block Island Wind Farm and Block Island Transmission System Environmental Report (ER), correspondence with USACE, TetraTech, and Deepwater Wind, and other sources of information provided to us through November 8, 2013.[3] A complete administrative record of this consultation will be kept on file at the NMFS Northeast Regional Office. Formal consultation was initiated on July 12, 2013.

2.0 CONSULTATION HISTORY

On July 5, 2012, the USACE requested consultation pursuant to section 7 of the ESA, on the effects of the construction and operation of the Deepwater Wind project. The USACE requested our concurrence with their determination that the proposed Deepwater Wind project "may affect, but is not likely to adversely affect" any listed species under our jurisdiction. To support their determination, USACE provided a May 2012 ER for the action. Based on our review of the May 2012 ER, in a letter dated August 7, 2012, we requested additional information on the proposed action.

In a letter dated October 9, 2012, the USACE provided us with additional information, as well as a revised ER (dated September 2012). On November 8, 2012, we held a follow up conference call with the USACE and Deepwater Wind, LLC, where Deepwater Wind, LLC, indicated that "take" of a small number of ESA listed whales and sea turtles, in the form of harassment from underwater noise, was likely. Based on this, we determined that formal consultation was necessary. At that time we also requested additional acoustic analysis for all ESA listed species in the project area. In a letter dated November 13, 2012, in response to the USACE's issuance of

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[1] Deepwater Wind Block Island, LLC, and Deepwater Wind Block Transmission, LLC, are both subsidiaries of Deepwater Wind Holdings, LLC; collectively termed "Deepwater Wind."

[2] ROW Grant request was submitted to BOEM on October 7, 2011. A General Activities Plan was submitted to BOEM on April 20, 2012. BOEM published a request for competitive interest in the Federal Register on May 23, 2012 (77 FR 30551). BOEM published a Determination of No Competitive Interest on August 7, 2012. BOEM has not yet approved the ROW easement for Deepwater Wind.

[3] The Environmental Reports constitutes the Biological Assessment for the proposed action.

a public notice on the proposed Deepwater Wind Project, we re-emphasized the need for additional information and analyses on the underwater acoustic footprint of the proposed project and its effects on our ESA listed species. In response to our November 2012 comments and request for additional information and analyses, we received a letter from the USACE, dated April 8, 2013, that included an updated acoustic report. On June 18, 2013, we received additional information on vibratory pile installation.

On July 12, 2013, we received the USACE request for formal consultation. In a letter dated July 24, 2013, we informed the USACE that we received all the information necessary to initiate formal consultation and as a result, July 12, 2013, serves as the initiation date for formal consultation. On September 26, 2013, we were notified by Deepwater Wind of project modifications, with additional details of these modifications provided on September 27, 2013. We requested additional information regarding these modifications on September 27, 2013. We received this information on November 8, 2013.

3.0 DESCRIPTION OF THE ACTION

The USACE is proposing to issue permits to Deepwater Wind Block Island, LLC, and Deepwater Wind Block Transmission, LLC, for the construction, operation, and decommissioning of a 30 mega-watt (MW) offshore wind farm (i.e., BIWF) and the BITS, respectively (see Appendix A).[4] Once constructed, the operational life of the BITS and BIWF is 25 years, with decommissioning expected to last an additional two years beyond that.

The BIWF will be constructed in Rhode Island State territorial waters approximately 3 miles southeast of Block Island and will consist of five 6-MW wind turbine generators (WTGs), spaced 0.5 miles apart; a submarine cable interconnecting the five WTGs (i.e., Inter-Array Cables; total length approximately 2 miles); and a 34.5 kilovolt (kV) submarine transmission cable (i.e., Export Cable), which will originate from the northernmost WTG and travel 6.2 miles to a manhole/interconnection point on the Block Island mainland where the submarine cable will be splice with a terrestrial cable that will continue on shore to the BIWF Generation Switchyard (BIWF Switchyard) which is part of the Block Island Substation located on the Block Island Power Company's property.[5]

In connection with the BIWF, the BITS, a 34.5-kV alternating current bi-directional submarine transmission cable, will be installed from Block Island to the Rhode Island mainland, in the Town of Narragansett. The BITS cable will originate from the BITS Island Switchyard, which is also part of the Block Island Substation. The terrestrial portion of the BITS cable will traverse a terrestrial cable route from the Switchyard to a manhole located in the parking lot of Block Island's Crescent Beach. From the manhole on Crescent Beach, the BITS terrestrial cable will be spliced with the BITS submarine cable that will traverse Rhode Island State territorial waters and

[4] Deepwater Wind Block Island, LLC, and Deepwater Wind Block Transmission, LLC, are both subsidiaries of Deepwater Wind Holdings, LLC; collectively termed "Deepwater Wind."

[5] The Block Island Substation will be the point of interconnection between the BIWF and the BITS. The Block Island Substation will consist of two adjoining switchyards, one dedicated to the BIWF (BIWF Switchyard) and the other dedicated to the BITS (BITS Island Switchyard).

Federal waters, to an interconnection point/manhole on the Rhode Island mainland (i.e., Scarborough State Beach (Scarborough Beach)). Once on the mainland, the terrestrial cable will continue to the Dillon's Corner Switchyard, and from there, traverse another 0.9 miles to an interconnection point at the exiting Narragansett Electric Company's National Grid Wakefield Substation, South Kingston, Rhode Island. In total, the submarine portion of the BITS will be 19.8 miles long. Because the transmission line crosses through Federal waters, the applicant must obtain a Right of Way from BOEM; this was issued in 2012.

Specific details on the construction, operation, maintenance, and decommissioning of the BIWF and BITS are provided below.

3.1 BIWF

3.1.1 Wind Turbine Generator Overview

The BIWF will consist of five, 6 MW WTGs arranged in a radial configuration spaced approximately 0.5 miles apart. Each turbine is pitch-regulated with active yaw to allow it to turn into the wind. Each WTG is comprised of a tower, a three bladed rotor, and a nacelle. The blades of the rotor are manufactured from fiberglass-reinforced epoxy, and will be approximately 253 feet to 271 feet in length. The blades are mounted, via pitch bearings, to the end of the nacelle and can be feathered 80 degrees for shutdown purposes.[6] The energy created from the rotation of the blades is relayed to the nacelle, which is the portion of the WTG that encompasses the drive train, supporting electromotive generating systems (e.g., yaw system, transmission system, and generator) that produce the wind-generated energy. The nacelle also contains the control and electrical systems of the WTG.[7] The nacelle would be mounted on a manufactured tubular conical steel tower. The steel towers will range in height between 269 feet and 328 feet and will be approximately 22 feet in diameter at the base and 15 feet in diameter at the top. A prefabricated access platform and service vessel landing (approximately 60 feet from mean low water (MLW) would be provided at the base of the tower, which will be supported by a 4-leg jacket foundation that is secured with four, through the leg foundation piles, between 42" and 54" in diameter. Control, lighting, and safety systems will be installed on each WTG as well.

Each WTG will be interconnected via a 34.5-kV submarine cable system connecting the WTGs in a serial radial inter-turbine (inter-array cable) configuration, with turbine "5" located closest to the Block Island shoreline. A separate 34.5-kV cable (termed "export cable") will connect the WTG array to the switchyard located on Block Island (the BIWF Generating Station situated in the Block Island Substation area); this cable will leave from WTG 5 and will land on the shoreline of Block Island to be interconnected with the terrestrial cables of the BIWF Generating Station switchyard.

[6] Feathering is the process of adjusting the pitch of the blade to stop the rotor when wind speeds reach or exceed the maximum rated speed for the turbine. For the Block Island WTG; the WTG will operate between a cut-in wind speed of 4.5 miles per hour (mph) and a cut-out wind speed of 67 mph.

[7] The WTGs will be equipped with a Supervisory Control and Data Acquisition (SCADA) system providing remote control and monitoring of the WTGs from an operations center onshore (i.e., Block Island Substation and Dillon's Corner Switchyard located on the Rhode Island Mainland).

3.1.2 Construction of the Offshore Wind Turbine Generator Array

Offshore construction will be completed according to the following sequence:

- Transportation of the foundations to the WTG installation site;
- Mobilization of equipment;
- Installation of the foundations;
- Installation of the cable systems; and
- Installation of the WTGs.

Details of each phase of construction are described in the following sections.

3.1.2.1 Foundation Transportation

The foundations of the WTGs, including piles, jackets, and transition decks will be fabricated in the U.S. Gulf of Mexico region, most likely in Texas or Louisiana. Once foundation components have been fabricated, the fabrication contractor will load out and tie down the structures for transportation on barges to Rhode Island. The jacket, deck, piles, and all other platform components and appurtenances will then be towed by ocean-going tugs to either the WTG installation sites (in Rhode Island Sound), where the installation vessels will be mobilized, or to one of the designated offshore support areas in Block Island Sound (See Appendix A).

3.1.2.2 Mobilization of Equipment

The WTGs and smaller secondary equipment will be transported to the staging facility on Quonset Point, Rhode Island, prior to construction. During construction, transportation barges, material barges, and other support vessels will transport the project components and equipment to the offshore construction sites. Appendix B provides a figure of the proposed vessel routes as well as a table of potential vessels that may be used for offshore construction.

3.1.2.3 Foundation Installation

Each WTG will be supported by a 50-foot x 50-foot four-leg jacket foundation that is secured to the sea floor with four, through the leg foundation piles that are between 42” and 54” in diameter. Each jacket member is joined together in a lattice structure, which sits on the seabed supporting the WTG.

Offshore installation of the jacket foundations will be carried out from 500-foot derrick barges moored to the seabed by an 8-point mooring system consisting of 10-ton anchors with a maximum penetration depth of 10 feet.[8] The derrick barge will be anchored at the location of the first foundation, most likely the most northern WTG. Prior to commencing installation activities, the seabed will be checked for debris and levelness within a 100-foot radius of the jacket installation location, and debris will be removed (e.g., via a grapnel) as necessary. Each jacket

[8] Alternatively the installation may be executed from the same jack-up vessel that will be used for the WTG installation.

will be lifted from the material barge, placed onto the seafloor, leveled, and made ready for pilings. The piles will then be inserted above sea level into each corner of the jacket in two segments. First, the lead sections of the piles will be inserted into the jacket legs and then driven into the seafloor. The second length of the piles will be placed on the lead pile section and welded into place. The foundation piles will then be driven into the seafloor to their final penetration design depth of 250 feet or until refusal, whichever occurs first. All piles will initially be driven with a 200 kilojoule (kJ) rated hydraulic hammer, followed by a 600 kJ rated hammer to reach final design penetration. Duration of pile driving is anticipated to be four days per jacket foundation (i.e., one pile per day; approximately 8 hours to install one pile), with all pile driving activities occurring only during daylight hours (i.e., starting approximately 30 minutes after dawn and ending 30 minutes prior to dusk unless a situation arises where ceasing the pile driving activity would compromise safety (both human health and environmental) and/or the integrity of the Project).

Once the pile driving is complete, the top of the piles will be welded to the jacket legs using shear plates and cut to allow for horizontal placement of the transition deck. Boat landing and transition decks will also be welded into place, and bags of sand and/or cement will be placed on the seafloor at the base of the jacket foundation to secure the inter-array cable between the J-tube exit point and subsea burial point at each WTG.[9] In total, each four-leg jacket foundation will require approximately 7 days to complete installation. Jackets foundations will be installed one at a time at each WTG location for a total of 5 weeks assuming no delays due to weather or other circumstances.

Throughout all phases of foundation installation, mats consisting of structured steel, steel plate, and/or wood beams and plates, will be attached to the jacket foundation to provide stability during pile installation. The foundation components (i.e., four circular legs, four linear braces between the legs, four triangular mud mats, and cable cement/sand bag armoring) will create a total footprint of approximately 0.07 acre on the seafloor per WTG (for a total of 0.35 acres).

3.1.2.4 Cable System Installation

Inter-Array Cable

The WTGs will be interconnected (serially; WTG 1 through 5) via a 34.5-kV submarine cable system connecting the WTGs in a radial inter-turbine configuration (Inter-Array Cable). In total, the inter-array cable system will be approximately two miles long and will be comprised of a single, three-core submarine cable that will carry 3-phase alternating current (AC) power. The cable will consist of three bundled aluminum or copper conductor cores surrounded by layers of insulating material within conducting and non-conducive metallic sheathing.[10] One or more fiber

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[9] The J-tube is a "J" shaped plastic tube that carries the power cable from each turbine to the cable trench in the seabed. As there are multiple WTGs at the BIWF, at each of foundation locations, the Inter-Array Cable from one turbine will be pulled into J-tubes located at the base of the adjacent WTG foundations.

[10] The metallic sheathing is typically comprised of a lead alloy covered by protective compound (typically polyethylene) designed to prevent direct contact between the metallic sheath and the surrounding water environment, thus effectively preventing the lead from corrosion as well as the dissolution of lead contaminants into the environment throughout operation and future abandonment.

optic cables will be included in the interstitial space between the three conductors and will be used to transmit data from each WTG or the BIWF Generation Switchyard as part of the SCADA system. The bundled cable will be approximately 6 in to 10 in (15.2 cm to 25.4 cm) in diameter.

The inter-array cable will be installed with a jet plow, which, via umbilical cords, will be connected to and operated from a dynamically positioned (DP) cable installation barge.[11] The jet plow will likely be a rubber-tired or skid-mounted plow that will be pulled along the seafloor behind the cable-laying barge with assistance of material barges.[12] High pressure water from vessel-mounted pumps will be injected into the sediments through nozzles situated along the plow, causing the sediments to temporarily fluidize and create a liquefied temporary trench approximately 5 feet wide. As the plow is pulled along the route behind the barge, the cable will be laid into the temporary, liquefied trench through the back of the plow, with the trench being backfilled by the water current and the natural settlement of the suspended material as the plow moves along. The target depth for cable burial is 6 feet below the sea floor, although actual burial depth may vary between 4 to 8 feet depending on substrate encountered along the cable installation route.[13] If less than 4 feet of burial is achieved, additional protection, such as concrete matting or rock piles, will be placed atop the buried cable. If the latter is necessary, anchored vessels will be used to install additional cable protection.

At each of the WTG foundation locations, the Inter-Array Cable will be pulled into the jacket foundation through J-tubes installed on the sides of the jacket foundation. At the submarine cable transition point at the J-tubes, additional cable armoring, such as rock piles, sandbags, and/or concrete mats, will be placed to protect the inter-array cable, especially those portions not completely buried at the junction point with the J-tube.

All equipment and materials necessary for cable installation will be loaded aboard the cable laying barge and material barges at the staging area in Quonset Point, Rhode Island. Once loaded, these vessels will leave Quonset Point, transit through the waters of Narragansett Bay and Rhode Island Sound to reach the area where the WTGs are to be installed (i.e., approximately 3 miles southeast of the Block Island shoreline). Depending on bottom conditions, weather, and other factors, installation of the inter-array cable is expected to take 2 to 4 weeks. This schedule assumes a 24-hour work window with no delays due to weather or other circumstances.

Submarine Export Cable

The submarine export cable will connect the WTG array (all 5 WTGS) to the BIWF Generation switchyard on Block Island. The export cable will consist of a 34.5 kV AC submarine cable that will originate from the northernmost WTG (i.e., WTG 5) and travel approximately 6.2 miles to a manhole on Block Island’s Crescent Beach where it will interconnect (i.e., be spliced) with the terrestrial cable that leads to the BIWF Switchyard at the Block Island Substation.

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[11] DP vessels maintain their position via thrusters instead of anchors.

[12] Two material barges are likely to be used. One barge will carry supporting equipment for the jet plow, while the other will serve to support the cable lay operations.

[13] Depth of burial is controlled by adjusting the angle of the plow relative to the bottom.

Prior to the installation of the submarine portion of export cable, the terrestrial (underground) portion of the export cable will be installed. The terrestrial portion of the export cable will run from the BIWF Generation Switchyard to Block Island's Crescent Beach, where it will eventually interconnect with the submarine portion of the export cable via a "landfall" location (i.e., manhole) that will be constructed at Crescent Beach. During landfall construction, a manhole will be established in the parking lot of Crescent Beach, and a temporarily trench (approximately 6 feet x 10 feet wide, 12 feet deep, and 60 feet long), that begins at the mean high water mark of Crescent Beach, will be excavated.[14] Via a horizontal directional drill (HDD), a cable conduit will be created that will enable to the two cables to be pulled through the conduit, anchored, and splice together.[15] Prior to HDD operations, steel sheet piling will be installed above the mean high water mark of Crescent Beach, via a vibratory hammer, to stabilize the excavated trench and support the HDD.[16] Once the sheet piles have been installed, the HDD will enter through the shore side of the excavated trench and the cable conduit will be installed between the trench and the manhole. Following the completion of HDD and cable conduit installation, the cable lay barge and its jet plow will transit to the shoreline of Crescent Beach. The end of the submarine cable will be pulled through the conduits and anchored and spliced with the terrestrial cable. Once the end of the submarine export cable has been spliced with the terrestrial cable, the jet plow will then be launch from the excavated trench on the shoreline, and installation of the submarine cables, below the seabed, will begin. The installation of the submarine portion of the export cable will use the same jet plow/DP cable installation barge technique as described above for the inter-array cable installation. The target burial depth is the same as described above for the inter-array cable. As with the installation of the inter-array cable, in those areas where the target burial depth is 4 feet or less, protective armoring (e.g., concrete matting) will be installed, via the use of anchored vessels, over the buried cable.

All equipment and materials necessary for installation of the submarine export cable will be loaded aboard the cable laying barge and material barges at Quonset Point and will transit through the waters of Narragansett Bay and Rhode Island Sound to reach the nearshore waters off Block Island's Crescent beach. Export cable lay operations will begin from this location and end at WTG 5, located approximately 3 miles southeast of Block Island. Depending on bottom conditions, weather, and other factors, installation of the submarine portion of the export is expected to take 2 to 4 weeks. This schedule assumes a 24-hour work window with no delays due to weather or other circumstances.

3.1.2.5 WTG Installation

The WTGs will be installed upon completion of the jacket foundations and the pull-in of the Inter-Array Cable. The WTGs will be transported to the offshore installation site from the storage facility at Quonset Point, Rhode Island, by jack-up material transportation barges. These transportation barges will set up at the installation site adjacent to the jack-up lift barges. The jack-up barge legs will be lowered to the seafloor to provide a level work surface and begin the

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[14] Spoils from the trench excavation will be stored on the beach and returned to the trench after the cables are installed.

[15] Deepwater Wind terms this operation the "short distance HDD landing" operation.

[16] All sheet pile installation will occur at low tide.

WTG installation. The WTGs will be installed in sections with the lower tower section lifted onto the transition deck followed by the upper tower section. The nacelle and each blade will then be lifted and connected to the tower. Pending final engineering, the tower sections and the full rotor might be pre-assembled at Quonset Point. Installation of each turbine will require 2 days to complete assuming a 24-hour work window and no delays due to weather or other circumstances. Occasional crew changes will be provided by the crew boat and/or helicopters. A derrick barge, moored to the seafloor by an 8-point mooring system consisting of 10-ton anchors, may also be used to install the WTGs.

3.2 BITS

3.2.1 BITS Overview

The BITS will serve to interconnect Block Island to the existing Narragansett Electric Company d/b/a National Grid distribution system on the Rhode Island mainland. Consisting of a single, 34.5 kV three-core cable that will carry 3-phase AC power, the BITS will originate on Block Island at the BITS Island Switchyard, located within the Block Island Substation, and terminate on the Rhode Island mainland.[17] The terrestrial portions of the cable, and facilities associated with the BITS, will be located on Block Island, at the BITS Island Switchyard within the Block Island Substation, and at the Dillon's Corner Switchyard, Narragansett, Rhode Island. The offshore portion of the BITS cable will be approximately 19.8 miles and will traverse Rhode Island State territorial waters and federal waters on the outer continental shelf (OCS) (approximately 9 miles of cable on the OCS). Installation of the BITS will begin at Narragansett, Rhode Island and end on Block Island.

3.2.2 BITS Installation

Prior to the installation of the submarine portion of the BITS cable, the terrestrial (underground) portion of the export cable will be installed. The terrestrial portion of the export cable, once constructed, will run from the Dillon's Corner Switchyard, Narragansett, Rhode Island to Scarborough Beach, Rhode Island where it will be spliced with the submarine portion of the BITS cable via a "landfall" location that will be constructed at Scarborough Beach. Deepwater Wind has proposed two alternative landfall methods: direct installation or long-distance horizontal directional drilling (HDD). At this time, a construction methodology has not been selected as pre-construction surveys have not been completed. However, in the event that pre-construction surveys indicate that landing the jet plow to the transition area between approximately MLW and MHW is not practicable, the cable will be installed via the long-distance HDD method. The following outlines the landing procedures and subsequent submarine cable laying procedures for both alternatives:

- ***Direct Installation (Deepwater Wind's preferred alternative)***

  The direct installation method is comparable to the export landing method described above in section 3.1.2.4; however, this method will involve the excavation of a trench between approximately mean low water (MLW) and the manhole in the Rhode Island

[17] The BITS cable is approximately 6 to 10 inches in diameter, and will consist of three bundled aluminum or copper conductor cores surrounded by layers of insulating material with conducting and non-conductive metallic sheathing.

Department of Environmental Management (RIDEM) parking lot, at Scarborough Beach, to install the conduit rather than using a HDD. The trench for the direct installation of the cable conduit would be a 5-foot to 8-foot wide excavated area across the berm of Scarborough Beach from MLW to Ocean Road. Excavation of the transition area trench will occur from approximately MLW to approximately MHW (similar to what has been described above for the landing of the export cable) to support connection of the submarine cable with the cable conduit (see below). The conduit trench will continue across Ocean Road and Burnside Avenue to the transition vault that will be installed in the RIDEM parking lot. Shoring will be placed in the trench to maintain the trench wall stability up to the water line during conduit installation. At the water line, metal sheeting will be utilized and installed via a vibratory hammer. Excavated sand from the conduit trench on the beach will be stored on the beach within the designated work area and returned to the trench after the conduit is installed.

Cable landfall construction will occur between October through May. Construction activities will occur over a period of approximately 6 weeks. Construction activities supporting the subsequent phase of pulling the cable through this installed infrastructure will commence upon arrival of the cable lay vessel and will occur over a period of 4 weeks. The pulling of the cable from the cable vessel will require approximately 2 days with 24 hours-per-day operation during the construction period. Other construction activities will generally occur up to 12 hours per day during daylight hours unless a situation arises where ceasing the activity would compromise safety (both human health and environmental) and/or the integrity of the installation.

- ***Long Distance HDD***

Installation of the cable conduit via the long-distance HDD method will involve a similar process as that described above for the landing of the export cable. However, unlike the landing process for the export cable, landing of the BITS cable via the long distance HDD method will require the installation of a 20 foot by 50 foot cofferdam, approximately 300 to 1,800 feet offshore of Scarborough Beach. The cofferdam will consist of steel sheet piles installed with a vibratory hammer. Installation will take approximately 2 days, with pile driving occurring for no more than 12 hours per day. Once cofferdam installation is complete, the area inside the cofferdam will be excavated in preparation for landing the cable. However, prior to excavation operations, a temporary silt curtain at a 50-foot radius around the cofferdam. The sheet pile cofferdam will remain in place for a period of 6 months, after which, the steel sheet piles will be removed, via a vibratory hammer, over a period of two days, with no more than 12 hours of pile driving operations to occur per day.

Installation of the conduit and manhole will occur over approximately 16 weeks with 24 hour per day HDD operation and up to 12 hours per day for the supporting construction activities. Construction activities supporting the cable pulling will occur subsequently over a period of 4 weeks.

After the conduit is installed, the cable lay barge, with its jet plow and the submarine portion of the BITS cable on board, will transit to the shoreline of Scarborough Beach. The end of the submarine cable will be brought ashore and pulled through the installed conduit to the transition vault in the RIDEM parking lot and spliced with the terrestrial cable. Once the end of the submarine BITS cable has been spliced with the terrestrial cable, the jet plow will then be launch from the excavated trench on the shoreline, and installation of the submarine cables, below the seabed, will begin. To accomplish the necessary burial, the jet plow will be positioned over the trench at the MLW mark and be pulled from shore by the cable installation vessel. The installation of the submarine portion of the BITS cable will use the same jet plow/DP cable installation barge technique as described above for the inter-array cable installation (see section 3.1.2.4). Target Burial depth is 6 feet below the sea floor; however, in those areas where the target burial depth is 4 feet or less, protective armoring will be installed, via the use of anchored vessels, over the buried cable. Additionally, where the BITS crosses two existing submarine cables on the OCS, the cable will be installed directly on the sea floor and will be protected from external aggression using a combination of sand bags and concrete mattresses. Anchored vessels will be used to install both the BITS and the associated cable armoring at these locations. Where the BITS cable crosses inactive cables, it is anticipated that the cables will be cut and cleared from the cable corridor during pre-lay grapnel runs.

Once the BITS submarine cable has been installed and reaches Crescent Beach (Block Island), landfall operations will need to occur to splice the submarine cable to the terrestrial BITS cable (installed previously). The terrestrial portion of the export cable will run from the BITS Island Switchyard to Block Island's Crescent Beach, where it will interconnect with the submarine portion of the BITS cable via a "landfall" location (i.e., manhole) that will be constructed at Crescent Beach adjacent to the landfall location for the BIWF export cable. During landfall construction a manhole will be established in the parking lot of Crescent Beach, and a temporarily trench (approximately 6 feet x 10 feet wide, 12 feet deep, and 60 feet long) that begins at the mean high water mark of Crescent Beach, will be excavated.[18] Via HDD, a cable conduit will be created that will enable the two cables to be pulled through the conduit, anchored, and splice together. Prior to HDD operations, steel sheet piling will be installed, via a vibratory hammer, to stabilize the excavated trench and support the HDD.[19] Once the sheet piles have been installed, the HDD will enter through the shore side of the excavated trench and the cable conduit will be installed between the trench and the manhole. Following the completion of HDD and cable conduit installation, the cable-lay barge and its jet plow will transit to the shoreline of Crescent Beach. The end of the submarine cable will be pulled through the conduits and anchored and be spliced with the terrestrial cable.

All equipment and materials necessary for installation of the submarine BITS cable will be loaded aboard the cable laying barge and material barges at the staging area in Quonset Point, Rhode Island. Once loaded, these vessels will leave Quonset Point, transit through the waters of Narragansett Bay and Rhode Island Sound to reach the nearshore waters off Block Island's Crescent Beach. BITS cable lay operations will begin from this location and end at the landfall

[18] Spoils from the trench excavation will be stored on the beach and returned to the trench after the cables are installed.

[19] All sheet pile installation will occur at low tide.

location on Scarborough Beach, Rhode Island. Depending on bottom conditions, weather, and other factors, installation of the submarine portion of the BITS cable is expected to take 4 to 6 weeks. This schedule assumes a 24-hour work window with no delays due to weather or other circumstances.

3.3 Commissioning and Post Construction Activities

Once all WTGs for the Project have been installed, Deepwater Wind will commence commissioning of the facility. This will entail testing the WTGs' and transmission system's capabilities to meet standards for safety and grid interconnection reliability. Technicians will travel to the turbines daily during the initial operating period following construction. Technicians will be transported to and from the WTGs via a dedicated crew workboat from Quonset Point or Port Judith, Rhode Island.

After the BITS submarine cable has been installed, but before connections to the terrestrial cables are completed, Deepwater Wind will perform a conductor continuity test and a voltage test. Once connections to the terrestrial cables are complete, additional commissioning tests will be performed, including a second continuity test and an AC voltage test. An optical time domain reflectometer (OTDR) will be used to verify the continuity of the fiber optic cable and that its terminations are in good working order. These testing and commissioning activities may be performed while the cable is energized.

Deepwater Wind will also conduct a post-construction inspection using multi-beam sonar and shallow sub-bottom profiler (chirp) to ensure cable burial depth was achieved to verify reconstitution of the trench. The sub-bottom profiler and the multi-beam survey will be located on one vessel and surveys will be conducted along the extent of each cable route. Surveys of the cable routes will not be done simultaneously. It is expected to take approximately two weeks to complete the post-construction inspection of the BITS, export, and inter-array cables. Based upon this post-construction inspection, Deepwater Wind will determine the need and frequency of additional inspections, via multi-beam sonar and/or a sub-bottom profiler, during the Operation and Maintenance phase to ensure the minimum safe burial depth is maintained.

3.4 Operations and Maintenance

Once construction of the BIWF and BITS is complete, the operational life of these structures will be 25 years. The following describes the operation and maintenance of the BIWF and BITS.

3.4.1 BIWF

Deepwater Wind Block Island, LLC will be responsible for operation of the BIWF. Prior to the commencement of operations, a facility-specific environmental compliance manual will be prepared for the BIWF. The manual will outline specific operating obligations and aid the staff regarding day-to-day regulatory and permit requirements.

3.4.1.1 Wind Turbine Generators and Foundations

The WTG will be maintained in accordance with a dedicated maintenance plan. It is anticipated that each WTG will require approximately three to five days of planned maintenance per year.

The timing of this maintenance will be coordinated with The Narragansett Electric Company in advance of execution.

For the foundation, an annual inspection program will be developed to ensure all nodes of the foundations are inspected within a 5-year time frame. Underwater inspection will include visuals and eddy currents tests with divers and/or remotely operated vehicles (ROVs). Any damage or cracks will be analyzed immediately and repaired accordingly.

3.4.1.2 Inter-Array and Export Cables

The Inter-Array cable and submarine and underground portions of Export Cable have no maintenance needs unless a fault or failure occurs. Cable failures are only anticipated as a result of damage from outside influences, such as boat anchors. The armoring of the Inter-Array Cable at the J-tubes, the target burial depth of six feet for the remainder of the offshore cable and burial depth onshore of up to seven feet are designed to ensure that damage would be an unlikely occurrence.

The cable burial depth along the route will be inspected using a sub-bottom profiler at least once every five years. The cable burial depth might be inspected more frequently based on the post-lay data. Operations-phase reporting for the submarine transmission cable will be implemented, as necessary, in accordance with the requirements specified in operating permits. If, following inspections, target cable burial depth has not been achieved along certain sections of the cable route, protective armoring (e.g., concrete matting, rock piles) will be placed along those sections of the cable route.

Both the overhead and underground sections of the terrestrial Export Cable will be maintained consistent with the Narragansett Electric Company d/b/a National Grid (TNEC) standards and will consist of periodic inspections and tree trimming in the vicinity of the overhead line right-of-way to prevent damage/interference from overgrown vegetation. The overhead poles, cross arms, insulators, and conductors will also be visually inspected on a regular basis and any damage will be noted and fixed as necessary per industry standards. If the overhead lines are damaged by a severe event (e.g., a storm) they will be repaired in accordance with TNEC procedures. If necessary, the WTGs will be shut down during the repair.

The standard industry life expectancy of the Inter-Array and Export Cables is 50 years; however, the equipment will be scheduled for decommissioning in advance of this timeframe (i.e., after 25 years) or replacement/upgrade in accordance with this standard.

3.4.2 BITS

Deepwater Wind Block Island Transmission, LLC will initially be responsible for operation of the BITS. It is anticipated that TNEC will purchase the BITS and will become responsible for all operations and maintenance of the BITS.

The operation and maintenance of the BITS transmission cable will be as described for the BIWF Export Cable in Section 3.4.1.2.

3.5 Decommissioning

After 25 years of operation, the BIWF and BITS will be decommissioned. Decommissioning of the BITS and BIWF is expected to take two years to complete (i.e., occur in 2041 and 2042). The activities associated with the decommissioning of these structures are described below.

3.5.1 BIWF

Decommissioning will follow the same relative sequence as construction, but will occur in reverse. The WTG components will be removed by a jack-up lift vessel or a derrick barge and lifted onto a material barge. The material barge will transport the components to a recycling yard where the components will be disassembled and prepared for re-use and/or recycling for scrap steel and other materials. The foundations will be cut by an internal abrasive water jet cutting tool at approximately 3 meters below the seabed. The balance of the foundations will be removed using 500-ton derrick barges and lifted onto material barges. The submarine cables will be abandoned in place. The substations associated with the BIWF will be disconnected, dismantled, and recycled in accordance with applicable permits and regulations.

3.5.2 BITS

Deepwater Wind proposes to allow the BITS submarine cable to remain in place at decommissioning. Abandoning decommissioned submarine cables in place is standard industry practice.

3.6 Construction, Operation and Maintenance Facilities

3.6.1 Quonset Point Port Facility

Quonset Point is small peninsula that juts out into Narragansett Bay in the Town of North Kingston, Rhode Island. Deepwater Wind has executed a land lease option, under which it has secured the rights to parcels at the Quonset Point port facility, specifically at the Port of Davisville, which provides 4,500 linear feet of berthing space consisting of two piers, a bulkhead, on-dock rail, and a 14-acre laydown area. Deepwater Wind will use existing piers for offloading, staging, pre-assembly, and load-out for the WTGs and some other smaller components of the BIWF and the BITS. Deepwater Wind does not anticipate that improvements or land-disturbing activities will be necessary to utilize the site for construction and staging of the Project.

3.6.2 Operation and Maintenance Facility

Deepwater Wind expects to locate an Operation and Maintenance (O&M) facility, including a shore operations center and a control room on an existing waterfront parcel in Point Judith, Rhode Island. The facility will be a combination of office, maintenance shop, and a small dockside facility. These facilities will house the Project’s administrative support offices, the warehouse facility and maintenance shop for all offshore generating units, and a marine terminal for the Project’s offshore support and logistics vessels.

The O&M facility and switchgear buildings located within the newly proposed Block Island Substation, on the Block Island mainland, and the Dillon’s Corner Switchyard, on the Rhode

Island mainland, will contain remotely operated SCADA control systems for use during operation of the Project.

3.7 BITS/BIWF Project Timeline

The timeline for construction and commissioning of the BITS and BIWF is as follows:

Activity [a/]	Anticipated Schedule
BITS & BIWF: Contracting, mobilization, and verification	January 2014-December 2014
BIWF &BITS: Onshore short-distance (direct) HDD cable installation/landing [b/]	December 2014 –June 2015
BITS: Onshore/offshore long-distance HDD cable installation/landing [c/]	January 2015-June 2015
BITS &BIWF: Onshore cable installation	October 2015-May 2015
BITS & BIWF: Landfall demobilization and remediation	May 2015-June 2015
BITS & BIWF: Offshore cable installation	April 2015-August 2015
BIWF: Foundation fabrication and transportation	October 2015-September 2015
BIWF: WTG jacket foundation-non-pile driving activity	Last week of April 2015-July 2015; or August 2015-October2015
BIWF:WTG jacket foundation-pile driving activity	May 2015-July 2015; or August 2015-October 2015
BIWF:WTG installation and commissioning	July 2015-November 2015

[a/] All project activities will be done sequentially and will not occur, in general, concurrently. That is, BITS cable installation will occur first. For landing operations at Scarborough Beach, should long distance HDD landing methods be used, cofferdam installation will occur prior to DP cable lay vessel movements. Once installed, DP cable lay vessels will transit to the landing area; submarine cable installation will proceed, followed by landing at Block Island. Once the BITS is installed, export cable installation will begin after several WTG foundations have been installed, specifically the WTG that will directly interconnect with the export cable, which will eventually end with the final installation of the last WTG and inter-array cable installation.

[b/] For the BITS: the short distance (direct) HDD installation is the cable landfall method that will used for landing the BITS cable on Block Island; however, although it is the preferred cable landfall method on Scarborough Beach, Rhode Island, it may not be used for landing the cable in this region.

[c/] For the BITS: the long-distance HDD method is the alternative method for cable landing at Scarborough Beach. This alternative will not be used for landing the BITS on Block Island.

Following the construction and commissioning of the BITS and BIWF (i.e., beginning in 2015), the operational life of the BIWF and BITS will be 25 years (i.e., through 2040). Following the operational life of these structures, the BITS and BIWF will be decommissioned over a two-year period (i.e., 2041-2042).

3.8 Mitigation Measures

The USACE and Deepwater Wind have agreed to implement the following mitigation measures to reduce the exposure of ESA-listed species (see section 4.0 for species information) to elevated levels of underwater noise and minimize the potential for vessel collisions during the construction of the BWIF and BITS.

3.8.1. BIWF and BITS Underwater Noise Mitigation

3.8.1.1 Exclusion and Monitoring Zones

Exclusion and monitoring zones will be established around acoustically active project components (i.e., pile driving (vibratory and impact) and DP thruster use for cable lay operations). These zones will be established to monitor for ESA listed species of sea turtles and whales that may enter the project area and to adjust project operations accordingly to prevent exposure of these animals to potentially injurious levels of underwater noise. Exclusion and monitoring zones are not being established for Atlantic sturgeon because this species occurs only under the water surface and visual observers will not be able to detect the presence of Atlantic sturgeon in the project area and no remote sensing technology that could detect Atlantic sturgeon is feasible for deployment in the area.

An exclusion zone will be established based on the estimated distances to the underwater noise levels believed to result in injury to marine mammals(i.e., 180 dB re 1 μPa RMS (180 dB_{RMS}); NMFS 1995; Southall *et al*. 2007).[20] A monitoring zone, extending further from the sound source than the exclusion zone, will be established based on the estimated distance to the underwater noise level believed to result in behavioral disturbance (i.e., 160 dB re 1 μPa RMS (160 dB_{RMS}; impulsive noise) or 120 dB re 1 μPa RMS (120 dB_{RMS}; non-impulsive); Malme *et al.* 1983, 1984; Richardson *et al*. 1990,1995,1986; Southall *et al*. 2007; NMFS 1995; Tyack 1998).

Noise analysis performed by TetraTech for Deepwater Wind has indicated that both vibratory pile driving and DP vessel thruster use will produce sound levels of 180 dB_{RMS} extending no further than 1 meter (m) from the source (TetraTech 2013 a,b). For DP vessel thruster use and vibratory pile driving, Deepwater Wind will establish a monitoring zone equivalent to the size of the predicted 160 dB_{RMS} isopleth, not the 120 dB_{RMS} isopleth. This is because the distance to the 120 dB_{RMS} isopleth will result in zones too large to effectively monitor (i.e., 89.9 km for vibratory pile driving operations; 4.75 km for DP vessels).

Exclusion and/or monitoring zones established for impact pile driving, DP vessel thruster use and vibratory pile driving activities are as follows:

- ***Impact Pile Driving of WTG Foundations***- Prior to the onset of pile driving, when the 200 kJ impact hammer is in use, an initial 200-meter radius exclusion zone will be established around each jacket foundation. In addition, an initial monitoring zone extending 3.6 kms (radius) from the pile will be monitored for each pile during impact

[20] The exclusion and monitoring zones that will be established are applicable to sea turtles as well. Sea turtle underwater acoustic injury and behavioral thresholds are believed to occur at 207 dB_{RMS} and 166 dB_{RMS}, respectively. As the marine mammal injury and behavioral disturbance thresholds encompass the sea turtle thresholds, the exclusion and monitoring zones to be established by Deepwater Wind will also be inclusive of the sea turtle injury and behavioral disturbance thresholds and therefore, protective of these species. For the definition of "RMS," see Section 7.1.3.

pile driving activities utilizing the 200 kJ impact pile driving hammer. During the final phases of pile installation, when a 600 kJ impact hammer will be used, the exclusion zone will be expanded to the maximum radial distance of approximately 600 meters. The monitoring zone will be expanded to the maximum radial distance of approximately 7 km. These distances are expected to equate to where 180 dB_{RMS} and 160 dB_{RMS} isopleth extend. Deepwater Wind will follow ramp up and shut down procedures in accordance with these monitoring zones (see below for further details).

- ***Vibratory Pile Driving of Cofferdam*** – Cofferdam installation and removal will produce sound levels of 180 dB_{RMS} within 10 m from the source (TetraTech 2013b) and thus, an exclusion zone will not be established. A 200 meter radius monitoring zone, based on TetraTech's modeled distance to the 160 dB_{RMS} isopleth, will be monitored during all vibratory pile driving activities. All marine mammal sightings, including those beyond the 160 dB_{RMS} isopleth, will be recorded.

- ***DP Vessel during Cable Installation*** – DP vessel use during cable installation will not produce sound levels at 180 dB_{RMS} beyond 1 m from the source (TetraTech 2013a,b) and thus, an exclusion zone will not be established. A monitoring zone, based on the extent to the 160 dBRMS isopleth, will be established around the DP vessel. The monitoring zone will extend an estimated 21 m from the source (i.e., DP vessel).[21] All marine mammal sightings, including those beyond the 160 dB_{RMS} isopleth will be recorded.

Field verification of both the monitoring and exclusion zones will be conducted to determine whether the proposed preliminary zones are adequate to encompass the 180 and 160 dB_{RMS} isopleths. Field verification of these zones will be conducted as follows for activities involving pile driving or DP thruster:

- ***Impact Pile Driving of WTG Foundations*** – Field verification of the initial 200 meter radius exclusion zone and the 3.6 km radius monitoring zone for the 200kJ impact pile driving hammer as well as the 600 meter radius exclusion zone and 7 km radius monitoring zone for 600 kJ impact pile driving hammer will be conducted. Acoustic measurements will include the driving of the last half (deepest pile segment) for any given open-water pile and will include measurements from two reference locations at two water depths (a depth at mid-water and a depth at approximately 1 meter above the seafloor). If the field measurements determine that the actual 180 dB_{RMS} and 160 dB_{RMS} ZOIs are less than or extend beyond the proposed exclusion zone and monitoring zone radii, a new zone(s) will be established accordingly. The USACE and NMFS will be notified within 24 hours whenever any new exclusion and/or monitoring zone are established by Deepwater Wind that extends beyond the initially proposed radii.

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[21] NMFS estimated the extent to the 160 dB_{RMS} isopleth. NMFS estimated using the Practical Spreading Loss Model; $R2=R1*10^{((\text{measured or calculated sound level-Noise Threshold})/15)}$ (Bastasch *et al.* 2008; Stadler and Woodbury 2009), where: R2= the distance (in meters) to the threshold; R1=distance of the measured or calculated sound level. For our calculations, R1=the source level for DP thruster use (i.e., 180 dB_{RMS}); Sound level (i.e., RMS, cSEL, peak)= noise level measured or calculated at distance R1; and Noise Threshold= depending on species of interest, NMFS thresholds for potential injury or behavioral response.

Implementation of the revised zone(s) smaller than the proposed radii will be contingent upon USACE and NMFS review and approval. In the event that a smaller zone(s) is determined to be appropriate, Deepwater Wind will continue to use the originally proposed zone(s) until agency approval is given.

- ***Vibratory Pile Driving of Cofferdams*** – Should the long-distance HDD landing option be selected, field verification of the preliminary 200 meter radius monitoring zone (i.e., confirmation that 200 meters =160 dB_{RMS}), and any modification to the zone, will be performed as described for impact pile driving.

- ***DP Vessel during Cable Installation*** – Field verification of the preliminary 21 meter radius monitoring zone (i.e., that the 160 dB_{RMS} isopleth does not extend beyond 21 meters) associated with DP vessel thruster use during cable installation will be performed using acoustic measurements from two reference locations at two water depths (a depth at mid-water and a depth at approximately 1 meter above the seafloor). As necessary, the monitoring zone will be modified and implemented as described for impact and vibratory pile driving).

3.8.1.2 Protected Species Observers

All observations for whales and sea turtles in the exclusion and monitoring zones will be performed by NMFS approved protected species observers (PSO). Observer qualifications will include direct field experience on a marine mammal/sea turtle observation vessel and/or aerial surveys in the Atlantic Ocean/Gulf of Mexico. It is anticipated a minimum of two PSOs will be stationed aboard each noise producing construction support vessel (e.g., derrick barge, jack-up barge, and cable lay vessel). Given the small size of the exclusion zones, the observers will be able to fully monitor the area and detect any marine mammals or sea turtles in the area and therefore ensure no work occurs while they are present in the exclusion zone. Observers in the monitoring zone are not likely to be able to detect every marine mammal or sea turtle that may be present given the larger size of these zones. To increase the potential for detection, given the distance of the monitoring zone associated with the impact pile driving, at least two additional PSOs will be stationed aboard an observation vessel dedicated to patrolling the monitoring zone while continuously searching for the presence of ESA listed species (i.e., whales and sea turtles; in the offshore marine environment, visual surface detection of Atlantic sturgeon is not feasible). As an alternative to a dedicated observation vessel, Deepwater Wind is also considering the use of aerial based observations of the established monitoring zone for impact pile driving during construction activities. Each PSO will monitor 360 degrees of the field of vision. Each PSO will follow the specified monitoring period for each of the following construction activities:

- ***Impact Pile Driving of WTG Foundations*** – The PSOs will begin observation of the monitoring zone for at least 30 minutes prior to the soft start of impact pile driving (see below for further details). Use of pile driving equipment will not begin until the associated exclusion zone is clear of all ESA listed whales and sea turtles for at least 30 minutes. Initial monitoring of the exclusion and monitoring zones prior to soft start will be conducted with the assistance of night vision equipment to account for dark conditions at or just prior to dawn. In addition, soft start of construction equipment, as described below, will not be initiated if the monitoring zone cannot be adequately monitored (i.e., obscured by fog, inclement weather) for a 30-minute period. If a soft start has been initiated before the onset of inclement weather, activities may continue through these periods if deemed necessary to ensure the safety and integrity of the Project. Observation of both the exclusion zones and the monitoring zones will continue throughout the construction activity and will end approximately 30 minutes after use of noise-producing equipment stops operation.

- ***DP Vessel during Cable Installation*** – PSOs stationed on the DP vessel will begin observation of the monitoring zone as the vessel initially leaves the dock. Observations of the monitoring zone will continue throughout the construction activity and will end after the DP vessel has returned to dock.

- ***Vibratory Pile Driving of Cofferdam*** – The PSOs will begin observation of the monitoring zone at least 30 minutes prior to vibratory pile driving. Use of noise-producing equipment will not begin until the associated monitoring zone is clear of all marine mammals and sea turtles for at least 30 minutes. In addition, soft start of construction equipment, as described below, will not be initiated if the monitoring zone cannot be adequately monitored (i.e., obscured by fog, inclement weather, poor lighting conditions) for a 30-minute period. If a soft start has been initiated before the onset of inclement weather, activities may continue through these periods if deemed necessary to ensure the safety and integrity of the Project. Observation of both the exclusion zones and the monitoring zones will continue throughout the construction activity and will end approximately 30 minutes after use of noise-producing equipment is completed.

For each of the three construction activities (impact pile driving, vibratory pile driving, DP thruster use during cable installation) PSOs, using binoculars, will estimate distances to whales and sea turtles either visually, using laser range finders, or by using reticle binoculars during daylight hours. It is important to note that all pile driving activity will occur only during daylight hours. As cable-laying activities will operate 24 hours a day, during night operations, night-vision binoculars will be used. If higher vantage points (greater than 25 feet) are available, distances can be measured using inclinometers. Position data will be recorded using hand-held or vessel global positioning system (GPS) units for each sighting, vessel position change, and any environmental change.

For monitoring established exclusion and monitoring zones, each PSO stationed on or in proximity to the noise-producing vessel or location will scan the surrounding area for visual indication of whale and sea turtle presence that may enter the zones. Observations will take place from the highest available vantage point on the associated operational platform (e.g., support vessel, barge or tug; estimated to be over 20 or more feet above the waterline). General 360-degree scanning will occur during the monitoring periods, and target scanning by the PSO will occur when alerted of the presence of a whale or sea turtle.

Data on all observations will be recorded based on standard PSO collection requirements. This will include dates and locations of construction operations; time of observation, location and weather; details of whale and sea turtle sightings (e.g., species, age classification [if known], numbers, behavior); and details of any observed behavioral disturbances or injury/mortality. In addition, prior to initiation of construction work, all crew members on barges, tugs and support vessels, will undergo environmental training, a component of which will focus on the procedures for sighting and protection of whales and sea turtles. A briefing will also be conducted between the construction supervisors and crews, the PSOs, and DWBI. The purpose of the briefing will be to establish responsibilities of each party, define the chains of command, discuss communication procedures, provide an overview of monitoring purposes, and review operational procedures. The Deepwater Wind Construction Compliance Manager (or other authorized individual) will have the authority to stop or delay impact pile driving activities, if deemed necessary. New personnel will be briefed as they join the work in progress.

3.8.1.3 Ramp-up/Soft Start Procedures

A ramp-up (also known as a soft-start) will be used for noise producing construction equipment capable of adjusting energy levels (i.e., pile driving operations).[22] The ramp-up procedure for noise-producing equipment utilized during impact pile driving of the WTG foundations and the vibratory pile driving of cofferdams is described below:

- ***Impact Pile Driving of the WTG Foundations***: The ramp-up procedure for noise-producing equipment utilized during impact pile driving of the WTG foundations will not be initiated if the monitoring zone cannot be adequately monitored (i.e., obscured by fog, inclement weather, poor lighting conditions) for a 30-minute period. If a soft start has been initiated before the onset of inclement weather, activities may continue through these periods if deemed necessary to ensure the safety and integrity of the Project. A ramp-up will be used at the beginning of each pile segment during impact pile driving in order to provide additional protection to Atlantic sturgeon, whales and sea turtles near the Project Area by allowing them to vacate the area prior to the commencement of pile-driving activities. The ramp-up procedures require 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. The procedure will be repeated two additional times. If whales or sea turtles are sighted within the impact pile driving monitoring zone prior to or during the soft-start, activities will be delayed until the animal(s) has moved outside the monitoring zone and no whales or sea turtles are sighted for a period of 30 minutes.

- ***Vibratory Pile Driving of Cofferdam*** – The ramp-up procedure will not be initiated if the monitoring zone cannot be adequately monitored (i.e., obscured by fog, inclement weather, poor lighting conditions) for a 30-minute period. A ramp-up or soft-start will be used at the beginning of each pile segment during vibratory pile driving in order to provide additional protection to marine mammals and sea turtles near the Project Area by allowing them to vacate the area prior to the commencement of vibratory pile-driving activities. The ramp-up requires an initial set of 3 strikes from the vibratory hammer at 40 percent energy with a one-minute waiting period between subsequent three-strike sets. The procedure will be repeated two additional times. If marine mammals or sea turtles are sighted within the vibratory pile driving monitoring zone prior to or during the soft-

[22] The DP vessel thrusters will be engaged from the time the vessel leaves the dock. Therefore, there is no opportunity to engage in a ramp up procedure.

start, activities will be delayed until the animal(s) has moved outside the monitoring zone and no marine mammals or sea turtles are sighted for a period of 30 minutes.

3.8.1.4 Shut-Down Procedures

The monitoring zone around the noise-producing activities (impact pile driving, vibratory pile driving, and DP thruster use during cable installation) will be monitored, as previously described, by PSOs for the presence of whales and sea turtles before, during and after any noise-producing activity. PSOs will work in coordination with Deepwater Wind's Construction Compliance Manager (or other authorized individual) to stop or delay any construction activity, if deemed necessary. The following outlines the shut-down procedures:

- ***Impact Pile Driving of WTG Foundations*** – For impact pile driving, from an engineering standpoint, any significant stoppage of driving progress will allow time for displaced sediments along the piling surface areas to consolidate and bind. Attempts to restart the driving of a stopped piling may be unsuccessful and create a situation where a piling is permanently bound in a partially driven position. In the event that a whale or sea turtle is observed within or approaching the monitoring zone during impact pile driving, PSOs will immediately report the sighting to the on-site Construction Compliance Manager (or other authorized individual). Upon this notification, Deepwater Wind proposes that the hammer energy will be reduced by 50 percent to a "ramp-up" level. This reduction in hammer energy will effectively reduce the potential for exposure of whales, sea turtles, and Atlantic sturgeon to sound energy, proportional to the reduction in force; however, established exclusion and monitoring zones will remain constant for monitoring purposes. By maintaining impact pile driving at a reduced energy level, momentum in piling penetration can be maintained minimizing risk to both Project integrity and to whales, Atlantic sturgeon, and sea turtles.

  After decreasing impact pile driving energy, PSOs will continue to monitor whale and/or sea turtle behavior and determine if the animal(s) is moving towards or away from the exclusion zone. If the animal(s) continues to move towards the sound source then impact piling operations will be halted prior to the animal entering the exclusion zone. Ramp-up procedures for impact pile driving may be initiated when PSOs report that the monitoring zone has remained clear of whales and/or sea turtles for a minimum of 30 minutes since the last sighting.

- ***DP Vessel during Cable Installation*** – During cable installation a constant tension must be maintained to ensure the integrity of the cable. Any significant stoppage in vessel maneuverability during jet plow activities has the potential to result in significant damage to the cable. Therefore, during DP vessel operations if whales or sea turtles enter or approach the established exclusion zone, Deepwater Wind proposes to reduce DP thruster to the maximum extent possible, except under circumstances when ceasing DP thruster use would compromise safety (both human health and environmental) and/or the integrity of the Project. As with reduced hammer force for pile driving operations, reducing thruster energy will effectively reduce the potential for exposure of whales and sea turtles to sound energy. Normal use may resume when PSOs report that the monitoring zone has remained clear of whales and/or sea turtles for a minimum of 30 minutes since last the sighting.

- ***Vibratory Pile Driving of Cofferdams*** – Cofferdam construction will produce sound levels of 180 dB_{RMS} extending no further than 10 m from the source (TetraTech 2013b); therefore, no exclusion zone for this activity has been established. However, if ESA listed species are observed entering or approaching the 200 m radius monitoring zone for vibratory pile driving, DWBI proposes to halt vibratory pile driving as a precautionary measure to minimize noise impact on the animal(s). Ramp-up procedures for vibratory pile driving may be initiated when PSOs report that the monitoring zone has remained clear of marine mammals and/or sea turtles for a minimum of 30 minutes since the last sighting.

3.8.1.5 Time of Day Restrictions

Impact pile driving for jacket foundation installation and vibratory pile driving cofferdams will occur during daylight hours starting approximately 30 minutes after dawn and ending 30 minutes prior to dusk unless a situation arises where ceasing the pile driving activity would compromise safety (both human health and environmental) and/or the integrity of the Project. If a soft-start has been initiated prior to the onset of inclement weather (e.g., fog, severe rain events), the pile driving of that segment may be completed. No new pile driving activities will be initiated until 30 minutes after dawn or after the inclement weather has passed. Cable installation will be conducted 24 hours per day. Night vision equipment will be used by PSOs to monitor the DP thruster monitoring zone.

3.8.1.6 Reporting

Deepwater Wind will provide the following reports during construction activities:

- Deepwater Wind will contact the USACE and NMFS at least 24 hours prior to the commencement of construction activities and again within 24 hours of the completion of the activity.

- Deepwater Wind will contact the USACE and NMFS within 24 hours of establishing any exclusion and/or monitoring zone. Within 7 days of establishing exclusion and/or monitoring zones, Deepwater Wind will provide a report to the USACE and NMFS detailing the field-verification measurements. This report will include the following information: a detailed account of the levels, durations, and spectral characteristics of the impact and vibratory pile driving sounds, DP thruster use, and the peak, RMS, and energy levels of the sound pulses and their durations as a function of distance, water depth, and tidal cycle.

- Deepwater Wind most notify USACE and NMFS within 24 hours of receiving any field monitoring results which indicate that any exclusion or monitoring zones should be modified (i.e., due to in-field sound monitoring suggesting that model results were too big or too small). No changes will be made to the exclusion or monitoring zones without written (e-mail) approval from NMFS and USACE.

- Any observed behavioral reactions (e.g., animals departing the area) or injury or mortality to any marine mammals, Atlantic sturgeon, or sea turtles must be reported to USACE and NMFS within 24 hours of observation. If any sturgeon are observed, these instances will also be reported to USACE and NMFS (incidental.take@noaa.gov) within 24 hours.

- A final technical report will be provided to USACE and NMFS within 120 days after completion of the construction activities. This report must provide full documentation of methods and monitoring protocols (including verification of the sound levels actually produced within the exclusion and monitoring zones), summarizes the data recorded during monitoring, and comparing these values to the estimates of listed marine mammals and sea turtles that were expected to be exposed to disturbing levels of noise during construction activities, and provides an interpretation of the results and effectiveness of all monitoring tasks.

3.8.2 Strike Avoidance

All vessels associated with the construction, operation, maintenance and repair, and decommissioning of the BITS and BIWF will adhere to NMFS guidelines for marine mammal ship strike avoidance (see (http://www.nmfs.noaa.gov/pr/pdfs/education/viewing_northeast.pdf **),** including maintaining a distance of at least 500 yards from right whales, at least 100 feet from all other whales, and having dedicated lookouts and/or protected species observers posted on all vessels who will communicate with the captain to ensure that all measures to avoid whales are taken.[23]

3.8.3 Geophysical Surveys Mitigation and Monitoring

Deepwater Wind will use the following measures during all geophysical surveys (i.e., multi-beam sonar and sub-bottom profiler (chirp)) (TetraTech 2011):

- **Implementation of Ramp-Up:** At the start of each survey day, instruments which have the capability of running at variable power levels and operate at a frequency detectable by ESA listed species will initially be operated at low levels, then gradually increased to minimum necessary power requirements for quality data collection. This allows any listed species capable of detecting this noise to depart the area before full power surveying commences. Surveys will not commence (i.e., ramp up) when the exclusion zone cannot be effectively monitored.

[23] PSOs will be placed on vessels with noise producing equipment and (e.g., vessels with the pile driver and the DP vessels) vessels assigned to actively observe the project's established exclusion and monitoring zones through construction. Other vessels will have a dedicated lookout to watch for whales and sea turtles and to communicate with the captain.

- **Establishment of Exclusion Zone:** Whenever multi-beam sonar or the chirp is in use, a 300-meter radius exclusion zone (from the source) will be established around the operating vessel or the towed survey device. The sounds produced by this equipment cannot be perceived by sea turtles or Atlantic sturgeon because the frequency is too high. Therefore, the exclusion zone will be maintained for listed whales. For example, if a sound source is towed 30 meters behind the survey vessel, the monitored area from the vessel will be out to 330 meters (or 300 meters from the source). The 300-meter exclusion zone encompasses the 160 dB_{RMS} isopleth, which for either geophysical survey device, is expected to occur within 150 meters or less from the operating device.

- **Visual Monitoring of the Exclusion Zones:** The exclusion zone will be monitored by a trained Environmental Compliance Monitor.[24] The Environmental Compliance Monitor will keep vigilant watch for the presence of marine mammals within the exclusion zone. The exclusion zone will be monitored for 30 minutes prior to the ramp up of sound sources. If the exclusion zone is obscured by fog or poor lighting conditions, surveying utilizing noise producing equipment will not be initiated until the entire exclusion zone is visible for the 30 minute period. If marine mammals are observed within the 300 meter safety exclusion zones during 30 minute period and before the ramp up begins, surveying utilizing noise producing equipment will be delayed until they move out of the area.

All sightings of NMFS listed species will be recorded on an established NMFS-approved log sheet by the Environmental Compliance Monitor. The following data will be recorded:

- Dates and location of operations
- Weather and sea-state conditions;
- Time of observation;
- Approximate location (latitude and longitude) at the time of the sighting;
- Details of sighting (species, numbers, behavior);
- General direction and distance of sighting from the vessel;
- Activity of the vessels at the time of sighting; and
- Action taken by the Environmental Compliance Monitor.

All observation data will be provided to NMFS within 60 days of the completion of surveys. In addition, during all survey operations, Deepwater Wind will report all sightings of ESA listed species, regardless of condition, to NMFS (incidental.take@noaa.gov) within 24 hours of the observation and record as much information as possible (e.g., species, size, decomposition state, obvious injuries etc.)

- **Shut-Down:** If a listed whale is spotted within or transiting towards the exclusion zone when equipment is operating that can be heard by that individual (i.e., the chirp) , an

[24] The Environmental Compliance monitor assigned to the survey vessel, as well as all individuals on board the survey vessel responsible for navigation duties will receive training on marine mammal and sea turtle sighting and reporting and vessel strike avoidance measures. The training course will be provided by TetraTech and will be modeled after a NMFS approved marine mammal and sea turtle training program. The training will include details on the Federal laws and regulations for protected species (ship strike information, migratory routes, and seasonal abundance) as well as training on species identification (TetraTech 2011).

immediate shutdown of the equipment will occur. Subsequent restart or ramp-up of equipment will occur only after the whale has cleared the safety exclusion zone.

3.9 Action Area

The action area is defined in 50 CFR 402.02 as "all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action." The action area includes the footprint of the energy project where the BIWF WTGs will be installed 3 miles southeast of Block Island; the submarine export/inter-array and BITS cable routes within Rhode Island Sound (BITS cable route: approximately 19.8 miles; export and inter-array cable route: approximately 6.2 miles); the route and waters traversed by project vessels between the staging and operations areas on Block Island and the Rhode Island mainland and the project sites (i.e., cable routes and wind farm; Block Island Sound, Rhode Island Sound); and, the underwater area where effects of the project (e.g., increases in suspended sediment (within approximately 1,000 feet from area of cable lay or pile disturbance in Rhode Island Sound), habitat modification (BIWF foundation: a total (construction plus operation) of approximately 29.9 acres of Rhode Island Sound benthos affected; cable lay areas: a maximum of 39.64 acres of Rhode Island Sound benthos affected; and underwater noise (Rhode Island Sound and portions of Vineyard Sound (confluence of Vineyard Sound and Rhode Island Sound) and coastal waters off Rhode Island (area south of Block Island to approximately 40°45.3'N)) will be experienced during construction, operations, and decommissioning.[25] The project location is illustrated in Appendix A.

4.0 STATUS OF THE SPECIES IN THE ACTION AREA

This section presents information on NMFS listed species in the action area and the 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.

The following listed species under NMFS jurisdiction are expected to occur in the action area and thus, may be exposed to the direct and indirect effects of the action:

Cetaceans	
North Atlantic right whale (*Eubalaena glacialis*)	Endangered
Humpback whale (*Megaptera novaeangliae*)	Endangered
Fin whale (*Balaenoptera physalus*)	Endangered
Sea Turtles	
Northwest Atlantic DPS of loggerhead sea turtle (*Caretta caretta*)	Threatened

[25] To define the maximum extent of underwater noise that would extend into the Atlantic Ocean, we considered the noise producing activity that would occur at the most southern extent of Block Island (impact pile driving: WTG foundation installation) and the isopleths of noise that would be produced from this activity. As whales, sea turtles and Atlantic sturgeon have different thresholds for injury or behavioral disturbance (see section 7.1.3), we then considered the isopleths at which these thresholds would be met during impact pile driving and considered the threshold that extended the farthest to represent the maximum extent of underwater noise that would extend into the Atlantic Ocean, and thus, potentially affect our species. We used Google Earth Pro to plot isopleths and estimate coordinates (last accessed December 16, 2013).

Leatherback sea turtle (*Dermochelys coriacea*)	Endangered
Kemp's ridley sea turtle (*Lepidochelys kempi*)	Endangered
Green sea turtle (*Chelonia mydas*)	Endangered/Threatened[26]
Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*)	
Gulf of Maine DPS	Threatened
New York Bight DPS	Endangered
Chesapeake Bay DPS	Endangered
South Atlantic DPS	Endangered
Carolina DPS	Endangered

4.1 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 southern and northern 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).

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). The species is also designated as depleted under the Marine Mammal Protection Act (MMPA).

The International Whaling Commission (IWC) recognizes two right whale populations in the North Atlantic: a western and eastern population (IWC 1986). It is thought that the eastern population migrated along the coast from northern Europe to northwest Africa. The current distribution and migration patterns of the eastern North Atlantic right whale population, if extant, are unknown. Sighting surveys from the eastern Atlantic Ocean suggest that right whales present in this region are rare (Best *et al.* 2001) and it is unclear whether a viable population in the eastern North Atlantic still exists (Brown 1986, NMFS 1991a). Photo-identification work has shown that some of the whales observed in the eastern Atlantic were previously identified as western Atlantic right whales (Kenney 2002). This Opinion will focus on the western North Atlantic right whale (*Eubalaena glacialis*), which occurs in the action area.

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.* 2013). 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).

[26] 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.

The distribution of right whales seems linked to the distribution of their principal zooplankton prey, calanoid copepods (Winn *et al.* 1986; NMFS 2005a; Baumgartner and Mate 2005; Waring *et al.* 2012). 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 Jeffreys Ledge, as well as Canadian waters including the Bay of Fundy and 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 note high interannual variability in right whale use of some habitats. Calving is known to occur in the winter months in coastal waters off of 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 off 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, 2010, and 2011, right whales were sighted on Jeffreys and Cashes Ledges, Stellwagen Bank, and Jordan Basin during December to February (Khan *et al.* 2009, 2010, 2011, 2012). 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.* 2012). 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 southeastern United States. The frequency with which right whales occur in offshore waters in the southeastern United States remains unclear (Waring *et al.* 2012).

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 as a result of 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 much 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 October 21, 2011 indicated that 425 individually recognized whales were known to be alive during 2009 (Waring *et al.* 2013). Whales catalogued by this date included 20 of the 39 calves born during that year. Adding the 19 calves not yet catalogued brings the minimum number alive in 2009 to 444. This number represents a minimum population size. The minimum number alive population index for the years 1990-2009 suggests a positive and slowly accelerating trend in population size. These data reveal a significant increase in the number of catalogued whales with a geometric mean growth rate for the period of 2.6% (Waring *et al.* 2013).

A total of 316 right whale calves were born from 1993 to 2010 (Waring *et al.* 2012). The mean calf production for this 18-year period is estimated to be 17.5/year (Waring *et al.* 2012). 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.* 2012). 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 2000-2010 calving seasons were remarkably better with 31, 21, 19, 17, 28, 19, 23, 23, 39, and 19 births, respectively (Waring *et al.* 2012). 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 2000/2001-2006/2007 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 (like that described in Henry *et al.* 2011, below). Of the 12 serious injuries and mortalities in 2005-2009, 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 recorded calf over a 25-year period (Kraus *et al.* 2007). In contrast, one of the largest right whales on record, "Stumpy," as a prolific breeder, successfully rearing calves in 1980, 1987, 1990, 1993, and 1996 (Moore *et al.* 2007). Stumpy was killed in February 2004 of an apparent ship strike (NMFS 2006a). 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 additional important information for assessing the effects of a proposed action on the species. As described in previous 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 the 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 with female survival particularly affected (Best *et al.* 2001). In 2002, NMFS 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 affecting females disproportionately (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), in conflict with the 2.6% positive trend from 1990-2009 noted above by Waring *et al.* (2013). Despite the preceding, examination of the minimum number alive population index calculated from the individual sightings database for the years 1990-2009 suggest a positive and slowly accelerating trend in population size (Waring *et al.* 2013). These data reveal a significant increase in the number of catalogued right whales alive during this period (Waring *et al.* 2013). 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 how the populations would fare without entanglement mortalities as compared to the status quo. Only two of 1,000 projections (with the status quo simulation) ended with a smaller total population size than they started, and no projections resulted in extinction. As described above, the mean growth rate estimated in the latest stock assessment report was 2.6% (Waring *et al.* 2012).

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 five years between 1998-2003, and then decreased to just over three years in 2004 and 2005.

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 that 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 underway to assess this relationship further and to examine 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. Similarly, 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 PCB and DDT concentrations were lower than those found in other affected marine mammals (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 is 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 conclude that right whales are being exposed to measurable quantities of paralytic shellfish poisoning (PSP) toxins and domoic acid via trophic transfer from their prey upon which they feed (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) suggests 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 abundance, right whales had significantly thinner blubber than during years of greater prey abundance. 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 Clapham *et al*. (2002) also suggests it affects calf survival. Greene *et al.* (2003) described the potential oceanographic processes linking climate variability to 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 1982-1992, 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 a function of both food availability and 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

The potential biological removal (PBR)[27] for the Western Atlantic stock of North Atlantic right whale is 0.9 (Waring *et al.* 2013). Right whale recovery is negatively affected by anthropogenic mortality. From 2006 to 2010, right whales had the highest proportion relative to their population of reported entanglement and ship strike events of any species (Waring *et al.* 2012). 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.* 2012). For the period 2006-2010, the annual human-caused mortality and serious injury rate for the North Atlantic right whale averaged 3.0 per year (2.4 in U.S. waters; 0.6 in Canadian waters) (Waring *et al.* 2013). Nineteen confirmed right whale mortalities were reported along the U.S. East Coast and adjacent Canadian Maritimes from 2006 to 2010 (Henry *et al.* 2012). 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.* 2012).

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 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.* 2012). Of the 19

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27 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.

total confirmed right whale mortalities (2006-2010) described in Henry *et al.* (2012), four were confirmed to be entanglement mortalities and five were confirmed to be ship strike mortalities. Serious injury involving right whales was documented for five entanglement events and one ship strike event.

Although disentanglement is often unsuccessful or not possible for many cases, there were at least two documented cases of entanglements for which the intervention of disentanglement teams averted a likely serious injury from 2006 to 2010 (Waring *et al.* 2012). Even when entanglement or vessel collision does not cause direct mortality, it may weaken or compromise an individual so that subsequent injury or death is more likely (Waring *et. al* 2012). 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.* 2012).

Entanglement records from 1990 to 2010 maintained by NMFS include 74 confirmed right whale entanglement events (Waring *et al.* 2012). Because whales often free themselves of gear following an entanglement event, scarification analysis of living animals may provide better indications of fisheries interactions rather than entanglement records (Waring *et al.* 2012). 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 six 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 right whales of 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 exhibits signs of injury from vessel strikes.

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, taking 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) overuse 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, an October 2011 review of the photo-ID recapture database indicated that 444 individually recognized right whales were known to be alive in 2009 (Waring *et al.* 2013). The 2000/2001-2009/2010 calving seasons had relatively high calf production (31, 21, 19, 17, 28, 19, 23, 23, 39, and 19 calves, respectively) and included additional first time mothers (*e.g.*, eight new mothers in 2000/2001) (Waring *et al.* 2009, 2012).

Over the five-year period 2006-2010, 55 confirmed events involved right whales, 33 were confirmed entanglements and 13 were confirmed ship strikes. There were 19 verified right whale mortalities, four due to entanglements, and five due to ship strikes (Henry *et al.* 2012). 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.* 2012).

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 October 21, 2011 for the years 1990-2009 suggest a positive trend in numbers of right whales (Waring *et al.* 2013). 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.

4.2 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 southern and northern 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 between 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-era 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 which 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 bays. 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.* 2012). 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, including the waters of Stellwagen Bank, year-round. They feed on 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 on capelin (Waring *et al.* 2010; 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.* 2012). 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 1991b).

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 between 1985 and 1992 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% CI. = 8,000-13,600) (Stevick *et al.* 2003; Waring *et al.* 2013). 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.* 2012). The minimum population estimate for the Gulf of Maine stock is 823 whales, derived from a 2008 mark-recapture based count (Waring *et al.* 2013).

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.* 2012). However, it is unclear whether the apparent decline in growth rate is a bias 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.* 2012). Regardless, calf survival appears to have increased since 1996, presumably accompanied by an increase in population growth (Waring *et al.* 2012). Stevick *et al.* (2003) calculated an average population growth rate of 3.1% in the North Atlantic population overall for the period 1979-1993.

Anthropogenic Injury and Mortality

The PBR for the Gulf of Maine stock of humpback whale is 2.7. 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 2006-2010, the minimum annual rate of human-caused mortality and serious injury to the Gulf of Maine humpback whale stock averaged 7.8 animals per year (U.S. waters, 7.2; Canadian waters, 0.6) (Waring *et al.* 2013). Between 2006 and 2010, humpback whales were involved in 101 confirmed entanglement events and 21 confirmed ship strike events (Henry *et al.* 2012). Over the five-year period, humpback whales were the most commonly reported entangled whale species; entanglements accounted for nine mortalities and 20 serious injuries (Henry *et al.* 2012). Of the 21 confirmed ship strikes, 10 of the events were fatal (Henry *et al.* 2012). 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.* 2012; Waring *et al.* 2012).

Based on photographs taken from 2000-2002 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, and 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 were 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 is listed as "undetermined," and the investigation has been closed, though could be re-opened if new information becomes available.

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.* 2012). Shifts in relative finfish species abundance correspond to changes in observed humpback whale movements (Stevick *et al.* 2006). However, whether humpback whales were adversely affected by these trophic changes is unknown.

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 823 whales (Waring *et al.* 2013). 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.* 2013). 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.

4.3 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 1998b). 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, 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 1987-2003 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.* 2012). 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 United States, 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.* 2012).

During the 1978-1982 aerial surveys, fin whales accounted for 24% of all cetaceans and 46% of all large cetaceans sighted over the continental shelf between Cape Hatteras and Nova Scotia (Waring *et al.* 2012). 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 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).

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 2012 Stock Assessment Report (SAR) gives a best estimate of abundance for fin whales in the western North Atlantic of 3,522 (CV = 0.27). 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.* 2012). The minimum population estimate for the western North Atlantic fin whale is 2,817 (Waring *et al.* 2012). However, there are insufficient data at this time to determine population trends for the fin whale (Waring *et al.* 2012). The PBR for the western North Atlantic fin whale is 5.6. 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 in U.S. and Canadian waters from 2006 to 2010 was 2.0 (U.S. waters, 1.8; Canadian waters, 0.2) (Waring *et al.* 2012). During this five-year period, there were 15 confirmed entanglements (two fatal; two serious injuries) and eight ship strikes (six fatal) (Henry *et al.* 2012). 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), seven 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 factors affecting geographic distribution of cetaceans, water temperature appears to be the main influence, with other factors primarily influencing how individuals are distributed within their ranges (MacLeod 2009). Cetacean species most likely to be affected by increases in water temperature are those with ranges restricted to non-tropical waters and with a preference for shelf waters. 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 3,522 and the minimum population estimate is 2,817. The 2012 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 Ocean 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.

4.4 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 rangewide 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 1998a, 2008), Kemp's ridley sea turtle (NMFS *et al.* 2011c), leatherback sea turtle (NMFS and USFWS 1992b, 1998b), Kemp's ridley sea turtle (NMFS *et al.* 2011c) and green sea turtle (NMFS and USFWS 1991, 1998c).

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 cleanup 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.

4.4.1 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 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 and recommendations have been made regarding its ESA listing status. Based on a 2007 five-year status review of the species, which discussed the range of threats to loggerheads including climate change, NMFS and USFWS determined that loggerhead sea turtles should not be delisted or reclassified as endangered. However, the 2007 status review also determined that an analysis and review of the species should be conducted to determine whether DPSs should be identified for the loggerhead sea turtle (NMFS and USFWS 2007a). This initiative was supported by studies showing that 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 USFWS 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 showing 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) that would 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 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 would be made and solicited new information and analysis. 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.

On September 22, 2011, NMFS and USFWS issued a final rule (76 FR 58868) determining that the loggerhead sea turtle population is composed of nine DPSs (as defined in Conant *et al.*, 2009). 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, that the overall nesting population remains widespread, that the trend for the nesting population appears to be stabilizing, and that 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 U.S. waters (NWA DPS and North Pacific DPS) would 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 were 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 nine 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, and west of 40°W l; Northeast Atlantic Ocean (NEA) DPS – north of the equator, south of 60°N, east of 40°W, and west of 5°36' W; South Atlantic DPS – south of the equator, north of 60°S, west of 20°E, and east of 60°W; Mediterranean DPS – the Mediterranean Sea east of 5°36'W. 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 U.S. 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 (LaCasella *et al.* In Review). Given that the action area is a subset of the area fished by U.S. 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). 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 five-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).

In the western Atlantic, waters as far north as southern Canada and the Gulf of Maine are used for foraging by juveniles and adults (Shoop 1987; Shoop and Kenney 1992; Ehrhart *et al.* 2003; Mitchell *et al.* 2003; NMFS NEFSC 2011a, 2012, 2013). In U.S. Atlantic waters, loggerheads most commonly occur throughout the inner continental shelf from Florida to Cape Cod, MA 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°-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. Surveys of continental shelf waters north of Cape Hatteras, NC indicated that loggerhead sea turtles were most commonly sighted in waters with bottom depths ranging from 22 to 49 meters deep (Shoop and Kenney 1992). Loggerheads were observed in waters ranging in depth from 0 (*i.e.*, on the beach) to 4,481 meters (Shoop and Kenney 1992). 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 1: Typical values of life history parameters for loggerheads nesting in the U.S.

Life History Parameter	Data
Clutch Size	100-126 eggs[28]
Egg incubation duration (varies depending on time of year and latitude)	42-75 days[29,30]
Pivotal temperature (incubation temperature that produces an equal number of males and females)	29.0˚C[31]
Nest productivity (emerged hatchlings/total eggs) x 100 (varies depending on site specific factors)	45-70%[2,3]
Clutch frequency (number of nests/female/season)	3-5.5 nests[32]
Internesting interval (number of days between successive nests within a season)	12-15 days[33]
Juvenile (<87 cm CCL) sex ratio	65-70%[34]
Remigration interval (number of years between successive nesting migrations)	2.5-3.7 years[35]
Nesting season	Late April-Early September
Hatching season	Late June-early November
Age at sexual maturity	32-25 years[36]
Life span	>57 years[37]

Population Dynamics and Status

The majority of Atlantic nesting occurs on beaches of the southeastern United States (NMFS and USFWS 2007a). For the past decade, 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; (2) a south Florida group of nesting females that nest from

[28] Dodd (1988).
[29] Dodd and Mackinnon (1999, 2000, 2001, 2002, 2003, 2004).
[30] Blair Witherington, FFWCC, personal communication, 2006 (information based on nests monitored throughout Florida beaches in 2005, n=865).
[31] Mrosovsky (1988).
[32] 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).
[33] Caldwell (1962); Dodd (1988).
[34] National Marine Fisheries Service (2001); Allen Foley, FFWCC, personal communication (2005).
[35] Richardson *et al.* (1978); Bjorndal *et al.* (1983); Ehrhart, unpublished data.
36 Melissa Snover, NMFS, personal communication (2005).
37 Dahlen *et al.* (2000).

29°N 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, FL 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, FL), (3) the Dry Tortugas Recovery Unit (DTRU: islands located west of Key West, FL), (4) the Northern Gulf of Mexico Recovery Unit (NGMRU: Franklin County, FL through Texas), and (5) the Greater Caribbean Recovery Unit (GCRU: Mexico through French Guiana, Bahamas, Lesser Antilles, and Greater Antilles).

The Loggerhead 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.

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 to 23 years. These analyses used different analytical approaches, but all found that there had been a significant overall nesting decline within the NWA DPS. However, with the addition of nesting data from 2008 to 2012, the trend line changes, showing a strong positive trend since 2007 (http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trends/). The nesting data presented in the Recovery Plan (through 2008) are 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 to 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 trend was analyzed using nesting data available as of October 2008. The NRU dataset included 11 beaches with an uninterrupted 20-year time series; 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). The trend was analyzed using nesting data available as of October 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 (1989-2008) with approximately 1,272 females nesting per year; (2) for the PFRU, a mean of 64,513 nests per year (1989-2007) with approximately 15,735 females nesting per year; (3) for the DTRU, a mean of 246 nests per year (1995-2004, excluding 2002) with approximately 60 females nesting per year; and (4) for the NGMRU, a mean of 906 nests per year (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 (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 estimates of the number of nesting females per year for any nesting assemblage in this recovery unit. 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 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 differences 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 conducted in some areas of the Northwest Atlantic provide data by which to assess the relative abundance of loggerhead sea turtles and changes in abundance over time (Maier *et al.* 2005; Morreale *et al.* 2005; Mansfield 2006; Ehrhart *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, no discernible trend at one site, and a decreasing at two sites in the northeast United States. 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.* (2005) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the southeast coast of the United States (Winyah Bay, SC to St. Augustine, FL) 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.* 2005). 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 four 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 to 2004 show an increasing trend of loggerheads at the 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 compared to the period 1987-1992. Only two loggerheads (of a total 54 turtles) were observed captured in pound net gear during the period 2002-2004, while the previous decade's study recorded 11 to 28 loggerheads 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 and annual reports for 2010, 2011, and 2012 have been produced. AMAPPS is a multi-agency initiative to assess marine mammal, sea turtle, and seabird abundance and distribution in the Atlantic. As presented in NMFS NEFSC (2011a), 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 2011a). 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, NC, 30% in the southern Mid-Atlantic Bight, and 6% in the northern Mid-Atlantic Bight. 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 results from aerial surveys and research to improve the abundance estimates are anticipated through 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 five-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). Among natural threats, 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, 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 breeders 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 consultations. 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). A 2002 section 7 consultation on the U.S. South Atlantic and Gulf of Mexico shrimp fisheries 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 design, interactions between loggerheads and the shrimp fishery had been declining because of reductions in fishing effort unrelated to fisheries management actions. The 2002 South Atlantic and GOM Shrimp Opinion (NMFS 2002a) take estimates were based in part on fishery effort levels. In recent years, low shrimp prices, rising fuel costs, competition with imported products, and the impacts of 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 were substantially less than were projected in the 2002 Opinion. In 2008, the NMFS Southeast Fisheries Science Center (SEFSC) estimated annual number of interactions between loggerheads and shrimp trawls in the Gulf of Mexico shrimp fishery to be 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). In August 2010, NMFS reinitiated section 7 consultation on southeastern state and federal shrimp fisheries based on a high level of strandings, elevated nearshore sea turtle abundance as measured by trawl catch per unit of effort, and lack of compliance with TED requirements. The 2012 section 7 consultation on the shrimp fishery was unable to estimate the current total annual level of take for loggerheads. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in at least thousands and possibly tens of thousands of interactions annually, of which at least hundreds and possibly thousands are expected to be lethal (NMFS 2012a).

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 reduction of sea turtle captures in fishing operations is identified in recovery plans and five-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 to 2008 (Warden 2011a). NEFOP data from 1994 to 2008 were used to develop a model of interaction rates that 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 in waters < 50 meters 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 nine-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 evaluated loggerhead sea turtle interactions in scallop dredge gear from 2001 to 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 is applied to trips with chain mats, the estimated number of observable and inferred interactions of hard-shelled sea turtles after chain mats were implemented is 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). Turtle Deflector Dredges (TDDs) are required in the scallop fishery as of May 1, 2013, and are expected to further decrease serious injuries to sea turtles.

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 to 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 (>7 inch/17.8 cm) gillnets (Murray 2009a). In the spring of 2000, a total of 275 loggerhead carcasses were recovered from North Carolina beaches. The cause of death for most of the turtles was unknown, but NMFS suspects that the mass mortality event was caused by a large-mesh gillnet fishery for monkfish and dogfish operating offshore in the preceding weeks (67 FR 71895, December 3, 2002).

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 2012). In 2010, there were 40 observed interactions between loggerhead sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2012). All of the loggerheads were released alive, with 29 out of 40 (72.5%) released with all gear removed. A total of 344.4 (95% CI: 236.6-501.3) loggerhead sea turtles were estimated to have interacted with the longline fisheries managed under the HMS FMP in 2010 based on the observed bycatch events (Garrison and Stokes 2012). The 2010 estimate is considerably lower than those in 2006 and 2007 and is well below the historical highs that occurred in the mid-1990s (Garrison and Stokes 2012). 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 interactions also occur in other fishery gear types and by non-fishery mortality sources (*e.g.*, hopper dredges, power plants, vessel collisions), although quantitative/qualitative estimates are only available for activities on which NMFS has consulted.

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) 2007a). 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 effects to sea turtles resulting from climate change are not quantifiable at this time (Hawkes *et al.* 2009). For analysis on the potential effects of climate change on loggerhead sea turtles, see Section 5.0 below.

Summary of Status for Loggerhead Sea Turtles

Loggerheads continue to be affected by many factors 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. Of the nine DPSs defined in the NMFS and USFWS final rule (75 FR 12598), only the NWA DPS is considered in this Opinion.

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 2012 are analyzed, researchers found no demonstrable trend, indicating a reversal of the post-1998 decline (http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trends/). Loggerhead nesting has been on the rise since 2008, and Van Houton and Halley (2011) suggest that nesting in Florida, which contains by far the largest loggerhead rookery in the DPS, could substantially increase over the next few decades. For the purposes of this Opinion, we consider that loggerhead nesting in the NWA DPS will continue to show no discernible trend, and perhaps more importantly, no decline over the period that data are available.

In-water data is conflicting, with some sites showing an increase while others indicating a possible decrease. Given the limited sampling locations and durations, differences in methodology, and conflicting information to date, we anticipate that the in-water data results will continue to be variable. For the purposes of this Opinion, we interpret the in-water data for the NWA DPS to show no discernible trend.

In terms of population numbers, the 2010 AMAPPS aerial line transect surveys provided a preliminary regional abundance estimate of about 588,000 loggerheads along the U.S. Atlantic coast, with an inter-quartile range of 382,000-817,000 (NEFSC 2011a). 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 sea turtle sightings. 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. However, a more recent loggerhead population estimate prepared by Richards *et al.* (2011) using data from 2001-2010 states that the loggerhead adult female population in the Northwest Atlantic is 38,334 individuals (SD =2,287). They estimated adult female recovery unit sizes to range from a minimum of 258 females for the DTRU to a maximum of 45,048 females for the PFRU. For the purposes of this Opinion, we consider the number of adult female loggerheads in the NWA DPS to be 38,334 turtles.

Based on the information presented above, for purposes of this Opinion, we consider that the status of NWA DPS of loggerheads over the next 29 years will be no worse than it is currently. Actions have been taken to reduce anthropogenic impacts to loggerhead sea turtles from various sources, particularly since the early 1990s. These include lighting ordinances, predation control, and nest relocations to help increase hatchling survival, as well as measures to reduce the mortality of pelagic immatures, benthic immatures, and sexually mature age classes from various fisheries and other marine activities (Conant *et al.* 2009). Recent actions have taken significant steps towards reducing the recurring sources of mortality and improving the status of all nesting stocks. For example, TED, chain mat, and TDD regulations represent a significant improvement in the baseline effects of trawl and dredge fisheries on loggerheads in the Northwest Atlantic, although shrimp trawling is still considered to be one of the largest sources of anthropogenic mortality on loggerheads (SEFSC 2009, NMFS 2012b).

4 4.2 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.* 2011a).

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.* 2011a). Females lay an average of 2.5 clutches within a season (TEWG 1998, 2000) and the mean remigration interval for adult females is two 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 *Sargassum* 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 suggest that benthic immature developmental areas occur along the U.S. coast and that these areas may change with resource quality and quantity (TEWG 2000). Developmental habitats are defined by several characteristics, including sheltered coastal areas such as embayments and estuaries, and nearshore temperate waters shallower than 50 meters (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 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 meters 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 three-day period in May 2007 and more than 4,000 of those nested at Rancho Nuevo (NMFS and USFWS 2007c). In 2008, 17,882 nests were documented on Mexican nesting beaches (NMFS *et al.* 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 *et al.* 2011). 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).

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 use the more northern habitats of Cape Cod Bay and Long Island Sound. In the last six years (2007-2013), the number of cold-stunned turtles ranged from a low in 2007 of 66 (40 Kemp's ridleys, seven loggerheads, 16 greens, and three unknown) to a high in 2013 of 491 (273 Kemp's ridleys, 167 loggerheads, 43 greens, and eight unknown). Annual cold stunning events vary in magnitude; the magnitude of episodic major cold stunning events may be associated with numbers of turtles using 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 are 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 egg exploitation and fishery interactions. From the 1940s through the early 1960s, nests from Rancho 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 (NMFS and USFWS 1992a). Subsequently, NMFS worked with the industry to reduce sea turtle takes in shrimp trawls and other trawl fisheries in several ways, including through the development and use of TEDs. As described above, there is lengthy regulatory history on the use of TEDs in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries (NMFS 2002a; Epperly 2003; Lewison *et al.* 2003).

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, has occurred annually after 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. The 2012 section 7 consultation on the shrimp fishery was unable to estimate the total annual level of take for Kemp's ridleys at present. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in at least tens of thousands and possibly hundreds of thousands of interactions with Kemp's ridleys annually, of which at least thousands and possibly tens of thousands are expected to be lethal (NMFS 2012a).

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 was unknown, but NMFS suspects that the mass mortality event was caused by 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 NEFSC also documented 14 Kemp's ridleys entangled in or impinged on Virginia pound net leaders from 2002 to 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. 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 to 2006 (NMFS 2006c).

The recovery plan for Kemp's ridley sea turtles (NMFS *et al.* 2011a) 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.* 2011a). Models (Davenport 1997, Hulin and Guillon 2007, Hawkes *et al.* 2007, all referenced in NMFS *et al.* 2011a) 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 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). For analysis on the potential effects of climate change on Kemp's ridley sea turtles, see Section 5.0 below

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.* 2011a). 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.* 2011a). 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 BP Deepwater Horizon oil spill, 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 also contribute to annual human-caused mortality, but the levels are unknown. Based on their five-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 Secretary of Environment and Natural Resources, Mexico (SEMARNAT) released the second revision to the Kemp's ridley Recovery Plan. Based on the information presented above, for purposes of this Opinion, we consider that the status of Kemp's ridleys over the next 29 years will be no worse than it is currently and that the species may actually be in the early stages of recovery, although this should be viewed in the context of a much larger population in the mid-20th century.

4 4.3 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 green sea turtles 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 located throughout the Pacific and along the southwestern U.S. coast (NMFS and USFWS 1998c). In the western Pacific, major nesting rookeries at four sites including Heron Island (Australia), Raine Island (Australia), Guam, and Japan were evaluated. Three where determined to be increasing in abundance, while the population in Guam appears stable (NMFS and USFWS 2007d). In the central Pacific, nesting occurs on French Frigate Shoals, HI, which has also been reported as increasing, with a mean of 400 nesting females annually from 2002 to 2006 (NMFS and USFWS 2007d). In 2012, we received a petition to delist the Hawaiian population of green sea turtles, and our 90-day finding determined that the petition, viewed in context of information readily available in our files, presents substantial scientific and commercial information indicating that the petition action may be warranted (77 FR 45571). A status review is currently underway. The main nesting sites for green sea turtles 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, more 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 caught for food in many areas of the Pacific. They also were commercially exploited, which, coupled with habitat degradation, led to their decline in the Pacific (NMFS and USFWS 1998c). 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 1998c; 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). 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 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. However, a declining trend is apparent along the coast of Israel, where 300-350 nests were deposited each year in the 1950s (Sella 1982) compared to a mean of six nests per year from 1993 to 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 Syrian coast was surveyed in 1991, but nesting activity was attributed to loggerheads) bodes well for the speculation that the unsurveyed coast of Libya may also host substantial nesting.

Atlantic Ocean

Distribution and Life History

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). Adult females may nest multiple times in a season (average three 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

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 five-year status review for the species identified eight geographic areas considered to be primary nesting sites 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) Trindad Island, 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 nesting sites except that nesting in Florida was reviewed in place of Trindad Island, Brazil. He concluded that all sites in the central and western Atlantic showed increased nesting except 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 to 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 to 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 Trindad Island, Brazil 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 five-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 (2000-2012) have shown an increasing trend of green sea turtle nesting, with a low of 581 in 2001 to a high of 15,352 in 2011 (NMFS and USFWS 2007d, FWC 2013). 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, and Florida panhandle beaches (Meylan *et al.* 1995). More recently, green sea turtle nesting occurred on Bald Head Island, NC (just east of the mouth of the Cape Fear River), Onslow Island, NC 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 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 lagoons, areas with low water turnover, 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, and may cause death (George 1997).

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 turtles 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.*, 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. The 2012 section 7 consultation on the shrimp fishery was unable to estimate the total annual level of take for green sea turtles. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in at least hundreds and possibly low thousands of interactions with green sea turtles annually, of which hundreds are expected to be lethal (NMFS 2012a).

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 and 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 most recent five-year status review for green sea turtles (NMFS and USFWS 2007d) notes that global climate change is affecting the species and will likely continue to be a threat. There is an increasing female bias in the sex ratio of green sea 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. For analysis on the potential effects of climate change on green sea turtles, see Section 5.0 below.

Summary of Status of Green Sea Turtles
A review of 32 Index Sites distributed globally revealed a 48-67% decline in the number of mature females nesting annually over the last three generations (Seminoff 2004).[38] An evaluation of green sea turtle nesting sites was also conducted as part of the five-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 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 of 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).

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

Seminoff (2004) and NMFS and USFWS (2007d) came to comparable conclusions for four nesting sites in the western Atlantic, finding that sea turtle abundance is increasing in the Atlantic Ocean. Both 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 at Tortuguero had increased markedly since the 1970s (Seminoff 2004; NMFS and USFWS 2007d).

However, the five-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).

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 also contribute to human-caused mortality, though the level is unknown. Based on its five-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 to determine whether DPSs should be identified (NMFS and USFWS 2007d). Based on the information presented above, for purposes of this Opinion, we consider that the status of green sea turtles over the next 29 years will be no worse than it is currently and that the status of the species in the Atlantic Ocean may actually be improving due to increased nesting.

4.4.4 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 1998b, 2007b; Sarti *et al.* 2000). The western Pacific major nesting beaches are 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 has been 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 noted throughout the western Pacific region, where observers report that nesting groups are well below abundance levels observed several decades ago (*e.g.*, Suárez 1999). 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). In the 2003-2004 season, only 120 nests on the four primary index beaches (combined) were counted (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. An 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, Colombia, 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. 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 to be 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 1992b). 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, NC 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 meters, but 84.4% of sightings were in waters less than 180 meters (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 than 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. NMFS is currently reviewing whether the addition of waters adjacent to a major nesting beach in Puerto Rico to the critical habitat designation is warranted. USFWS also plans to address this region during a future planned status review. 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 July 16, 2010, which found that the petition did not present substantial scientific information indicating that the revision was warranted. The original petitioners submitted a second petition on November 2, 2010 to revise the critical habitat designation to include waters adjacent to a major nesting beach in Puerto Rico, and this time included additional information on the usage of the waters. On May 5, 2011, NMFS determined that a revision to critical habitat off Puerto Rico may be warranted, but on June 4, 2012 issued a decision denying the petition due to a lack of reasonably defined physical or biological features that are essential to the leatherback sea turtle's conservation and that may require special management considerations or protection (77 FR 32909). Note that on August 4, 2011, USFWS 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. 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 nine 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 centimeters 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 two to three 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. Leatherback hatchlings enter the water soon after hatching. Based on a review of all sightings of leatherback sea turtles of <145 centimeters CCL, Eckert (1999) found that leatherback juveniles remain in waters warmer than 26°C until they exceed 100 centimeters CCL.

Population Dynamics and Status

As described earlier, sea turtle nesting survey data is important because 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 five-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 U.S., 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) to 1,712 recorded in 2012 (FWC 2013). 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 to 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 five of the seven populations or groups of populations, with the exceptions of the Western Caribbean and West Africa groups. 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 also seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests in 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 a positive population growth rate was found for French Guinea and Suriname using nest numbers from 1967 to 2005, a 39-year period, and that there was 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 to 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. 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 suggested that the true abundance of leatherbacks may be 4.27 times higher (Palka 2000).

Threats

The five-year status review (NMFS and USFWS 2007b) and TEWG (2007) reports both 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, particularly trap/pot gear. 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, associated seawater ingestion, and stress.

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 after 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. The most recent section 7 consultation on the shrimp fishery, completed in May 2012, was unable to estimate the total annual level of take for leatherbacks at present. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in a few hundred interactions annually, of which a subset are expected to be lethal (NMFS 2012a). Leatherbacks have been documented interacting with longline, trap/pot, trawl, and gillnet fishing gear. For instance, an estimated 6,363 leatherback sea turtles were caught by the U.S. Atlantic tuna and swordfish longline fisheries between 1992 and 1999 (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 three-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 2012). All leatherbacks were released alive, with all gear removed in 14 (53.8%) of the 26 captures. A total of 170.9 (95% CI: 104.3-280.2) leatherback sea turtles are estimated to have interacted with the longline fisheries managed under the HMS FMP in 2010 based on the observed takes (Garrison and Stokes 2012). The 2010 estimate continues a downward trend since 2007 and remains well below the average prior to implementation of gear regulations (Garrison and Stokes 2012). 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 (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).

Leatherbacks are susceptible to entanglement in the lines associated with trap/pot gear used in several fisheries. From 1990 to 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). From 2002 to 2011, NMFS received 159 reports of sea turtles entangled in vertical lines from Maine to Virginia, with 147 events confirmed (verified by photo documentation or response by a trained responder; NMFS 2008a). Of the 147 confirmed events during this period, 133 events involved leatherbacks, 13 involved loggerheads, and 1 involved a green sea turtle. NMFS identified the gear type and fishery for 93 of the 147 confirmed events, which included lobster (51[39]), whelk/conch (23), black sea bass (10), crab (7), and research pot gear (2). 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 2002a). Leatherbacks are likely to encounter shrimp trawls working in the coastal waters off the U.S. Atlantic coast (from Cape Canaveral, FL 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). Modified TEDs are now required in order to exclude leatherbacks as well as large benthic immature and sexually mature loggerhead and green sea turtles. With these gear modifications, Epperly *et al.* (2002) anticipated an average of 80 leatherback mortalities a year in shrimp gear interactions, but dropped the estimate to 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). The most recent Opinion, issued in 2012, does not give a numerical ITS for leatherbacks, but instead monitors TED compliance and fishery effort to monitor and limit take (NMFS 2012a).

Other trawl fisheries are also known to interact with leatherback sea turtles on a much smaller scale. For example, NMFS fisheries observers documented leatherbacks taken in trips targeting *Loligo* squid off Delaware in 2001 and off Connecticut in 2009, and targeting little skate off Connecticut in 2011. 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.

39 One case involved both lobster and whelk/conch gear, but this animal is listed only under the lobster group.

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. NEFOP data from 1994 to 1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in pelagic drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 64% to 99% (Waring *et al.* 2000). 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 leatherbacks were removed from an 11-inch (28.2-cm) monofilament shark gillnet set in the nearshore waters off Cape Hatteras (STSSN unpublished data reported in NMFS SEFSC 2001). Murray (2009a) reports five observed leatherback captures in Mid-Atlantic sink gillnet fisheries between 1994 and 2008.

Fishing gear interactions can occur throughout the leatherback's range, including in Canadian waters. Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland/Labrador were entangled in salmon nets, herring nets, gillnets, trawl lines, and crab pot lines. 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 seen in the leatherback sea turtle population in French Guiana from 1973 to 1998 (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 *et al.*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 Trinidad and Tobago, with mortality estimated to be between 50% and 95% (Eckert and Lien 1999). Many of the sea turtles do not die as a result of drowning, but rather because the fishermen butcher them to remove them from 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 1985 and 2007) reported plastic within the turtles' stomach contents, and in some cases (8.7% of cases in which plastic was reported), blockage of the gut 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 drifting movements, 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 (Mrosovsky *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.* 2009). Leatherbacks have expanded their range in the Atlantic north by 330 kilometers in the last 17 years as warming has caused the northerly migration of the 15°C 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). For analysis on the potential effects of climate change on leatherback sea turtles, see Section 5.0 below.

Summary of Status for Leatherback Sea Turtles

In the Pacific Ocean, the abundance of leatherback sea turtles on nesting beaches has declined dramatically during 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 due to human activities that have reduced the number of nesting females and reduced the reproductive success of females (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 beaches in Suriname and French Guiana that support the majority of leatherback nesting in this region

(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, mortality due to fisheries interactions 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 anthropogenic mortality. The long-term recovery potential of this species may be further threatened by observed low genetic diversity, even in the largest nesting groups (NMFS and USFWS 2007b).

Based on its five-year status review of the species, NMFS and USFWS (2007b) determined that endangered leatherback sea turtles should not be delisted or reclassified. However, it also was 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). Based on the information presented above, for purposes of this Opinion, we consider that the status of leatherbacks over the next 29 years will be no worse than it is currently and that the status of the species in the Atlantic Ocean may actually be improving due to increased nesting.

4.5 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 provides information specific to the status of each DPS of Atlantic sturgeon. Below, we also provide a description of the Atlantic sturgeon DPSs likely to occur in the action area and their use of the action area.

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, FL (Scott and Scott 1988; ASSRT 2007;). NMFS has divided U.S. populations of Atlantic sturgeon into five DPSs (77 FR 5880 and 77 FR 5914). These are: the Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs (see).[40]

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 that 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.

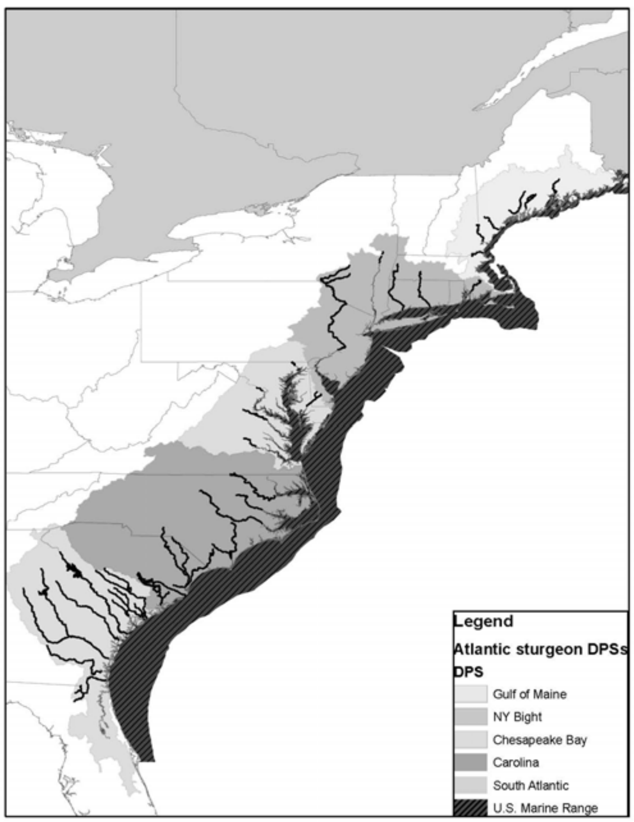
On February 6, 2012, we published notice in the Federal Register that we were listing the New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs as "endangered," and the Gulf of Maine DPS as "threatened" (77 FR 5880 and 77 FR 5914). The effective date of the listings is April 6, 2012. The DPSs do not include Atlantic sturgeon spawned in Canadian rivers. Therefore, fish that originated in Canada are not included in the listings. As described below, individuals originating from all five listed DPSs may occur in the action area. Information general to all Atlantic sturgeon, as well as information specific to each of the DPSs, is provided

[40] 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."

below.

Atlantic Sturgeon Life History
Atlantic sturgeon are long-lived (approximately 60 years), late maturing, estuarine dependent, anadromous fish (Bigelow and Schroeder 1953; Vladykov and Greeley 1963; Mangin 1964; Pikitch *et al.* 2005; Dadswell 2006; ASSRT 2007).[41] They are a relatively large fish, even among sturgeon species (Pikitch *et al.* 2005) and can grow to over 14 feet weighing 800 pounds. Atlantic sturgeon are bottom feeders that suck food into a ventral 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). Juvenile Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007).

Figure 1: Geographic Locations for the Five ESA-listed DPSs of Atlantic Sturgeon



[41] 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).

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. The largest recorded Atlantic sturgeon was a female captured in 1924 that measured approximately 4.26 meters (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). The lengths of Atlantic sturgeon caught since the mid-late 20th century have typically been less than three meters (Smith *et al.* 1982; Smith and Dingley 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). While females are prolific, with egg production ranging from 400,000 to 4 million eggs per spawning year, females spawn at intervals of two to five 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% of the maximum lifetime egg production is achieved is estimated to be 29 years (Boreman 1997). Males exhibit spawning periodicity of one to five 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°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 centimeters per second and depths are 3-27 meters (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 1984; 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 four weeks old, with total lengths (TL) less than 30 millimeters; Van Eenennaam *et al.* 1996) are assumed to mostly live on or near the bottom 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 juvenile 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 both high salinity and 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 meters 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.* 2004a; 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 meters during winter and spring, and in the northern portion of the Mid-Atlantic Bight at depths less than 20 meters 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, NC 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, with the majority of these tag returns from relatively shallow nearshore fisheries, with few fish reported from waters in excess of 25 meters (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 meters (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.* 2004a; 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, FL. We have considered the best available information to determine from which DPSs individuals in the action area are likely to have originated. We have determined that

Atlantic sturgeon in the action area likely originate from all five DPSs at the following frequencies: Gulf of Maine (GOM) 11%; New York Bight (NYB) 51%; Chesapeake Bay (CB) 13%; Carolina 2%, and South Atlantic (SA) 22%. Approximately 1% of the Atlantic sturgeon in the action area originate from Canada. These percentages are based on genetic sampling of all individuals (n=173) captured during observed fishing trips along the Atlantic coast from Maine through North Carolina, and the results of the genetic analyses for these 173 fish were compared against a reference population of 411 fish and results for an additional 790 fish from other sampling efforts. Therefore, they represent the best available information on the likely genetic makeup of individuals occurring in the action area. The genetic assignments have a plus/minus 5% confidence interval. However, for purposes of section 7 consultation, we have selected the reported values without their associated confidence intervals. The reported values, which approximate the mid-point of the range, are 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).

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 River, 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 17 U.S. rivers are known to support spawning (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 five rivers (Kennebec, Androscoggin, Hudson, Delaware, James) are known to currently support spawning from Maine through Virginia, where historical records show that there used to be 15 spawning rivers (ASSRT 2007). Thus, there are substantial gaps between Atlantic sturgeon spawning rivers among northern and Mid-Atlantic States which could make recolonization of extirpated populations more difficult.

At the time of the listing, there were no current, published population abundance estimates for any of the currently known spawning stocks or for any of the five DPSs of Atlantic sturgeon. An estimate of 863 mature adults per year (596 males and 267 females) was calculated for the Hudson River based on fishery-dependent data collected from 1985 to 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 and Altamaha Rivers 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).

Lacking complete estimates of population abundance across the distribution of Atlantic sturgeon, the NEFSC developed a virtual population analysis model with the goal of estimating bounds of Atlantic sturgeon ocean abundance (see Kocik *et al*. 2013). The NEFSC suggested that cumulative annual estimates of surviving fishery discards could provide a minimum estimate of abundance. The objectives of producing the Atlantic Sturgeon Production Index (ASPI) were to characterize uncertainty in abundance estimates arising from multiple sources of observation and process error and to complement future efforts to conduct a more comprehensive stock assessment (Table 2). The ASPI provides a general abundance metric to assess risk for actions that may affect Atlantic sturgeon in the ocean; however, it is not a comprehensive stock assessment. In general, the model uses empirical estimates of post-capture survivors and natural survival, as well as probability estimates of recapture using tagging data from the United States Fish and Wildlife Service (USFWS) sturgeon tagging database, and federal fishery discard estimates from 2006 to 2010 to produce a virtual population. The USFWS sturgeon tagging database is a repository for sturgeon tagging information on the Atlantic coast. The database contains tag, release, and recapture information from state and federal researchers. The database records recaptures by the fishing fleet, researchers, and researchers on fishery vessels.

In additional to the ASPI, a population estimate was derived from the Northeast Area Monitoring and Assessment Program (NEAMAP) (Table 2). NEAMAP trawl surveys are conducted from Cape Cod, Massachusetts to Cape Hatteras, North Carolina in nearshore waters at depths up to 18.3 meters (60 feet) during the fall since 2007 and spring since 2008. Each survey employs a spatially stratified random design with a total of 35 strata and 150 stations. The ASMFC has initiated a new stock assessment with the goal of completing it by the end of 2014. NOAA Fisheries will be partnering with the ASMFC to conduct the stock assessment, and the ocean population abundance estimates produced by the NEFSC will be shared with the stock assessment committee for consideration in the stock assessment.

Table 2: Description of the ASPI Model and NEAMAP Survey Based Area Estimate Method.

Model Name	Model Description
A. ASPI	Uses tag-based estimates of recapture probabilities from 1999 to 2009. Natural mortality based on Kahnle *et al.* (2007) rather than estimates derived from tagging model. Tag recaptures from commercial fisheries are adjusted for non-reporting based on recaptures from observers and researchers. Tag loss assumed to be zero.
B. NEAMAP Swept Area	Uses NEAMAP survey-based swept area estimates of abundance and assumed estimates of gear efficiency. Estimates based on average of ten surveys from fall 2007 to spring 2012.

Table 3: Modeled Results

Model Run	Model Years	95% low	Mean	95% high
A. ASPI	1999-2009	165,381	417,934	744,597
B.1 NEAMAP Survey, swept area assuming 100% efficiency	2007-2012	8,921	33,888	58,856
B.2 NEAMAP Survey, swept area assuming 50% efficiency	2007-2012	13,962	67,776	105,984
B.3 NEAMAP Survey, swept area assuming 10% efficiency	2007-2012	89,206	338,882	588,558

As illustrated by Table 3 above, the ASPI model projects a mean population size of 417,934 Atlantic sturgeon and the NEAMAP Survey projects mean population sizes ranging from 33,888 to 338,882 depending on the assumption made regarding efficiency of that survey. As noted above, the ASPI model uses empirical estimates of post-capture survivors and natural survival, as well as probability estimates of recapture using tagging data from the United States Fish and Wildlife Service (USFWS) sturgeon tagging database, and federal fishery discard estimates from 2006 to 2010 to produce a virtual population. The NEAMAP estimate, in contrast, is more empirically derived and does not depend on as many assumptions. For the purposes of this Opinion, while the ASPI model is considered as part of the ASMFC stock assessment, we consider the NEAMAP estimate as the best available information on population size.

Once we have selected the NEAMAP method, we must then determine the most appropriate estimate of the efficiency of that survey. Atlantic sturgeon are frequently encountered during the NEAMAP surveys. The information from this survey can be used to calculate minimum swept area population estimates within the strata swept by the survey. The estimate from fall surveys ranges from 6,980 to 42,160 with coefficients of variation between 0.02 and 0.57, and the estimates from spring surveys ranges from 25,540 to 52,990 with coefficients of variation between 0.27 and 0.65 (Table 4). These are considered minimum estimates because the calculation makes the assumption that the gear will capture (i.e. net efficiency) 100% of the sturgeon in the water column along the tow path and that all sturgeon are with the sampling domain of the survey. We define catchability as 1) the product of the probability of capture given encounter (i.e. net efficiency), and 2) the fraction of the population within the sampling domain. Catchabilities less than 100% will result in estimates greater than the minimum. The true catchability depends on many factors including the availability of the species to the survey and the behavior of the species with respect to the gear. True catchabilities much less than 100% are common for most species. The ratio of total sturgeon habitat to area sampled by the NEAMAP survey is unknown, but is certainly greater than one (i.e. the NEAMAP survey does not survey 100% of the Atlantic sturgeon habitat, i.e. does not include rivers, northernmost and southernmost portions of range or depths beyond 18.3m).

Table 4: Annual minimum swept area estimates for Atlantic sturgeon during the Spring and Fall from the Northeast Area Monitoring and Assessment Program Survey.[42]

Year	Fall Number	CV	Spring Number	CV
2007	6,981	0.015		
2008	33,949	0.322	25,541	0.391
2009	32,227	0.316	41,196	0.353
2010	42,164	0.566	52,992	0.265
2011	22,932	0.399	52,840	0.480
2012			28,060	0.652

Available data do not support estimation of true catchability (i.e., net efficiency X availability) of the NEAMAP trawl survey for Atlantic sturgeon. Thus, the NEAMAP swept area biomass estimates were produced and presented in Kocik *et al*. (2013) for catchabilities from 5 to 100%. In estimating the efficiency of the sampling net, we consider the likelihood that an Atlantic sturgeon in the survey area is likely to be captured by the trawl. Assuming the NEAMAP surveys have been 100% efficient would require the unlikely assumption that the survey gear captures all Atlantic sturgeon within the path of the trawl and all sturgeon are within the sampling area of the NEAMAP survey. In estimating the fraction of the Atlantic sturgeon population within the sampling area of the NEAMAP, we consider that the NEAMAP-based estimates do not include young of the year fish and juveniles in the rivers. Additionally, although the NEAMAP surveys are not conducted in the Gulf of Maine or south of Cape Hatteras, NC, the NEAMAP surveys are conducted throughout the majority of the action area from Cape Cod to Cape Hatteras at depths up to 18.3 meters (60 feet), which includes the preferred depth ranges of subadult and adult Atlantic sturgeon. NEAMAP surveys take place during seasons that coincide with known Atlantic sturgeon coastal migration patterns in the ocean. Therefore, the NEAMAP estimates are minimum estimates of the ocean population of Atlantic sturgeon but are based on sampling in much of the action area, in known sturgeon coastal migration areas during times that sturgeon are expected to be migrating north and south.

Based on the above, we consider that the NEAMAP samples an area utilized by Atlantic sturgeon, but does not sample all the locations and times where Atlantic sturgeon are present and the trawl net captures some, but likely not all, of the Atlantic sturgeon present in the sampling area. Therefore, we assumed that net efficiency and the fraction of the population exposed to the NEAMAP survey in combination result in a 50% catchability. The 50% catchability assumption seems to reasonably account for the robust, yet not complete sampling of the Atlantic sturgeon oceanic temporal and spatial ranges and the documented high rates of encounter with NEAMAP survey gear and Atlantic sturgeon. For this Opinion, we have determined that the best available data at this time are the population estimates derived from NEAMAP swept area biomass resulting from the 50% catchability rate.

42 Estimates assume 100% net efficiencies. Estimates provided by Dr. Chris Bonzek, Virginia Institute of Marine Science (VIMS).

The ocean population abundance of 67,776 fish estimated from the NEAMAP survey assuming 50% efficiency was subsequently partitioned by DPS based on genetic frequencies of occurrence (Table 5). Given the proportion of adults to subadults in the observer database (approximate ratio of 1:3), we have also estimated a number of subadults originating from each DPS. However, this cannot be considered an estimate of the total number of subadults because it only considers those subadults that are of a size vulnerable to capture in commercial sink gillnet and otter trawl gear in the marine environment and are present in the marine environment.

Table 5: Summary of calculated population estimates based upon the NEAMAP survey swept area assuming 50% efficiency

DPS	Estimated Ocean Population Abundance	Estimated Ocean Population of Adults	Estimated Ocean Population of Subadults (of size vulnerable to capture in fisheries)
GOM (11%)	7,455	1,864	5,591
NYB (51%)	34,566	8,642	25,925
CB (13%)	8,811	2,203	6,608
Carolina (2%)	1,356	339	1,017
SA (22%)	14,911	3,728	11,183
Canada (1%)	678	170	509

Threats Faced by Atlantic Sturgeon Throughout Their Range

Atlantic sturgeon are susceptible to over-exploitation given their life history characteristics (e.g., late maturity and 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 19th and 20th centuries (Taub 1990; Smith and Clugston 1997; Secor and Waldman 1999).

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 could 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) loss of unique haplotypes; (5) 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, emigration to marine habitats to grow, and return of adults to natal rivers to spawn.

Based on the best available information, NMFS has concluded that unintended catch in fisheries, vessel strikes, poor water quality, fresh 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 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 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, because Atlantic sturgeon depend on a variety of habitats, every life stage is likely affected by one or more of the identified threats.

Atlantic sturgeon are particularly sensitive to bycatch mortality 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. Based on these life history traits, Boreman (1997) calculated that Atlantic sturgeon can only withstand the annual loss of up to 5% of their population to bycatch mortality without suffering population declines. Mortality rates of Atlantic sturgeon taken as bycatch in various types of fishing gear range between 0 and 51%, with the greatest mortality occurring in sturgeon caught by sink gillnets. Atlantic sturgeon are particularly vulnerable to being caught in sink gillnets; therefore, fisheries using this type of gear account for a high percentage of Atlantic sturgeon bycatch. 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, including the prohibition on possession, have addressed impacts to Atlantic sturgeon through directed fisheries, the listing determination concluded that the mechanisms in place to address the risk posed to Atlantic sturgeon from commercial bycatch were insufficient.

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 their parts in or from the EEZ 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 captured 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.

Bycatch in U.S. waters is one of the primary threats faced by all five 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 DPSs. This is because of (1) the small number of data points and, (2) the lack of information on the percent of incidents 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 (NMFS NEFSC 2011b). The analysis estimates that from 2006 through 2010, there were averages of 1,548 and 1,569 encounters per year in observed gillnet and trawl fisheries, respectively, with an average of 3,118 encounters combined annually. Mortality rates in gillnet gear were approximately 20%. Mortality rates in otter trawl gear are generally lower, at approximately 5%.

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 affect the South Atlantic and Carolina DPSs. Implications of climate change to the Atlantic sturgeon DPSs have been speculated, yet no scientific data are available on past trends related to climate effects on this species, and current scientific methods are not able to reliably predict the future magnitude of climate change and associated impacts or the adaptive capacity of these species. Impacts of climate change on Atlantic sturgeon are uncertain at this time, and cannot be quantified. Any prediction of effects is made more difficult by a lack of information on the rate of expected change in conditions and a lack of information on the adaptive capacity of the species (i.e., its ability to evolve to cope with a changing environment). For analysis on the potential effects of climate change on Atlantic sturgeon, see Section 5.0 below.

4.5.1 Status of Gulf of Maine DPS

The GOM DPS includes the following: all anadromous Atlantic sturgeon that spawn or are spawned in the watersheds from the Maine/Canadian border and, extending southward, all watersheds draining into the GOM as far south as Chatham, MA. The marine range of Atlantic sturgeon from the GOM DPS extends from Hamilton Inlet, Labrador, Canada, to Cape Canaveral, FL. The riverine range of the GOM DPS and the adjacent portion of the marine range are shown in Figure 1. Within this range, Atlantic sturgeon historically spawned in the Androscoggin, Kennebec, Merrimack, Penobscot, and Sheepscot Rivers (ASSRT 2007). Spawning still occurs in the Kennebec and Androscoggin Rivers, 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. 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% 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) (Kieffer 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 ongoing to determine whether Atlantic sturgeon are spawning in the Penobscot and Saco Rivers. Atlantic sturgeon that are spawned elsewhere continue to use habitats within these rivers as part of their overall marine range (ASSRT 2007).

At its mouth, the Kennebec River drains an area of 24,667 square kilometers, and is part of a large estuarine system that includes the Androscoggin and Sheepscot Rivers (ASMFC 1998a; NMFS and USFWS 1998d; Squiers 1998). The Kennebec and Androscoggin Rivers flow into Merrymeeting Bay, a tidal freshwater bay, and exit as a combined river system through a narrow channel, flowing approximately 32 kilometers (20 miles) to the Atlantic Ocean as the tidal segment of the Kennebec River (Squiers 1998). This lower tidal segment of the Kennebec River forms a complex with the Sheepscot River estuary (ASMFC 1998a; Squiers 1998).

Substrate type in the Kennebec estuary is largely sand and bedrock (Fenster and Fitzgerald 1996; Moore and Reblin 2010). Main channel depths at low tide typically range from 17 meters (58 feet) near the mouth to less than 10 meters (33 feet) in the Kennebec River above Merrymeeting Bay (Moore and Reblin 2010). Salinities range from 31 parts per thousand at Parker Head (5 kilometers from the mouth) to 18 parts per thousand at Doubling Point during summer low flows (ASMFC 1998a). The 14-kilometer river segment above Doubling Point to Chops Point (the outlet of Merrymeeting Bay) is an area of transition (mid estuary) (ASMFC 1998a). The salinities in this section vary both seasonally and over a tidal cycle. During spring this section is entirely fresh water but during summer low flows, salinities can range from 2 to 3 parts per thousand at Chops Point to 18 parts per thousand at Doubling Point (ASMFC 1998a). The river is essentially tidal freshwater from the outlet of Merrymeeting Bay upriver to the site of the former Edwards Dam (ASMFC 1998a). Mean tidal amplitude ranges from 2.56 meters at the mouth of the Kennebec River estuary to 1.25 meters in Augusta near the head of tide on the Kennebec River (in the vicinity of the former Edwards Dam) and 1.16 meters at Brunswick on the Androscoggin River (ASMFC 1998a).

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 1998a; NMFS and USFWS 1998d). 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 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 four ripe males and one ripe female captured on July 26,1980; and, (3) capture of nine adults during a gillnet survey conducted from 1977 to 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 1998d; ASMFC 2007). The low salinity of waters above Merrymeeting Bay are consistent with values found in other rivers where successful Atlantic sturgeon spawning is known to occur.

Age to maturity for GOM 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 11 to 21 years for Atlantic sturgeon originating from the Hudson River (Young *et al.* 1998), and 22 to 34 years for Atlantic sturgeon that originate from the Saint Lawrence River (Scott and Crossman1973). Therefore, age at maturity for Atlantic sturgeon of the GOM DPS likely falls within these values. Of the 18 sturgeon examined from the commercial fishery that occurred in the Kennebec River in 1980, all of which were considered mature, age estimates for the 15 males ranged from 17-40 years, and from 25-40 years old for the three females (Squiers *et al.* 1981).

Several threats play a role in shaping the current status of GOM 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 were caught in the Kennebec River by local fishermen (Squiers *et al.*, 1979). After the collapse of sturgeon stock in the 1880s, the sturgeon fishery was almost non-existent. All directed Atlantic sturgeon fishing as well as retention of Atlantic sturgeon bycatch has been prohibited since 1998. Nevertheless, mortalities associated with bycatch in fisheries in state and federal waters still occur. In the marine range, GOM DPS Atlantic sturgeon are incidentally captured in federal and state-managed fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (Stein *et al.* 2004b; 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 affected by dredging and other in-water activities, disturbing spawning habitat and also altering the benthic forage base. Many rivers in the GOM DPS have navigation channels that are maintained by dredging. Dredging outside of federal channels and in-water construction occurs throughout the GOM DPS. 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. 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, and 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 historical natural falls and likely represent the maximum upstream extent of sturgeon occurrence even if the dams were not present. Because no Atlantic sturgeon 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. The extent that Atlantic sturgeon are affected by operations of dams in the Gulf of Maine region is currently unknown; however, 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 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 Dam, which prevents Atlantic sturgeon from accessing approximately 29 kilometers of habitat, including the presumed historical spawning habitat located downstream of Milford Falls, the site of the Milford Dam. While removal of the Veazie Dam is anticipated to occur in the near future, the presence of this dam is currently preventing access to significant habitats within the Penobscot River. Atlantic sturgeon are known to occur in the Penobscot River, but it is unknown whether spawning is currently occurring or whether the presence of the Veazie Dam 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. As with the Penobscot, it is unknown how the Essex Dam affects the likelihood of spawning in this river.

GOM 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 pulp and paper mills' industrial 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.

There are no direct in-river abundance estimates for the GOM DPS. The Atlantic Sturgeon Status Review Team (ASSRT) (2007) presumed that the GOM DPS was comprised of less than 300 spawning adults per year, based on extrapolated abundance estimates from the Hudson and Altamaha 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. As described earlier in Section 4.4, we have estimated that there are a minimum of 7,455 GOM DPS adult and subadult Atlantic sturgeon of size vulnerable to capture in federal marine fisheries. We note further that this estimate is predicated on the assumption that fish in the GOM DPS would be available for capture in the NEAMAP survey which extends from Block Island Sound (RI) southward.

Summary of the Gulf of Maine DPS

Spawning for the GOM DPS is known to occur in two rivers (Kennebec and Androscoggin). Spawning may be occurring in other rivers, such as the Sheepscot, Merrimack, and Penobscot, but has not been confirmed. There are indications of potential increasing abundance of Atlantic sturgeon belonging to the GOM 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 GOM 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 GOM 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). In Maine state waters, there are strict regulations on the use of fishing gear that incidentally catches sturgeon. In addition, in the last several years 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% (e.g., 7 of 84 fish) of interactions observed south of Chatham being assigned to the GOM DPS (Wirgin and King 2011). Tagging results also indicate that GOM DPS fish tend to remain within the waters of the Gulf of Maine and only occasionally venture to points south.

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 % originated from the GOM DPS (Wirgin *et al.* 2012). Thus, a significant number of the GOM DPS fish appear to migrate north into Canadian waters where they may be subjected to a variety of threats including bycatch.

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). We have determined that the GOM 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.

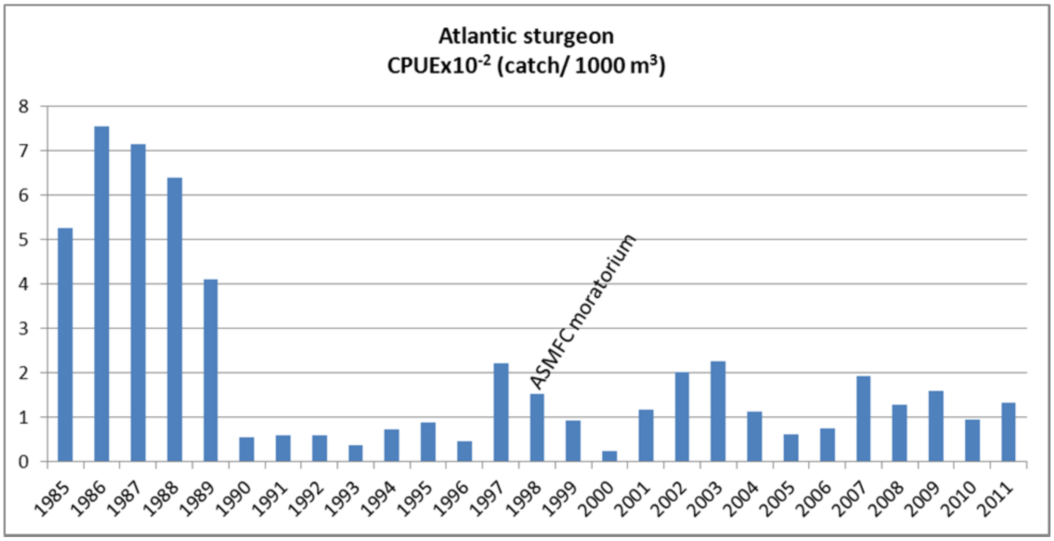
4.5.2 Status of New York Bight DPS

The NYB DPS includes the following: all anadromous Atlantic sturgeon that spawn or are spawned in the watersheds that drain into coastal waters from Chatham, MA to the Delaware-Maryland border on Fenwick Island. The marine range of Atlantic sturgeon from the NYB DPS extends from Hamilton Inlet, Labrador, Canada, to Cape Canaveral, FL. The riverine range of the NYB DPS and the adjacent portion of the marine range are shown in

Figure 1. 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 before the over-exploitation of the 1800s is unknown but has been conservatively estimated at 6,000 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 to 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. All available data on abundance of juvenile Atlantic sturgeon in the Hudson River Estuary indicate a substantial drop in production of young since the mid 1970s (Kahnle *et al.* 1998). A decline appeared to occur in the mid to late 1970's followed by a secondary drop in the late 1980s (Kahnle *et al.* 1998; Sweka *et al.* 2007; ASMFC 2010) CPUE data suggests that recruitment has remained depressed relative to catches of juvenile Atlantic sturgeon in the estuary during the mid-late 1980s (Sweka *et al.* 2007; ASMFC 2010). The CPUE data from 1985 to 2011 show significant fluctuations. There appears to be a decline in the number of juveniles between the late 1980s and early 1990s and then a slight increase in the 2000s, but, given the significant annual fluctuation, it is difficult to discern any real trend. Despite the CPUEs from 2000 to 2011 being slightly higher than those from 1990 to 1999, they are low compared to the mid to late 1980s (see **Figure 2**)

Figure 2: Hudson River Atlantic Sturgeon CPUE Juvenile Index (1985-Present).



There is no overall, empirical abundance estimate for the Delaware River population of Atlantic sturgeon. Harvest records from the 1800s 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 millimeters 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 these YOY indicates that at least three 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 small.

Several threats play a role in shaping the current status and trends observed in the Delaware River and Estuary. Mortalities associated with bycatch in fisheries in state and federal waters occur. In the marine range, NYB DPS Atlantic sturgeon are incidentally captured in federal and state-managed fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (Stein *et al.* 2004b; 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. 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 and may be detrimental to the long-term viability of the NYB DPS, as well as other DPSs (Brown and Murphy 2010).

Summary of the New York Bight DPS

Atlantic sturgeon originating from the NYB 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 relatively high between these rivers. Some of the impact from the threats that contributed to the decline of the NYB 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 NYB DPS.

In its marine range, NYB DPS Atlantic sturgeon are incidentally captured in federal and state-managed fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (Stein *et al.* 2004a; ASMFC 2007). Based on mixed stock analysis results presented by Wirgin and King (2011), more than 40% of the Atlantic sturgeon bycatch interactions in the Mid Atlantic Bight region were sturgeon from the NYB 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 NYB DPS (Wirgin *et al.* 2012). At this time, we are not able to quantify the impacts from threats other than fisheries 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 to document fish mortalities, many do not. We have reports of one Atlantic sturgeon entrained during hopper dredging operations in Ambrose Channel, NJ. We recently consulted on two dredging projects: the ACOE Delaware River Federal Navigation Channel deepening project and on the New York and New Jersey Harbor Deepening Project. In both cases, we determined that while the proposed actions may adversely affect Atlantic sturgeon, they were not likely to jeopardize the continued existence of any DPS of Atlantic sturgeon (NMFS 2012c and NMFS 2012d).

In the Hudson and Delaware Rivers, dams do not block access to historical habitat. The Holyoke Dam on the Connecticut River blocks passage past the dam at Holyoke; however, the extent that Atlantic sturgeon would historically have used habitat upstream of Holyoke is unknown. The first dam on the Taunton River may block access to historical spawning habitat. Connectivity also 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 to which Atlantic sturgeon are affected by operations of dams in the New York Bight region is currently unknown. Atlantic sturgeon may also be impinged or entrained at power plants in the Hudson and Delaware Rivers, and may be adversely affected by the operation of the power plants, but the power plants have not been found to jeopardize their continued existence.

NYB DPS Atlantic sturgeon may also be affected by degraded water quality. Rivers in the NYB region, including the Hudson and Delaware, have been heavily polluted by industrial and sewer discharges. In general, water quality has improved in the Hudson and Delaware over the past several decades (Lichter *et al.* 2006; EPA 2008). While water quality has improved and most discharges are limited through regulations, it likely that pollutants persist in the benthic environment. This can be particularly problematic if pollutants are present on spawning and nursery grounds, where developing eggs and larvae are particularly susceptible to exposure to contaminants.

Vessel strikes are known to 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 NYB 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 NYB DPS. As described in Section 4.4, we have estimated that there are a minimum of 34,566 NYB DPS adult and subadult Atlantic sturgeon of size vulnerable to capture in federal marine fisheries. We have determined that the NYB DPS is currently at risk of extinction due to: (1) 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.

4.5.3 Status of Chesapeake Bay DPS

The CB DPS includes the following: all anadromous Atlantic sturgeons that spawn or 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. The marine range of Atlantic sturgeon from the CB DPS extends from Hamilton Inlet, Labrador, Canada, to Cape Canaveral, FL. The riverine range of the CB DPS and the adjacent portion of the marine range are shown in

Figure 1. 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 % 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 *et al.* 2009). However, conclusive evidence of current spawning is only available for the James River, where a recent study found evidence of Atlantic sturgeon spawning in the fall (Balazik *et al.* 2012). 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 (Vladykov and Greeley 1963; ASSRT 2007; Wirgin *et al.* 2007; Grunwald *et al.* 2008).

Age to maturity for CB 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 CB DPS likely falls within these values.

Several threats play a role in shaping the current status of CB 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 1998b; 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 1998a; ASSRT 2007; EPA 2008). These conditions contribute to reductions in DO levels throughout the Bay. The availability of nursery habitat, in particular, may be limited given the recurrent hypoxic (low DO) conditions within the Bay (Niklitschek and Secor 2005; 2010). Heavy industrial development during the twentieth century in rivers inhabited by sturgeon impaired water quality and impeded these species' recovery.

Although there have been improvements in the some areas of the Bay's health, the ecosystem remains in poor condition. EPA gave the overall health of the Bay a grade of 45% based on goals for water quality, habitats, lower food web productivity, and fish and shellfish abundance (EPA CBP 2010). This was a 6% increase from 2008. According to EPA, the modest gain in the health score was due to a large increase in adult blue crab population, expansion of underwater grass beds growing in the Bay's shallows, and improvements in water clarity and bottom habitat health as highlighted below:

- 12% of the Bay and its tidal tributaries met Clean Water Act standards for DO between 2007 and 2009, a decrease of 5% from 2006-2008.
- 26% of the tidal waters met or exceeded guidelines for water clarity, a 12% increase from 2008.
- Underwater bay grasses covered 9,039 more acres of the Bay's shallow waters for a total of 85,899 acres, 46% of the Bay-wide goal.
- The health of the Bay's bottom dwelling species reach a record high of 56% of the goal, improving by approximately 15 Bay-wide.
- The adult blue crab population increased to 223 million, its highest level since 1993.

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 CB DPS.

In the marine and coastal range of the CB 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.* 2004b; ASMFC 2007; ASSRT 2007).

Summary of the Chesapeake Bay DPS

Spawning for the CB 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 CB 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). As described in Section 4.4, we have estimated that there is a minimum ocean population of 8,811 CB DPS Atlantic sturgeon, of which 2,319 are adults and 6,608 are subadults of size vulnerable to capture in federal marine fisheries.

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 CB DPS of Atlantic sturgeon. Of the 35% of Atlantic sturgeon incidentally caught in the Bay of Fundy, about 1% were CB DPS fish (Wirgin *et al.* 2012). Studies have shown that Atlantic sturgeon can only sustain low levels of bycatch mortality (Boreman 1997; ASMFC 2007; Kahnle *et al.* 2007). The CB DPS is currently at risk of extinction given (1) 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.

4.5.4 Status of the Carolina DPS

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, FL. The riverine range of the Carolina DPS and the adjacent portion of the marine range are shown in

Figure 1.

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 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. 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. Fish from the Carolina DPS likely use other river systems than those listed here for their specific life functions.

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. Prior 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 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, are estimated to be less than 3% of what they were historically (ASSRT 2007). As described in Section 4.4, we have estimated that there are a minimum of 1,356 Carolina DPS adult and subadult Atlantic sturgeon of size vulnerable to capture in federal marine fisheries.

Table 6: 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.

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	

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 more than 60% of the historical sturgeon habitat upstream of the dams in the Cape Fear and Santee-Cooper River systems. Water quality (velocity, temperature, and 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 also 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 and other resource agencies. Since the 1993 legislation requiring certificates for transfers took effect, 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 in the mid to late 19th century, from which they have never rebounded. Continued bycatch of Atlantic sturgeon in commercial fisheries is an ongoing impact to the Carolina DPS. More robust fishery independent data on bycatch are available for the northeast and mid-Atlantic than in the Southeast where high levels of bycatch underreporting are suspected.

Though there are statutory and regulatory regulations that authorize reducing the impact of dams on riverine and anadromous species, these mechanisms have proven inadequate for preventing dams from blocking access to habitat upstream and degrading habitat downstream. Water quality continues to be a problem in the Carolina DPS, even with existing controls on some pollution sources. Current regulatory regimes are not 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 are 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 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% of historical population sizes) for 100 years. Small numbers of individuals resulting from drastic reductions in populations, such as that which occurred 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 late-maturing species such as Atlantic sturgeon, and they continue to face a variety of other threats that contribute to their risk of extinction. Their late age at maturity provides more opportunities for individual Atlantic sturgeon to be removed from the population before reproducing. While a long life-span also allows multiple opportunities to contribute to future generations, it also increases the time frame 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.

Summary of the Status of the Carolina DPS of Atlantic Sturgeon

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 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 impede their recovery.

The presence of dams has resulted in the loss of more than 60% 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 may 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 either reduced ability to perform major life functions, such as foraging and spawning, or post-capture mortality. While some of the threats to the Carolina DPS have been ameliorated or reduced due to existing regulatory mechanisms, such as the moratorium on directed fisheries for Atlantic sturgeon, bycatch and habitat alterations are currently not being addressed through existing mechanisms. Further, despite NMFS' authority under the Federal Power Act to prescribe fish passage and existing controls on some pollution sources, access to habitat and improved water quality continues to be a problem. The inadequacy of regulatory mechanisms to control bycatch and habitat alterations is contributing to the status of the Carolina DPS.

4.5.5 Status of South Atlantic DPS

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, FL. The marine range of Atlantic sturgeon from the South Atlantic DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, FL. The riverine range of the South Atlantic DPS and the adjacent portion of the marine range are shown in

Figure 1.

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. 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 populations present in the St. Johns, are 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. Fish from the South Atlantic DPS likely use other river systems than those listed here for their specific life functions.

Table 7: 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.

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	

The riverine spawning habitat of the South Atlantic DPS occurs within the South Atlantic Coastal Plain ecoregion, 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. The primary threats to biological diversity in the South Atlantic Coastal Plain listed by The Nature Conservancy (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's 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 before the collapse of the fishery in 1890. However, because fish from South Carolina are included in both the Carolina and South Atlantic DPSs, it is likely that some of the historical 8,000 fish would be attributed to both the Carolina DPS and the South Atlantic DPS. The sturgeon fishery had been the third largest fishery in Georgia. 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 population in at least two river systems within the South Atlantic DPS has been extirpated. As described in Section 4.4, we have estimated that there are a minimum of 14,911 SA DPS adult and subadult Atlantic sturgeon of size vulnerable to capture in federal marine fisheries.

The South Atlantic DPS was listed as endangered under the ESA as a result of a combination of habitat curtailment and modification, overuse (i.e., being taken as bycatch) in commercial fisheries, and the inadequacy of regulatory mechanisms in addressing 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 River. Reductions in water quality from terrestrial activities also 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 highly sensitive to low DO and the negative (metabolic, growth, and feeding) effects caused by low DO increase when water temperatures are concurrently high, such as those found within the range of the South Atlantic DPS. Additional stressors arising from water allocation and climate change threaten to exacerbate existing water quality problems throughout the range of the South Atlantic DPS. Large water withdrawals of more than 240 mgd of water are known to be removed from the Savannah River for power generation and municipal uses. However, permits for users withdrawing less than 100,000 gallons per day (gpd) are not required, so actual water withdrawals from the Savannah and other rivers within the range of the South Atlantic DPS are unknown, but 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.

The directed Atlantic sturgeon fishery caused initial severe declines in southeast Atlantic sturgeon populations. Although the directed fishery is closed, bycatch in other commercial fisheries continues to impact the South Atlantic DPS. Statutory and regulatory mechanisms exist that authorize reducing the impact of dams on riverine and anadromous species such as Atlantic sturgeon, but 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 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.

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

The population of mature adult Atlantic sturgeon in the South Atlantic DPS is estimated to be at least 3,728. The DPS's 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, FL. 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 and DO are also contributing to the status of the South Atlantic DPS, particularly during times of high water temperatures, which increase the detrimental effects on Atlantic sturgeon habitat. Interbasin water transfers and climate change may exacerbate existing water quality issues. Bycatch also contributes to the South Atlantic DPS's 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 use 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 either reduced ability to perform major life functions, such as foraging and spawning, or post-capture mortality. While some 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 and habitat alteration are currently not being adequately addressed through existing mechanisms. Further, access to habitat and good water quality continues to be a problem even with NMFS' authority under the Federal Power Act to prescribe 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 (i.e., construction through decommissioning) of the proposed action (i.e., 29 years). For the following reasons, effects will only be considered over the 29 year life of the project as effects of the action are not expected to extend beyond this timeframe. Construction of the BITS and BIWF will result in the most significant direct and indirect effects to these species and their habitat. Effects from the construction of these structures will occur during the construction itself which is likely to take approximately three years . Any effects resulting from the operation, maintenance and repair of the BITS and BIWF are expected to be experienced at most, a few months after any disturbance, and thus, confined to the 25 year operational life of the BITS and BIWF. As explained in the effects analysis section, the portions of the project that may affect listed species are restricted to the construction phase and these effects are temporary only and will not extend beyond that phase of the project. Additionally, at the end of the operational life of the BITS and BIWF, all cables will remain in place, and only the WTG foundations will be removed, via cutting. Cutting operations may result in minor disturbances to benthic sediments, which are expected to settle within several hours after cutting operations are complete, and prolonged effects are not expected. All other decommissioning activities will occur above the surface of the water (i.e., removal of the WTGs). Based on this information, we expect any effects from decommissioning to remain within the timeframe of these activities (i.e. from 2041-2042).

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.

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 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°-5°C (5°-9°F) on average in the next 100 years which is more than the projected global increase (NAST 2000). A warming of about 0.2°C (0.4°F) per decade is projected for the next two decades over a range of emission scenarios (IPCC 2007a). 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 2007a).

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 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, and Fin 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. 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 and 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). For example, 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 (Greene *et al*. 2003). More information, is therefore, 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).

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 2007a). 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.* 2011a) 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 seawater. 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.* 2011a). Models (Davenport 1997, Hulin and Guillon 2007, Hawkes *et al.* 2007, all referenced in NMFS *et al.* 2011a) 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 salt wedge 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 salt wedge. It is unlikely that shifts in the location of the salt wedge 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, and Fin 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 (to 2043) 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 1.5°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 to the year 2043.[43] As noted previously, the anticipated impacts from the proposed project are concentrated during the construction phase which is expected to be completed by 2019. Given the slow rate of climate change, it is even

[43] Frumhoff *et al*. 2007 predicted Northeast ocean sea surface temperatures to increase somewhere between 2.8 and 4.4°C by 2100. As predictive models on sea surface temperature changes in Rhode Island Sound were not available, the latter serves as the best available information on sea surface temperature changes in the action area as a result of climate change.

more unlikely, therefore, that whales will experience any significant effect from climate change between now and 2019. As such, we do not anticipate any shifts in the species range within the next five years that would change the way we have conducted our effects analysis in this Opinion.

Mother-calf pairs are not a common occurrence in the action area. Since 1986, only 18 pairs have been documented in the action area (i.e., Block Island Sound:2 pair; Rhode Island Sound: 7 pairs; Atlantic Ocean (area south of Block Island to approximately 40°45.3'N): 8 pairs; Vineyard Sound: 1 pair; http://www.nefsc.noaa.gov/psb/surveys/SASInteractive2.html (last accessed December 18, 2013).[44] However, changes in sea temperature have the potential to increase the occurrence of calves in the action area. Right and humpback whales calve in the winter months (i.e., between approximately December through March), within warm waters (i.e., 13 to 17°C) off the southeastern United States or the West Indies, respectively (calving and calving areas for fin whales are unknown at this time; SARS 2012; Katona and Beard 1990; Clapham *et al*. 1993; Palsbøll *et al*. 1997; Stevick *et al*. 1998; Mate *et al.* 1997; Garrison 2007; Good 2008; Patrician *et al*. 2009; Keller *et al*. 2012**).** Calving is thought to occur in these areas because calves have less blubber and are less insulated against cold temperatures (Keller *et al*. 2012; Garrison 2007) and thus, the absence of mother-calf right whale pairs from New England waters before April is thought to be primarily related to water temperature (see Keller *et al*. 2012). However, should climate change affect New England sea surface temperatures in the winter months, such that they increase to levels that will support a calf (i.e., between 13 and 17°C), then mother-calf pairs could occur sooner and more frequently in the action area. We considered climate change impacts in the action area over the next 29 years to provide context within which the effects of the action will occur from present to 2043. The model projections are for sea surface temperatures to increase somewhere between 2.8-4.4°C by 2100 (Frumhoff *et al.* 2007). Assuming that there is a linear trend in increasing water temperatures, one could anticipate a 0.03-0.05°C increase each year, with an increase in temperature of approximately 1.5°C between now and 2043. We conclude that given this small increase, it is not likely that over the proposed 29-year life of the project that any water temperature changes would be significant enough to change the distribution, abundance or behavior of whales in the action area such that the conclusions reached by us in this consultation are invalid. Further, after 2019 the only effects of the action will be limited to the presence of the WTGs and the cable; even if there were shifts in the distribution or abundance of right whales in the action area between 2019 and 2043, this would not change our assessment of effects. As noted above, water temperatures for calving habitat need to be between 13 and 17°C (Garrison 2007; Good 2008). Temperatures in the action area during the calving season are significantly colder, ranging between 0 and 10°C in the winter. We are not aware of any models that predict large enough temperature increases to make New England waters, including Block Island and Rhode Island Sound, as warm as the southern calving habitat during the winter. During the 29-year life of the BIWF and BITS, we do not anticipate sea surface temperatures will increase to such a level that more mothers will bring very young calves to, or even give birth in, the action area. As such, we do not, over the life of the project, expect more numbers of calves to be present in the action area. It is also important to note that our analysis considers the potential for mothers and calves to be present in the action area, based on their occasional occurrence in the past.

[44] Years of documented mother/calf pairs in the action area were 1986 (1); 1998 (4); 1994 (4); 1998 (1); 2010 (1); 2011 (5); 2012 (1); 2013 (1).

Climate change may also affect the abundance and distribution of prey species. Currently, the action area is not a prime foraging ground for listed species of whales. While whales forage widely opportunistically, areas with consistently high levels of food visited by a large percentage of the population on a regular basis are considered prime feeding grounds. In the Northeast, primary foraging grounds are located in the Massachusetts Bay (primarily the area of Stellwagen Bank), Cape Cod Bay, the Great South Channel, and other parts of the Gulf of Maine. These areas combine the presence of large amounts of copepods with oceanographic features that concentrate the copepods into patches that are sufficient densities to trigger feeding. The Rhode Island Sound region has not reliably and consistently contained that combination of features to support predictable feeding and therefore, has not been considered a right whale foraging ground. However, conditions in the action area have resulted in periodic, temporary, episodes of prey abundance and thus, concentrations of whale species in the action area than normally expected. For example, in April 1998 and April 2010, high rain fall events resulted in high runoff and nearshore phytoplankton/zooplankton blooms in the action area and thus, increased numbers of foraging right whales in the action area for a period of several weeks (Kenney 2010). As there have only been two times in which such an event has occurred, and there is not enough data to predict a trend, it is difficult to predict if and when the next such event may occur in the action area. Until the frequency of such events increases, enabling us to predict a trend, the 1998 and 2010 events only demonstrate how unforeseen climatic events can influence and affect the distribution and abundance of prey species and the animals that forage upon these species. Thus, over the life of the action (i.e., 29 years), although we cannot discount the possibility that another event such as those that occurred in 1998 or 2010 will occur over the life the project, we cannot with confidence state that the frequencies of such events over the next 29 years will be such that the action area will become an essential foraging ground for listed species of whales. Therefore, until further information and climatic trends can be identified for the action area, it is likely that the action area will remain an area of opportunistic foraging. In our analysis we have considered that whales may be feeding in the action area.

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 and 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 time period considered in this Opinion (i.e., 29 years), sea surface temperatures are expected to rise less than 1.1°C.[45] Warming temperatures would likely result in a shift in the seasonal distribution of sea turtles in the action area, such that sea turtles may begin northward migrations from their southern overwintering grounds earlier in the spring and thus would be present in the action area earlier in the year. Likewise, if water temperatures were warmer in the fall, sea turtles could remain in the action area later in the year. Sea turtles are known to enter the waters of New England when sea surface temperatures are at or above 15°C (Morreale 1999;

[45] See Footnote 31

Morreale 2003; Morreale and Standora 2005; Shoop and Kenney 1992). As increases in sea surface temperatures are expected to be small over the next 29 years (i.e., approximately 1.1°C), it is unlikely that a shift in sea turtle distribution will be seen over the timeframe of the action.

It has also been speculated that the nesting range of some sea turtle species may shift northward with increasing temperature. No nesting has been documented along Rhode Island, or other adjacent New England shorelines (i.e., Massachusetts, Connecticut). 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 between now and 2043 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. 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.

Changes in water temperature may also alter the forage base and thus, foraging behavior of sea turtles. Changes in the foraging behavior of sea turtles in the action area could lead to either an increase or decrease in the number of sea turtles in the action area, depending on whether there was an increase or decrease in the forage base and/or a seasonal shift in water temperature. For example, if there was a decrease in sea grasses in the action area resulting from increased water temperatures or other climate change related factors, it is reasonable to expect that there may be a decrease in the number of foraging green sea turtles in the action area. Likewise, if the prey base for loggerhead, Kemp's ridley or leatherback sea turtles was affected, there may be changes in the abundance and distribution of these species in the action area. 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 changes to the foraging behavior of sea turtles over the next 29 years. If sea turtle distribution shifted along with prey distribution, it is likely that there would be minimal, if any, impact on the availability of food. Similarly, if sea turtles shifted to areas where different forage was available and sea turtles 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 sea turtles shifted to an area or time where insufficient forage was available; however, the likelihood of this happening seems low because sea turtles feed on a wide variety of species and in a wide variety of habitats.

Based on the information presented above, over the 29-year life of the project, it is unlikely that climate change will reach such levels that there will be significant change in the distribution and use of the action area by sea turtles. As a result, it is unlikely, that over the time period considered here, that there will be a significant change in sea turtle numbers and population sizes in the action area as a result of climate change.

5.3.3 Atlantic Sturgeon

Although climate change has the potential to impact Atlantic sturgeon in various ways (see section 5.2.6), due to the location of the action area (i.e., Sound; coastal, offshore waters), the most likely effect to Atlantic sturgeon in the action area from climate change would be if warming temperatures led to changes in their range and migratory patterns. Warming temperatures predicted to occur over the next 100 years could likely result in a northward shift/extension of their range while truncating the southern distribution, thus effecting the recruitment and distribution of sturgeon rangewide. However, over the life of the action (i.e., to 2043), this increase in sea surface temperature would be minimal (i.e., approximately 1.1^{o}C) and thus, it is unlikely that a potential shift in range will be observed over the next 29 years. 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 Atlantic sturgeon or a significant modification to the number of sturgeon likely to be present in the action area over the life of the action.

Although the action area is not a spawning ground for Atlantic sturgeon, sturgeon are likely to migrate through the action area to reach the natal rivers located in the northern part of their range (i.e., Hudson River, Kennebec River, Penobscot (possibly), and Androscoggin River) to spawn. Elevated temperatures could modify cues for spawning migration, resulting in an earlier spawning season, and thus, altering the time of year sturgeon may be present within the action area. This may cause a change in the timing in the number of sturgeon present in the action area. 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 alone will affect the seasonal movements of sturgeon through the action area.

In addition, changes in water temperature may also alter the forage base and thus, foraging behavior of Atlantic sturgeon. Any forage species that are temperature dependent may also shift in distribution as water temperatures warm and thus, potentially cause a shift in the distribution of Atlantic sturgeon. 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.

5.3.4 Summary of Climate Change

As discussed above, we considered the potential impact of climate change on listed species in the action area. Available information would indicate that temperatures in the action area may increase up to 1.1°C over the life of this proposed action. This may result in some minor changes in distribution of listed species in the action area. It is important to note, however, that the effects of the project are largely concentrated in the first three years during construction. No detectable changes in distribution, abundance or behavior of listed species are anticipated as a result of climate change in that timeframe. In our analysis we considered that listed species may be present in the action area and may be conducting a variety of behaviors and this broad analysis encompasses any anticipated changes as a result of climate change.

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

We have 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 adverse impacts of the action on listed species.

6.1.1 Authorization of Fisheries through Fishery Management Plans

We have authorized 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. While the action area is mostly in State waters, it includes some Federal waters. Fishermen who fish in State waters, but also have Federal permits, are required to follow the Federal rules pertaining to the fishery if those Federal rules are more restrictive than State rules. Commercial and recreational fisheries in the action area employ gear that is known to 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 and the Atlantic sea scallop FMP fisheries and we completed one Biological Opinion ("Batch Fishery BiOp") considering effects of the following seven FMPs: Atlantic bluefish, Atlantic mackerel/squid/ butterfish, monkfish, northeast multispecies, spiny dogfish, summer flounder/scup/black sea bass fisheries. These consultations have considered effects to loggerhead, green, Kemp's ridley and leatherback sea turtles, Atlantic sturgeon as well as ESA-listed whales. 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 Atlantic sturgeon, sea turtle or whale species or DPS. Each Opinion included an incidental take statement (ITS) exempting a certain amount of lethal and/or non-lethal take (i.e., capture and/or injury) of Atlantic sturgeon and/or sea turtles resulting from interactions with the fishery. These ITSs are summarized for the American lobster and Atlantic Sea Scallop FMP fisheries in the table below (Table 8). The ITS for the "batch" Opinion, is discussed below. In each Opinion, we concluded that the potential for interactions between listed species and fishing vessels (*i.e.*, vessel strikes) was extremely low. In all of these consultations we have also concluded that any effects to 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.

Table 8: Information on Fisheries Opinions conducted by NMFS NERO for American lobster and Atlantic Sea Scallop fisheries that operate in the action area

FMP	**Date of Most Recent Opinion**	**Atlantic Sturgeon(All DPS)**	**Loggerhead (NWA DPS)**	**Kemp's ridley**	**Green**	**Leather back**
American lobster	August 3, 2012	0	1	0	0	5
Atlantic sea scallop	July 12, 2012	1 lethal from any of the 5 DPS over a 20 year period	2013 and beyond: 301 (115 lethal)	3	2	2

On December 16, 2013, NMFS completed a formal Biological Opinion on seven FMPs (Batch Fishery BiOp) managing the Northeast Multispecies, Monkfish, Spiny Dogfish, Atlantic Bluefish, Northeast Skate Complex, Mackerel/Squid/Butterfish, and Summer Flounder/Scup/Black Sea Bass fisheries. In the Batch Fishery BiOp NMFS determined that the fisheries may adversely affect, but are not likely to jeopardize, the continued existence of North Atlantic right whales, humpback whales, fin whales, and sei whales, or loggerhead (specifically, the NWA DPS), leatherback, Kemp's ridley, and green sea turtles, any of the five DPSs of Atlantic sturgeon, or GOM DPS Atlantic salmon. The ITS exempts the incidental take of ESA-listed species as follows:

- for loggerhead sea turtles from the NWA DPS, NMFS anticipates (a) the annual take of up to 269 individuals over a five-year average in gillnet gear, of which up to 167 per year may be lethal; (b) the annual take of up to 213 individuals over a four-year average in bottom trawl gear, of which up to 71 per year may be lethal; and (c) the annual take of up to one individual in trap/pot gear, which may be lethal or non-lethal;
- for leatherback sea turtles, NMFS anticipates (a) the annual take of up to four individuals in gillnet gear, of which up to three per year may be lethal; (b) the annual take of up to four individuals in bottom trawl gear, of which up to two per year may be lethal; and (c) the annual take of up to four individuals in trap/pot gear, which may be lethal or non-lethal;
- for Kemp's ridley sea turtles, NMFS anticipates the annual take of up to three individuals in gillnet gear, of which up to two per year may be lethal, and the annual take of up to three individuals in bottom trawl gear, of which up to two per year may be lethal;
- for green sea turtles, NMFS anticipates the annual take of up to four individuals in gillnet gear, of which up to three per year may be lethal, and the annual take of up to three individuals in bottom trawl gear, of which up to two per year may be lethal;
- for Atlantic sturgeon from the GOM DPS, NMFS anticipates (a) the annual take of up to 137 individuals over a five-year average in gillnet gear, of which up to 17 adult equivalents per year may be lethal; (b) the annual take of up to 148 individuals over a five-year average in bottom trawl gear, of which up to 5 adult equivalents per year may be lethal;

- for Atlantic sturgeon from the NYB DPS, NMFS anticipates (a) the annual take of up to 632 individuals over a five-year average in gillnet gear, of which up to 79 adult equivalents per year may be lethal; (b) the annual take of up to 685 individuals over a five-year average in bottom trawl gear, of which up to 21 adult equivalents per year may be lethal;
- for Atlantic sturgeon from the CB DPS, NMFS anticipates (a) the annual take of up to 162 individuals over a five-year average in gillnet gear, of which up to 21 adult equivalents per year may be lethal; (b) the annual take of up to 175 individuals over a five-year average in bottom trawl gear, of which up to 6 adult equivalents per year may be lethal;
- for Atlantic sturgeon from the Carolina DPS, NMFS anticipates (a) the annual take of up to 25 individuals over a five-year average in gillnet gear, of which up to four adult equivalents per year may be lethal; (b) the annual take of up to 27 individuals over a five-year average in bottom trawl gear, of which up to one adult equivalent per year may be lethal;
- for Atlantic sturgeon from the SA DPS, NMFS anticipates (a) the annual take of up to 273 individuals over a five-year average in gillnet gear, of which up to 34 adult equivalents per year may be lethal; (b) the annual take of up to 296 individuals over a five-year average in bottom trawl gear, of which up to 9 adult equivalents per year may be lethal;
- The annual take of up to five individuals from the GOM DPS of Atlantic salmon over a five-year average in gillnet gear, of which up to two takes may be lethal;
- The annual take of up to five individuals from the GOM DPS of Atlantic salmon over a five-year average in bottom trawl gear, of which up to three takes may be lethal.

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 whales, 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 whales, 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 Non-Federally Regulated Fishery Operations

State fisheries operate in the state waters of Rhode Island. Very little is known about the level of interactions with listed species in fisheries that operate strictly in state waters. Impacts on Atlantic sturgeon and sea turtles from state fisheries may be greater than those from Federal activities in certain areas due to the distribution of these species in these waters. Depending on the fishery in question, however, many state permit holders also hold Federal licenses; therefore, section 7 consultations on Federal actions in those fisheries address some state-water activity. Impacts of state fisheries on endangered whales are addressed as appropriate through the MMPA take reduction planning process. We are actively participating in a cooperative effort with the Atlantic States Marine Fisheries Commission (ASMFC) and member states to standardize and/or implement programs to collect information on level of effort and bycatch of protected species in state fisheries. When this information becomes available, it can be used to refine take reduction plan measures in state waters.

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^{O}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 Conservation Management Areas (FLCMAs) designated in the Federal lobster regulations (the action area is considered to be located in FLCMA 2). 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 exert 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 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: completed July 2013; and
- Develop and publish final rule to implement risk reduction from vertical lines. Time frame: by Mid-2014.

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.

We, in consultation with the ALWTRT, have 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 whales can happen anywhere along the U.S. eastern seaboard, including the action area. In response to this fact, we created the Whale Disentanglement Network. The Network is managed by us, 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 us 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 we issued a permit to PCCS to disentangle large whales. Additionally, we 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 us 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. We have established agreements with many coastal states to collaboratively monitor and respond to entangled whales. As a result of the success of the disentanglement network, we believe 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 below.

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/Federal 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 bothr egulatory 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 ships trikes, 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, including the action area, 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. DMAs can be designated anywhere along the U.S. eastern seaboard, including the action area, 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. When DMAs are designated, 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 10knots or less. Compliance with these zones is voluntary.

On December 9, 2013, NMFS issued a final rule to eliminate the expiration date (or "sunset clause") contained in regulations requiring vessel speed restrictions to reduce the likelihood of lethal vessel collisions with North Atlantic right whales (78 FR73726).

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 along the eastern seaboard of the U.S. from Florida to Maine (action area within Northeast aerial survey track; http://whale.wheelock.edu/whalenet-stuff/reportsRW_NE/, last visited December 17, 2013). 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. 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. As noted above, and SMA has been designated within the action area (Block Island Sound SMA) from November 1 through April 30 of any year. As such, SAS will assist mariners transiting the action area, specifically during this time frame.

Marine Mammal Health and Stranding Response Program (MMHSRP)
Marine mammals can strand anywhere along the eastern seaboard of the U.S. In response to this fact, 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 to help 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 issued under the authority of 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. A complete listing of fishery regulations, including those fisheries in the action area, can be found at : 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. 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.

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, which can occur anywhere along these coastlines, including the action area. 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. 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. In the 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.

7.0 EFFECTS OF THE ACTION

This section of the Opinion assesses the direct and indirect effects of the proposed construction, operation, maintenance, and decommissioning of the BIWF and BITS on threatened and endangered species or critical habitat, together with the effects of other activities that are interrelated or interdependent. Indirect effects are those that are caused later in time, but are still reasonably certain to occur. This Opinion examines the likely effects of the proposed actions on ESA listed species of whales, sea turtles, and Atlantic sturgeon (all five DPSs) and their habitat in the action area within the context of the species' current and projected status, the environmental baseline and cumulative effects. Because there is no critical habitat in the action area, none will be affected. We have not identified any interrelated or interdependent activities.

A brief summary of information related to sea turtle, Atlantic sturgeon, and whale presence in the action area is as follows:

- Loggerhead, Kemp's ridley, leatherback, and green sea turtles are likely to be present in the action area when water temperatures are at least 15^{o}C, which typically coincides with the months of June through October, although, some may remain through the first week of November (Morreale 1999; Morreale 2003; Morreale and Standora 2005; Shoop and Kenney 1992). The action area is not a concentration area for sea turtles but sea turtles are routinely documented in the waters of Rhode Island Sound and Block Island Sound (OBIS SEAMAP online database mapper, accessed on November 25, 2013). Sea turtles in these waters are likely to be found swimming through the action area as they complete northward migrations in the spring and southward migrations in the fall. Sea turtles may also be found transiting the action area while moving into or out of nearby foraging areas (i.e., Cape Cod Bay (Lazell 1980) or Long Island Sound (Burke *et al.* 1991, 1994; Morreale and Standora 1998)).

- We expect Atlantic sturgeon to be in the action area from June 1 through the first week of November (Savoy and Pacileo 2003). The action area is not a known area for Atlantic sturgeon to forage, spawn, or overwinter, and thus, concentrate. The action area is likely to be used as a migratory route to reach foraging, overwintering, and/or spawning grounds located along other portions of the Eastern Seaboard of the United States.

- Large whales are seasonally present in the action area. Most whales in this area are making seasonal northward (to foraging grounds) or southward (to calving grounds) migratory movements. Humpback whales are primarily found in the action area during the spring, summer and fall, while fin whales may be present year round. Right whales have been observed in these waters during all seasons of the year, with most sightings in the spring and fall (i.e., November 1 through April 30). Feeding by right whales is occasionally observed in the Rhode Island region, but is likely an opportunistic response to relatively rare occurrences of appropriate prey patches resulting from natural events (i.e., high rain fall events resulting in high runoff and nearshore phytoplankton/zooplankton blooms; April 1998 and April 2010 event). Outside of these unpredictable natural events, the action area is not area considered to be a foraging ground for right whales. Right whale foraging grounds are located in Cape Cod Bay, Great South Channel, and other parts of the Gulf of Maine. These areas combine the presence of large amounts of copepods with oceanographic features that concentrate the copepods into patches that are of sufficient density to trigger feeding. The Rhode Island Sound region has not reliably and consistently contained that combination of features to support predictable feeding and therefore, has not been considered a right whale foraging ground (Pace and Merrick 2008).

The sections below will outline potential direct and indirect effects to ESA listed whales, sea turtles, and Atlantic sturgeon from the proposed action. The analysis is organized by the three following categories of impacts (1) forage and habitat modifications; (2) water quality; and (3) acoustic impacts.

7.1 Construction of the BIWF and BITS

Land-Based Activities

Portions of the project will occur on land or on the beach, where ESA listed species under our jurisdiction do not occur. Components of the onshore construtional phase of cable installation (e.g., terrestrial cable lay operation, excavation of shoreline trenches) will occur above the mean high water mark, or on the portions of the beach between the MLW and MHW. In addition, construction activities at the Block Island (BITS and BIWF) and Dillon's Corner Switchyards or Substations will not expose any listed species or their prey to any effects, as all work will occur on land. This onshore work is not expected to affected coastal waters where ESA listed whales, sea turtle or sturgeon occur. As a result, no listed species will be exposed to any effects of activities that occur on land or above the high water mark of the beach. Because listed species under our jurisdiction only occur in the water, the remainder of this Opinion will only consider effects from in-water activities. This includes in-water jet plowing operations and the installation of an offshore cofferdam to assist in BITS cable landing.

Water-Based Activities

The major constructional aspects of the BIWF and BITS will involve cable lay operations/installation (i.e., BITS and BIWF's inner-array and export cables) and the installation of WTG and their foundations. The construction of the BIWF and the installation of the BIWF's export, inter-array and BITS cables, via jet plowing, have the potential to affect ESA listed species of sea turtles, Atlantic sturgeon, and/or whales via:

- changes to habitat and thus, potential prey availability;
- changes in water quality, including total suspended solid concentrations (TSS) from cable-lay operations and WTG installation;
- exposure to increased underwater noise resulting from pile installation (WTG foundations, cofferdam installation) and DP thruster use (cable lay operations); and
- vessel and/or equipment interactions throughout all constructional aspects of the action.

7.1.1 Forage and Habitat Modification

7.1.1.1 BITS, Export, and Interarry Cable Installation:Impacts to Habitat

During the installation of the BITS, export, and inter-array cables, the benthic habitat and its associated benthic community along the cable route will be affected, both directly and indirectly.

As a result of exposure to the high pressure water jets, surface dwelling (e.g., species of amphipods and bivalves) and infaunal (e.g. species of polychaetes) organisms within the pathway of the plow will be removed, displaced, and/or killed during the trenching process. Additionally, as the jet plow moves along the benthos, any infaunal or surface dwelling organisms located in the path of the jet plow's skids or wheels that span the trench are expected to be crushed. Any infaunal or surface dwelling organisms located within or near jet plow operations may also be buried by the redeposition of sediment on either side of the trench.

Based on sediment transport modeling done for cable installation operations, sediment redeposition is not expected to exceed 1 millimeter (mm) at a distance of 130 feet, 250 feet, and 330 feet from either side of the trench centerline along the inter-array, export, and BITS cable routes, respectively (RPS ASA 2012). Although studies have indicated that many types of benthic fauna (e.g., polychaetes, clams, and amphipods) particularly those that inhabit highly dynamic ecosystems, such as Rhode Island Sound, are able to withstand burial under 3-inches of sediment, some mortality to benthic faunal species is possible, particularly earlier life stages of those species (CRMC 2010; Maurer *et al*. 1986). Cable lay operations will result in the temporary disturbance and loss of benthic resources along the cable routes in Rhode Island Sound (i.e., approximately 3.67 acres, 11.27 acres, and 39.64 acres of benthos will be disturbed via jet plow operations along the inter-array cable, export cable, and BITS cable routes, respectively).

Installation of the sheet piles off Scarborough Beach will disturb and displace benthic infaunal and surface dwelling organisms. As described in section 3.2.2, once the cofferdam has been installed, the area inside the cofferdam will be excavated in preparation for the cable to be pulled ashore. Excavation will result in the removal of surface dwelling and infaunal organisms located inside the confines of the cofferdams, resulting in the temporary loss of benthic resources from approximately 0.2 acres of Rhode Island Sound.

The placement of concrete mats or rock piles will result in the permanent conversion of soft substrate to hard substrate along the cable routes, and will modify the benthic community in these areas from primarily infaunal organisms (e.g., amphipods, polychaetes, bivalves), to sessile or highly mobile organisms (e.g., sponges, hydroids, crustaceans)). It is estimated that no more than one percent of the entire length of each submarine cable will require concrete matting or rock pile placement.[46] This translates into approximately 0.10 acres, 0.29 acres, and up to 1.21 acres of habitat along the inter-array cable, export cable, and BITS cable, respectively, being converted permanently from soft substrate to hard substrate. In order to install concrete matting or rock piles along these sections of the cable, an 8-point anchored barge will be used. Placement of the anchor will crush any benthic organisms beneath the anchor. While installation activities are underway he associated anchor chain , will rest or sweep across the sea floor, resulting in the disturbance of the top few inches of benthos and to the organisms residing in these areas. The anchor and anchor chains will temporarily disturb approximately 0.12 acres of benthic habitat per anchoring event.

[46] Submarine portions of the: Inter-array cable=2.1 miles long; Export cable= 6.2 miles long; and BITS=19.8 miles long.

7.1.1.2 WTG Foundation Construction and Installation: Impacts to Habitat

Preparation for, and construction of, the BIWF will involve multiple activities that will impact the benthic habitat within and near the BIWF. Prior to actual WTG construction, components of the WTG will be transported to the offshore WTG installation site via a jack-up transportation barge, which, once on location will be secured by inserting spuds within the benthos. Once installation of WTGs is ready to begin, offshore installation of the each WTG jacket foundation, will be carried out from derrick barges moored to the seafloor by an 8-point mooring system consisting of 10-ton anchors with a maximum penetration depth of 10 meters. As described in Section 3.1.2.3, each of the four through- the- leg- foundation piles (each approximately 42" to 52" in diameter) will be installed via an impact hammer. Following the construction of each WTG foundation, installation of the actual WTG will begin and will be completed from a jack-up transportation barge. In total, the WTG construction and operational footprint will affect 28.9 acres of benthic habitat in Rhode Island Sound (this number also takes into consideration the anchors and anchor chains of barges associated with the construction of the WTGs). Of this 28.9 acres, the WTG foundations alone (not considering the presence of barges) will permanently impact 0.07 acres of benthic habitat per WTG (this includes the placement of sand/cement bag as protective armoring at the base of each foundation as noted above) or 0.35 acres of benthic habitat for all 5 WTG foundations.

In those areas of the WTG foundation where sand/concrete bags will be placed for additional inter-array/J-tube cable protection, infaunal or surface dwelling organisms will be removed and displaced during the installation of these structures; the placement of these armoring devices will also result in the permanent conversion of soft substrate to hard substrate. The benthic community at these sites will be modified (i.e., from primarily infaunal and surface mobile organisms (e.g., amphipods, polychaetes, bivalves), to sessile or highly mobile organisms (e.g., sponges, hydroids, crustaceans)). In addition, in order to install the WTG foundation and WTG's themselves, 8-point anchored derrick barges, as well as jack-up barges will be needed to support these activities. As a result, placement of the anchor/spud piles will crush any sessile organisms beneath the anchor/spud pile, and the associated anchor chain, while installation activities are underway, will rest or sweep across the sea floor, resulting in the disturbance of the top few inches of benthos and to the organisms residing in these areas. During pile driving operations, piles being driven will crush any sessile organisms in the footprint of the pile.

7.1.1.3 BITS and BIWF Habitat Modification: Effects to ESA Listed Species

The activities associated with installation of the BITS and the construction of the BIWF have the potential to impact some NMFS ESA listed species in the action area by reducing prey species through the alteration and/or loss of the existing biotic assemblages. As listed species of whales and leatherback sea turtles forage upon pelagic prey items (e.g., whales: krill, copepods, sand lance; leatherbacks: jellyfish), cable lay operations (i.e., jet plowing, installation and excavation of cofferdam for cable landing) and their associated impacts on the benthic environment are not expected to have any direct or indirect effects on whale and leatherback sea turtle foraging items or the foraging ability of these species. Green sea turtles feed almost exclusively on sea grasses. All cable routeshave been adjusted to avoid any seagrass beds present in Block or Rhode Island Sound; therefore, we do not anticipate any impacts to foraging green sea turtles or their prey base. [47] The remainder of this section will discuss the effects of cable lay and pile installation operations on loggerhead and Kemp's Ridley sea turtles and Atlantic sturgeon forage and foraging habitat.

Atlantic Sturgeon: **Forage Potential in the Action Area**

Atlantic sturgeon aggregate in several distinct areas along the Mid-Atlantic coastline; Atlantic sturgeon are most likely to occur in areas adjacent to estuaries and/or coastal features formed by bay mouths and inlets (Stein *et al.* 2004a; Laney *et. al* 2007; Erickson *et al*. 2011; Dunton *et al*. 2010). These aggregation areas are located within the coastal waters off North Carolina; waters between the Chesapeake Bay and Delaware Bay; the New Jersey Coast; and the southwest shores of Long Island (Laney *et. al* 2007; Erickson et al. 2011; Dunton *et al*. 2010). Based on five fishery-independent surveys, Dunton *et al*. (2010) identified several "hotspots" for Atlantic sturgeon captures, including an area off Sandy Hook, New Jersey, and off Rockaway, New York. These "hotspots" are aggregation areas that are most often used during the spring, summer, and fall months (Erickson *et al*. 2011; Dunton *et al*. 2010). These aggregation areas are believed to be where Atlantic sturgeon overwinter and/or forage (Laney *et. al* 2007; Erickson *et al*. 2011; Dunton *et al*. 2010). Areas between these sites serve as migration corridors to and from these areas, as well as to spawning grounds found within natal rivers.

The action area is over 100 nautical miles away from the nearest identified aggregation area (i.e., nearshore waters off Rockaway, New York, or Sandy Hook, New Jersey). Based on the distribution and location of known aggregation areas, as well as available information on the benthic habitat within the action area, it is extremely unlikely that the areas where sediment disturbing activities will occur are used for overwintering and/or foraging aggregations. While opportunistic foraging may occur within the action area, it is more likely that the action area is used by migrating individuals as they move from foraging, overwintering, and spawning grounds located in coastal waters of the Eastern Seaboard. If any foraging does occur in the action area, Atlantic sturgeon would feed on benthic invertebrates (e.g., amphipods, polychaetes, decapods) and occasionally on small fish such as sand lance (Savoy 2007).

Sea Turtles: Forage Potential in the Action Area

Satellite tracking studies of loggerhead, and Kemp's ridley sea turtles in coastal New York waters found that foraging turtles mainly occurred in areas where the water depth was between approximately 16 and 49 feet (Morreale and Standora 1990; Ruben and Morreale 1999). This depth was interpreted not to be as much an upper physiological depth limit for turtles, as a natural limiting depth where light and food are most suitable for foraging turtles (Morreale and Standora 1990). The depths where cable installation and WTG foundations will be installed range from 0 to 50 feet, and thus, overlap with the depths preferred by sea turtles, suggesting that if suitable foraging items were present (i.e., crabs and mollusks; Morreale and Standora 1992; Bjorndal 1997), loggerhead and Kemp's ridley sea turtles may be foraging along portions of the BITS or export cable route. Although surveys conducted in the action area indicate that sea turtle

[47] Eelgrass surveys were conducted in August 2010 at the landfall locations for the BITS and BIWF export cables near Block Island, and the BITS cable off of Scarborough State Beach. No eelgrass was identified at the BITS cable landfall location off Scarborough State Beach. An existing eelgrass bed was confirmed along the southern margin of Block Island. To avoid impacts, Deepwater Wind adjusted the proposed landing location for the BIWF Export Cable and BITS cable to a location approximately 2,000 feet north of this confirmed bed.

foraging items exist in the action area, the action area is not known to be an area where sea turtles concentrate to forage; therefore, foraging in the action area is expected to be limited to opportunistic events by individuals.

BITS and BIWF Alteration of Foraging Habitat: Overall Impacts to Atlantic Sturgeon and Sea Turtles

Based on the information above, the alteration of benthic habitat and the loss of benthic resources during the construction/installation of the cable routes and WTGs may affect loggerhead and Kemp's ridley sea turtles and Atlantic sturgeon due to the loss of potential forage. The total combined area of impacts associated with the installation of the BITS and BIWF is approximately 85.27 acres of Rhode Island Sound (e.g., takes into consideration all areas impacted by cable installation routes, WTGs foundation sites, and barge anchors, anchor sweeps, and scour matts/rock piles associated with the BITS and BIWF), with 2.07 acres of this associated with permanent impacts to the benthos (i.e., those regions converted from soft to hard substrate). As Rhode Island Sound is approximately 617,763 acres, the proposed action will only temporarily affect 0.013% of the available habitat in Rhode Island Sound and permanently affect 0.0003% of the available habitat in Rhode Island Sound. As such, while there is likely to be some loss of forage items for sea turtles or sturgeon, based on the above information, the amount of habitat affected by the proposed action represents a very small percentage of the potential foraging habitat in Rhode Island Sound and, thus, is likely to have an insignificant effect on the foraging ability of sea turtles and sturgeon. In addition, as suitable foraging items will continue to be available throughout other regions of Rhode Island Sound, as well as within adjacent waters off New England, and the proposed action will not alter the habitat in any way that prevents Atlantic sturgeon or sea turtles from using the action area as a migratory pathway to reach those areas that are undisturbed, we do not expect the foraging ability of sea turtles or Atlantic sturgeon to be significantly impaired as a result of the proposed action.

Although we are assuming that sea turtles or Atlantic sturgeon will temporarily shift their foraging efforts to other undisturbed foraging areas of Rhode Island Sound, this movement to other undisturbed areas is likely to be temporary, and is not likely to significantly affect the behavior or ability of sea turtles or sturgeon to find adequate nourishment. However, in those ecosystems that are highly dynamic (e.g., have strong bottom currents that continually move surface sediments around), the benthic organisms that comprise these ecosystems are adapted to frequent disturbances and it estimated that in these communities, where substrate composition is primarily sand, complete recolonization of the benthos following a major disturbance can occur within 2 to 3 years following a disturbance. As the action area is such an ecosystem, it is believed that once the construction/installation of the BITS (including removal of the cofferdam and placement of excavated material back into excavated area) and BIWF has been completed, the benthic community will completely re-establish itself within 3 years.

The placement of concrete matting and/or rock piles over sections of the BITS or export cable or around the WTG foundations will result in the permanent conversion of the benthos within these sections from soft substrate habitat to hard substrate habitat. This conversion may have a beneficial effect on Atlantic sturgeon or sea turtles by causing an increase in available prey items, and potentially, the availability of preferred prey items previously not found within these sites. On a small scale, it has been found that larger diameter stone used for rip-rap or fill is correlated with an increase in invertebrate taxa found within the area of stone placement and that riprap areas have an increase in species richness and density when compared to natural banks or sand-bed systems (Shields *et al.* 1995), as these areas create new microhabitats and large annual spaces previously not available. We can assume that something similar to this is likely to occur in the offshore waters of the project site. As hard substrate areas already exist within other portions of the project area (i.e., near cable landfall locations and in the southwest portion of the BIWF), it is likely that recruitment of organisms from these areas will occur on these newly established hard substrate areas of Rhode Island Sound and thus, over the period of benthic recovery (i.e., up to 3 years), not only will the species associated with the soft bottom substrates reestablish themselves, but additional species, associated with hard bottom substrates, will also become newly established in areas of the project they were previously not found. As a result, species abundance and diversity may increase following the recovery of the ecosystem and thus, may afford additional foraging opportunities for sea turtles and Atlantic sturgeon.

We anticipate that while activities associated with the construction and installation of the BITS and the BIWF may temporarily disrupt normal feeding behaviors for sea turtles and Atlantic sturgeon, the action is not likely to remove critical amounts of prey resources from the action area and any disruption to normal foraging or migration is likely to be insignificant. In addition, the installation of the BITS and BIWF, as well as the activities associated with the construction of each, are not likely to alter the habitat in any way that prevents sea turtles or Atlantic sturgeon from using the action area as a migratory pathway to other near-by areas that may be more suitable for foraging. Therefore, effects to sea turtle or Atlantic sturgeon foraging and migration as a result of the construction and installation of the BITS and BIW are insignificant.

7.1.2 Water Quality: Turbidity and Release of Sediment Contaminants

7.1.2.1 Turbidity

Increased turbidity and resuspension of sediments can be expected from the following activities:

- BITS and BIWF export and inter-array cable installation (Jet plowing; offshore and nearshore/landfall operations (e.g., cofferdam installation/excavation));
- WTG foundation installation (Pile driving);
- Vessel anchoring (anchor placement and chain sweep); and
- Placement of scour protection (along cable routes and WTG foundations).

Of these activities, the installation of the BITS and BIWF export and inter-array cables are expected to generate the most turbidity and disturbance to the bottom sediments. Simulations of sediment transport and deposition from jet plow embedment of the BIWF export and inter-array cables and the BITS were performed and reported in the ER.[48] Results of the sediment transport modeling demonstrate that suspended sediment levels during inter-array cable installation will: not exceed 100 mg/L; decrease to 10 mg/L or less within an hour; and, be confined to an area within 160 feet of the jet plow /trench. This is also expected to be true during the installation of the offshore portion of the export cable, except in regions where there is significant quantities of

[48] The sediment transport and deposition simulations used two models: HYDROMAP to calculate currents and SSFATE to calculate suspended sediments in the water column and bottom deposition that could result from jet plow operations.

silt and clay. In these areas, total suspended sediment (TSS) concentrations are expected to be higher, ranging from 200 mg/L to 500 mg/L at distances of approximately 650 feet and 260 feet, respectively, from the trench/jet plow, and decreasing to 10 mg/L within approximately 3,200 feet from the jet plow. Within these regions of silt and clay, TSS levels greater than 200 mg/l will persist for no more than 10 minutes, while levels reaching 10 mg/l will persist in the area for approximately 2 hours. For the BITS, results of the sediment transport modeling indicate that during the installation of the offshore portion of the BITS cable, TSS concentrations will vary along the cable route due to differences in current and sediment type along particular sections of the cable route. Regardless of this variability, concentrations along the offshore portion of the route will not exceed 200 mg/L to 500 mg/L beyond 1,100 feet and 1,000 feet, respectively from the jet plow/trench, and typically will dissipate to less than 10 mg/L within 12 hours.

Sediment transport models were also completed for the installation of the BITS and export cable installation within the nearshore/tidal zone of BlockIsland's Crescent Beach and/or Rhode Island's Scarborough Beach. In this area, the models demonstrate that elevated levels of suspended sediment and sediment accumulation will be confined to the area near the jet plow/track and that concentrations of suspended sediment of 100 mg/L will cover an area of 7.1 acres for approximately 10 minutes, cumulatively, during jet plowing in the nearshore/tidal zone. Model results for the area off Scarborough Beach, Rhode Island, indicate that excess water column concentrations and sediment accumulation in the nearshore/tidal zone will be similar to those described above for the offshore phase of BITS cable installation and will be confined to the area near the jet plow/track in the nearshore/tidal zone. Concentrations of 100 mg/L will be confined to an area within 196 feet from the jet plow, while concentrations of 10 mg/L will extend further, up to approximately 410 feet, on average, but up to 755 feet where the currents are closer to shore and do not mix greatly with the surrounding waters. It is estimated that concentrations of suspended sediment of 100 mg/L will cover an area of 10 acres for approximately 10 minutes, while concentrations of 10 mg/l will cover an area of 35 acres for approximately 35 minutes during BITS installation in the nearshore/tidal zone off Scarborough Beach.

Other activities associated with the construction and installation of the BITS and BIWF (i.e., WTG foundation installation, cofferdam installation/excavation, vessel anchoring, placement of scour protection) will also disturb offshore bottom sediments. However, suspended sediment levels produced by these activities are expected to be minor in comparison to cable-lay operations (i.e., jetting operations), or in the case of cofferdam installation/excavation, are expected to be non-detectable as a silt curtain will be in place throughout cofferdam installation and excavation. Available information indicates that pile driving activities (including removal of piles) will produced turbidity levels less than 50 mg/l (NMFS 2013), with concentrations of TSS reaching levels of approximately 5 to 10 mg/L above background levels (i.e., 1 mg/L to 2 mg/L) within a few hundred feet of the pile being driven (NMFS 2013; FHWA 2011b). Anchoring activities and placement of scour protection along sections of the cable routes and/or the WTG foundation are expected to produce similar to lower levels of turbidity in the project area as pile driving.

No information is available on the effects of (TSS) on juvenile and adult sea turtles or whales; however, studies of the effects of turbid waters on fish suggest that concentrations of suspended solids can reach thousands of milligrams per liter before an acute toxic reaction is expected (Burton 1993). TSS is most likely to affect sea turtles or whales if a plume causes a barrier to normal behaviors or if sediment settles on the bottom affecting sea turtle or sturgeon prey. As Atlantic sturgeon, sea turtles and whales are highly mobile they are likely to be able to avoid any sediment plume by modifying their movements around the area experiencing turbidity. While the increase in suspended sediments may cause Atlantic sturgeon, sea turtles or whales to alter their normal movements, any change in behavior is likely to be insignificant as it will only involve minor, temporary movements to alter their course out of the sediment plume. Based on this information, any increase in suspended sediment is not likely to affect the movement of Atlantic sturgeon, sea turtles or whales between foraging areas or while migrating or otherwise negatively affect listed species in the action area. Additionally, the TSS levels expected from the construction of the BITS and BIWF (see above) are below those shown to have an adverse effect on fish (580.0 mg/L for the most sensitive species, with 1,000.0 mg/L more typical (Breitburg 1988 in Burton 1993; Summerfelt and Moiser 1976 and Combs 1979 in Burton 1993)) and benthic communities (390.0 mg/L (EPA 1986)); therefore, effects to sturgeon and/or benthic resources that sturgeon or sea turtles may eat are unlikely. Based on this information, and the fact that any suspended sediment will be temporary and of relatively short duration, it is likely that the effect of the suspension of sediment resulting from BITS and BIWF export and inter-array cable installation, WTG foundation installation, anchoring operations, and placement of scour protection, on sea turtles, Atlantic sturgeon, or whales will be insignificant.

7.1.2.2 Sediment Contaminant Release

AECOM Marine and Coastal Center (AECOM), and Ocean Surveys, Inc., performed an environmental sediment survey and analysis for Deepwater Wind (AECOM 2012, ER 2012). Sediment cores were taken from the proposed trench areas and along the BITS and BIWF cable routes in order to obtain an overall representation of sediment type and quality (i.e., presence of contaminants or heavy metals) within regions of Rhode Island Sound affected by the proposed BIWF and BITS. Results of the sediment analysis showed that all chemical parameters (i.e., total organic carbon, metals, and polynuclear aromatic hydrocarbon compounds) were below the Coastal Resources Management Council (CRMC) dredged material suitability limits for subaqueous Confined Aquatic Disposal (CAD) capping purposes, as well as below Rhode Island Department of Environmental Management water quality criteria for toxic pollutants, and were below Effects Range Low (ERL) concentrations, which are biological effects-based sediment quality guidelines.[49] Based on these results, it was determined that benthic sediments were non-toxic (i.e., non-detectable to extremely low levels of contaminants) and there will be no release of any contaminants within the water column throughout the construction or operation of the BITS or BIWF. These results are consistent with the sediment survey results, which showed that over 90% of the sampled areas in Rhode Island Sound are comprised of sand, a sediment grain

[49] In environmental toxicology, effects range low (ERL) is a specific chemical concentrations that is derived from compiled biological toxicity assays and sampling of marine sediment. These numerical values are sediment quality guidelines that were developed by Long and Morgan (1990) for the National Oceanic and Atmospheric Administration's National Status & Trends program as informal tools in screening sediment for trace metals and organic contaminants. Concentrations below the ERL value represent a range in which effects to aquatic species will be minimal or in other words, a range in which effects would be rarely observed (O'Connor 2008). Only concentrations equal to and above the ERL, represent a range in which effects to aquatic species are possible ((O'Connor 2008).

size known to contain extremely low to no contaminant concentration. This is because dissolved heavy metals and contaminants primarily bind to/associate with small grain sized sediment (i.e., silts and clays) and remain strongly bound to these sediments (Wilson *et al*. 2007; Engstrom 2004). As a result, the chemical/contaminant(s) concentration in fine grained sediments will be greater than those found in coarse grained sediment, such as the sands found in Rhode Island Sound (O'Connor 2008).

Whales, Atlantic sturgeon, and sea turtle's exposure to contaminants within their environment occurs almost exclusively through their food sources, with contaminants bioaccumulating in their systems via a process of biomagnification. Based on the above information, the temporary and localized disturbance of these sediments during the proposed action's construction activities are not anticipated to result in increased contaminants in lower trophic levels. Therefore, sea turtles, Atlantic sturgeon, and whales are not likely to experience increased bioaccumulation of chemical contaminants in their tissues from the consumption of prey items in the vicinity of the proposed action. Any effects to whales, Atlantic sturgeon or sea turtles from the disturbance of these sediments will be discountable. Since other sources of turbidity and seafloor disturbance (i.e., cable installation, pile installation, cofferdam installation, and scour protection placement) will be minimal compared to that caused by cable installation, the overall effect of project construction on listed species due to turbidity and exposure to contaminants is insignificant or discountable.

7.1.3 Acoustic Impacts

Sources of noise associated with the proposed action include pile driving (impact and vibratory), vessel operations (DP thruster use and support vessel transits), geophysical surveys, and operations of the wind turbines. It is important to note that most in-water work will be done sequentially, and thus, only one source of noise will be produced at a time. However, there is the potential that during the final stages of export cable installation and WTG foundation installation some overlap of construction will occur, and thus, an overlap in sound fields is possible (see section 7.1.3.4 below for details). Below, we present background information on underwater acoustics, characterize the sound sources associated with the proposed action and analyze the effects of exposure to these sound sources by species group (i.e., whales, sea turtles and Atlantic sturgeon). These activities will occur in the construction, operations and maintenance phases of the project; however, for ease of analysis, all acoustic impacts of the proposed action are discussed comprehensively below.

Background Acoustic Information and Terminology

Frequency (i.e., number of cycles per unit of time, with hertz (Hz) as the unit of measurement) and amplitude (loudness, measured in decibels (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 $1\mu Pa^2$-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 or continuous vibration over a period of time; the cSEL value is not a measure of the instantaneous or maximum noise level, but is a measure of the accumulated energy over a period of time to which an animal is exposed during any kind of signal. The cSEL value can be estimated using either one of the following equations: cSEL (dB) = RMS pressure level + 10 Log (duration of exposure, in seconds) or cSEL(dB) = Single-strike SEL + 10 Log (N); where N is the number of strikes. The latter equation is primarily used to calculate the cSEL value for impulsive noise sources; however, if information is unavailable on the number of strikes and/or the single strike SEL for the pile to be installed, the former equation may be used to calculate the cSEL.
- Root Mean Square (RMS): the square root of the average squared pressures over the duration of a pulse; most pile-driving impulses occur over a 50 to 100 millisecond (msec) period, with most of the energy contained in the first 30 to 50 msec (Illingworth and Rodkin, Inc. 2001, 2009). Therefore, RMS pressure levels are generally "produced" within seconds of the operations, and represent the effective pressure, and its resultant intensity (in dB re: 1 μPa;), produced by a sound source.

Information on Noise Sources Associated with the Proposed Action

BIWF WTG Foundation Installation (Impact Pile Driving)
As described in Section 3.1.2.3, 42" and 54" diameter foundation piles will be installed, via an impact hammer (200kJ and 600 kJ rated hammers), to support the 4-leg foundation of each WTG, for a total of 20 piles. Source levels associated with the driving of piles, and the extent to which injury or behavioral modification thresholds for Atlantic sturgeon, sea turtles, or whales will be attained have been modeled by us or by TetraTech (TetraTech 2013a) for Deepwater Wind and are presented below in
Table 9.

Table 9: Source levels of underwater noise produced by the installation of 42" and 54" diameter foundation piles and resultant distance to injury or behavioral modification thresholds for sea turtles, Atlantic sturgeon, and whales.

			Whales		**Sea Turtles**		**Atlantic Sturgeon**		
Pile	**Impact Hammer**	**Source Level (dB re 1 µPa)**	**Distance (m) to 160 dBRMS**	**Distance (m) to 180 dBRMS**	**Distance (m) to 166 dBRMS**	**Distance (m) to 207 dBRMS**	**Distance (m) to 206 dBPeak**	**Distance (m) to 187 dBcSEL**	**Distance (m) to 150 dBRMS**
42" and 54" Foundation	200kJ	213	3,600	200	1359**	2.5**	2.92**	8,576**	15,849**
	600kJ	219	7,000	600	3414**	6.31**	7.36**	116,591**	39,810**

** NMFS estimated using the Practical Spreading Loss Model; $R2=R1*10^{((\text{measured or calculated sound level-Noise Threshold})/15)}$ (Bastasch *et al.* 2008; Stadler and Woodbury 2009), where: R2= the distance (in meters) to the threshold; R1=distance of the measured or calculated sound level; Sound level (i.e., RMS, cSEL, peak) = noise level measured or calculated at distance R1. If the cSEL was calculated , NMFS used the following equation to calculate the cSEL value: cSEL= dB_{RMS} + 10 Log (duration to install the pile, in seconds); where the dB_{RMS} value for impact pile driving was considered the source levels of underwater for either the 200 kJ or 600 kJ impact hammer; duration = 1800 seconds, per pile with a 200kJ hammer or 27,000 seconds, per pile, for installation with a 600kJ hammer ; and Noise Threshold=depending on species of interest, NMFS thresholds for injury or behavioral modification.

It will take up to 8 hours to install each pile, with no more than 1 pile installed per day. Pile driving will occur for a total of no more than 160 non-continuous hours over 20 non-consecutive days. The pile installation will occur over a 5-week period (i.e., between May-July2015 or August to October 2015).

Cable Installation (DP Thruster Use)

DP thrusters will be operational for a 24-hour period during cable lay operations. However, during this 24-hour period, thrusters will never be operating at full power (i.e., 100%). Thrusters will be operated at a power level of 50% or less in order to maintain vessel position and movements along the cable route. As the power levels will be variable throughout cable lay operations, there will be variability in underwater noise levels produced by the DP thrusters, with the highest levels produced at power levels of 50% and the lowest levels produced at levels below 50%. As a result, the information presented in
Table 10 reflects the worst case scenario of thrusters always operating at a 50% power level. Thruster use will occur over a period of 4 to 6 weeks for the installation of the BITS followed by a period of 2 to 4 weeks for the BIWF export and inter-array cable. All thruster use will occur between April and August. Sound levels associated with the DP thruster use, have been modeled by us or TetraTech (TetraTech 2013 a,b) at various depths (i.e., 7m, 10m, 20m, and 40m) and are presented below in
Table **10**.

Table 10: DP thruster source levels and resultant distance to a variety of isopleths.

	Whales		Sea Turtles		Atlantic Sturgeon		
Source Level (dB re 1 µPa @ 1m)	**Distance (m) to 120 dBRMS**	**Distance (m) to 180 dBRMS**	**Distance (m) to 166 dBRMS**	**Distance (m) to 207 dBRMS**	**Distance (m) to 206 dBPeak**	**Distance (m) to 187 dBcSEL**	**Distance (m) to 150 dBRMS**
180	4,750	1	8.6*	Not attained**	Not attained**	630	100

* NMFS estimated using the Practical Spreading Loss Model; $R2=R1*10^{((\text{measured or calculated sound level-Noise Threshold})/15)}$ (Bastasch *et al.* 2008; Stadler and Woodbury 2009), where: R2= the distance (in meters) to the threshold; R1=distance of the measured or calculated sound level. For our calculations, R1=the source level for DP thruster use (i.e., 180 dB_{RMS}); Sound level (i.e., RMS, cSEL, peak)= noise level measured or calculated at distance R1; and Noise Threshold= depending on species of interest, NMFS thresholds for potential injury or behavioral response.

**Calculations were based on a 24 hour period of operations as the source will move every 24 hour as will animals transiting through the project area on a daily basis. Please note however, when calculations are made, it is assumed that the source is stationary as there is currently no methods to estimate the acoustic footprint of moving sources of noise.

Offshore Cofferdam Installation (Vibratory Pile Driving)

As described in section 3.2.1, landing of the BITS on Scarborough Beach may require the Long-Distance HDD method of landfall construction, which requires the installation and removal of an offshore cofferdam made of sheet piles. The sheet piles will be installed with a vibratory hammer over a period of 2 days, with no more than 12 hours of pile driving operations to occur per day. Installation of the sheet piles is expected to occur sometime between January 1 and the end of March 2015, as terrestrial and submarine cable splicing need to occur prior to April 1, 2015, when submarine cable lay operations are to begin. Removal of the sheet piles will also occur between January 1 and May 1, 2015; pile removal will occur over two days with the vibratory hammer operating for no more than 12 hours each day.

Sound levels associated with the installation of sheet piles, via a vibratory hammer, and the distance to the 120 and 180 dB RMS isopleths have been modeled by TetraTech (TetraTech 2013b) and are presented below in Table 11.

Table 11: Sheet pile source levels and resultant distance to120 and 180 dB RMS isopleths.

Source Level (dB re 1 µPa @ 1m)	**Distance (m) to 120 dBRMS**	**Distance (m) to 180 dBRMS**
194	89,850	10

Support/Crew Vessel Noise

Up to 16 support vessels (e.g., anchor handling and towing tugs; material, derrick, jack-up and transportation barges; work and crew vessels) will be used throughout the construction of the BIWF and up to 7 support vessels will be used throughout the construction of the BITS. These support vessels will regularly transit the action area, at various stages and times, to assist or aid in installation and construction of the project.

Vessels transmit noise through water. The dominant source of vessel noise 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. In general, 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. Based on this information, vessels associated with the proposed action are expected to produce noise of approximately 150 to 170 dB re 1 μPa-m at frequencies below 1,000 Hz.

Geophysical Surveys

Following cable installation (BITS, export, inter-array), Deepwater Wind will conduct an inspection of the cable route to ensure cable burial depth is achieved. Inspections will be done via a high resolution geophysical survey using a multi-beam (sonar) survey and a shallow sub-bottom profiler (i.e., chirper). The survey ships will be approximately 60 feet in length and will travel speeds of approximately 3 to 4 knots. The survey ship will be 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. In addition to the post-installation survey, every 5 years, cable burial depth along the BITS, export, and inter-array cable will be checked with a sub-bottom profiler. Operations and vessel requirements will be the same as that described for the initial survey.

In a memo prepared by TetraTech, estimates of the distance from the source to the 180 dB_{RMS} radius and the 160 dB_{RMS} radius for the different survey instruments were provided (TetraTech 2011). The source levels and operating frequencies and the distances to the 180 dB and 160 dB_{RMS} isopleth radii are noted in

Table 12.

Table 12: Source levels, operating frequencies, and distance to 180 and 160 dB_{rms} isopleths for the multi-beam sonar and chirp sub-bottom profiler.

Source	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
Chirp sub-bottom Profiler	198	2-16kHz	11	150	**Yes**	No	No
Multi-beam sonar	162	455kHz	65	109	No	No	No

The above modeling scenarios undertaken by TetraTech (2011) to estimate the radial distance to the 160 dB_{RMS} and 180 dB_{RMS} isopleths were based on a 17.4 Log R spreading loss model. The distance presented in the table above represent the maximum distances to attenuation of 160 dB_{RMS} or 180 dB_{RMS} and can be considered as a "worst case" representation.

Operational Noise of the Wind Turbine Generators

Once installed, the operation of the WTGs is not expected to generate substantial underwater sound levels above baseline sound in the area. Preliminary results from noise studies conducted at offshore wind farms in Europe suggest that in general, the level of noise created during the operation of a offshore wind farm is very low. Even in the area directly surrounding the wind turbines, noise, in general, was not found above the level of background noise (Nedwell 2011, reported in BOEM 2008). Source levels of underwater noise from these studies were generally with the range of 150 dB re 1uPa or lower, with underwater noise levels between 112-115 dB found within 330 feet or less of the wind turbines (levels of underwater noise reaching 120 dB were estimated to occur within 110 to 170 feet of the turbine).[50]

Acoustic modeling of underwater operational noise produced by proposed wind farms has also been performed within the waters off Massachusetts (Nantucket Sound) and the nearshore waters surrounding Block Island (Block Island Sound, Atlantic Ocean). Within Nantucket Sound, the models predicted that sound levels of a WTG would be approximately 109.1 dB at 20 meters from the WTG monopile, and that this sound level would fall off to 107.5 dB at 50 meters

[50] Distance to the 120 dB threshold were estimated using the available data and the following equation: Received Level= Source Level-15 Log R (NMFS 2012b).

(BOEM 2010). These predicted sound levels were only 0.3 to 1.9 dB above baseline/ambient underwater sound levels of 107.2 dB (BOEM 2010), and thus, did not greatly exceed, and therefore, contribute significant levels of underwater noise to ambient underwater noise levels in the waters of Nantucket Sound. Similar results were found in the acoustic modeling studies done in the waters off Block Island. The modeling suggested that the operation of a wind farm in the waters south of Block Island would fall within ambient noise levels (Miller *et al*. 2010).[51] The study defined the noise budget in the waters surrounding Block Island, with the main contributors identified to be shipping (97 dB re1uPa^2), wind (97 dB re1uPa^2), rain (92 dB re1uPa^2), and biological noise (87 dB re1uPa^2) (Miller *et al*. 2010). Modeling results of the proposed wind farm predicted that operational noise would contribute (88 dB re1uPa^2) little to any additional noise, as the additional noise from the wind turbines would be less than noise from shipping, wind, and rain, (Miller *et al*. 2010).

7.1.3.1 Effects of Noise Exposure to Right, Humpback and Fin Whales

Background Information on Acoustics and Marine Mammals

When anthropogenic disturbances elicit responses from 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 behavior.

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.
6.

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

[51] In order for marine life to detect new sources of underwater noise, the frequency and associated decibel level of that new source must exceed the ambient underwater noise levels within the affected area.

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 potential impacts to marine mammals are as follows:

Injury	Behavioral Disturbance
180 dB RMS	120 dB RMS (continuous noise source)
	160 dB RMS (non-continuous noise source (impulsive))

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 (NMFS 1995; Southall *et al*. 2007; Malme *et al.* 1983, 1984; Richardson *et al*. 1990,1995,1986; Tyack 1998). 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, and Fin Whale Hearing

In order for whales to be affected by 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 ranges from 7 Hz to 30 kHz. 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 and 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 (Table 13) summarizes the range of sounds produced by right, humpback, and fin, whales (from Au *et al*. 2000):

Table 13: Summary of known right, humpback, and fin 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)

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 and fin whale hearing and the source levels and dominant frequencies of the project 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.

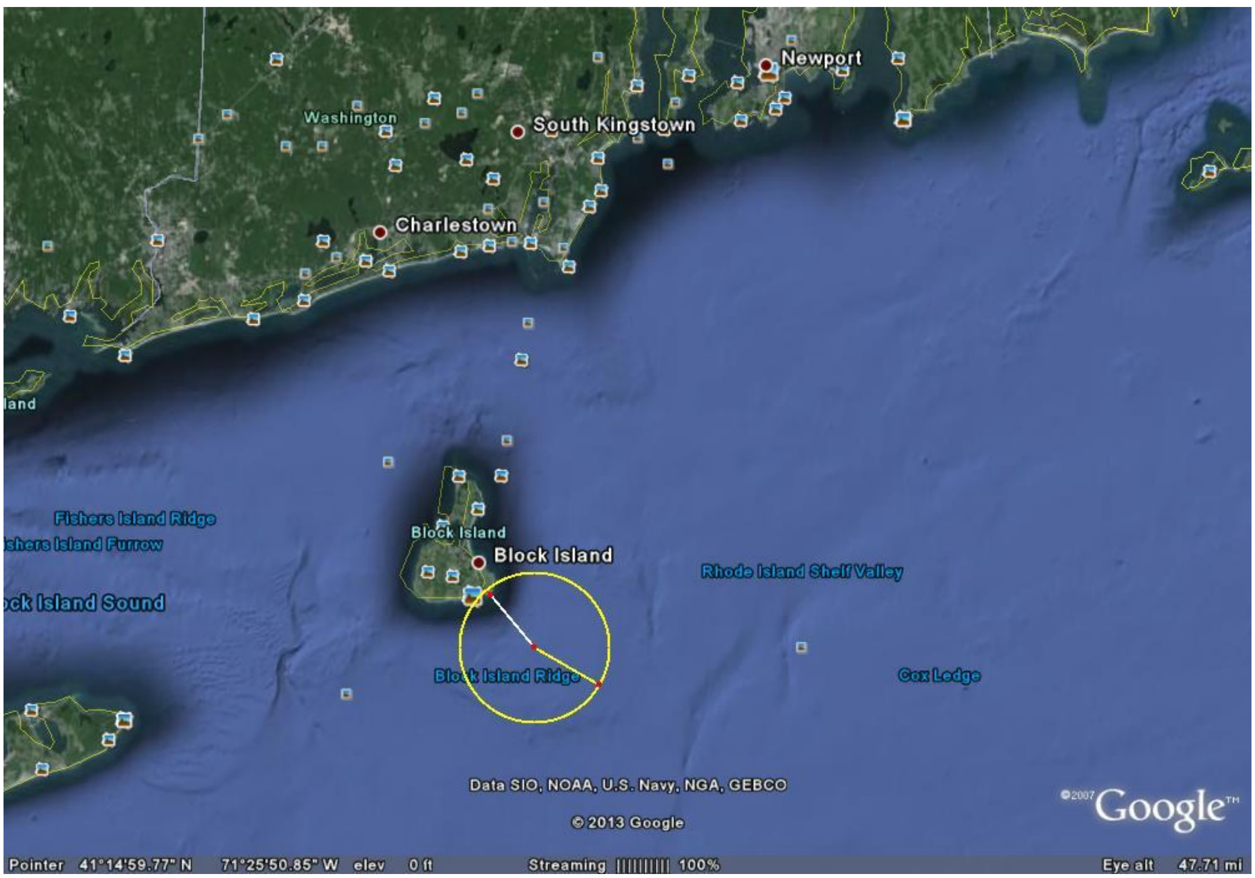
Effects to Whales from Exposure to Impact Pile Driving Noise

As noted above, injury can result to whales upon exposure to impulsive noises, such as pile driving with an impact hammer, above 180 dB re 1µPa RMS. According to the best available estimates (see Table 13), noise levels greater than 180 dB re 1µPa RMS will be experienced only very close to the pile being driven with noise attenuating to less than 180 dB re 1µPa RMS within 200 meters of the pile when the 200 kJ hammer is being used and within 600 m when the 600 kJ hammer is used. An exclusion zone extending from the pile being installed to the estimated distance of attenuation to 180 dB will be established prior to pile installation. This exclusion zone (extending either 200 or 600 m from the pile depending on the hammer being used) will be monitored for at least 30 minutes prior to the beginning of pile driving. Pile driving will not begin until the exclusion zone is free of whales for at least 30 minutes. Given the small area of the exclusion zone and the shallow depths and the dive time of whales in the area (right whales 10-15 minute maximum, humpback 6-7 minutes typical, fin 20 minutes), it is reasonable to expect that monitoring the exclusion zone for at least 30 minutes will allow the observers to detect any whales that may be submerged in the exclusion zone. Once pile driving begins, should a whale be detected within the exclusion zone, all operations will be halted or delayed until the exclusion zone is clear of whales for at least 30 minutes. Based on this, it is extremely unlikely that a whale will be present within 200 m or 600 m of the piledriving when the 200 or 600 kJ hammer is operating; therefore, it is extremely unlikely that any whale will be exposed to noise that could cause injury.

In the event that in-field monitoring indicates that the 180 dB_{RMS} isopleth is greater than or less than 200 m or 600 m, then a new exclusion zone will be established. No changes to the size of the exclusion zone will be made without USACE and NMFS approval.

As noted in Table 13 and illustrated below in Figure 3, underwater noise levels of 160 dB_{RMS} will extend a maximum of 7 km from the pile being driven, resulting in a maximum ensonified area of 89.6 km^2 (see Figure 3).

Figure 3: Impact Pile Driving: Ensonified Area to the 160 dB_{RMS} Isopleth



Available information suggests that impulsive noise above 160 dB re 1µPa RMS may trigger a behavioral response in whales; behavioral responses could range from a startle with immediate resumption of normal behaviors to complete avoidance of the area where noise is elevated above 160 dB re 1µPa RMS and could also include changes in foraging behavior. Any whales present in the area illustrated above during any of the pile driving (total of 160 hours) is occurring may react behaviorally to this noise.

Pile driving will occur either between May and July or August and October; these times of year have been selected by the applicant to minimize the potential for exposure of right whales to pile driving noise. During those times of year, right whales are typically located outside of the action area. A review of right whale sightings data for the May-July and August-October period (recorded since January 1, 1999; available at: http://www.nefsc.noaa.gov/psb/surveys/) shows no documented right whale sightings within the area where noise will be above 160 dB re 1uPa RMS during pile driving. There are a few records of right whales, including mother and calf pairs, in nearby waters, suggesting that occasional right whales may be present in the general area when pile driving will occur. Since 1986, only 18 pairs have been documented in the action area (i.e., Block Island Sound: 2 pair; Rhode Island Sound: 7 pairs; Atlantic Ocean (area south of Block Island to approximately 40o45.3'N): 8 pairs; Vineyard Sound: 1 pair; http://www.nefsc.noaa.gov/psb/surveys/SASInteractive2.htlm (last accessed December 18, 2013). During the time of year when pile driving will occur, right whale sightings are limited to solitary individuals or single mother-calf pairs. Maximum SPUE densities in the area where pile driving will occur are 0.07 right whales/100 km^2 (Kenney and Vigness-Raposa 2009). While feeding aggregations have been recorded in Rhode Island Waters, these have not been observed during the time of year pile driving will occur and given the seasonal distribution of copepods in the action area, it is not reasonable to anticipate that they would occur during the May-July or August-October period when pile driving occurs. Therefore, based on past sightings data, we expect there to be very few right whales exposed to pile driving noise and that the individuals exposed would be solitary individuals or single mother-calf pairs.

A review of sightings of humpback whales (as recorded in the OBIS database, with data from 1986-2012: http://seamap.env.duke.edu/species/180532) indicates 6 sightings in the area with 4 (total of 5 animals) occurring at the time of year when pile driving would occur. With the exception of a pair of humpbacks sighted on June 4 and 7, 1988, the other instances (July 1888 and August 1992) were of single animals. A similar query for fin whale sightings (http://seamap.env.duke.edu/species/180532, data from 1986-2012) indicates similar results, with fewer than 10 individuals sighted within the area where noise will be above 160 dB during pile driving. All sightings were of individuals or small groups (less than 5 individuals). Maximum SPUE densities in the area where pile driving will occur are 0.11 humpback whales/100 km^2 and 1.92 fin whales/100 km^2 (Kenney and Vigness-Raposa 2009).

It is difficult to predict the number of whales that may be exposed to potentially disturbing levels of noise associated with impact pile driving. In their application for an MMPA Incidental Harassment Authorization (IHA), Deepwater Wind has calculated an estimate based on sightings per unit effort (SPUE) data in the affected area. SPUE used for these estimates was calculated by Kenney and Vigness-Raposa (2009). Kenney and Vigness-Raposa (2009) derived the SPUE data from a number of sources including: 1) North Atlantic Right Whale Consortium database (NARWC); 2) CeTAP (CeTAP, 1982); 3) sightings data from the Coastal Research and Education Society of Long Island, Inc. (CRESLI) and Okeanos Ocean Research Foundation; 4) the Northeast Regional Stranding (NERS) network (marine mammals); and 5) the NOAA Fisheries Sampling Branch (Woods Hole, MA).

Estimates of animals exposed to potentially disturbing levels of noise were computed according to the following formula:

> *Estimated Number of Animals Exposed = D x ZOI x (1.5) x (d)*
> Where:
> D = average highest species density (number per 100 km2)
> ZOI = maximum ensonified area to 160 dB (impulsive noise) or 120dB (continuous noise)
> 1.5 = Correction factor to account for marine mammals that may be underwater
> d = number of days

This method is also likely to overestimate the number of animals exposed because it uses the maximum SPUEs, regardless of season, to predict exposure and assumes that all pile driving will be accomplished with the higher energy 600 kJ hammer; it also rounds up to whole animals any calculated fractions of animals exposed. Estimates of exposure to impact pile driving noise are based on ZOI of 34.6 mi^2 (89.6 km^2) and a total construction period of 20 days (assumes 4 days of pile driving for each of the five jacket foundations). Using this method, Deepwater calculates that a total of 2 right whales, 3 humpback whales and 52 fin whales will be exposed to potentially disturbing levels of noise (between 180 dB and 160 dB) over the 20 days of impact pile driving. Below, we consider the effects of that exposure to this small number of whales. We expect any whales within 7 km of the piles being driven will react behaviorally. Available information on behavioral responses to underwater noise indicates that a range of behaviors could be experienced, ranging from a temporary startle with immediate resumption of pre-disturbance behaviors to evasive movements resulting in departure from the area ensonified above 160 dB_{RMS}. Whales exposed to pile driving noise are expected to be transiting the area while participating in north-south or south-north migrations and may forage opportunistically if appropriate forage is present. Animals that are disturbed would make adjustments to their behaviors, resulting in an energetic cost. This energetic cost could be minor to non-existent if the whale was near the edge of the ensonified area, or could be larger if it was closer to the pile being driven and needed to swim over 7 km to escape the noise.

Whales migrating through the area when pile driving occurs are expected to adjust their course to avoid the area where noise is elevated above 160 dB re 1uPa RMS. Depending on how close the individual is to the pile being driven, this could involve swimming beyond 7 km, assuming that they take a direct route. Given that this is a single sound source, that is of low intensity, we believe this is a reasonable assumption. The whale may experience physiological stress during this avoidance behavior, but this stressed state would resolve once the whale had swam away from the area with disturbing levels of noise. Right whales typically swim at speeds of 1.3 km/hour (Hain *et al.* 2013; including individuals, groups and mother-calf pairs) while humpback whales and fin whales swim considerably faster (Humpbacks normally swim (4.8-14 km/h), but can go up to 24-26.5 km/h) in bursts; fin whales swim at speeds of 9–15 km/h and can swim at burst speeds of up to 42 km/h; Society for Maine Mammology, accessed December 2013). This suggests that even at a normal, non-agitated, swimming speed, right whales would be able to swim out of the area with disturbing levels of noise within approximately three hours and fin and humpback whales would swim out of the area in one to two hours. Thus, the stressed state would be temporary. Similarly, any disruption or delay in opportunistic foraging or resting would be temporary and persist only as long as it took the whale to swim away from the noisy area. Resting or opportunistic foraging would resume once the whale left the noisy area. Even if a whale wanted to return to the area it was displaced from, it would be displaced for no more than the 8 hours a day. Migration is expected to continue with the avoidance representing a minor disruption to the migratory path.

While in some instances temporary displacement from an area may have significant consequences to individuals or populations, this is not the case here. For example, if whales were prevented from accessing calving grounds or were precluded from foraging for an extensive period, there could be impacts to reproduction and the health of individuals, respectively. However, in this case the area where noise may be at disturbing levels is a small portion of the coastal area used for north-south and south-north migrations and is a tiny subset of the coastal Northeast waters used by foraging whales. Therefore, although in the worst case, whales may avoid or be temporarily excluded from the area with disturbing levels of sound for the duration of pile driving operations (i.e., 8 hours a day for a total of 20 non-consecutive days), the area from which an individual is being excluded is not a considered to be especially important or unique,and the behaviors that would have been carried out in the area can be carried out elsewhere with only minor, short term costs to the individuals affected.

All behavioral responses to a disturbance, such as those described above, will have an energetic or metabolic consequence to the individual reacting to the disturbance (e.g., adjustments in migratory movements or foraging). It is believed that short-term interruptions of normal behavior are likely to have little effect on the overall health, reproduction, and energy balance of an individual or population (Richardson *et al.* 1995). As the disturbance will occur for only 8 hours a day, over a period of 20 non-consecutive days, whales are not expected to be exposed to chronic levels of underwater noise and thus, chronic levels of disturbance that significantly impair essential behavior patterns. Thus, although there will be a temporary energetic consequence to any whale disturbed by impact pile driving noise, due to the temporary nature of the disturbance, the additional energy expended is not likely to significantly impair essential life functions (i.e., foraging, migration, nesting, rearing) or impair the health, survivability, or reproduction of an individual.

Based on this analysis, we have determined that any changes in behavior resulting from exposure to increased underwater noise associated with pile driving will temporarily disrupt whale behavior (e.g., migratory movements, resting, foraging) , but the individual's ability to carry out these behaviors will resume as soon as the animal swims out of the noisy area or the pile driving ceases. Therefore, any impairment will be temporary and limited to a short term stress response and temporary shift in energy expenditures from the pre-disturbance behavior to evasive movements. For those reasons, any impairment will not rise to the level of a significant impairment of any essential behaviors such as resting, foraging,or migrating and we do not expect the fitness of any individuals to be affected. Additionally, while there will be a short term increase in energy expenditure, this is not expected to have any detectable effect on the physiology of any individuals or any future effect on growth, reproduction, or general health.

Based on the above analyses, although on an individual level, we expect temporary adjustments in individual behaviors, we do not expect the exposure of impact pile driving noise to result in injury or death by significantly impairing essential behavioral patterns for individual whales. No population level effects are likely.

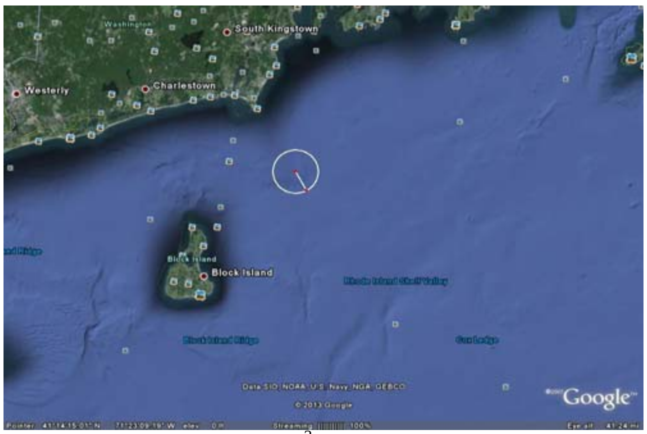
Effects to Whales – DP Thrusters

As described above in

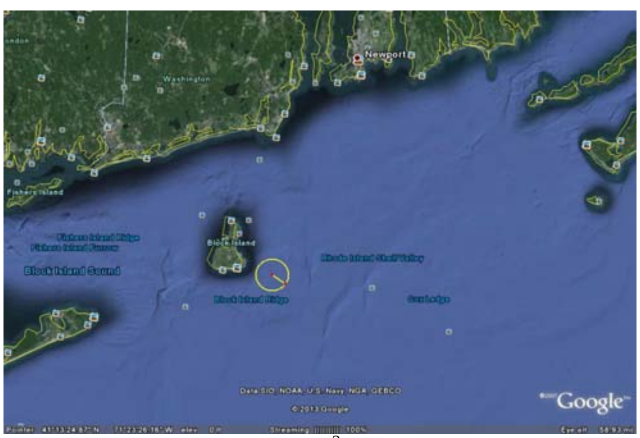
Table 12, underwater noise levels of 180 dB_{RMS} or greater are expected within 1 meter of the DP vessel. ESA listed species of whales are not expected to occur within 1 meter of vessel and thus, no whales are expected to be exposed to injurious levels of underwater noise.

DP thruster operation is considered a continuous noise source. Based on modeling performed by TetraTech (TetraTech 2013 a, b), the average ensonified area at the 120 dB_{RMS} isopleth extends 4.75 km from the source, with the total size of the area experiencing noise of 120 dB_{RMS} or greater being 23 km^2 or 25.1 km^2 along the BITS or BIWF export or inter-array cable route, respectively (see Figure 4).[52]

Figure 4: Area of Ensonification for DP Thruster Operation along the BITS or BIWF Export and Inter-array Cable



**BITS: representative 23km² ZOI along one portion of the BITS cable route. Rectangular objects artifact of creating map.*



** BIWF:Representative 25.1km² ZOI along one portion of the export or interarray cable. Rectangular objects artifact of creating map.*

[52] The estimated average ensonified area took into account the three representative water depths (i.e., 10m, 20m, and 40m) along the BIWF cable routes and the four representative water depths (i.e., 7m, 10m, 20m, and 40m) along the BITS cable route in which the 120dB isopleth was modeled, as well as took into consideration the continuous movement of the vessel along the cable route.

As the DP vessel is continually moving along the cable route over a 24-hour period, the area within the 120 dB_{RMS} isopleth is constantly moving and shifting within a 24-hour period. Therefore, no single area in Rhode Island Sound will have noise levels above 120 dB_{RMS} for more than a few hours.

Available information suggests that continuous noise above 120 dB re 1µPa RMS may trigger a behavioral response in whales; behavioral responses could range from a startle with immediate resumption of normal behaviors to complete avoidance of the area where noise is elevated above 120 dB re 1µPa RMS and could also include changes in foraging behavior. Any whales present in the area where noise is elevated above 120 dB_{RMS} when the DP thruster is operational may react behaviorally to this noise.

Operation of the DP thrusters will occur along the cable installation routes between April and August. A review of right whale sightings data (recorded since January 1, 1999; available at: http://www.nefsc.noaa.gov/psb/surveys/) shows few documented right whale sightings within the area where noise will be above 120 dB re 1uPa RMS during cable installation. All of these sightings were in April and with the exception of a single individual sighted in April 1979, the remainder were sighted in April 2010. During April 2010, large aggregations of right whales were observed in Rhode Island Sound; based on behaviors displayed by these animals, we assume they were feeding on copepods. Based on historic sightings data, in April we expect right whales to occur in the area where DP thrusters will be used.

A review of sightings of humpback whales (as recorded in the OBIS database: http://seamap.env.duke.edu/species/180532) indicates 3 sightings in the area that will experience increased noise due to DP thruster use with 4 (total of 5 animals) occurring at the time of year when DP thruster use would occur (April – August). With the exception of a pair of humpbacks sighted on June 4 and 7, 1988, the other instances (July 1888 and August 1992) were of single animals. A similar query for fin whale sightings (http://seamap.env.duke.edu/species/180532) indicates similar results, with fewer than 10 individuals sighted within the area where noise will be above 120 dB during DP thruster use. All sightings were of individuals or small groups (less than 5 individuals).

Using the method for calculating the number of right, humpback and fin whales exposed to potentially disturbing levels of noise explained above, and the highest seasonal SPUEs reported for the area where DP thrusters will be used (0.06 right whales/100km^2; 0.11 humpback whales/100km^2; and, 2.15 fin whales/100km^2), Deepwater calculates that a total of 1 right whale, 2 humpback whales and 23 fin whales will be exposed to potentially disturbing levels of noise (greater than 120 dB_{RMS}) over the entire duration of DP thruster use. Below, we consider the effects of that exposure to this small number of whales.

We expect any whales within 4.75 km of the DP thruster will react behaviorally. Available information on behavioral responses to underwater noise indicates that a range of behaviors could be experienced, ranging from a temporary startle with immediate resumption of pre-disturbance behaviors to evasive movements resulting in departure from the area ensonified by continuous noise above 120 dB_{RMS}. Whales exposed to the DP thruster noise are expected to be transiting the area while participating in north-south or south-north migrations and may forage opportunistically if appropriate forage is present. Animals that are disturbed would make adjustments to their behaviors, resulting in an energetic cost. This energetic cost could be minor to non-existent if the whale was near the edge of the ensonified area, or could be larger if it was closer to the DP vessel and needed to swim nearly 5 km to escape the noise.

Whales migrating through the area when the DP thruster is in use are expected to adjust their course to avoid the area where noise is elevated above 120 dB re 1uPa RMS. Depending on how close the individual is to the DP thruster, this could involve swimming up to 5 km. The whale may experience physiological stress during this avoidance behavior but this stressed state would resolve once the whale had swam away from the area with disturbing levels of noise. Right whales typically swim at speeds of 1.3 km/hour (Hain *et al.* 2013; including individuals, groups and mother-calf pairs) while humpback whales and fin whales swim considerably faster (Humpbacks normally swim (4.8-14 km/h), but can go up to 24-26.5 km/h) in bursts; fin whales swim at speeds of 9–15 km/h and can swim at burst speeds of up to 42 km/h; Society for Maine Mammology, accessed December 2013). This suggests that even at a normal, non-agitated, swimming speed, right whales would be able to swim out of the area with disturbing levels of noise within approximately 3 hours and fin and humpback whales would swim out of the area in less than an hour. Thus, the stressed state would be temporary. Similarly, any disruption or delay in foragingor resting would be temporary and persist only as long as it took the whale to swim away from the noisy area. Resting or foraging would resume once the whale left the noisy area. Even if a whale wanted to return to the area it was displaced from, it would be displaced for no more than the few hours when the DP vessel was operating in a particular area. Migration is expected to continue with the avoidance representing a minor disruption to the migratory path.

As noted above, whales are expected to forage opportunistically in the action area. There have been rare instances in the action area; however, where prey abundance is high due to climatic changes in the environment, resulting in area that is favorable for whale foraging and thus, aggregations. As described above, in April 1998 and April 2010, high rain fall events resulted in high runoff and nearshore phytoplankton/zooplankton blooms in Rhode Island Sound and thus, increased numbers of foraging right whales in the area for a period of several weeks (Kenney 2010). Similar events have not occurred since this time. However, should such an event occur during the 4 to 6 weeks of cable installation (between the months of April and August), based on the 1998 and 2010 sites of aggregation, DP thruster use would overlap with this area and thus, effect foraging right whales should such an event occur again during this phase of construction. As such, we have considered the effects to right whales if DP thruster use occurred during a foraging event similar to those experienced in Rhode Island Sound in 1998 and 2010. The most severe consequence would be abandonment of feeding activities by right whales. This could have short term negative impacts to right whales; however, because the DP thruster vessel is constantly moving, the area experiencing noise above 120 dB_{RMS} is constantly shifting. Copepods occur in large dense patches; given the patchy distribution of copepods and the constant movement of the DP vessel, it appears to be extremely unlikely that the ensonified area would overlap more than temporarily with the area where food resources were present should such an event occur. Due to the constant movement of the DP vessel, it seems unlikely that right whales would abandon the area if copepods were present as any one area would have potentially disturbing levels of noise for no more than a few hours. Further, baleen whales, including right whales, which only feed during part of the year and must satisfy their annual energetic needs during the foraging season, are more likely to continue foraging in the face of disturbance (NMFS 2013). Based on the nature of the DP vessel noise (i.e.,. constantly moving resulting in a transient sound field), and the lessened response to disturbance during foraging, we do not expect right whales to abandon foraging activities if foraging areas overlapped with the area ensonified with the DP thruster; rather, we expect foraging to continue with affected individuals experiencing a mild stressed response and perhaps increased vigilance during this period which could result in less efficient foraging. Due to the seasonal occurrence of right whales in the action area (primarily from November 1 through April 30), if such an event were to occur, we would only expect these possibile effects to right whales to exist during the April timeframe of construction.

All behavioral responses to a disturbance, such as those described above, will have an energetic or metabolic consequence to the individual reacting to the disturbance (e.g., adjustments in migratory movements or foraging). It is believed that short-term interruptions of normal behavior are likely to have little effect on the overall health, reproduction, and energy balance of an individual or population (Richardson *et al.* 1995). Any exposure of whales to DP thruster noise is expected to be temporary and limited to either the time it takes a restingor migrating right, humpback or fin whale to move away from the disturbing level of noise (one to three hours, depending on species) or the time it takes the DP vessel to move away from an area where a whales may be foraging opportunistically. Whales are not expected to be exposed to chronic levels of underwater noise and thus, chronic levels of disturbance that significantly impair essential behavior patternswill not occur. Thus, although there will be a temporary energetic consequence to any migrating or resting whale disturbed by DP thruster noise, due to the temporary nature of the disturbance, the additional energy expended is not likely to significantly impair essential life functions or impair the health, survivability, or reproduction of an individual. Any whales that may be foraging in the action area and are exposed to DP thruster noise are expected to continue foraging, but may forage less efficiency due to increased energy spent on vigilance behaviors. This may have short term metabolic consequences for individual animals and may result in a period of physiological stress; however, this stressed state and less efficient foraging is only expected to last as long as prey distribution overlaps with the area ensonified above 120 dB_{RMS}, which is expected to be temporary and due to the constant movement of the DP vessel, would never persist more than a few hours.

Based on this analysis, we have determined that any changes in behavior resulting from exposure to increased underwater noise associated with DP thruster use will temporarily disrupt behaviors including resting, foraging and migrating but the individual's ability to carry out these behaviors will resume as soon as the animal swims out of the noisy area or the DP thruster use ceases. Therefore, any impairment will be temporary and limited to a short term stress response and temporary shift in energy expenditures from the pre-disturbance behavior to evasive movements. For those reasons, any impairment will not rise to the level of a significant impairment of any essentialbehaviors, such as resting, foraging,or migrating, and we do not expect the fitness of any individuals to be affected. Additionally, while there will be a short term increase in energy expenditure, this is not expected to have any detectable effect on the physiology of any individuals or any future effect on growth, reproduction, or general health. In general, it is believed that short-term interruptions of normal behavior are likely to have an insignificant effect on essential behavioral patterns and thus, an insignificant effect on the overall health, reproduction, and energy balance of an individual or population (Richardson *et al.* 1995).

Whales (sheet piles) (Vibratory Pile Driving)

As described above in Table 12, underwater noise levels of 180 dB_{RMS} or greater are expected only within 10 meters of the sheet pile to be installed/removed. Due to the shallow, nearshore location of the area where sheet piles are to be installed/removed, no individual whales will occur within 10 m of the steel sheet piles. Therefore, there is no potential for exposure to noise that may result in injury.

As described above, whales are expected to react behaviorally to exposure to continuous noise sources (i.e., non-impulsive) resulting in underwater noise levels of 120 dB_{RMS}. Underwater noise levels of 120 dB_{RMS} will extend 89 km from the shoreline where the sheet piles are being installed (see Table 11), resulting in an ensonified area of 4,352 km^2. Vibratory pile driving to install and remove sheet piles will occur over four 12-hour periods between January 1 and May 1, 2015.

A review of right whale sightings data (recorded since January 1, 1999; available at: http://www.nefsc.noaa.gov/psb/surveys/) for the January–May 11 period shows few documented right whale sightings within the area where noise will be above 120 dB re 1uPa RMS during January and February. The majority of sightings occurred in April, with fewer in March. These sightings include single whales, groups and occasional mother/calf pairs. Reviews of humpback and fin whale sightings indicate few sightings during the winter months with occasional individuals and small groups sighted in the area during April.

Using the method for calculating the number of right, humpback and fin whales exposed to potentially disturbing levels of noise explained above, and the highest seasonal SPUEs reported for the area ensonified by the vibratory pile driver (0.026 right whales/100km^2; 0.057 humpback whales/100km^2; 0.46 fin whales/100km^2), Deepwater estimates that 7 right whales, 15 humpback whales, and 121 fin whales may be exposed to behaviorally disturbing levels of underwater noise over the entirety of the four 12-hour periods when sheet pile driving and removal will occur.

The area where noise will be elevated above 120 dB_{RMS} during sheet pile installation is very large; extending through Rhode Island Sound and into the Atlantic Ocean, 89 km from the shoreline. However, the duration of the ensonification is short; occurring over a total of four 12-hour periods. We expect any whales within 89 km of the pile installation and removal may react behaviorally. Available information on behavioral responses to underwater noise indicates that a range of behaviors could be experienced, ranging from a temporary startle with immediate resumption of pre-disturbance behaviors to evasive movements resulting in departure from the area ensonified by continuous noise above 120 dB_{RMS}. Whales exposed to the vibratory pile driver noise are expected to be transiting the area while participating in north-south or south-north migrations and may forage opportunistically if appropriate forage is present. Animals that are disturbed would make adjustments to their behaviors, resulting in an energetic cost. This energetic cost could be minor to non-existent if the whale was near the edge of the ensonified area, or could be larger if it was closer to the shore and needed to swim greater than 89 km to escape the noise.

Whales migrating through the area when pile driving occurs are expected to adjust their course to avoid the area where noise is elevated above 120 dB re 1uPa RMS. Depending on how close the individual is to the pile being driven, this could involve swimming over 89 km. The whale may experience physiological stress during this avoidance behavior but this stressed state would resolve once the whale had swam away from the area with disturbing levels of noise. Right whales typically swim at speeds of 1.3 km/hour (Hain *et al.* 2013; including individuals, groups and mother-calf pairs) while humpback whales and fin whales swim considerably faster (Humpbacks normally swim (4.8-14 km/h), but can go up to 24-26.5 km/h) in bursts; fin whales swim at speeds of 9–15 km/h and can swim at burst speeds of up to 42 km/h; Society for Maine Mammology, accessed December 2013). This suggests that at a normal, non-agitated, swimming speed, right whales close to the source at the start of pile installation may not be able to swim out of the area where noise is greater than 120 dB_{RMS} within the 12 hour period that pile installation occurs. Therefore, we expect that these whales would experience a stress response during that 12 hour period. Right whales closer to the edge of the ensonified area would experience a disturbance only for the time it took them to swim out of the area. In regards to humpback and fin whales, assuming that disturbed individuals would swim at least as fast as the high range of their normal non-burst swim speed, we would expect humpback and fin whales to swim out of the area within 6 hours. Thus, the stressed state would be temporary. Similarly, any disruption or delay in foraging or resting would be temporary and persist only as long as it took the whale to swim away from the noisy area. Restingor foraging would resume once the whale left the noisy area. Even if a whale wanted to return to the area it was displaced from, it would be displaced for no more than the few hours it took the individual humpback or fin whale to swim out of the area. Migration is expected to continue with the avoidance representing a minor disruption to the migratory path.

Given the extensive size of the area where noise greater than 120 dB_{RMS} will be experienced, it appears that during vibratory pile installation there could be a disruption to the migratory route of whales moving along the Rhode Island coast. Whales would swim around the area, expending additional energy to do so. However, because the area will only be ensonified for a short period of time, no more than four 12-hour periods, we do not expect that individual whales migrating along the coast will be affected on more than one day and do not expect this one-time exposure to result in any future changes to the migratory route.

While in some instances temporary displacement from an area may have significant consequences to individuals or populations this is not the case here. For example, if whales were prevented from accessing calving grounds or were precluded from foraging for an extensive period, there could be impacts to reproduction and the health of individuals, respectively. As explained above, most whales in the area are expected to be transients, moving through the area during springtime south-north migrations. Whales will adjust their migratory movements to avoid the area with disturbing levels of sound. In addition, there have been rare instances in the action area, where prey abundance is high due to climatic changes in the environment, resulting in area that is favorable for whale foraging and thus, aggregations. As described above, in April 1998 and April 2010, high rain fall events resulted in high runoff and nearshore phytoplankton/zooplankton blooms in Rhode Island Sound and thus, increased numbers of foraging right whales in the area for a period of several weeks (Kenney 2010). Should such an event occur during this phase of construction, effects to foraging right whales are possible. As such, we have considered the effects to right whales if vibratory pile driving occurred during a foraging event similar to those experienced in Rhode Island Sound in 1998 and 2010. If copepods were present in the area ensonified by vibratory pile driving, there is the potential that some right whales may not access the area due to the disturbing levels of noise. However, any delay in accessing the area would be limited to 12 hours, which is unlikely to have significant impacts on the health of any individual right whale. Whales already foraging in the area when pile driving begins are unlikely to abandon foraging due to the presence of disturbing levels of noise (NMFS 2013) but are likely to be stressed during that 12 hour period and may forage less efficiently. However, due to the temporary nature of this disturbance there are unlikely to be any health or fitness consequences.

All behavioral responses to a disturbance, such as those described above, will have an energetic or metabolic consequence to the individual reacting to the disturbance (e.g., adjustments in migratory movements or foraging). It is believed that short-term interruptions of normal behavior are likely to have little effect on the overall health, reproduction, and energy balance of an individual or population (Richardson *et al.* 1995). Any exposure of whales to vibratory pile driving noise is expected to be temporary and limited to either the time it takes a resting or migrating humpback or fin whale to move away from the disturbing level of noise (up to six hours, depending on location of the individual during onset of pile driving activities) or the time when a right whale may be foraging opportunistically on copepods during the month of April and is exposed to increased noise (up to 12 hours). Whales are not expected to be exposed to chronic levels of underwater noise and thus, chronic levels of disturbance that significantly impair essential behavior patterns. Thus, although there will be a temporary energetic consequence to any migrating or resting whale disturbed by vibratory pile driver noise, due to the temporary nature of the disturbance, the additional energy expended is not likely to significantly impair essential life functions or impair the health, survivability, or reproduction of an individual. Foraging right whales exposed to vibratory pile driver noise are expected to continue foraging, but may forage less efficiency due to increased energy spent on vigilance behaviors. This may have short-term metabolic consequences for individual animals and may result in a period of physiological stress; however, this stressed state and less efficient foraging is only expected to last as long as copepod distribution overlaps with the area ensonified above 120 dB_{RMS}, which is expected to be temporary (no more than 12 hours).

Due to the extensive area of ensonification, significant adjustments in behavior (e.g., deflecting movements away from the affected area) to avoid the noise are likely; however, these significant adjustments (swimming up to 89 km to avoid the noise) will be short lived (no more than 12 hours). Based on this analysis, we have determined that any changes in behavior resulting from exposure to increased underwater noise associated with vibratory pile installation use will temporarily disrupt behaviors including resting, foraging and migrating but the individual's ability to carry out these behaviors will resume as soon as the animal swims out of the noisy area or the pile driving ceases. Therefore, any impairment will be temporary and limited to a short term stress response and temporary shift in energy expenditures from the pre-disturbance behavior to evasive movements. For those reasons, any impairment will not rise to the level of a significant impairment of any essential behaviors, such as resting, foraging or migrating, and we do not expect the fitness of any individuals to be affected. Additionally, while there will be a short term increase in energy expenditure, this is not expected to have any detectable effect on the physiology of any individuals or any future effect on growth, reproduction, or general health. In general, it is believed that short-term interruptions of normal behavior are likely to have an insignificant effect on essential behavioral patterns and thus, an insignificant effect on the overall health, reproduction, and energy balance of an individual or population (Richardson *et al.* 1995).

Effects to Whales - Surveys

As described previously, right, humpback, and fin whales are expected to only perceive sound emitted by the chirp sub bottom profiler. The multi beam sonar operates outside of the hearing frequency of these species and cannot be perceived. According to the information provided by TetraTech (2011), noise levels of 180 dB_{RMS} or greater will be experienced within 11 meters of the chirp and noise will attenuate to 160 dB_{RMS} within 150 meters. Deepwater Wind is implementing and the USACE is requiring a 300-meter radius exclusion zone during the survey. This exclusion zone will be monitored for at least 30 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 30 minutes. As whales typically surface at least once every 30 minutes, it is reasonable to expect that monitoring the exclusion zone for at least 30 minutes will allow the PSO and the compliance monitor to detect any whales that may be submerged in the exclusion zone. Once the equipment is turned on, should a whale be detected within 300 meters of the survey vessel, all operations will be halted or delayed until the exclusion zone is clear of whales for at least 30 minutes. Because the exclusion zone will be monitored throughout operations and the survey will stop if a whale is detected within 300 m of the source, it is extremely unlikely that a whale will be present within 300 m of the source while the geophysical survey equipment is operating and thus, exposed to injurious or behaviorally disturbing levels of underwater noise. Based on this information, and the fact that the exclusion zones will be continuously maintained and no surveys will occur if whales are near enough to experience noise above 160 dB_{RMS}, we do not anticipate that any whales will be exposed to noise loud enough to result in injury or a behavioral response.

Operational Noise

The noise producing components of the WTG are at the nacelle, hundreds of feet above the water surface. Underwater noise is expected to be at or near ambient noise levels. Studies of operating wind farms in Europe indicate that operating wind farms do not cause avoidance of the area by marine species (Nedwell 2011; Miller *et al*. 2010; Westerberg 1994, Degan 2000, Henriksen 2001, Betke 2004, Ingemansson 2003, Thomas 2006, and Nedwell 2011 in Marmo *et al*. 2013). Because the underwater noise associated with the operation of the wind turbines is masked by other natural underwater noises, whales are not expected to be able to detect the operational noise of the WTGs. Because individuals will not perceive the noise, there will be no effects to any whales.

Vessel Noise

Vessels transmit noise through water; the dominant source of vessel noise from the proposed action is propeller cavitation, although other ancillary noises may be produced. As noted above, 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.

Exposure to individual vessel noise by whales within the action area would be transient and temporary as vessels moved along their route. Whale 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. Restrictions on vessel approaches near whales will ensure that project vessels are never within 500 meters of right whales and 100 meters from all other whales; this is a sufficient separation distance to avoid any exposure of whales to potentially disturbing noise associated with the operation of all project related vessels. As such, no whales are expected to be exposed to injurious or disturbing 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 project related vessels and any effects will be insignificant or discountable.

Masking

In addition to the behavioral effects discussed above, when exposed to loud anthropogenic noises that overlap with the frequency of their calls, whales may experience "masking." Here, we consider the potential for masking from all of the sound sources considered in this Opinion.

Masking, which refers to the reduction in an animals' ability to detect communication or other relevant sound signals due to elevated levels of background noise, is a natural phenomenon which marine mammals must cope with even in the absence of man-made noise (Richardson *et al*. 1995). 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). Some, if not all, of the whales exposed to increased underwater noise associated with the proposed activity may experience masking. However, in all instances this will be limited to the time it takes for the animal to swim away from the disturbing levels of noise, which is limited to a period of several minutes to several hours. These whales may make temporary shifts in calling behavior to reduce the effects of masking. The energy expended to adjust calls is expected to be minor. 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. As evidenced by sightings data, right, humpback, and fin whales typically occur in the action area as individuals or small groups. There are very few instances of aggregations of right whales in the action area and these species are not considered to be highly concentrated in the area where increased underwater noise will be experienced. Based on the temporary nature of any masking, masking effects to whales are expected to be insignificant.

Acoustically Induced Stress

Acoustically induced stress is a condition that whales can experience upon chronic exposure to anthropogenic noise. Here, we consider the potential for whales in the action area to experience acoustically induced stress due to noise associated with the proposed action.

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. Typical adaptive responses to 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. Acute noise exposure may cause inhibited growth (in a young animal), or reproductive or immune responses. 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.

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 drilling rig 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. 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 any of the sound sources associated with the proposed action, we do not anticipate this type of stress response from these activities, and thus, any stress response likely to be experienced by a whale as a result of exposure to the noise sources discussed here is expected to be insignificant.

7.1.3.2 Effects of Noise Exposure on Sea Turtles

Background Information on 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). We assume that these sensitivities to sound apply to all of the sea turtles in the action area (i.e., green, Kemp's ridley, leatherback and loggerhead sea turtles).

Thresholds for Assessing the Potential for Physiological and Behavioral Effects
Currently, there are no NMFS established criteria for injury or behavioral disturbance or harassment for sea turtles. As described above, the hearing capabilities of sea turtles are poorly known and there is little available information on the effects of noise on sea turtles. Some studies have demonstrated that sea turtles have fairly limited capacity to detect sound, although all results are based on a limited number of individuals and must be interpreted cautiously. Most recently, McCauley *et al*. (2000) noted that decibel levels of 166 dB re 1µPa$_{RMS}$ (166 dB$_{RMS}$) were required before any behavioral reaction (*e.g*., increased swimming speed) was observed, and decibel levels above 175 dB re 1µPa$_{RMS}$ elicited avoidance behavior of sea turtles. The study done by McCauley *et al*. (2000), as well as other studies done to date, used impulsive sources of noise (*e.g*., air gun arrays) to ascertain the underwater noise levels that produce behavioral modifications in sea turtles. As no studies have been done to assess the effects of impulsive and continuous noise sources on sea turtles, McCauley *et al*. (2000) serves as the best available information on the levels of underwater noise that may produce a startle, avoidance, and/or other behavioral or physiological response in sea turtles. Based on this and the best available information, NMFS believes any sea turtles exposed to underwater noise greater than 166 dB$_{RMS}$ may experience behavioral disturbance/modification (*e.g.*, movements away from ensonified area).

While there is some data suggesting noise levels from exposure to underwater explosives might result in injury to sea turtles, no such information is available for pile driving; however, studies on the effects of explosions on sea turtles recommend that an empirically based safety range developed by Young (1991) and Keevin and Hempen (1997) be used for guidance in estimating possible injury thresholds for sea turtles. Using the safety range formulas developed by Young (1991), and Keevin and Hempen (1997), and converting back to sound pressure levels using the "Ross Formula (Ross 1987)," SVT Engineering Consultants (2010) calculated a value of 222 dB re 1µPa_{Peak} as a conservative estimate of the underwater noise levels that may cause injury to sea turtles during pile driving operations. The study by SVT Engineering Consultants (2010); however, did not provide an estimated RMS value of underwater noise levels that may result in injury to sea turtles. As the sea turtle behavioral thresholds noted above are measured using the RMS of the sound source, to be consistent, we estimated the RMS value from the estimated PEAK level of underwater noise associated with possible sea turtle injury *(i.e.,* 222 dB re 1µPa_{Peak}). The RMS of a sound source is approximately 15 dB lower than the PEAK level of underwater noise for that sound source (developed by J. Stadler and D. Woodbury for NMFS pile driving calculations; see http://www.dot.ca.gov/hq/env/bio/fisheries_bioacoustics.htm). Based on this information, we have estimated an RMS value for injury of 207 dB re 1µPa_{RMS} (207 dB_{RMS}). This value, like the PEAK value estimated by SVT Engineering Consultants (2010), is a conservative estimate of the level of underwater noise, resulting from pile driving, that may cause injury to sea turtles. Based on this, we believe that underwater noise levels at or above 207 dB_{RMS} have the potential to injure sea turtles.

In summary, based on the best available information, we believe underwater noise at, or above, the following levels have the potential to cause injury or behavioral modification to sea turtles:

Organism	**Injury**	**Behavioral Modification**
Sea Turtle	207dB re 1µPa_{RMS}	166 dB re 1µPa_{RMS}

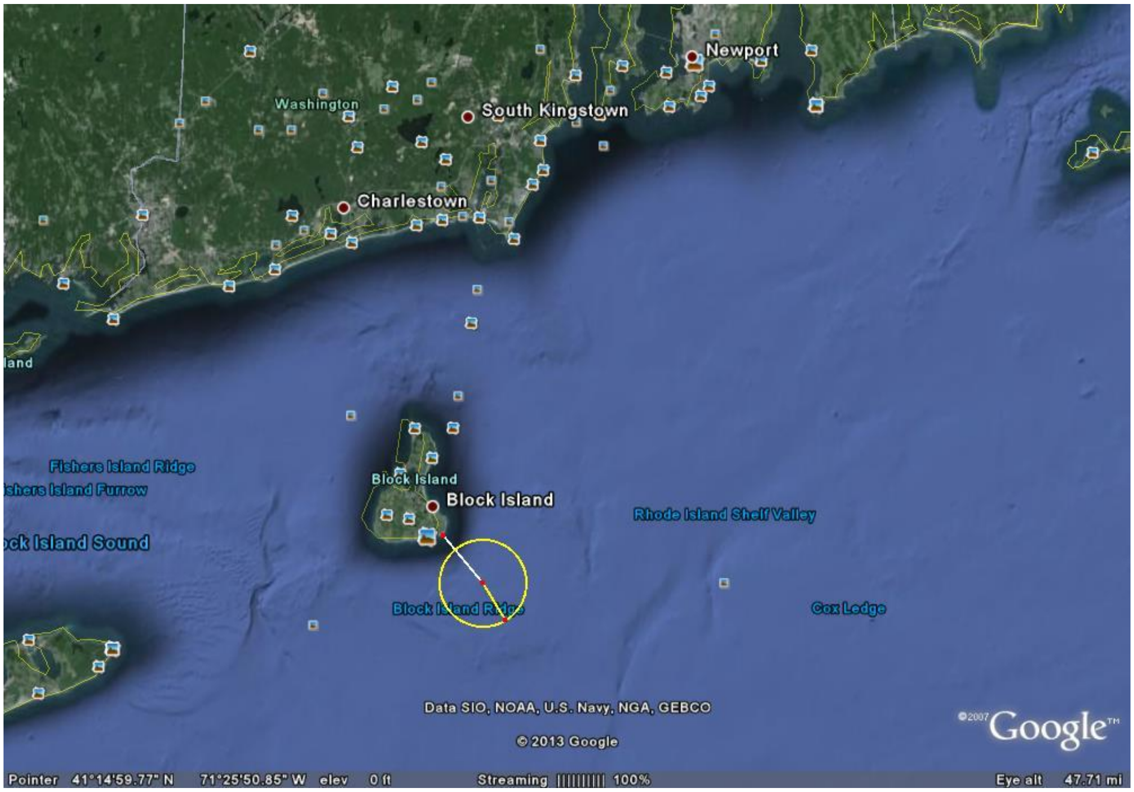
Effect of Exposure to Pile Driving Noise (Impact Hammer)

As noted above, we expect potential injury to sea turtles upon exposure to pile driving noises greater than 207 dB re 1µPa RMS. When the 200 kJ hammer is used, noise attenuates to below 207 dB re 1µPa RMS within 3 meters of the pile being driven; when the 600 kJ hammer is used, the area where noise is above 207 dB extends less than 7 meters from the source. Deepwater will maintain a 200 or 600-meter exclusion zone during pile driving (depending on the hammer being used). This exclusion zone will be monitored for at least 30 minutes prior to startup of the survey equipment. Pile driving will not be started until the exclusion zone is free of sea turtles for at least 30 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). Given the small area encompassed by the exclusion zone (i.e., extending only 200 or 600 m from the source) and the relatively shallow depths in the action area (i.e., less than 30 meters), it is reasonable to expect that monitoring the exclusion zone for at least 30 minutes will allow the observer 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 the exclusion zone, pile driving will be halted or delayed until the exclusion zone is clear of turtles for at least 30 minutes. Based on this, it is extremely unlikely that a sea turtle will be present within 7 m of any pile being driven. Additionally, given the noise levels produced during pile driving and given the expected behavioral response of avoiding noise levels greater than 166 dB re 1 µPa RMS, it is extremely unlikely that any sea turtles would swim towards the pile being installed once pile driving begins. Therefore, we do not anticipate any sea turtles will be exposed to pile driving noise that could result in injury.

As explained above, the best available information indicates that sea turtles will respond behaviorally to impulsive noises greater than 166 dB re 1 µPa RMS and will actively avoid areas with this noise level. It is reasonable to assume that sea turtles, on hearing the sound produced during pile driving, would either not approach the source or would move around it/away from it. When considering the potential for behavioral effects, we need to consider the geographic and temporal scope of any impacted area. For this analysis, we consider the area where noise levels greater than 166 dB re 1 µPa RMS will be experienced and the duration of time that those underwater noise levels could be experienced. Behavioral responses could range from a startle with immediate resumption of normal behaviors to complete avoidance of the area and could also include changes in diving patterns or changes in foraging behavior.

The 166 dB re 1 µPa RMS isopleth (radius) would extend 1,359 to 3,414 m from the pile being driven (resulting in a maximum ensonified area of 36.6 km^2 (see Figure 5) and would persist for the duration of pile driving activities (up to 8 hours per day, for 20 non-consecutive days). Sea turtles are present in the action area during the warmer months, typically from May or June through October or November, depending on weather and water temperatures in particular years. This time period overlaps with the period when pile driving will occur. There are no available estimates of the number of sea turtles specifically in the action area generally or the area where noise will be greater than 166 dB re 1uPa RMS during pile driving. Sea turtles in the area could be migrating, resting or foraging; sea turtles within 3.4 km of the pile being driven are expected to temporarily stop these behaviors and make evasive movements (changes in diving or swimming patterns) until they are outside the area where noise is elevated above 166 dB re 1uPa RMS. Given that the piles will be installed in an open ocean environment with no impediments to movement, we do not expect any instances where a sea turtle would not be able to avoid the sound source.

Figure 5: Impact Pile Driving: Ensonified Area to the 166 dB_{RMS} Isopleth



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[53]) 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.

[53] 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.

Kraus *et al.* (2013 DRAFT), presents SPUE-based density estimates for loggerhead and leatherback sea turtles in the Massachusetts Wind Energy Area which is in close proximity to the action area. During this survey effort, 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 in nearby waters ($0.510/km^2$ for loggerheads and $0.037/km^2$ for leatherbacks), and the area where noise levels greater than 166 dB re 1uPa will be experienced during impact pile driving (36.6 km^2), 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 18 loggerheads and 2 leatherback sea turtle are likely to be exposed to potentially disturbing levels of noise during each day of pile driving. Over the 20-day pile driving period, we would expect that up to 360 loggerheads and 40 leatherbacks 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. This assumption is supported by the sightings data reported by Kraus et al. (2013 DRAFT) of no green sea turtles and only 6 Kemp's ridleys (compared to 93 leatherbacks and 76 loggerheads). Therefore, during each day of pile driving, no more than 2 Kemp's ridley and 2 green sea turtles are likely to experience potentially disturbing levels of noise. In total, we expect no more than 40 Kemp's ridleys and 40 green sea turtles to be exposed to potentially disturbing levels of noise from the impact pile driving. 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, and it uses the maximum distances modeled for noise attenuation. 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 impact pile driver.

Sea turtles migrating through the area when pile driving occurs are expected to adjust their course to avoid the area where noise is elevated above 166 dB re 1uPa RMS. Depending on how close the individual is to the pile being driven, this could involve swimming up to 3.4 km. The turtle may experience physiological stress during this avoidance behavior but this stressed state would resolve once the sea turtle had swam away from the area with disturbing levels of noise. Sea turtles typically cruise (i.e., swim at their normal speed) at speeds of 1.4-2.25 km per hour. This suggests that even at a normal, non-agitated, swimming speed, sea turtles would be able to swim out of the area with disturbing levels of noise within less than 2 hours assuming that they take a direct route. Given that this is a single sound source, that is of low intensity, we believe this is a reasonable assumption. Thus, the stressed state would be temporary. Similarly, any disruption or delay in foraging or resting would be temporary and persist only as long as it took the sea turtle to swim away from the noisy area. Resting or foraging would resume once the sea turtle left the noisy area. Even if a sea turtle wanted to return to the area it was displaced from, it would be displaced for no more than the 8 hours it took to install the pile. Migration is expected to continue with the avoidance representing a minor disruption to the migratory path.

While in some instances temporary displacement from an area may have significant consequences to individuals or populations this is not the case here. For example, if individual turtles were prevented from accessing nesting beaches and missed a nesting cue or were precluded from a foraging area for an extensive period, there could be impacts to reproduction and the health of individuals, respectively. However, the area where noise may be at disturbing levels is a small portion of the coastal area used for north-south and south-north migrations and is a tiny subset of the coastal Northeast waters used by foraging sea turtles. Therefore, although in the worst case, sea turtles may avoid or be temporarily excluded from the area with disturbing levels of sound for the duration of pile driving operations (i.e., 8 hours a day), the area from which an individual is being excluded is not essential to any turtle and the behaviors that would have been carried out in the area can be carried out elsewhere with only minor, short term costs to the individuals affected.

All behavioral responses to a disturbance, such as those described above, will have an energetic or metabolic consequence to the individual reacting to the disturbance (e.g., adjustments in migratory movements or foraging). It is believed that short-term interruptions of normal behavior are likely to have little effect on the overall health, reproduction, and energy balance of an individual or population (Richardson *et al.* 1995). As the disturbance will occur for only 8 hours a day, over a period of 20 non-consecutive days, sea turtles are not expected to be exposed to chronic levels of underwater noise and thus, chronic levels of disturbance that significantly impair essential behavior patterns. Thus, although there will be a temporary energetic consequence to any sea turtle disturbed by impact pile driving noise, due to the temporary nature of the disturbance, the additional energy expended is not likely to significantly impair essential life functions (i.e., foraging, migrations, nesting) or impair the health, survivability, or reproduction of an individual.

Based on this analysis, we have determined that any changes in behavior resulting from exposure to increased underwater noise associated with pile driving will temporarily disruptbehaviors including resting, foraging and migrating but the individual's ability to carry out these behaviors will resume as soon as the animal swims out of the noisy area or the pile driving ceases. Therefore, any impairment will be temporary and limited to a short term stress response and temporary shift in energy expenditures from the pre-disturbance behavior to evasive movements. Because of the short term nature of this disturbance, no sea turtles will be precluded or significantly impaired from completing any normal behaviors such as resting, foraging or migrating and we do not expect the fitness of any individuals to be affected. Additionally, while there will be a short term increase in energy expenditure, this is not expected to have any detectable effect on any present or future effect on growth, reproduction, or general health.

Based on the above analyses, although on an individual level, we expect temporary adjustments in individual behaviors, we do not expect the exposure of impact pile driving noise to result in injury or death by significantly impairing essential behavioral patterns for individual sea turtles. No population level effects are likely.

Effects to Sea Turtles – DP thruster

Underwater noise levels produced by DP vessel operation will produce underwater noise levels below those that may result in injury to sea turtles from a single exposure (i.e., 207dB re

$1 \mu Pa_{RMS}$). As a result, no sea turtles will be exposed to potentially injurious levels of underwater noise. Potentially disturbing levels of noise (greater than 166 dB RMS) extend a maximum of 8.6 m from the source. As the DP vessel is continually moving along the cable route over a 24 hour period, the ensonified area is constantly moving.

Assuming the worst case behaviorally, that individuals would avoid an area with underwater noise greater than 166 dB re 1 μPa, there would never be an area larger than $0.0002 km^2$ (232 square meters;) from which sea turtles might be temporarily excluded. Additionally, because the DP vessel is constantly moving, any one area is impacted for only a few minutes. Thus, the time period when an individual sea turtle could be expected to react behaviorally in an area is similarly limited to this short period.

Individual sea turtles in the action area are likely to be migrating through the area and may forage opportunistically while migrating. An individual migrating through the area when the DP vessel is being operated may change course to avoid the area where noise levels are above 166 dB re 1 μPa RMS; however, the furthest a turtle would need to swim to avoid the ensonified area would be less than 9 meters. This type of minor adjustment to movements is expected to happen without any stress response, increase in energy expenditure, or other physiological response. Because any changes in movements would be limited to momentary avoidance of an extremely small area, any disturbance is likely to have an insignificant effect on the individual. Similarly, any disruption to foraging or resting would be limited to no more than the few seconds it took the individual to move 9 meters and would quickly resume without any impact to the individual.

Effects to Sea Turtles-Installation and Removal of Sheet Piles
Sea turtles will only be present in the action area from June to October (although, some may remain through the first week of November) of any year (Morreale 1999; Morreale 2003; Morreale and Standora 2005; Shoop and Kenney 1992). All installation and removal of sheet piles is expected to occur between January 1 and May 1, when sea turtles do not occur in the action area. Therefore, no sea turtles will be exposed to any effects of sheet pile installation or removal.

Effects to Sea Turtles-Geophsyical Surveys
The multi-beam sonar and the chirper operate at frequencies outside the hearing bandwidths of sea turtles (i.e., between 100-2000 Hz for sea turtles; Ridgway *et al.* 1969; Lenhardt 1994; Bartol *et al.* 1999). Because sea turtles cannot perceive the sound associated with these surveys, there will be no effects to any sea turtles from the acoustic sources operated during the initial post-installation survey or any of the five-year maintenance surveys.

Effects to Sea Turtles-Vessel Noise
Noise levels that may elicit a behavioral response will only be experienced within several meters of the project related vessels. We do not expect sea turtles to be that close to any project vessel; therefore, we do not anticipate any behavioral disturbance from noise associated with the operations of the project vessels.

Effects to Sea Turtles-Operation of WTGs

The noise producing components of the WTG are at the nacelle, hundreds of feet above the water surface. Underwater noise is expected to be at or near ambient noise levels. Because of this, sea turtles will not be able to detect the operational noise of the WTGs as it is masked by other natural underwater noises. Because individuals will not perceive the noise, there will be no effects to any sea turtles.

7.1.3.3 Effects of Noise Exposure to Atlantic Sturgeon

Background Information on Underwater Noise and Sturgeon

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 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 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.* 2012)

Criteria for Assessing the Potential for Physiological Effects to Sturgeon
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) (206 dB_{Peak}).
- cSEL: 187 decibels relative to 1 micro-Pascal-squared second (dB re 1μPa^2-s) for fishes above 2 grams (0.07 ounces) (187 dBcSEL).
- cSEL: 183 dB re 1μPa^2-s for fishes below 2 grams (0.07 ounces) (183 dBcSEL).

At this time, they represent the best available information on the thresholds at which physiological effects to sturgeon from exposure to impulsive noise such as pile driving, 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 dBcSEL. 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, we consider the potential for physiological effects upon exposure to impulsive noise of 206 dB_{Peak} and 187 dBcSEL. Use of the 183 dBcSEL 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.

Available Information for Assessing Behavioral Effects on Sturgeon
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} at 1,000 to 2,000Hz.

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.

Anderson *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 (see below), it does demonstrate that significant noise-induced effects on behavior are possible, and that in addition to avoidance, fish may react to increased noise with a startle response or reduced foraging efficiency during the time of sound exposure.

For purposes of assessing behavioral effects of pile driving at several projects, NMFS has employed a 150 dB_{RMS} sound pressure level (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_{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_{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.

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, provided the operational frequency of the source falls within the hearing range of the species of concern. 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 a potential, upon exposure to noise at this level, to experience some behavioral response. We expect that behavioral responses could range from a temporary startle to avoidance of an area with disturbing levels of sound. The effect of any anticipated response on individuals will be considered in the effects analysis below.
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, so we will rely on sound pressure level criteria. Although we agree that more research is needed, the studies noted above support the 150 dB_{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 Atlantic sturgeon in response to pile driving noise. However, given the available information from studies on other fish species, we consider 150

dB_{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_{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_{RMS} as a threshold for examining the potential for behavioral responses. We will use 150 dB_{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.

Effects to Atlantic Sturgeon – Impact Hammer

Atlantic sturgeon in the area where piles will be installed are limited to adults and subadults making coastal migrations. As noted above, we expect potential injury to Atlantic sturgeon upon exposure to pile driving noises greater than 206 dB re 1µPa peak or 187 dB re 1uPa cSEL. When the 200 kJ hammer is used, noise attenuates to below 206 dB re 1µPa peak within 3 meters of the pile being driven; when the 600 kJ hammer is used, the area where noise is above 206 dB peak extends less than 7.5 meters from the source. To experience noise loud enough to cause injury with just a single exposure (i.e., one strike of the hammer), a sturgeon would need to be within 3 or 7.5 meters of the pile being driven. There are several factors that make this extremely unlikely to occur. First, Atlantic sturgeon are dispersed throughout the action area in relatively low numbers, making the likelihood of their occurrence in any particular area low. Only one Atlantic sturgeon has been captured in Rhode Island Sound during a trawl survey carried out by the Rhode Island Department of Environmental Management annually since 1997 (Greene et al. 2009) and no Atlantic sturgeon have been captured in a monthly trawl survey that has been ongoing in the action area since it began in September 2012.

Even if a sturgeon was very close to the pile installation site, all pile driving operations will be initiated with a "soft" start or a system of "warning" strikes that are designed to create enough noise to cause fish to leave the area prior to full energy pile driving; that is, the impact hammer will be operated at 40 percent of its total energy, which will result in the production of underwater noise levels at or above 150 dB_{RMS} (within seconds of the initiation of pile driving operations), but below 206 dB_{Peak}. At this energy level, warning strikes will consist of a set of 3 strikes on the pile, followed by a one minute waiting period; this will be performed two subsequent times. As described above, sturgeon are expected to respond behaviorally, via avoidance, upon exposure to bothersome levels of noise (greater than 150 dB re 1uPa RMS; see below for further assessment of behavioral effects). As a result, we expect any sturgeon that are close to the piles when pile driving begin, will detect the warning strikes and begin to move away from the noise source. Because the soft-start will take 3-5 minutes, we expect sturgeon to move more than 8 meters from the pile and therefore, never be exposed to a single strike peak noise of 206 dB re 1uPa.

In addition to the "peak" exposure criteria which relates to the energy received from a single pile strike, the potential for injury exists for multiple exposures to lesser noise. That is, even if an individual fish is far enough from the source to not be injured during a single pile strike, the potential exists for the fish be exposed to enough smaller-impact strikes to result in physiological impacts (this is the cSEL criteria). As described above, the cSEL is not an instantaneous maximum noise level, but is a measure of the accumulated energy over a specific period of time (e.g., the period of time it takes to install a specific structure, such as a pile). For the proposed action, it will take approximately 8 hours to install each pile with an impact hammer, with only one pile being driven per day. As such, it will take approximately 8 hours to attain cSEL values of 187dBcSEL, with this level being reached at a distance 8.576 km or 116.6 km from the pile to be driven with a 200 kJ or 600 kJ impact hammer, respectively. For an Atlantic sturgeon to be exposed to this level of underwater noise, the sturgeon would have to be present at the onset of pile driving operations within 8.6 km or 116.6 km of the pile, and would have to remain within this distance, for the full duration of pile installation (i.e., 8 hours), to experience this injurious level of underwater noise (i.e., 187 dBcSEL).

It is extremely unlikely that a sturgeon would remain within this distance of the pile being driven for the entire eight hour period. From the initiation to the completion of pile driving, disturbing levels of underwater noise will be produced within seconds of each strike of the pile and thus, well before any energy is accumulated to a level in which injury may occur. As described above, a soft start will be undertaken prior to the initiation of pile driving at full energy, and thus, will result in underwater noise levels (150 dB_{RMS}) that will result in the movement of Atlantic sturgeon away from the pile being installed. As each strike of the pile intensifies, the extent at which the 150 dB_{RMS} will be experienced will also increase; that is at full energy, underwater noise levels of 150 dB_{RMS} will be experienced at a distance of 39.8 km from the source. Thus, sturgeon that left the area during the initiation of pile driving will continue to divert their movements away from the sound source as pile driving operations continue and the area of behaviorally disturbing levels of noise increases. As a result, any sturgeon that may have been present at the onset of pile driving operations is not expected to be found within 8.6 km or 116.6 km of the pile, and thus, are not expected to remain within the area long enough to accumulate injurious pressure levels.

Based on this analysis, we do not expect any Atlantic sturgeon to be exposed to noise resulting from impact pile driving that could result in physiological effects including injury or mortality.

As described above, Atlantic sturgeon are expected to react behaviorally to underwater noise levels of 150 dB_{RMS} by demonstrating avoidance behaviors. Underwater noise levels of 150dB_{RMS} will extend a maximum of 39.8 km from the source, resulting in a maximum ensonified area of 4,979 km^2 (see
Table 9).

The action area is primarily used by Atlantic sturgeon transiting these waters as they complete coastal marine migrations, with migratory movements generally shifting southward in the fall, for overwintering purposes, and generally shifting northward in the spring, as adults return to natal rivers to spawn. Individual sturgeon that are within 40 km of the pile being driven are expected to make evasive movements to avoid the area where noise is disturbing. This will result in increased energy expenditure and a delay of resting and foraging. However, due to the temporary nature of the disturbance (i.e., 8 hours a day, over 20 non-consecutive days) and the transient nature of individuals in the action area, an individual Atlantic sturgeon is only likely to experience this disturbance once. One eight-hour period of increased energy expenditure to swim away from the noisy area will have short term costs to the animals energy budget, but would not result in a significant delay of any individual in accessing areas that are necessary essential behavioral functions (e.g., spawning grounds in natal rivers, such as the Hudson, or overwintering grounds off North Carolina) because this disturbance will be short lived. Further, during the time of year when pile driving will occur (May – October), Atlantic sturgeon are not likely to be moving to riverine spawning grounds (these movements would already be completed) or overwintering aggregations (these movements do not typically occur until water temperatures drop in the late Fall). However, they will be undertaking coastal marine migrations at this time, foraging and resting opportunistically. Thus, the behaviors that are most likely to be disrupted are migration, resting and foraging. However, because any disruption is expected to be temporary and limited in scope, we don't anticipate a significant impairment of the essential behavior functions of migration, resting and foraging. There is not expected to be any significant physiological consequence to increased energy exertion for a one-time eight hour period or an eight hour disruption to resting, migrating, or foraging.

All behavioral responses to a disturbance, such as those described above, will have an energetic or metabolic consequence to the individual reacting to the disturbance (e.g., adjustments in migratory movements or foraging). It is believed that short-term interruptions of normal behavior are likely to have little effect on the overall health, reproduction, and energy balance of an individual or population (Richardson *et al.* 1995). As the disturbance will occur for only 8 hours a day, for a period of 20 non-consecutive days, Atlantic sturgeon are not expected to be exposed to chronic levels of underwater noise and thus, chronic levels of disturbance that significantly impair essential behavior patterns. Thus, although there will be a temporary energetic consequence to any Atlantic sturgeon disturbed by impact pile driving noise, due to the temporary nature of the disturbance, the additional energy expended is not likely to significantly impair essential life functions (i.e., foraging, migrations, spawning, overwintering) or impair the health, survivability, or reproduction of an individual. Although on an individual level, we expect temporary adjustments in individual behaviors, we do not expect the exposure of impact pile driving noise to result in injury or death by significantly impairing essential behavioral patterns for individual Atlantic sturgeon. No population level effects are likely..

Effects to Atlantic Sturgeon – DP Thruster

Underwater noise levels produced by DP vessel operation will produce underwater peak underwater noise levels below those that may result in physiological impacts to Atlantic sturgeon from a single exposure (i.e., 206 dB re 1μPa_{peak}). However, we have considered whether Atlantic sturgeon could be exposed to lower levels of noise over time and also experience physiological impacts. As noted in Table 11, an Atlantic sturgeon would need to stay within 630 meters of the DP thruster for a period of 24 hours in order to accumulate enough energy to experience physiological impacts. Given the disperse and transient nature of Atlantic sturgeon in the action area, it is extremely unlikely that an individual would remain within 630 meters of the source for an entire 24 hours. This likelihood is further reduced by the transitory nature of the vessel; because the vessel is moving, an individual sturgeon would have to not only stay within 630 meters of the vessel but move along with it for the entire 24 hour period. Because Atlantic sturgeon in the action area are migrating through, it is not reasonable to anticipate that an individual would behave this way. Therefore, we have determined it is extremely unlikely any Atlantic sturgeon will be exposed to noise reaching 187 dB re 1uPa cSEL from the DP thrusters.

As noted above, 150 dB_{RMS} is believed to be a reasonable estimate of the noise level at which exposure may result in behavioral modifications to Atlantic sturgeon. This noise level may be experienced within 100 meters of the DP vessel. Any sturgeon within 100 meters of the DP thruster is expected to move away until it is outside of the area where noise is disturbing. However, the furthest an Atlantic sturgeon would need to swim to avoid the ensonified area would be 100 meters. This type of minor adjustment to movements is expected to happen without any stress response, increase in energy expenditure, or other physiological response. Because any changes in movements would be limited to momentary avoidance of an extremely small area, any disturbance is likely to have an insignificant effect on the individual. Similarly, any disruption to foraging, migrating or resting would be limited to no more than the few seconds it took the individual to move 100 meters and would quickly resume without any impact to the individual.

Effects to Atlantic Sturgeon-Installation and Removal of Sheet Piles

Atlantic sturgeon will only be present in the action area from June to early November of any year (Savoy and Pacileo 2003). All installation and removal of sheet piles is expected to occur between January 1 and May 1, when Atlantic sturgeon do not occur in the action area. Therefore, no sturgeon will be exposed to any effects of sheet pile installation or removal.

Effects to Atlantic Sturgeon-Geophysical Surveys

The multi-beam sonar and the chirper operate at frequencies outside the hearing bandwidths of Atlantic sturgeon (i.e., between 100-1000 Hz, see Meyer and Popper 2002; Popper 2005; Lovell *et al.* 2005; Meyer *et al.* 2010). Because Atlantic sturgeon cannot perceive the sound associated with these surveys, there will be no effects to any individuals from the acoustic sources operated during the initial post-installation survey or any of the five-year maintenance surveys.

Effects to Atlantic Sturgeon-Vessel Noise

Noise levels that may elicit a behavioral response will only be experienced within several meters of the project related vessels. We do not expect Atlantic sturgeon to be that close to any project vessel; therefore, we do not anticipate any behavioral disturbance from noise associated with the operations of the project vessels.

Effects to Atlantic Sturgeon-Operation of WTGs

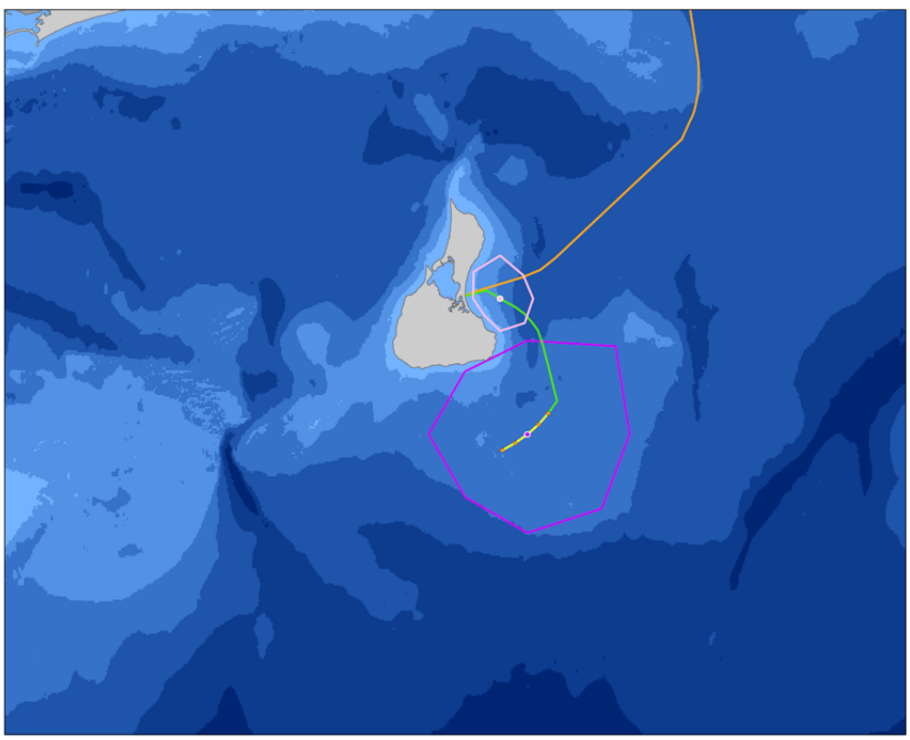
The noise producing components of the WTG are at the nacelle, hundreds of feet above the water surface. Underwater noise is expected to be at or near ambient noise levels. Because of this, Atlantic sturgeon will not be able to detect the operational noise of the WTGs as it is masked by other natural underwater noises. Because individuals will not perceive the noise, there will be no effects to any Atlantic sturgeon.

Effects of Noise Exposure: Cumulative Sound Effects from Pile Driving Operations and DP Thruster Use

During the final stages of export cable installation (DP thruster use) and WTG foundation installation (impact pile driving) some overlap of construction will occur, and therefore, an overlap in sound fields is possible. Should this occur, an individual could be exposed to both noise sources at the same time.

Based on information provided to us from Deepwater Wind (November 8, 2013, memo), due to the logarithmic nature of the decibel scale, sound energy added together results in a 3 dB increase, not a doubling of sound energy. Therefore, if the sound contour/ZOI of the DP vessel and impact pile driving intercept, while an incremental increase in sound is possible, a doubling of sound energy would not result (Deepwater Wind 2013). As the underwater noise produced from WTG foundation installation is stationary, relative to the movement of the DP vessel, the only period of time in which there will be an overlap of underwater noise sources will be when a DP vessel is approaching within or near the area where the WTG foundations are being installed. This is a relatively small area, both in time and space, for these cumulative noise impacts to be experienced. For example, Figure 6 depicts the separate ZOI for the 120 dB isopleth (DP thruster use: 25.1 km^2 and the 160 dB RMS isopleth for installation of piles with the 600 kJ impact hammer: 89.6 km^2.

Figure 6: Marine Mammal Behavioral Threshold ZOI during DP Thruster (Non-impulsive Noise)(Light Pink) and WTG Foundation Installation (Impulsive Noise)(Dark Pink)



*DP thruster ZOI modeled at a depth of 20 meters.
**Impact Pile Driving ZOI modeled during the use of the 600kJ impact pile driver

Assessing Figure 6, even if the DP vessel is positioned so that the 120 dB_{RMS} ZOI lies entirely within the 160 dB_{RMS} ZOI, the net increase in underwater sound levels that would potentially extend the distance to the 160 dB_{RMS} ZOI would be small and in fact may not extend at all due to the differences in characteristics between the pulse versus continuous waveform such as energy of the signal and length of the pulse. Additionally, as the ZOI for impact pile driving is always larger than that for DP thruster operations, anywhere there is overlap between the two sources, the ZOI for impact pile driving will encompass that of the DP thrusters. As such, the cumulative ZOI for both activities would be reflective of the ZOI estimated for impact pile driving on its own; that is, during concurrent overlap of activities, the ZOI is not expected to be greater than or extend father than the already established ZOI for solely impact pile driving. The same conclusions would hold true for sea turtles and Atlantic sturgeon and their associated ZOI for DP thruster use and impact pile driving operations (see Sections 7.1.3.1 and 7.1.3.2).

Although there will be a minor (approximately 3 dB) net increase in underwater noise levels where the DP thruster and impact pile driving sound fields overlap, this increase in underwater noise level will not be significantly different than those described and assessed above and the distance to the isopleths of concern does not change. Therefore, effects to our listed species will remain as described above for impact pile driving or DP thruster noise; that is, we no do not expect any cumulative acoustic impacts to our species that will differ from that what we described above (see Sections 7.1.3.1 and 7.1.3.2) for each activity operating on its own.

7.1.4 Effects of Exposure to Project Vessels

The construction and maintenance of the proposed project requires the use of a variety of vessels. During the construction of the BIWF (from foundation transport to actual construction), approximately 17 vessels, of varying types (e.g., tugs, transportation barges, derrick barges, work vessels, cable lay barge, jack up barges) will be present in the waters of Block Island and Rhode Island Sound. These vessels will transit to the project site from the staging area in Quonset, Rhode Island. Vessels will be present in these waters for a period of 27 days to up to a year, and will remain in one particular location, as construction needs warrant, for a period of 14 to 24 hours a day.

During the construction of the BITS, approximately 8 vessels, of varying types (e.g., tugs, material barges, work vessels, cable lay barge) will be present in the waters of Block Island and Rhode Island Sound. These vessels will be present in these waters for a period of 27 to 125 days, and will remain in one particular location, as construction needs warrant, for a period of 14 to 24 hours a day. These time periods do not characterize the amount of time that vessels will be operational within the action area as the vessels are expected to be largely stationery once they reach the project site or the staging areas; the vessels will be operating for only a fraction of this time.

Once all the WTGs for the BIWF have been installed, commissioning of the WTG will involve testing the WTGs and transmission system's capabilities to meet standards for safety and grid interconnection reliability. This testing will require technicians to frequently travel to the WTGs over the approximately five month commissioning period. As such, a crew workboat will frequently transport technicians to the WTGs from Point Judith, Rhode Island, where support vessels are stationed. The number of vessels necessary for commissioning is expected to be small (e.g., no more than five).

As described in Section 3.6.2, a small number of vessels used for maintenance and repair activities will be stationed out of Point Judith, Rhode Island. Approximately five vessels will be used to make occasional trips to each of the five WTGs.

Several measures are being undertaken to minimize the potential of interactions between project vessels and listed species. During the period of November 1-April 30, the mid-Atlantic Seasonal Management Area for right whales is effective; in this area, which overlaps with a portion of the action area, including the area where the WTGs will be built, the speed of all vessels greater than 65 feet in length must be no greater than 10 knots. In addition, USACE will require that all project vessels, regardless of length, operate at speeds less than 10 knots during the November 1 – April 30 time period, regardless of whether they are inside or outside of the designated SMA. During the May 1 – October 30 time period, smaller crew support vessels may operate at speeds of up to 15 knots. Tugs and barges, especially when transporting a full load, will travel at considerably slower speeds (less than 5 knots). The vessel carrying out surveys along the cable route will also travel slowly, at speeds of approximately 3 knots; as will the vessel laying down the cable.

All vessels associated with the construction, operation, maintenance and repair, and decommissioning of the BITS and BIWF will adhere to NMFS guidelines for marine mammal ship strike avoidance (see (http://www.nmfs.noaa.gov/pr/pdfs/education/viewing_northeast.pdf**)**, including maintaining a distance of at least 500 yards from right whales, at least 100 feet from all other whales, and having dedicated lookouts and/or protected species observers posted on all vessels who will communicate with the captain to ensure that all measures to avoid whales and sea turtles are taken. These measures can include slowing down or maneuvering away from any whales or sea turtles that are observed.

Collision with vessels remains a source of anthropogenic mortality for listed species of sea turtles, whales, and sturgeon. The proposed project will lead to increased vessel traffic in the action area that would not exist but for the proposed action. We have considered whether this increase in vessel traffic could result in an increased risk of vessel strike to listed species. 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 a cause of mortality to Atlantic sturgeon and 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.

Effects to Whales

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 2003). 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 2003; Laist *et al.* 2001; Vanderlaan and Taggart 2006). Based on ship strike records from 1998 to the present, only 5 whales (3 fin, 1 humpback, and 1 blue whale) have been documented as being struck by a vessel within a 29 mile radius of Block Island (pers. Comm; David Morin, NMFS Marine Resources Management Specialist, December 5, 2013; radius established by establishing an initial 3 mile radius of the southern tip of Block Island and adding an additional 26 miles, which accounts for the distances from shipping channels in which animals that may have been struck are likely to be found (Knowlton and Kraus 2001)). This is a rate of approximately 0.3 strikes/year in this area.

There are no estimates of ship traffic on a daily or annual basis for the action area specifically. However, as part of the development of the Rhode Island Ocean SAMP, an effort was made to characterize vessel traffic in that area, of which the action area is a significant portion. The action area is frequented by a wide variety of commercial and recreational boat traffic. Vessels transiting through this area include vessels accessing the commercial ports of Quonset, Providence and Fall River as well as passenger ferry and cruise ship terminals and Naval port facilities in Newport and Quonset. For the year 2007, an estimated 2,600 commercial transits occurred through this area. There are also five ferry companies operating 2-18 trips each per day as well as 120 cruise ship transits and multiple Naval and USCG transits as well as hundreds of recreational and commercial vessel traffic transits each year. This indicates that there at least 3,000 vessel moving to and from Narragansett Bay through the action area each year. In addition, part of the commercial traffic moving through the Ocean SAMP area consists of vessels traveling coastwise. Many of these ships are tug and barge units carrying petroleum products; these vessels originate in the Port of New York and New Jersey or points south and travel to and from Buzzards Bay and the Cape Cod Canal. There are also ships transiting to and from Long Island Sound via Block Island Sound Exact numbers of coastwise transits through the Ocean SAMP area are not available; however, traffic data from Long Island Sound and the Cape Cod Canal provide an approximation of traffic traveling through this area associated with surrounding East Coast ports. In 2006, the U.S. Coast Guard estimated that there may be 2,004,000 transits through Long Island Sound each year; those transits leaving the eastern end of Long Island Sound must pass through the Ocean SAMP area. In 2005, 443 foreign-flagged vessels were recorded traveling through the SAMP area, destined for ports within Long Island Sound (U.S. Coast Guard 2006). And in 2007, 649 foreign vessels were recorded passing through the Cape Cod Canal (U.S. Army Corps of Engineers 2007), thus passing through Buzzards Bay into the Ocean SAMP area.

This information suggests that there are many thousands of vessels transiting through the action area each year. The proposed additional of no more than 17 vessels at any one time represents an extremely small fraction of the existing vessel traffic in the action area. As noted above, there have been an average of 0.3 interactions between whales and vessels in the general area with at least 3,000 vessel transits just to and from ports within Narragansett Bay each year. Even assuming that the risk of ship strike is proportional to vessel traffic and using the maximum number of project vessels and assuming they were transiting the action area every day for a year, the risk of a strike is 0.0017 whales/year. The use of best management practices including reduced speeds and dedicated lookouts is expected to lower this even further. As such, we have determined that a vessel strike is extremely unlikely to occur.

Although there is the potential that some vessels may attain speeds of up to 15 knots, given the required separation distances from whales (at least 100 meters), in combination with 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. As a result, we have concluded that the potential interaction between a vessel and a listed species of whale is discountable.

Effects to 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 do not occur in the action area, therefore there would be no impacts to this life stage.

Information is lacking on the type or speed of vessels involved in turtle vessel strikes. However, there does appear to be a correlation between the number of vessel struck turtles and the level of recreational boat traffic (NRC 1990). Although little is known about a sea turtle's reaction to vessel traffic, it is generally assumed that turtles are more likely to avoid injury from slower-moving vessels since the turtle has more time to maneuver and avoid the vessel. Hazel *et al.* (2007) reported that green sea turtles ability to avoid an approaching vessel decreases significantly as the vessel speed increases. As vessels in the action area will be operating at slow speeds (i.e., no more than 15 knots) and with a designated lookout, vessel interactions with sea turtles are not expected. Based on this information, the potential for a strike to sea turtles from these vessels is discountable.

Effects to 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 very 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 (i.e., 20 to up 129 feet) interactions between surface vessels and fish at or near the bottom are extremely unlikely. Based on these factors, the potential for an increased risk of strikes to Atlantic sturgeon from the increase in vessel traffic is discountable.

7.1.5 Interactions Cable Lay Equipment

As described in sections 3.1.2.4 and 3.2.2, the installation of the BITS, export, and inter-array cables will also require the deployment, from the cable lay barge, of a jet plow and cable. Jet plows move along the benthos at slow speeds (i.e., < 1 knot). As sea turtles and sturgeon are highly mobile, any sturgeon or sea turtle that may be present at or near the benthos will be able to move out of the way of the device, thereby avoiding an interaction. Although any sea turtles or sturgeon present in the vicinity of the jet plow may be displaced, displacement would be temporary (i.e., for the duration of the jet pass; approximately several minutes) and will only result in a temporary shift in swimming direction away from the area affected by the jet plow for up to several minutes. This displacement is not likely to affect the ability of the individual to complete any essential life functions (i.e., opportunistic foraging, resting, migrating) that may take place along the cable route as any animals that may have moved from the affected area will be able to continue normal life functions in other nearby unaffected areas and will also be able to resume these behaviors once the jet plow has passed. Additionally, as the cable will be taut as it is unrolled and laid in the trench, there is no risk of entanglement. Based on this information, we believe that it is extremely unlikely that sea turtles or sturgeon will directly interact with cable laying and jetting equipment and thus, believe that an interaction with these pieces of equipment is discountable.

In regards to listed species of whales in the action area, interactions with the jet plow or cable are not expected. For an interaction with the jet plow to occur, a whale would have to be at the benthos within the vicinity of the jet plow. Listed species of whales will not occur on the benthos and thus, any interactions with the jet plow will not occur. In addition, as noted above, as the cable will be taut as it is unrolled and laid in the trench, there is no risk of entanglement. Based on this information, we have concluded that a whales interaction with a jet plow or cable piece of cable-lay equipment is discountable.

7.2 Operations and Maintenance and Repair

7.2.1 Operations

7.2.2.1 Electromagnetic Field

The cable system for the BIWF inner-array and export cables and BITS is a dielectric AC cable, consisting of a core of 3-phase conductors encased by grounded metallic (i.e., lead) shielding that effectively blocks any electric field generated by the operating cabling system. Since the electric field will be completely contained within those shields, impacts are limited to those related to the magnetic field emitted from the BITS and export and inter-array cables; however, the magnetic field produced by these cables is expected to be weak as the containment of all three phases of each circuit within the submarine cable results in the significant cancellation of the magnetic fields produced by the circuit as a whole.[54] As the magnetic field of a cable decreases rapidly with increasing distance from the source, the magnetic field of the BITS, export, and inter-array cables is also expected to be weak due to the burial depth of these cables (i.e. between 4 and 8 feet, depending on benthic conditions or presence of utility crossings) (TetraTech 2012). Additionally, the frequency of the magnetic field of the BITS, export and

[54] Induced eddy currents in conductive sheathing materials will create opposing magnetic fields that partially cancels the magnetic field from the core (TetraTech 2012).

inter-array cables will be 60-Hertz(Hz); at this frequency most marine species, such as sea turtles, whales, and sturgeon, are not likely to sense very low intensity magnetic fields, such as those likely to be produced by the proposed action (Normandeau *et al.* 2011).

Research on EMF also indicates that although high sensitivity has been demonstrated by certain species (especially sharks) for weak electric fields, this sensitivity is limited to steady and slowly-varying fields (Cape Wind Tech Report; ICNIRP 2010; Adai 1994; Valberg *et al*. 1997 in BOEM 2008; Normandeau *et al*. 2011). The proposed action produces 60-Hz time-varying fields and no steady or slowly-varying fields. Likewise, evidence exists for marine organisms utilizing the geomagnetic field for orientation, but again, these responses are limited to steady and slowly-varying fields. 60-Hz alternating power-line EMF fields, such as those generated by the proposed action, have not been reported to disrupt marine organism behavior, orientation, or migration. Based on the body of scientific evidence, there are no anticipated adverse impacts expected from the undersea power transmission cables or other components of the proposed action on the behavior, orientation, or navigation of marine organisms, including listed sea turtle, Atlantic sturgeon, or whale species, or their prey species. Based on this and the best available information, the magnetic fields associated with the operation of the cable systems are not anticipated to result in any adverse impacts to listed sea turtles, whales or Atlantic sturgeon during the normal operation of the inner-array, export, or BITS cables and thus, any effects to these species are expected to be insignificant or discountable.

The burial depth of the cables also minimizes potential thermal impacts from operation of these cable systems. In addition, the inner-array and submarine cable systems utilize solid dielectric AC cable designed for use in the marine environment that does not require pressurized dielectric fluid circulation for insulating or cooling purposes. There will be no direct impacts to Atlantic sturgeon, sea turtle, or whale species during the normal operation of the export, inner-array or BITS cable systems. There will also be no impacts to prey species of sea turtles, sturgeon, or whales during the normal operation of theses cable systems.

7.2.2 Maintenance and Repair

Periodic maintenance and/or repairs to the BITS or to the BIWF's WTGs, export or inter-array cables will be necessary throughout the 29 year life of the project. Annually, the WTG and its foundation will be inspected (the latter with divers and/or remotely operated vehicles (ROVs), with each WTG requiring 3 to 5 days for inspection. The submarine cables will also be inspected annually, via a survey vessel towing a sub-bottom profiler (chirper), to ensure cable burial depths are maintained. The majority of maintenance and repair activities will thus, involve a limited number of small vessels similar to the support vessels used during construction or previous cable geophysical surveys.

As noted above, in addition to vessels, equipment involved in routine maintenance operations includes ROVs and towed sub-bottom profilers. Hand operated devices, such as ROVs , move at slow speeds as do sub-bottom profilers, which are towed slowly behind the survey vessel. As listed species of whales, sturgeon, and sea turtles are highly mobile, they are likely to be able to avoid contact with the ROV or towed sub-bottom profiler. Although avoidance of the maintenance/repair equipment may result in the temporary displacement of the species from the area, however, there is no evidence to suggest that whales, sea turtles, or Atlantic sturgeon are more attracted to the resources along the BITS, export, or inter-array cable routes or WTGs foundations than to those in surrounding waters and thus, similar to foraging impacts experienced during construction (see section 7.1.1 Foraging and Habitat Modification), the temporary displacement to neighboring areas is not likely to have a significant impact on foraging success or the completion of any other essential life functions of any listed species. Based on this and the best available information, the likelihood of listed species colliding or directly interacting with maintenance or repair equipment is discountable, and the effect of any associated displacement will be insignificant.

7.2.2.3 Habitat Disturbance

As described above, maintenance and repair activities will involve the use of different types of support vessels, similar to those used during construction, and may also involve jetting techniques to re-bury any cables. Support vessels are likely to use anchors to stabilize the vessels during maintenance and repair operations and thus, the placement of the anchor and the anchors associated anchor chain sweep, is likely to disturb the benthos (i.e., increase levels of TSS) and remove any benthic infaunal or surface dwelling organisms in the pathway of the anchor and its chain. In addition, although geophysical surveys themselves will not affect the benthic habitat of the action area, the resultant findings of the survey may. That is, should surveys reveal sections of the cable route where the cable has not attained target burial depths, concrete matting or rock piles will be placed on top of those sections. Effects of these activities to listed species of sea turtles, whales and Atlantic sturgeon; however, are not expected to be greater than those resulting from construction activities. As a result, effects to listed species of whales, sea turtles, and Atlantic sturgeon from habitat modification are expected to be similar to those described above in Sections 7.1.1 (Foraging and Habitat Modification resulting from Construction) and Section 7.1.2 (Water Quality resulting from Construction) and thus, are expected to be insignificant (please see above for further analysis).

7.3 Decommissioning

Decommissioning of the BIWF will follow the same relative sequence as construction, but will occur in reverse. The WTG components will be removed by a jack-up lift vessel or a derrick barge and lifted onto a material barge. The material barge will transport the components to a recycling yard where the components will be disassembled and prepared for re-use and/or recycling for scrap steel and other materials. The foundations will be cut by an internal abrasive water jet cutting tool, which will involve the placement of cutting tool within the pile and lowering it to area approximately 10 feet below the seabed. Once cutting is complete, the balance of the foundations will be removed using 500-ton derrick barges and lifted onto material barges. In regards to BIWF's export and inter-array cables and the BITS, all cables will be abandoned in place.

Based on the above, as all cutting operations will occur within the foundation piles, and at a distance within the pile located 10 feet below the sea bed, no elevated levels of underwater noise will be produced during the cutting of the piles. As a result, listed species will not be exposed to any disturbing or injurious levels of underwater noise. In addition, significant disturbances to the sea floor is not expected. Small disturbances to the sea floor may occur when the foundations piles are lifted from the water, however, any disturbance is expected to be small and is expected to remain confined to the area of disturbance. As the cables will remain in place, significant disturbances to the seafloor will not be incurred via the decommissioning of these structures. Although vessels will be present during the decommissioning of the WTG, the number of vessels to be used during decommissioning is not expected to be any greater than that which was used during construction (i.e., approximately 18 vessels). As such, we expect any vessel related impacts to listed species to be similar to those described in section (7.1.4). Based on this information, we expect any effects to sea turtles, Atlantic sturgeon, and whales from decommissioning activities to be insignificant.

7.4 Other Project Related Impacts

7.4.1 Light Pollution

Most construction activities (pile driving, WTG assembly) will be limited to daylight hours. However, cable laying operations would take place 24 hours per day, 7 days a week during installation. The submarine transmission cable will take approximately 2-4 weeks to complete and the inner array cable will be installed over several months. Construction and support vessels would be required to display lights when operating at night and deck lights would be required to illuminate work areas. However, lights would be down shielded to illuminate the deck, and would not intentionally illuminate surrounding waters. If sea turtles, Atlantic sturgeon, whales, or their prey are attracted to the lights, it could increase the potential for interaction with equipment or associated turbidity. However, due to the nature of project activities and associated seafloor disturbance, turbidity, and noise, listed species and their prey are not likely to be attracted by lighting because they are disturbed by these latter disturbances. As such, we have determined that any effects of project lighting on sea turtles, sturgeon, or whales will be insignificant.

In addition to vessel lighting, the WTGs will be lit for navigational and aeronautical safety. 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 Massachusetts, and hatchlings are not known to be present in Massachusetts waters. As result, surface lighting on the WTGs will have no impacts to nesting or hatchling sea turtles.

7.4.2 Air Emissions from Project Vessels

Air emissions are not produced by the BIWF and BITS; however, the vessels associated with construction, maintenance/repair, and decommissioning of the structures will produce air emissions; however, based on the information presented in TetraTech's Environmental Report for Deepwater Wind, any emissions will be minor and short-term, and overall, will not negatively affect air quality in Rhode Island. The United States Environmental Protection Agency (EPA) has conducted an air emissions analysis resulting from the construction, maintenance/repair, and decommissioning of the structures, pursuant to the Clean Air Act, the EPA has determined that as the state of Rhode Island has attained EPA's new ozone standard of 0.075 parts per million (ppm) (EPA 2012), emissions associated with BIWF and BITS (i.e., those air emissions attributed to vessels constructing, servicing, decommissioning the WTGs or BITS) will not be subject to EPA's General Air Conformity Requirements of the Clean Air Act (42 USC & 7401 et. seq). The EPA has also explained that the project's peak emissions will not result in any exceedance of any currently attained primary or secondary National Ambient Air

Quality Standards (NAAQS). Primary NAAQS are set to protect public (human) health with an adequate margin of safety, including the health of "sensitive" populations such as asthmatics, children, and the elderly. Secondary NAAQS set limits to protect public welfare, including protection against decreased visibility, damage to animals, crops, vegetation, and buildings. In addition, while there will be some emissions associated with the construction, maintenance/repair, and decommissioning of the BIWF and BITS, according to the Rhode Island Ocean Special Area Management Plan, offshore wind facilities (including the associated submarine cables) will produce far fewer emissions of criteria pollutants, hazardous air pollutants, and greenhouse gases than fossil fuel burning generators currently operating in Rhode Island (e.g., 98% of the Green House Gases (GHG) emissions emitted in Rhode Island are from fossil fuel combustion and GHG emissions associated with electricity imported into the state represent 50% of the emissions generated within the state (Brown University 2000; the BIWF will represent approximately 1.2 % of Rhode Island's forecasted generation of fossil fuels (TetraTech 2012)). As a result, overall, the BIWF and BITS, is expected to provide Rhode Island, including Block Island, with "measurable environmental benefits" including, but not limited to, a regional reduction in air pollution (RIDEM 2010).

Based on this and the best available information, any effects to air quality from the proposed action are likely to be insignificant. At this time, there is no information on the effects of air quality on listed species that may occur in the action area. However, as the emissions regulated by EPA and the State will have insignificant effects on air quality, it is reasonable to conclude that any effects to listed species from these emissions will also be insignificant.

7.4.3 WTG Foundation: Habitat Shift

The presence of five WTG foundations, with 4 piles a piece, in Rhode Island Sound and their associated scour control sand/cement bags have the potential to shift the area immediately surrounding each pile foundation 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 pile of the foundation and an increased availability of shelter among the pile structure. The WTG 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 WTG 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 WTG 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 WTG foundations 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. If the WTG foundations led to an increase in schooling fish around the piles, it is possible that individual whales could be attracted to the foundations. However, the small number of foundations and total number of piles associated with all 5 WTG foundations (i.e., total of 20 piles over the entirety Rhode Island Sound, an area of more than 617,763 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 or movements of whales 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 5 WTG foundations. Despite possible localized changes in prey abundance and distribution, any changes are expected to be small due to the small number of WTG foundations and the distance between them (i.e., 0.5 miles apart). Therefore, any effects to Atlantic sturgeon foraging are expected to be minor and localized.

Although the WTG 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 20 piles over an 617,763 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 WTG foundations (0.5 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 WTG foundations may affect the localized movements of sea turtles and sturgeon 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.

7.4.4 Marine Debris

Personnel will be present onboard the vessels throughout construction, commissioning, maintenance and repair, and decommissioning activities, thus presenting some potential for accidental releases of debris overboard. ESA listed species of whales, sea turtles, and Atlantic sturgeon can be adversely affected by such debris should they become entangled in or ingest debris, particularly plastics that are mistaken for prey items. The discharge and disposal of garbage and other solid debris from vessels by lessees is prohibited the USCG (MARPOL Annex V, Public Law 100-220 [Statute 1458]). The discharge of plastics is strictly prohibited. Deepwater Wind will also ensure all crew supporting the construction, operation, maintenance, repair, and decommissioning of the BIWF and BITS will undergo marine debris awareness training. Based on this training, during construction, operation/maintenance/repair, and decommissioning activities, individual crew members will be responsible for ensuring that debris is not discharged into the marine environment. Additionally, training of construction crews will include a requirement explaining that the discharge of trash and debris overboard is harmful to the environment, and is illegal under the Act to Prevent Pollution from Ships and the Ocean Dumping Ban Act of 1988. Therefore, discharge of debris will be prohibited, and violations will be subject to enforcement actions. Therefore, activities associated with the construction, operation and maintenance, and decommissioning of the BIWF and BITS are not likely to result in increased marine debris, and thus, are not expected to affect ESA listed species of sea turtles, whales, or Atlantic sturgeon.

7.4.5 Pre-lay Grapnel Run

Prior to submarine cable installation, a pre-lay grapnel run will occur to remove any obstructions of debris along the cable route. The pre-lay grapnel run will involve towing a grapnel, via the main cable laying vessel, along the benthos of the cable burial route. During the pre-lay grapnel run, the cable-lay vessel will operate and thus, tow the grapnel at slow speeds (i.e., approximately 1 knot or less) to ensure all debris is removed. As sea turtles and sturgeon are highly mobile, any sea turtle or sturgeon that may be present at the bottom will be able to move out of the way device, thereby avoiding an interaction. Additionally, as the cable of the grapnel run will remain taught as it is pulled along the benthos, there is not risk for sea turtle, whales, and Atlantic sturgeon entanglement. Disturbance of the benthos/sediments (e.g., turbidity) and removal of benthic invertebrates are also likely during this phase of the project; however, the degree of this disturbance is expected to be no greater than those assessed for jetting operations and thus, for the same reasons provided with regard to the effects of jetting operations, we have concluded that effects to ESA listed species of sturgeon, sea turtles, and whales from pre-lay grapnel run activities are insignificant (see Sections 7.1.1 and 7.1.2).

7.5 Unexpected Project Events

7.5.1 Fuel or Oil Spill

A fuel or oil spill could result from damage to vessels used during construction, operation/maintenance, or decommissioning or from the unexpected collapse (due to a storm event, a large vessel interaction) of a WTG. Any oil or fuel spill; however, would be an unintended, unpredictable event and should an event occur, Deepwater Wind will follow their Spill Prevention and Control and Counter Measures Plan as well as the USCG's oil spill prevention and response plans (in accordance with the Oil Pollution Act of 1990 (OPA-90) and MARPOL 73/78). As such, fuel and oil and spill events are extremely unlikely to occur and thus, any effects to our species is discountable.

7.5.2 HDD Drilling Fluid Release

As described above, during cable landing operations, a HDD will need to be used to establish the conduit for the terrestrial cable to be spliced with the submarine cable. Although these activities will occur above the mean high water mark, there is the potential, albeit unlikely, for HDD drilling fluid to be released into the water column. Additionally, HDD operations also create the potential for a frac-out, which occurs when drilling fluids migrate unpredictably to the surface through factures, fissures, or other conduits in underlying rock or unconsolidated sediments, thus, entering the water column. Based on information provide in the ER (ER), should such an incident occur, the fluid release, which is non-toxic and comprised of clays and rock particles, will be small and localized and is expected to result in a temporary increase in turbidity and sedimentation in the shallow nearshore environment where HDD operations will occur. To minimize the potential for drilling fluid release or potential risks associated with a frac-out, Deepwater Wind will implement best management practices, including a HDD Contingency Plan for the Inadvertent Release of Drilling Fluid prior to construction. Based on this, we have concluded that affects to ESA listed species of whales, sea turtles, and sturgeon and their prey from drilling fluid release are insignificant. Further, this release is extremely unlikely to occur.

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."

Given the nature of the action area (i.e., nearshore and offshore areas off the coast of Rhode Island and Block Island), 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 portions of the action area that overlaps with state waters include the BIWF (including the export and inter-array cables), portions of the transit routes that may be used by project vessels, and portions of the BITS submarine cable routes. 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, vessel interactions, and pollution. 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.

Vessel Interactions- As noted in the Environmental Baseline section, private vessel activities in the action area may adversely affect listed species in a number of ways, including entanglement, boat strike, or harassment. As vessel activities will continue in the future, the potential for a vessel to interact with a listed species exists; however, the frequency in which these interactions will occur in the future is unknown and thus, the level of impact to sea turtle, whale, or Atlantic sturgeon populations cannot be projected. 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/environmental baseline section.

Pollution and Contaminants – Human activities in the action area causing pollution are reasonably certain to continue in the future, as are impacts from them on Atlantic sturgeon, sea turtles, or whales. However, the level of impacts cannot be projected. Sources of contamination in the action area include atmospheric loading of pollutants, stormwater runoff from coastal development, groundwater discharges, and industrial development. Chemical contamination may have an effect on listed species reproduction and survival. 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/environmental baseline section.

9.0 INTEGRATION AND SYNTHESIS OF EFFECTS

The effects of the proposed action include: habitat disturbance resulting in potential impacts to water quality and prey; exposure to increased underwater noise; exposure to increased vessel traffic; exposure to cable lay equipment; electromagnetic fields; and unintended or unplanned events including oil spills. We have determined that the only stressor that is likely to result in adverse effects to listed species is noise. The source levels associated with the installation of the wind turbine support piles with an impact hammer result in large areas with noise levels that are potentially disturbing for right, humpback and fin whales, loggerhead, Kemp's ridley, green and leatherback sea turtles and Atlantic sturgeon. We expect these animals to alter their behavior from foraging, rearing, migrating, and resting to make evasive movements away from the area with disturbing levels of noise. This may result in stress to these animals and may come at a metabolic and energetic cost. However, because this response is limited to only a few hours, the stress will resolve and not result in distressed individuals and any disruption to essential behaviors will be temporary. We do not anticipate any injury or mortality immediately or in the future and do not anticipate any reduction in fitness. We have determined that this behavioral disturbance is considered "harassment" under the ESA definition of take. This harassment will occur in the form of avoidance or displacement from habitats and behavioral and/or metabolic/energetic compensation to deal with short term (hours) of stress resulting from exposure to disturbing levels of noise, which modify or degrade habitat used by whales, sea turtles, and Atlantic sturgeon. While these individuals may experience temporary disruption of behavior patterns, we do not anticipate that the habitat modification caused by noise will actually kill or injure listed whales, sea turtles, or Atlantic sturgeon by significantly impairing essential behavioral patterns due to the temporary nature of any effects and the ability of individuals to resume pre-disruption behaviors once the disturbance has ceased. In the effects of the action section of this Opinion, we determined that up to 2 right whales, 3 humpback whales and 52 fin whales are likely to be exposed to disturbing levels of noise over the 20 days of impact pile driving. We also anticipate the exposure of up to 360 loggerheads and 40 leatherbacks, 40 Kemp's ridley and 40 green sea turtles.

Sea turtles and sturgeon exposed to other acoustic sources during the proposed action will experience only minor and temporary effects limited to small (less than 100 meters) movements away from the sound source; these effects will be insignificant. We anticipate behavioral disturbance of whales upon exposure to disturbing levels of noise associated with the use of DP thrusters along the cable route and upon exposure to the noise associated with the installation and removal of the sheet pile cofferdams. As with exposure to the impact pile driving, we expect these animals to alter their behavior from foraging, migrating, and resting to make evasive movements away from the area with disturbing levels of noise. This will result in stress to these animals and will come at a metabolic and energetic cost. However, because this response is limited to only a few hours, the stress will resolve and not result in distressed individuals and any disruption to behaviors will be temporary. We do not anticipate any injury or mortality immediately or in the future and do not anticipate any reduction in fitness. We have determined that this behavioral disturbance is considered “harassment” under the ESA definition of take. This harassment will occur in the form of avoidance or displacement from habitats and behavioral and/or metabolic/energetic compensation to deal with short term (hours) of stress resulting from exposure to disturbing levels of noise. While these individuals may experience temporary disruption of behavior patterns, we do not anticipate that the habitat modification caused by noise will actually kill or injure listed whales, sea turtles, or Atlantic sturgeon by significantly impairing essential behavioral patterns due to the temporary nature of any effects and the ability of individuals to resume pre-disruption behaviors once the disturbance has ceased. As presented in the Effects of the Action, during DP thruster use, we expect 1 right whale, 2 humpback whales and 23 fin whales are likely to be exposed to disturbing levels of noise. We expect 7 right whales, 15 humpback whales and 121 fin whales to be exposed to disturbing levels of noise over the four 12-hour days the vibratory hammer will be used for pile installation and removal. 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 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, we consider 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

The area in which the windfarm will be built and operated will continue to be used by fishing vessels, recreational boaters, and the maritime industry. It will also continue to be subject to noise pollution, contaminants, increased turbidity, and benthic habitat disturbance. While the effects of construction and operation of the windfarm will be added to the baseline conditions that will exist over the course of the action, we do not anticipate that the effects of the action and baseline conditions will combine synergistically to produce effects greater than the sum of the parts. As explained in the Opinion, we do not anticipate any injury or mortality of any right whales to result from the proposed action itself. As explained below, we do not expect the effects of the action, even when added to baseline conditions over the course of the action, will result in injury or mortality of any whales.

With regard to any project-caused benthic habitat disturbance, prey loss, turbidity, and release of contaminants, these impacts are anticipated to be so small in area, low in severity, limited to the construction phase, and temporary even during that time, that we do not anticipate they will have a perceptible effect on baseline conditions. Any temporary prey loss for sea turtles and sturgeon might be off-set by slight increases in prey over time due to the addition of concrete matting and rock piles associated with the project. Given that there are a small number of project-related vessels, we expect project-related vessel strikes to be extremely unlikely to occur and the additional project-related vessels will have an insignificant effect on the baseline risk of vessel strikes over the course of the action.

Behavioral effects, such as disruption of feeding, resting, or other activities or alterations in breathing, vocalizing, or diving rates, are likely during exposure to underwater noise between 120 and 180 dB re 1uPa RMS resulting from the DP thrusters and vibratory hammer, and between 160 and 180 dB re 1uPa RMS for the impulsive noise of the impact hammer. Effects of other project-related sources of noise (geophysical surveys, vessels, turbine generators' operation) are anticipated to be extremely unlikely to occur or insignificant. The project-related acoustic effects from the impact hammer, vibratory hammer, and DP Thrusters will be temporary, short-term, and geographically limited to a very small portion of the overall species' range. Even when added to the existing acoustic baseline, these time and space limited project-related acoustic effects are not likely significantly impair any essential behaviors or to affect an individual's health, survival or reproductive success. Specifically, in concerning the combined effects of ambient noise plus a introduced sound signal, the overall sound level in the affected environment can be determined by the following expression: L_{x+y} (in dB) =10 Log ($10^{x/10} + 10^{y/10}$); where, L_{x+y}= the overall sound level; x= ambient noise level (in dB) and y =the introduced sound signal (Richardson et al. 1995). This expression demonstrates that, due to the logarithmic nature of the decibel scale, the summation of two sound sources does not result in simply a doubling of sound energy. Instead, when considered together, ambient noise has little to no effect on the overall sound level produced and in fact, is "over taken" by the introduced sound signal. That is, ambient noise is non-detectable in the presence of the sound source. As a result, the total sound produced is reflective of the introduced sound source. For instance, if we consider ambient conditions to be 100 dB and the introduced sound signal to be 180 dB, the overall/total sound in the affected area is 180 dB. Based on this information, project related sound levels, when considered in combination with ambient noise, will not result in overall sound levels that differ from those project specific source levels considered on their own. As such, the behavioral effects, and the extent that they will be experienced, will remain as described for each sound source.

Based on the information provided above, when added to baseline conditions, the proposed action, including the behavioral disturbance of no more than 10 right whales due to exposure to disturbing levels of noise due to vibratory pile installation and removal, DP thruster use and impact pile driving, 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; and (3) the action will have only a minor and temporary effect on the distribution of individual 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.

Since the proposed action is not likely to result in injury or mortality of any individuals, it is also not likely to reduce fitness or future reproduction of any individual and, therefore, is not expected to affect the persistence of the species. Likewise, since the proposed action is not likely to result in injury or mortality and will result in any reduction in numbers or reproduction, it is also not likely to delay the recovery timeline or otherwise decrease the likelihood of recovery. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of North Atlantic right whales.

9.2 Humpback Whales

The area in which the windfarm will be built and operated will continue to be used by fishing vessels, recreational boaters, and the maritime industry. It will also continue to be subject to noise pollution, contaminants, increased turbidity, and benthic habitat disturbance. While the effects of construction and operation of the windfarm will be added to the baseline conditions that will exist over the course of the action, we do not anticipate that the effects of the action and baseline conditions will combine synergistically to produce effects greater than the sum of the parts. As explained in the Opinion, we do not anticipate any injury or mortality of any humpback whales to result from the proposed action itself. As explained below, we do not expect the effects of the action, even when added to baseline conditions over the course of the action, will result in injury or mortality of any whales.

With regard to any project-caused benthic habitat disturbance, prey loss, turbidity, and release of contaminants, these impacts are anticipated to be so small in area, low in severity, limited to the construction phase, and temporary even during that time, that we do not anticipate they will have a perceptible effect on baseline conditions. Any temporary prey loss for sea turtles and sturgeon might be off-set by slight increases in prey over time due to the addition of concrete matting and rock piles associated with the project. Given that there are a small number of project-related vessels, we expect project-related vessel strikes to be extremely unlikely to occur and the additional project-related vessels will have an insignificant effect on the baseline risk of vessel strikes over the course of the action.

Behavioral effects, such as disruption of feeding, resting, or other activities or alterations in breathing, vocalizing, or diving rates, are likely during exposure to underwater noise between 120 and 180 dB re 1uPa RMS resulting from the DP thrusters and vibratory hammer, and between 160 and 180 dB re 1uPa RMS for the impulsive noise of the impact hammer. Effects of other project-related sources of noise (geophysical surveys, vessels, turbine generators' operation) are anticipated to be extremely unlikely to occur or insignificant. The project-related acoustic effects from the impact hammer, vibratory hammer, and DP Thrusters will be temporary, short-term, and geographically limited to a very small portion of the overall species' range. Even when added to the existing acoustic baseline, these time and space limited project-related acoustic effects are not likely significantly impair any essential behaviors or to affect an individual's health, survival or reproductive success. Specifically, in concerning the combined effects of ambient noise plus a introduced sound signal, the overall sound level in the affected environment can be determined by the following expression: L_{x+y} (in dB) =10 Log ($10^{x/10} + 10^{y/10}$); where, L_{x+y}= the overall sound level; x= ambient noise level (in dB) and y =the introduced sound signal (Richardson et al. 1995). This expression demonstrates that, due to the logarithmic nature of the decibel scale, the summation of two sound sources does not result in simply a doubling of sound energy. Instead, when considered together, ambient noise has little to no effect on the overall sound level produced and in fact, is "over taken" by the introduced sound signal. That is, ambient noise is non-detectable in the presence of the sound source. As a result, the total sound produced is reflective of the introduced sound source. For instance, if we consider ambient conditions to be 100 dB and the introduced sound signal to be 180 dB, the overall/total sound in the affected area is 180 dB. Based on this information, project related sound levels, when considered in combination with ambient noise, will not result in overall sound levels that differ from those project specific source levels considered on their own. As such, the behavioral effects, and the extent that they will be experienced, will remain as described for each sound source.

Based on the information provided above, when added to baseline conditions, the proposed action, including the behavioral disturbance of no more than 20 humpback whales due to exposure to disturbing levels of noise due to vibratory pile installation and removal, DP thruster use and impact pile driving, 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; and (3) the action will have only a minor and temporary effect on the distribution of individual 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.

Since the proposed action is not likely to result in injury or mortality of any individuals, it is also not likely to reduce fitness or future reproduction of any individual and, therefore, is not expected to affect the persistence of the species. Likewise, since the proposed action is not likely to result in injury or mortality and will result in any reduction in numbers or reproduction, it is also not likely to delay the recovery timeline or otherwise decrease the likelihood of recovery. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of humpback whales.

9.3 Fin Whales

The area in which the windfarm will be built and operated will continue to be used by fishing vessels, recreational boaters, and the maritime industry. It will also continue to be subject to noise pollution, contaminants, increased turbidity, and benthic habitat disturbance. While the effects of construction and operation of the windfarm will be added to the baseline conditions that will exist over the course of the action, we do not anticipate that the effects of the action and baseline conditions will combine synergistically to produce effects greater than the sum of the parts. As explained in the Opinion, we do not anticipate any injury or mortality of any fin whales to result from the proposed action itself. As explained below, we do not expect the effects of the action, even when added to baseline conditions over the course of the action, will result in injury or mortality of any whales.

With regard to any project-caused benthic habitat disturbance, prey loss, turbidity, and release of contaminants, these impacts are anticipated to be so small in area, low in severity, limited to the construction phase, and temporary even during that time, that we do not anticipate they will have a perceptible effect on baseline conditions. Any temporary prey loss for sea turtles and sturgeon might be off-set by slight increases in prey over time due to the addition of concrete matting and rock piles associated with the project. Given that there are a small number of project-related vessels, we expect project-related vessel strikes to be extremely unlikely to occur and the additional project-related vessels will have an insignificant effect on the baseline risk of vessel strikes over the course of the action.

Behavioral effects, such as disruption of feeding, resting, or other activities or alterations in breathing, vocalizing, or diving rates, are likely during exposure to underwater noise between 120 and 180 dB re 1uPa RMS resulting from the DP thrusters and vibratory hammer, and between 160 and 180 dB re 1uPa RMS for the impulsive noise of the impact hammer. Effects of other project-related sources of noise (geophysical surveys, vessels, turbine generators' operation) are anticipated to be extremely unlikely to occur or insignificant. The project-related acoustic effects from the impact hammer, vibratory hammer, and DP Thrusters will be temporary, short-term, and geographically limited to a very small portion of the overall species' range. Even when added to the existing acoustic baseline, these time and space limited project-related acoustic effects are not likely significantly impair any essential behaviors or to affect an individual's health, survival or reproductive success. Specifically, in concerning the combined effects of ambient noise plus a introduced sound signal, the overall sound level in the affected environment can be determined by the following expression: L_{x+y} (in dB) =10 Log ($10^{x/10} + 10^{y/10}$); where, L_{x+y}= the overall sound level; x= ambient noise level (in dB) and y =the introduced sound signal (Richardson et al. 1995). This expression demonstrates that, due to the logarithmic nature of the decibel scale, the summation of two sound sources does not result in simply a doubling of sound energy. Instead, when considered together, ambient noise has little to no effect on the overall sound level produced and in fact, is "over taken" by the introduced sound signal. That is, ambient noise is non-detectable in the presence of the sound source. As a result, the total sound produced is reflective of the introduced sound source. For instance, if we consider ambient conditions to be 100 dB and the introduced sound signal to be 180 dB, the overall/total sound in the affected area is 180 dB. Based on this information, project related sound levels, when considered in combination with ambient noise, will not result in overall sound levels that differ from those project specific source levels considered on their own. As such, the behavioral effects, and the extent that they will be experienced, will remain as described for each sound source.

Based on the information provided above, when added to baseline conditions, the proposed action, including the behavioral disturbance of no more than 196 fin whales due to exposure to disturbing levels of noise due to vibratory pile installation and removal, DP thruster use and impact pile driving, 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; and (3) the action will have only a minor and temporary effect on the distribution of individual 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.

Since the proposed action is not likely to result in injury or mortality of any individuals, it is also not likely to reduce fitness or future reproduction of any individual and, therefore, is not expected to affect the persistence of the species. Likewise, since the proposed action is not likely to result in injury or mortality and will result in any reduction in numbers or reproduction, it is also not likely to delay the recovery timeline or otherwise decrease the likelihood of recovery. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of fin whales.

9.4 Northwest Atlantic DPS of Loggerhead Sea Turtles

The area in which the windfarm will be built and operated will continue to be used by fishing vessels, recreational boaters, and the maritime industry. It will also continue to be subject to noise pollution, contaminants, increased turbidity, and benthic habitat disturbance. While the effects of construction and operation of the windfarm will be added to the baseline conditions that will exist over the course of the action, we do not anticipate that the effects of the action and baseline conditions will combine synergistically to produce effects greater than the sum of the parts. As explained in the Opinion, we do not anticipate any injury or mortality of any loggerhead sea turtles to result from the proposed action itself. As explained below, we do not expect the effects of the action, even when added to baseline conditions over the course of the action, will result in injury or mortality of any sea turtles.

With regard to any project-caused benthic habitat disturbance, prey loss, turbidity, and release of contaminants, these impacts are anticipated to be so small in area, low in severity, limited to the construction phase, and temporary even during that time, that we do not anticipate they will have a perceptible effect on baseline conditions. Any temporary prey loss for sea turtles and sturgeon might be off-set by slight increases in prey over time due to the addition of concrete matting and rock piles associated with the project. Given that there are a small number of project-related vessels, we expect project-related vessel strikes to be extremely unlikely to occur and the additional project-related vessels will have an insignificant effect on the baseline risk of vessel strikes over the course of the action.

Behavioral effects, such as disruption of feeding, resting, or other activities or alterations in breathing, vocalizing, or diving rates, are likely during exposure to underwater noise between 166 and 207 dB re 1uPa RMS resulting from the impact hammer. Effects of other project-related sources of noise (DP thruster operations, vibratory hammer, geophysical surveys, vessels, turbine generators' operation) are anticipated to be extremely unlikely to occur or insignificant. The project-related acoustic effects from the impact hammer will be temporary, short-term, and geographically limited to a very small portion of the overall species' range. Even when added to the existing acoustic baseline, these time and space limited project-related acoustic effects are not likely significantly impair any essential behaviors or to affect an individual's health, survival or reproductive success. Specifically, in concerning the combined effects of ambient noise plus a introduced sound signal, the overall sound level in the affected environment can be determined by the following expression: L_{x+y} (in dB) =10 Log ($10^{x/10} + 10^{y/10}$); where, L_{x+y}= the overall sound level; x= ambient noise level (in dB) and y =the introduced sound signal (Richardson et al. 1995). This expression demonstrates that, due to the logarithmic nature of the decibel scale, the summation of two sound sources does not result in simply a doubling of sound energy. Instead, when considered together, ambient noise has little to no effect on the overall sound level produced and in fact, is "over taken" by the introduced sound signal. That is, ambient noise is non-detectable in the presence of the sound source. As a result, the total sound produced is reflective of the introduced sound source. For instance, if we consider ambient conditions to be 100 dB and the introduced sound signal to be 180 dB, the overall/total sound in the affected area is 180 dB. Based on this information, project related sound levels, when considered in combination with ambient noise, will not result in overall sound levels that differ from those project specific source levels considered on their own. As such, the behavioral effects, and the extent that they will be experienced, will remain as described for each sound source.

Based on the information provided above, when added to baseline conditions, the proposed action, including the behavioral disturbance of no more than 360 loggerhead sea turtles due to exposure to disturbing levels of noise due to impact pile driving, 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; and (3) the action will have only a minor and temporary effect on the distribution of individual 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.

Since the proposed action is not likely to result in injury or mortality of any individuals, it is also not likely to reduce fitness or future reproduction of any individual and, therefore, is not expected to affect the persistence of the species. Likewise, since the proposed action is not likely to result in injury or mortality and will result in any reduction in numbers or reproduction, it is also not likely to delay the recovery timeline or otherwise decrease the likelihood of recovery. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of loggerhead sea turtles.

9.5 Leatherback Sea Turtles

The area in which the windfarm will be built and operated will continue to be used by fishing vessels, recreational boaters, and the maritime industry. It will also continue to be subject to noise pollution, contaminants, increased turbidity, and benthic habitat disturbance. While the effects of construction and operation of the windfarm will be added to the baseline conditions that will exist over the course of the action, we do not anticipate that the effects of the action and baseline conditions will combine synergistically to produce effects greater than the sum of the parts. As explained in the Opinion, we do not anticipate any injury or mortality of any leatherback sea turtles to result from the proposed action itself. As explained below, we do not expect the effects of the action, even when added to baseline conditions over the course of the action, will result in injury or mortality of any sea turtles.

With regard to any project-caused benthic habitat disturbance, prey loss, turbidity, and release of contaminants, these impacts are anticipated to be so small in area, low in severity, limited to the construction phase, and temporary even during that time, that we do not anticipate they will have a perceptible effect on baseline conditions. Any temporary prey loss for sea turtles and sturgeon might be off-set by slight increases in prey over time due to the addition of concrete matting and rock piles associated with the project. Given that there are a small number of project-related vessels, we expect project-related vessel strikes to be extremely unlikely to occur and the additional project-related vessels will have an insignificant effect on the baseline risk of vessel strikes over the course of the action.

Behavioral effects, such as disruption of feeding, resting, or other activities or alterations in breathing, vocalizing, or diving rates, are likely during exposure to underwater noise between 166 and 207 dB re 1uPa RMS resulting from the DP thrusters, impact hammer, and vibratory hammer. Effects of other project-related sources of noise (DP thruster operation, vibratory hammer, geophysical surveys, vessels, turbine generators' operation) are anticipated to be extremely unlikely to occur or insignificant. The project-related acoustic effects from the impact hammer will be temporary, short-term, and geographically limited to a very small portion of the overall species' range. Even when added to the existing acoustic baseline, these time and space limited project-related acoustic effects are not likely significantly impair any essential behaviors or to affect an individual's health, survival or reproductive success. Specifically, in concerning the combined effects of ambient noise plus a introduced sound signal, the overall sound level in the affected environment can be determined by the following expression: L_{x+y} (in dB) =10 Log $(10^{x/10} + 10^{y/10})$; where, L_{x+y}= the overall sound level; x= ambient noise level (in dB) and y =the introduced sound signal (Richardson *et al.* 1995). This expression demonstrates that, due to the logarithmic nature of the decibel scale, the summation of two sound sources does not result in simply a doubling of sound energy. Instead, when considered together, ambient noise has little to no effect on the overall sound level produced and in fact, is "over taken" by the introduced sound signal. That is, ambient noise is non-detectable in the presence of the sound source. As a result, the total sound produced is reflective of the introduced sound source. For instance, if we consider ambient conditions to be 100 dB and the introduced sound signal to be 180 dB, the overall/total sound in the affected area is 180 dB. Based on this information, project related sound levels, when considered in combination with ambient noise, will not result in overall sound levels that differ from those project specific source levels considered on their own. As such, the behavioral effects, and the extent that they will be experienced, will remain as described for each sound source.

Based on the information provided above, when added to baseline conditions, the proposed action, including the behavioral disturbance of no more than 40 leatherback sea turtles due to exposure to disturbing levels of noise due to impact pile driving, 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; and (3) the action will have only a minor and temporary effect on the distribution of individual 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.

Since the proposed action is not likely to result in injury or mortality of any individuals, it is also not likely to reduce fitness or future reproduction of any individual and, therefore, is not expected to affect the persistence of the species. Likewise, since the proposed action is not likely to result in injury or mortality and will result in any reduction in numbers or reproduction, it is also not likely to delay the recovery timeline or otherwise decrease the likelihood of recovery.

Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of leatherback sea turtles.

9.6 Kemp's Ridley Sea Turtles

The area in which the windfarm will be built and operated will continue to be used by fishing vessels, recreational boaters, and the maritime industry. It will also continue to be subject to noise pollution, contaminants, increased turbidity, and benthic habitat disturbance. While the effects of construction and operation of the windfarm will be added to the baseline conditions that will exist over the course of the action, we do not anticipate that the effects of the action and baseline conditions will combine synergistically to produce effects greater than the sum of the parts. 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 itself. As explained below, we do not expect the effects of the action, even when added to baseline conditions over the course of the action, will result in injury or mortality of any sea turtles.

With regard to any project-caused benthic habitat disturbance, prey loss, turbidity, and release of contaminants, these impacts are anticipated to be so small in area, low in severity, limited to the construction phase, and temporary even during that time, that we do not anticipate they will have a perceptible effect on baseline conditions. Any temporary prey loss for sea turtles and sturgeon might be off-set by slight increases in prey over time due to the addition of concrete matting and rock piles associated with the project. Given that there are a small number of project-related vessels, we expect project-related vessel strikes to be extremely unlikely to occur and the additional project-related vessels will have an insignificant effect on the baseline risk of vessel strikes over the course of the action.

Behavioral effects, such as disruption of feeding, resting, or other activities or alterations in breathing, vocalizing, or diving rates, are likely during exposure to underwater noise between 166 and 207 dB re 1uPa RMS resulting from the DP thrusters, impact hammer, and vibratory hammer. Effects of other project-related sources of noise (DP thruster operation, vibratory hammer, geophysical surveys, vessels, turbine generators' operation) are anticipated to be extremely unlikely to occur or insignificant. The project-related acoustic effects from the impact hammer will be temporary, short-term, and geographically limited to a very small portion of the overall species' range. Even when added to the existing acoustic baseline, these time and space limited project-related acoustic effects are not likely significantly impair any essential behaviors or to affect an individual's health, survival or reproductive success. Specifically, in concerning the combined effects of ambient noise plus a introduced sound signal, the overall sound level in the affected environment can be determined by the following expression: L_{x+y} (in dB) =10 Log $(10^{x/10} + 10^{y/10})$; where, L_{x+y}= the overall sound level; x= ambient noise level (in dB) and y =the introduced sound signal (Richardson *et al*. 1995). This expression demonstrates that, due to the logarithmic nature of the decibel scale, the summation of two sound sources does not result in simply a doubling of sound energy. Instead, when considered together, ambient noise has little to no effect on the overall sound level produced and in fact, is "over taken" by the introduced sound signal. That is, ambient noise is non-detectable in the presence of the sound source. As a result, the total sound produced is reflective of the introduced sound source. For instance, if we consider ambient conditions to be 100 dB and the introduced sound signal to be 180 dB, the overall/total sound in the affected area is 180 dB. Based on this information, project related sound levels, when considered in combination with ambient noise, will not result in overall sound levels that differ from those project specific source levels considered on their own. As such, the behavioral effects, and the extent that they will be experienced, will remain as described for each sound source.

Based on the information provided above, when added to baseline conditions, the proposed action, including the behavioral disturbance of no more than 40 Kemp's ridley sea turtles due to exposure to disturbing levels of noise due to impact pile driving, 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; and (3) the action will have only a minor and temporary effect on the distribution of individual 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.

Since the proposed action is not likely to result in injury or mortality of any individuals, it is also not likely to reduce fitness or future reproduction of any individual and, therefore, is not expected to affect the persistence of the species. Likewise, since the proposed action is not likely to result in injury or mortality and will result in any reduction in numbers or reproduction, it is also not likely to delay the recovery timeline or otherwise decrease the likelihood of recovery. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of Kemp's ridley sea turtles.

9.7 Green Sea Turtles

The area in which the windfarm will be built and operated will continue to be used by fishing vessels, recreational boaters, and the maritime industry. It will also continue to be subject to noise pollution, contaminants, increased turbidity, and benthic habitat disturbance. While the effects of construction and operation of the windfarm will be added to the baseline conditions that will exist over the course of the action, we do not anticipate that the effects of the action and baseline conditions will combine synergistically to produce effects greater than the sum of the parts. As explained in the Opinion, we do not anticipate any injury or mortality of any green sea turtles to result from the proposed action itself. As explained below, we do not expect the effects of the action, even when added to baseline conditions over the course of the action, will result in injury or mortality of any sea turtles.

With regard to any project-caused benthic habitat disturbance, prey loss, turbidity, and release of contaminants, these impacts are anticipated to be so small in area, low in severity, limited to the construction phase, and temporary even during that time, that we do not anticipate they will have a perceptible effect on baseline conditions. Any temporary prey loss for sea turtles and sturgeon might be off-set by slight increases in prey over time due to the addition of concrete matting and rock piles associated with the project. Given that there are a small number of project-related vessels, we expect project-related vessel strikes to be extremely unlikely to occur and the additional project-related vessels will have an insignificant effect on the baseline risk of vessel strikes over the course of the action.

Behavioral effects, such as disruption of feeding, resting, or other activities or alterations in breathing, vocalizing, or diving rates, are likely during exposure to underwater noise between 166 and 207 dB re 1uPa RMS resulting from the DP thrusters, impact hammer, and vibratory hammer. Effects of other project-related sources of noise (DP thruster operation, vibratory hammer, geophysical surveys, vessels, turbine generators' operation) are anticipated to be extremely unlikely to occur or insignificant. The project-related acoustic effects from the impact hammer will be temporary, short-term, and geographically limited to a very small portion of the overall species' range. Even when added to the existing acoustic baseline, these time and space limited project-related acoustic effects are not likely significantly impair any essential behaviors or to affect an individual's health, survival or reproductive success. Specifically, in concerning the combined effects of ambient noise plus a introduced sound signal, the overall sound level in the affected environment can be determined by the following expression: L_{x+y} (in dB) =10 Log $(10^{x/10} + 10^{y/10})$; where, L_{x+y}= the overall sound level; x= ambient noise level (in dB) and y =the introduced sound signal (Richardson *et al*. 1995). This expression demonstrates that, due to the logarithmic nature of the decibel scale, the summation of two sound sources does not result in simply a doubling of sound energy. Instead, when considered together, ambient noise has little to no effect on the overall sound level produced and in fact, is "over taken" by the introduced sound signal. That is, ambient noise is non-detectable in the presence of the sound source. As a result, the total sound produced is reflective of the introduced sound source. For instance, if we consider ambient conditions to be 100 dB and the introduced sound signal to be 180 dB, the overall/total sound in the affected area is 180 dB. Based on this information, project related sound levels, when considered in combination with ambient noise, will not result in overall sound levels that differ from those project specific source levels considered on their own. As such, the behavioral effects, and the extent that they will be experienced, will remain as described for each sound source.

Based on the information provided above, when added to baseline conditions, the proposed action, including the behavioral disturbance of no more than 40 green sea turtles due to exposure to disturbing levels of noise due to impact pile driving, 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; and (3) the action will have only a minor and temporary effect on the distribution of individual 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.

Since the proposed action is not likely to result in injury or mortality of any individuals, it is also not likely to reduce fitness or future reproduction of any individual and, therefore, is not expected to affect the persistence of the species. Likewise, since the proposed action is not likely to result in injury or mortality and will result in any reduction in numbers or reproduction, it is also not likely to delay the recovery timeline or otherwise decrease the likelihood of recovery. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of green sea turtles.

9.8 Atlantic Sturgeon

The area in which the windfarm will be built and operated will continue to be used by fishing vessels, recreational boaters, and the maritime industry. It will also continue to be subject to noise pollution, contaminants, increased turbidity, and benthic habitat disturbance. While the effects of construction and operation of the windfarm will be added to the baseline conditions that will exist over the course of the action, we do not anticipate that the effects of the action and baseline conditions will combine synergistically to produce effects greater than the sum of the parts. As explained in the Opinion, we do not anticipate any injury or mortality of any Atlantic sturgeon to result from the proposed action itself. As explained below, we do not expect the effects of the action, even when added to baseline conditions over the course of the action, will result in injury or mortality of any sturgeon.

With regard to any project-caused benthic habitat disturbance, prey loss, turbidity, and release of contaminants, these impacts are anticipated to be so small in area, low in severity, limited to the construction phase, and temporary even during that time, that we do not anticipate they will have a perceptible effect on baseline conditions. Any temporary prey loss for sea turtles and sturgeon might be off-set by slight increases in prey over time due to the addition of concrete matting and rock piles associated with the project. Given that there are a small number of project-related vessels, we expect project-related vessel strikes to be extremely unlikely to occur and the additional project-related vessels will have an insignificant effect on the baseline risk of vessel strikes over the course of the action.

Behavioral effects, such as disruption of feeding, resting, or other activities or alterations in breathing, vocalizing, or diving rates, are likely during exposure to underwater noise above 150 dB re 1uPa RMS resulting from the impact hammer. Effects of other project-related sources of noise (DP thruster operation, vibratory hammer, geophysical surveys, vessels, turbine generators' operation) are anticipated to be extremely unlikely to occur or insignificant. The project-related acoustic effects from the impact hammer will be temporary, short-term, and geographically limited to a very small portion of the overall species' range. Even when added to the existing acoustic baseline, these time and space limited project-related acoustic effects are not likely significantly impair any essential behaviors or to affect an individual's health, survival or reproductive success. Specifically, in concerning the combined effects of ambient noise plus a introduced sound signal, the overall sound level in the affected environment can be determined by the following expression: L_{x+y} (in dB) =10 Log ($10^{x/10} + 10^{y/10}$); where, L_{x+y}= the overall sound level; x= ambient noise level (in dB) and y =the introduced sound signal (Richardson *et al*. 1995). This expression demonstrates that, due to the logarithmic nature of the decibel scale, the summation of two sound sources does not result in simply a doubling of sound energy. Instead, when considered together, ambient noise has little to no effect on the overall sound level produced and in fact, is "over taken" by the introduced sound signal. That is, ambient noise is non-detectable in the presence of the sound source. As a result, the total sound produced is reflective of the introduced sound source. For instance, if we consider ambient conditions to be 100 dB and the introduced sound signal to be 180 dB, the overall/total sound in the affected area is 180 dB. Based on this information, project related sound levels, when considered in combination with ambient noise, will not result in overall sound levels that differ from those project specific source levels considered on their own. As such, the behavioral effects, and the extent that they will be experienced, will remain as described for each sound source.

Based on the information provided above, when added to baseline conditions, the proposed action, including the behavioral disturbance of Atlantic sturgeon due to exposure to disturbing levels of noise due to impact pile driving, 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 Atlantic sturgeon; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; and (3) the action will have only a minor and temporary effect on the distribution of individual Atlantic sturgeon 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.

Since the proposed action is not likely to result in injury or mortality of any individuals, it is also not likely to reduce fitness or future reproduction of any individual and, therefore, is not expected to affect the persistence of the species. Likewise, since the proposed action is not likely to result in injury or mortality and will result in any reduction in numbers or reproduction, it is also not likely to delay the recovery timeline or otherwise decrease the likelihood of recovery. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of Atlantic sturgeon.

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, leatherback or the Northeast Atlantic DPS of loggerhead sea turtles,North Atlantic right, humpback, or fin whales, or the GOM, NYB, CB, Carolina 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 – AMENDED OCTOBER 30, 2014

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 results from, but is not the purpose 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 the USACE and Deepwater Wind, for the exemption in section 7(o)(2) to apply. The USACE has a continuing duty to regulate the activity covered by this Incidental Take Statement. If the USACE (1) fails to assume and implement the terms and conditions consistent with its authority or (2) fails to require Deepwater Wind to adhere to the terms and conditions of the Incidental Take Statement, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the USACE 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).

11.1 Anticipated Amount or Extent of Incidental Take

Sea Turtles

We do not anticipate any injury or mortality of any loggerhead, leatherback, Kemp's ridley or green sea turtles to result from the proposed action. We anticipate the behavioral disturbance of (harassment) no more than 360 loggerhead, 40 leatherback, 40 Kemp's ridley and 40 green sea turtles due to exposure to disturbing levels of noise during impact pile driving. We do not anticipate any impacts to the health, survival or reproductive success of any individual loggerhead, leatherback, Kemp's ridley or green sea turtles. All other effects to sea turtles, including increased vessel traffic and impacts to benthic resources, will be insignificant and discountable.

As explained in the Opinion, the calculated number of sea turtles that may be behaviorally disturbed are likely to result in overestimates of the number of individuals exposed. For impact pile driving operations, 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 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 impact pile driving. Because both the distribution and numbers of sea turtles in the action area during 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, 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 USACE or Deepwater Wind will be able to monitor the behavior of all sea turtles in the action area in a manner which would detect responses to 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 harassment. Deepwater Wind will verify the extent in which behavioral disturbance thresholds are attained during the installation of each WTG foundation.

Atlantic Sturgeon

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 resulting from the impulsive noise of the impact hammer, such as disruption of feeding, resting, migration, or other activities 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 and impacts to benthic resources, will be insignificant and discountable. 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 impact 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, 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 impact 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 impact pile driving, and thus 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 impact pile driving underwater noise is elevated above 150 dB_{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 that would result in behaviors consistent with harassment. Deepwater Wind will verify the extent in which behavioral disturbance thresholds are attained during the installation of the eachWTG foundation.

Whales

NMFS has concluded that the construction of the BIWF and the BITS in the coastal and marine environment east of Block Island and in Rhode Island Sound is likely to result in incidental take of North Atlantic right (*Eubalaena glacialis*), humpback (*Megaptera novaeaengliae*), and fin (*Balaenoptera physalus*) whales in the form of acoustic harassment. The exposure to underwater noise between 120 and 180 dB re 1uPa RMS resulting from the DP thrusters and vibratory hammer, and between 160 and 180 dB re 1uPa RMS for the impulsive noise of the impact hammer may cause behavioral effects, such as disruption of feeding, resting, or other activities or alterations in breathing, vocalizing, or diving rates. The project-related acoustic effects from the impact hammer, vibratory hammer, and DP thrusters will be temporary, short-term, and geographically limited to a very small portion of the overall species' range.

The NMFS Office of Protected Resources (OPR) Permits, Conservation, and Education Division has issued two Incidental Harassment Authorizations (IHA) to Deepwater Wind Block Island, LLC and Deepwater Wind Block Transmission, LLC ("Deepwater Wind") for the harassment of a small number of marine mammals incidental to the construction activity for the Block Island Wind Farm (BIWF) and the Block Island Transmission System (BITS). The IHA for the BIWF is effective October 31, 2014 through October 30, 2015 and was issued on September 3, 2014 (79 FR 53409; Sep 9, 2014). The IHA for the BITS is effective November 1, 2014 through October 31, 2015 and was issued on August 22, 2014 (79 FR 51314; Aug 28, 2014).

Each IHA is effective for a period of one year, during which the maximum take authorized for both facilities combined may be up to 228 fin whales (75 BIWF, 153 BITS), 22 humpback whales (5 BIWF, 17 BITS), and 11 North Atlantic right whales (3 BIWF, 8 BITS). Each of these exposures will be considered a take by harassment.

The amount of exempted take will be exceeded if any right, humpback, or fin whales are harmed, injured, or killed as a result of the construction of the BITS or BIWF, or if the number of such whales taken by acoustic harassment as defined above exceeds the estimate of 75 fin whales, 5 humpback whales, and 3 North Atlantic right whales for the BIWF and 153 fin whales, 17 humpback whales, and 8 North Atlantic right whales for the BITS. For fin, humpback, and right whales, this ITS is only valid from October 31, 2014-October 30, 2015 for the BIWF and November 1, 2014-October 31, 2015 for BITS. Through acoustic monitoring, Deepwater Wind will verify the extent in which behavioral disturbance thresholds are attained during the installation of each WTG foundation.

11.2 Reasonable and Prudent Measures

Reasonable and prudent measures (RPMs) are those measures necessary and appropriate to minimize and monitor incidental take of a listed species. Section 3.8 of this Opinion identifies a number of mitigation measures included in the project description which are designed to avoid and minimize impacts to listed species. The applicant, Deepwater Wind, has committed to implementing these measures and they will be included as Special Conditions of permits issued by the USACE. Because they are part of the proposed action, we are not repeating them as Reasonable and Prudent Measures or Terms and Conditions. The most significant potential impacts from this project are from noise and the exclusion areas established avoid the potential exposure of listed species to levels of noise that may otherwise cause injury. This is an important project component and the amount and type of take is minimized as a result of this measure. Additionally, the monitoring of the zone with noise levels that may cause harassment of listed species provides an opportunity to cease operations when marine mammals or sea turtles are detected in this area which also avoids or minimizes impacts to listed species. The reasonable and prudent measures below are in addition to the measures established by Deepwater Wind and that will be adhered to throughout all phases of the project and will be included as special conditions of the USACE permits.

Failure to implement the listed species mitigation measures that were already considered part of the proposed action would trigger reinitiation of consultation under 50 CFR 402.16, and this ITS would not apply given that the action would be different than the action for which this ITS exempts take. The listed species mitigation measures outlined as part of this proposed action must be implemented in order for this ITS to exempt incidental take. We believe the following reasonable and prudent measures are necessary and appropriate to minimize and monitor impacts of incidental take of fin, humpback, and North Atlantic right whales; Kemp's ridley, green, loggerhead, and leatherback sea turtles; and Atlantic sturgeon. As noted above, these are in addition to the measures already being implemented as part of the proposed action.

1. The USACE must ensure that any endangered species observers contracted by Deepwater Wind are approved by NMFS.

2. The USACE must ensure that designated exclusion zones for all noise producing activities are monitored by NMFS-approved observers. The exclusion zone is considered that area ensonified by injurious levels (i.e., underwater noise levels greater than or equal to 180 re 1 μPa RMS).

3. The USACE must ensure that designated monitoring zones for all noise producing activities are monitored by NMFS-approved observers. The monitoring zone is considered that area ensonified by noise levels that may cause behavioral disturbance (160 re 1 μPa RMS).

4. The USACE must ensure that field verification of modeled noise levels for injury or mortality are undertaken and that monitoring is conducted throughout the work period to confirm modeled sound levels. This needs to be conducted for (1) impact pile driving operations; (2) installation and removal of cofferdams with vibratory pile driving; and, (3) DP thruster use.

5. The USACE must ensure that field verification of modeled noise levels for behavioral disturbance are undertaken and that monitoring is conducted throughout the work period to confirm modeled sound levels. This needs to be conducted for (1) impact pile driving operations; (2) installation and removal of cofferdams with vibratory pile driving; and, (3) DP thruster use. This RPM functions as a surrogate for monitoring incidental take.

6. Any sea turtle or Atlantic sturgeon observed during activities considered in this Opinion must be recorded, with information submitted to NMFS within 30 days. Any dead or injured sea turtle or Atlantic sturgeon must be reported to NMFS within 24 hours.

7. Reasonable attempts should be made to collect any dead sea turtles or sturgeon. These individuals must be held in cold storage until disposition can be discussed with NMFS.

8. Any whale taken in a manner not authorized by the Incidental Harassment Authorizations issued August 22, 2014 and September 3, 2014 (e.g., injury, serious injury, or mortality) must be reported immediately to NMFS Greater Atlantic Region (978-281-9328) and NMFS Office of Protected Resources (301-427-8401) and via email to incidental.take@noaa.gov, Jolie.Harrison@noaa.gov, John.Fiorentino@noaa.gov, and Mendy.Garron@noaa.gov. If a specified activity clearly causes an unauthorized take, the specified activity must cease immediately. Specified activity may not resume until NMFS has reviewed the circumstances of the prohibited take and the applicant is notified by NMFS that activities may resume. If an injured or dead whale is discovered, and the cause of injury is unknown or not associated with authorized activities, the incident must be reported immediately, as above, but activities may continue while NMFS reviews the circumstances.

9. Deepwater Wind must provide the following notifications to NMFS during construction activities:

    a. Beginning of construction activities (within 24 hours) and completion of construction of activities (within 24 hours)

b. Within 24 hours of receiving information that exclusion or monitoring zones should be changed
c. Within 24 hours of seeing behavioral responses by ESA-listed species

11.3 Terms and Conditions

In order to be exempt from prohibitions of section 9 of the ESA, USACE and Deepwater Wind 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, the USACE shall provide NMFS with the names and resumes of all endangered species monitors to be employed at the project site at least 30 days prior to the start of WTG construction. No observer shall work at the project site without written approval of NMFS. If during project construction or DP vessel operations, additional endangered species monitors are necessary, the USACE will provide those names and resumes to NMFS for approval at least 10 days prior to the date that they are expected to start work at the site.

2. To implement RPMs #2 and #3, during impact or vibratory pile driving operations, observers must begin monitoring the exclusion and monitoring zones at least 60 minutes prior to the initiation of soft start pile driving. Full energy pile driving must not begin until the zone is clear of all whales and sea turtles for at least 60 minutes. Monitoring will continue through the pile driving period and end approximately 60 minutes after pile driving is completed. Observers must notify operators if any whales or sea turtles appear to be moving toward the exclusion or monitoring zones, so that operations can be adjusted (i.e., pile driving energy reduced) to minimize the size of the exclusion and monitoring zones. If the latter occurs, the observer must monitor the area within and near the exclusion and monitoring zones for 60 minutes, and if clear after 60 minutes after the last sighting, notify the operator that full energy pile driving may resume.

3. To implement RPM#3, during DP vessel operations, observers will begin monitoring the monitoring zone as soon as the vessel leaves the dock and will continue throughout the construction activity. Observers must notify the vessel operator if any whales or sea turtles appear to be moving toward the monitoring zone, so that operations can be adjusted (i.e., reduced DP thruster energy) to minimize the size of the monitoring zone (i.e., underwater noise levels greater than or equal to 160 re 1 μPa RMS) If the latter occurs, the observer must monitor the area within and near the monitoring zone for 60 minutes, and if clear after 60 minutes of the last sighting, notify the vessel operator that full energy thruster use may resume. As DP vessels will be operational for 24 hours, at least two observers should be onboard the vessel, working a 12 hour on, 12 hour off schedule. That observer working the night shift must be provided night-vision binoculars.

4. To implement RPM #4, acoustic verification and monitoring must be conducted during impact pile driving (for the installation of each WTG foundation pile), DP thruster use, and vibratory pile driving (for cofferdam installation and removal) to ensure the

exclusion zone is appropriately defined and thus, monitored by the observer required in RPM #2. Acoustic monitoring must be sufficient to determine source levels (i.e., within 1 m of the source) as well as the radius of the following isopleths:

a. Atlantic sturgeon acoustic injury thresholds: Distance to the 206 dB re 1 μPa Peak and 187 dB re 1μPa^2-s cSEL isopleths.
b. Sea turtle acoustic injury threshold: Distance to the 207 dB re 1 μPa RMS isopleth.
c. Whale acoustic injury threshold: 180 dB re 1 μPa RMS isopleth

Results of this monitoring must be reported to NMFS at incidental.take@noaa.gov. For pile driving operations, results must be provided to NMFS prior to the installation of the next pile or within 24 hours of installation, whichever is sooner. For DP vessel operation, results must be provided every 24 hours. If there is any indication that injury thresholds have been attained in a manner not considered in this Opinion (i.e., extent of 206 dB re 1 μPa PEAK or 187 dB re 1μPa^2-s cSEL (Atlantic sturgeon); 207 dB re 1 μPa RMS (sea turtles) 180 dB re 1 μPa RMS (whales)), the following NMFS contacts must be notified immediately: NMFS Greater Atlantic Region (978-281-9328) and NMFS Office of Protected Resources (301-427-8401) and incidental.take@noaa.gov, Jolie.Harrison@noaa.gov, John.Fiorentino@noaa.gov, and Mendy.Garron@noaa.gov..

5. To implement RPM #5, acoustic verification and monitoring must be conducted during impact pile driving for the installation of each WTG foundation pile, DP thruster use, and vibratory pile driving (for cofferdam installation and removal). Acoustic monitoring must be sufficient to determine source levels (i.e., within 1 m of the source) as well as the following:
    a. Atlantic sturgeon acoustic behavioral disturbance thresholds: Distance to the 150 dB re 1 μPa RMS isopleth.
    b. Sea turtle acoustic behavioral disturbance threshold: Distance to the 166 dB re 1 μPa RMS isopleth.
    c. Whale acoustic behavioral disturbance threshold: Distance to the 160 dB re 1 μPa RMS isopleth for both impulsive and continuous noise.

    Results of this monitoring must be reported, via email to NMFS at incidental.take@noaa.gov. For pile driving operations, results must be provided to NMFS prior to the installation of the next pile or within 24 hours of installation, whichever is sooner.

6. To implement RPM #6, in the event of any observations of dead sea turtles or Atlantic sturgeon, dead specimens should be collected with a net and preserved (refrigerate or freeze) until disposal procedures are discussed with NMFS.

7. To implement RPM #7, USACE or Deepwater Wind must contact NMFS within 24 hours of any observations of dead or injured sea turtles or sturgeon. The take must be reported to NMFS Greater Atlantic Region Fisheries Office via email to incidental.take@noaa.gov.

8. To implement RPM #8, USACE or Deepwater Wind must contact NMFS immediately upon observing any dead or injured whale and cease all activity if authorized activities caused or may have caused the death or injury. The take must be reported to NMFS Office of Protected Resources (301-427-8401) and via email to incidental.take@noaa.gov, Jolie.Harrison@noaa.gov, John.Fiorentino@noaa.gov, and Mendy.Garron@noaa.gov.

The reasonable and prudent measures, with their implementing terms and conditions, are designed to minimize and monitor the impact of incidental take resulting 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 the USACE based on which NMFS has made conclusions regarding take.

RPM and Term and Condition #1 is necessary and appropriate because it is specifically designed to ensure that all endangered species monitors employed by Deepwater Wind are qualified to conduct the necessary duties. Including this review of endangered species monitors by NMFS staff is only a minor change because it is not expected to result in any delay to the project and will merely ensure endangered species monitors have the qualifications that are already required by the USACE.

RPM and Term and Condition #1 is necessary and appropriate because it is specifically designed to ensure that all endangered species monitors employed by the applicant are qualified to conduct the necessary duties. Including this review of endangered species monitors by NMFS staff is only a minor change because it is not expected to result in any delay to the project and will merely enforce the qualifications of the endangered species monitors that are already required by the USACE.

RPMs #2 and #3 and Terms and Conditions # 2, 3, and 4 are necessary and appropriate to ensure listed species are not exposed to injurious levels of noise throughout the proposed action and that project operations are adjusted accordingly to further avoid this exposure. These RPMs and their Terms and Conditions are not expected to result in any delay to the project and will merely enforce the qualifications and duties of the endangered species monitors that are already required by the USACE.

RPM #4 and 5 and Terms and Conditions #4 and 5 are necessary and appropriate because they are designed to verify that the sound levels modeled by for Deepwater Wind 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 and thus, it is not expected to result in any delay to the project or a significant change to the project.

RPMs #6, 7 and 8 and Terms and Conditions #6, 7, and 8 are necessary and appropriate to ensure the proper handling and documentation of any interactions with listed species as well as requiring that these interactions are reported to us in a timely manner with all of the necessary information. This is essential for monitoring the level of incidental take associated with the proposed action.

These RPMs and Terms and Conditions represent only minor changes as compliance will not result in any increased cost, delay of the project (unless unanticipated take occurs), or decrease in the efficiency of any activity.

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. The USACE should use its authorities to support research on the effects of pile driving, DP thruster operation, and WTG operational 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.

The applicant has also applied for an IHA and is submitting information to NMFS Office of Protected Resources in Silver Springs, Maryland as part of that process. If information and/or analysis from that process reveals effects of this action that may affect listed species in a manner or to an extent not considered in this Opinion, or the description of the proposed action is changed such that it causes an effect to listed species not considered here, this consultation must be reinitiated.

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APENNDIX A

BIWF and BITS Project Location

APENNDIX B

Construction Vessel and Vessel Routes