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

Agency: Army Corps of Engineers (USACE), New York District

Activity: Beach Nourishment Projects Utilizing the Sea Bright Offshore Borrow Area: Union Beach, Port Monmouth, and Elberon to Loch Arbour, New Jersey
(NER-2014-10606)
GARFO-2013-00017

Conducted by: National Marine Fisheries Service
Northeast Regional Office

Date Issued: 3/7/2014

Approved by: for John Bullard

Contents

1.0 INTRODUCTION

This constitutes the biological opinion (Opinion) of NOAA's National Marine Fisheries Service (NMFS) issued pursuant to Section 7 of the Endangered Species Act (ESA) of 1973, as amended, on the effects of the U.S. Army Corp of Engineers (USACE) conducting three beach nourishment projects utilizing the Sea Bright Offshore Borrow Area (SBOBA):

- Port Monmouth
- Union Beach
- Elberon to Loch Arbour

This Opinion is based on information provided in the Biological Assessments (BA) dated October 2013, past consultations with the USACE New York District, and scientific papers and other sources of information as cited in this Opinion. We will keep a complete administrative record of this consultation at our Northeast Regional Office. Formal consultation was initiated on October 29, 2013.

2.0 CONSULTATION HISTORY

The USACE submitted a biological assessment (BA) to us on August 26, 2013, along with a request to initiate consultation on three dredging projects, with supplemental information provided in a revised BA on November 1, 2013. The three proposed actions are in response to the impacts sustained from Hurricane Sandy on October 29, 2012. Because the projects are similar, they take place in the same geographic area, and affect the same species in the same manner, we determined it would be most efficient to combine the analysis of effects in one consultation. As such, while there are three independent actions considered here (i.e., beach nourishment projects for Port Monmouth, Union Beach, and Elberon to Loch Arbour), we are producing one Opinion. This type of "multi-action" consultation is contemplated in the NMFS-USFWS Section 7 Consultation Handbook (see page 5-5).

In the future, reinitiation of consultation may be necessary (see Section 14 and 50 CFR§ 402.16). Depending on the circumstances associated with the cause for reinitiation, it may not be necessary to reinitiate consultation for all of the actions considered here. For example, if a new species is listed that may be affected by dredging activities, it would likely be necessary to reinitiate consultation on all of the activities considered here. However, if the cause for reinitiation has effects that are limited to one action (for example, a change in dredge type, dredge volume or disposal area), reinitiation of consultation on only that action may be necessary. We expect that determinations about the scope of any future reinitiation(s) will be made in cooperation with the USACE and us.

3.0 DESCRIPTION OF THE ACTION

This Opinion considers the effects of three new beach nourishment projects located in New Jersey: Port Monmouth, Union Beach, and Elberon to Loch Arbour. The projects will use sand from the SBOBA which is located 1-3 miles offshore of the southern end of Sandy Hook, NJ. The mean water depth of the borrow area is 50 feet (USACE-NYD 2006). Each project will also construct structures along the shoreline that aim to reduce damages from future storm events. These activities are carried out by the USACE and their contractors as independent actions as detailed below. As described below, each of the three projects have different start dates with the durations ranging from 1 to 50 years.

3.1 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 for this consultation includes the SBOBA, Raritan Bay, and the Atlantic Ocean waters off Elberon to Loch Arbour, specifically those areas where dredging, beach nourishment, and construction events will be completed (i.e., SBOBA, Port Monmouth, Union Beach, Elberon to Loch Arbour) (See Figures 1 through 4. In addition, the action area also includes the waters between and immediately adjacent to these areas where project vessels will travel and dredged material will be transported to these sites. The action area will also encompass the underwater area where dredging or fill placement will result in increased suspended sediment and where sound pressure waves associated with pile driving will be experienced. The size of the sediment plume will vary depending on the type of dredge used and is detailed below. Effects of pile driving are expected to be limited to an area with a radius of 30 meters around the pile driving site.

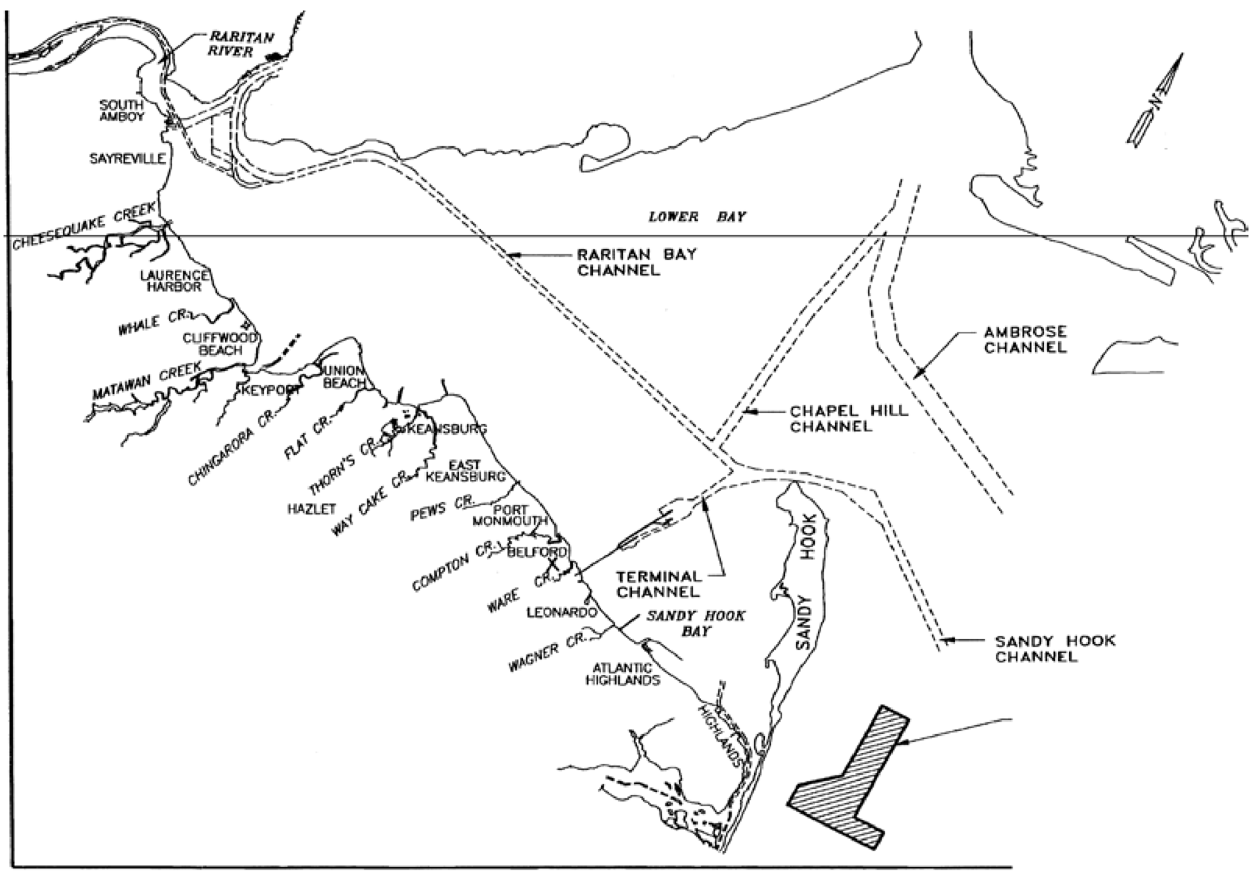


Figure 1: Location of the Sea Bright Offshore Burrow Area

3.2 Port Monmouth

Port Monmouth is located on Raritan Bay, NJ and is bordered by East Keansburg and Belford. The action is anticipated to begin in March, 2014, and will last for approximately 13 months.

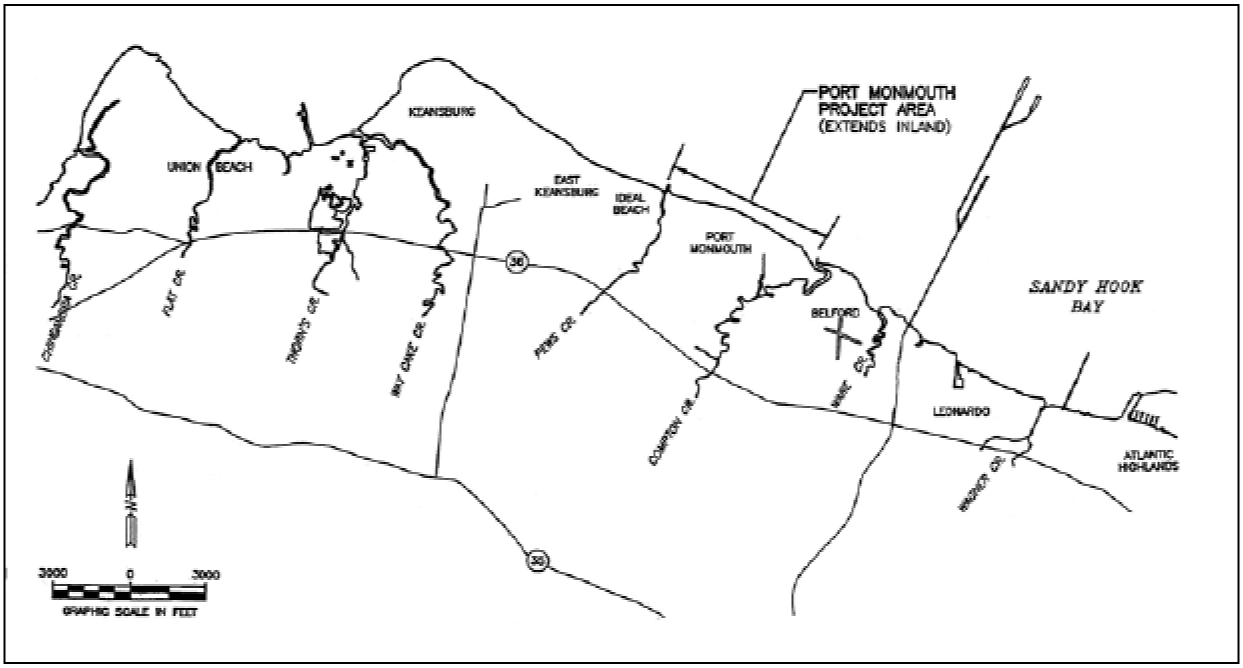


Figure 2. Location of Port Monmouth Project Area

The shoreline protection component of the proposed action aims to reduce damages from coastal erosion and tidal inundation along the project's bay shoreline. Approximately 391,000 cubic yards (CY) of sand will be dredged from SBOBA via a hopper dredge equipped with an unexploded ordnance (UXO) screen (longitudinal bar screens that typically have an opening of 1.25 - 1.5" x 6"). The hopper will dredge the material from approximately 46 acres of the SBOBA, with an average of 5.5 feet of dredged material removed, and then sail to a pumpout area. Resuspension of fine-grained dredged material during hopper dredging operations is caused by the dragheads as they are pulled through the sediment, turbulence generated by the vessel and its prop wash, and overflow of turbid water during hopper filling operations. Nearfield concentrations ranged from 80.0-475.0 mg/l. By a distance of 4,000 feet from the dredge, plume concentrations are expected to return to background levels.

The approximate distance from the SBOBA to the pump out station is anticipated to be approximately 16 miles. The approximate and typical transit speed during the nourishment projects operating in the SBOBA to Raritan Bay are expected to be: 9.8-10.8 mph (8.5-9.4 knots) between the borrow area to Raritan Bay; and 2-3 mph (1.7-2.6 knots) while dredging. The hopper will then connect to a pumpout barge where it will pump sand from the hopper onto the shoreline via a pipeline along approximately 3,300 linear feet of shoreline from Pews Creek to Compton Creek.

Dune integrity will be ensured by a extending a section of beach seaward of the dune through periodic nourishment beginning approximately 10 years after initial construction and continuing at 10 year intervals for 40 years after initial construction; the interval can be shorter or longer depending on the project conditions over time. The estimated amount of sand for renourishment will be 95,200 CY per event and the source of sand will be upland. The sand will be transported via truck to the site.

One 305 foot long stone terminal groin at the western end of the dune line will be constructed. The groin will extend seaward approximately 280 feet from the existing mean high water mark and approximately 0.57 acres of seafloor will be affected by the footprint of the groin. Approximately 6 tons of median size armor stone will be used to construct the onshore and offshore portions of the structure. The cross-section consists of one layer of 6-ton median armor underlain by two layers of 1200 lb. median underlayer stone, underlain by a 1 ft thick layer of 60-lb. median bedding stone on top of geotextile. A tugboat/barge will be used to place the stones from water, and a crane or dozer will be used to place the stones from land. The stones will stretch continuously along the groin structure and the barge/tugboat will relocate to a new position to place new layers of stones. The placement of stone (bedding, armor, and underlayer) during the construction of the groins will disturb shoreline sediments and may cause a temporary increase in suspended sediment in the nearshore area. Turbidity levels associated with any sediment plume are expected to be < 5mg/L.

An existing timber fishing pier will be modified to include a new access ramp and a 195 linear foot extension to the seaward end of the fishing pier. Approximately 40 timber piles, one foot in diameter will be installed via jetting. Jetting may be completed via land, up to approximately 5-6 feet. A barge with a tugboat will be used beyond approximately 5-6 feet seaward. The barge/tug will be stationary except when relocating to a new position to reach a new set of timber pile installation points.

A system of levees and floodwalls will be constructed to extend continuously from a levee in adjacent East Keansburg, NJ, across Pews Creek, to connect with the shore protection segment along the bay shore, and then along undeveloped lands adjoining Compton Creek to higher existing elevation (USACE-NYD 2000). With the exception of a sector gate at Pews Creek, this part of the project will be on land.

The sector gate at Pews Creek will have a 40 foot wide opening and will be 21 feet in height. The gate will be constructed across Pews Creek at approximately 91.5 meters south of the Pews Creek Bridge (e.g., where Port Monmouth road crosses the creek). This location is approximately 535 meters from where the creek meets Raritan Bay. The gate will connect to an existing concrete pile supported T-wall on the east side of Pews Creek for about 150 feet where it will join the existing Keansburg levee. During construction of the gate, steel sheet piling may be installed via a vibratory or impact hammer to support the structure.

In summary, the total amount of beach fill required for the shore protection construction events are as follows:

Construction Year	Estimated Beach Fill Quantity (CY)	Source of Sand
Initial Construction – 2013	391,000	SBOBA
10 Years Post Initial Construction	95,200	Upland (trucking)
20 Years Post Initial Construction	95,200	Upland (trucking)
30 Years Post Initial Construction	95,200	Upland (trucking)
40 Years Post Initial Construction	95,200	Upland (trucking)
TOTAL	**771,800**	

Table 1. Estimated dredged quantities for Port Monmouth beach fill.

3.3 Union Beach

Union Beach occupies a 1.8 square mile area of land, including approximately 3,000 feet of shoreline along the coast of Raritan Bay, NJ. Union Beach is bordered by the Borough of Keansburg to the east and Chingarora Creek to the west. The proposed action is expected to begin in August, 2014, and will last for approximately two years.

Figure 3. Location of the recommended plan for Union Beach

Approximately 688,000 CY of sand will be dredged from SBOBA via a hopper dredge equipped with a UXO screen similar to that described for Port Monmouth. The hopper will dredge the material and then sail to a pumpout area. The distance from the SBOBA to the pump out station is anticipated to be approximately 16 miles. The approximate transit speed of the dredge from the SBOBA to the project area is expected to be: 9.8-10.8 mph (8.5-9.4 knots); and 2-3 mph (1.7-

2.6 knots) while dredging. The hopper will then connect to a pumpout barge where it will pump sand from the hopper onto the shoreline via a pipeline along approximately 3,000 feet of shoreline.

Beach renourishment will occur every 9 years for 50 years at an expected volume of 21,000 CY of sand per event and the source of sand will be upland. Sand will be transported via truck to the site.

A 3,160 foot beach berm and dune system will be constructed using sand from the SBOBA. The dune will be at 17 feet National Geodetic Vertical Datum (NGVD) with a 50 foot wide crest extending down to the 9 feet NGVD berm elevation. The width of the berm would range from 15 m (50 feet) near the two terminal groins, to a maximum of 50 m (164 feet) between Beach Street and Florence Avenue. The beach and dune are designed to contain 688,000 CY of fill. The dune section will be stabilized with dune grass and fencing, and three wood overwalks will be constructed to protect dune vegetation and provide public access to beach areas. In addition, a walkway connecting the overwalks will run along the crest of the dune. The construction of the beach berm and dune system will take place on land.

A 228 foot eastern terminal groin, with an associated 630 foot revetment, and a 245 foot western terminal groin, with a 405 foot revetment will be constructed. The heads of the groins will be constructed of 4 ton quarry stone placed over 2 to 40 lb core and bedding stone. The trunks of the groins will be constructed of 11 ton quarry stone and 2,200 lb underlayer stone placed on 6 to 110 lb core and bedding stone. The armor layers and underlayers will be two units thick. The bedding layers will be two feet thick. The total amount of acreage of seafloor to be affected by groin placement would be .09 acres. The groin construction method described for Port Monmouth in Section 3.2 also applies to this project.

Multiple levee/floodwalls will be constructed. With the exception of storm surge barriers, all levee/floodwall elements of the project will be built on land and will not affect any ESA-listed species. Therefore, this part of the project will not be considered in this Opinion

Storm surge barriers (across Flat Creek and East Creek) with pump stations and sluice gates will be constructed. On East Creek Tributary, the existing bridge on the Henry Hudson Trail will be removed and replaced with a gate structure containing three 6 foot by 6 foot box culverts with sluice gates. The existing bridge is 18.4 feet wide and the proposed opening with three 6 foot by 6 foot sluice gates will be 18 feet wide. Steel sheet piles will be installed via a vibratory or impact hammer during the construction of sluice gates and storm surge barriers.

In summary, the total amount of beach fill required per shore protection construction event for the Union Beach project is listed in the table below.

Construction Year	Beach Fill Quantity (CY)	Total SBOBA source (CY)	Total Upland source (CY)
Initial Beach Nourishment – date TBD	688,000	688,000	0
Beach Berm and Dune System Construction – date TBD	18,000	18,000	0
9 Years Post Initial Construction	21,000	0	21,000
18 Years Post Initial Construction	21,000	0	21,000
27 Years Post Initial Construction	21,000	0	21,000
36 Years Post Initial Construction	21,000	0	21,000
45 Years Post Initial Construction	21,000	0	21,000
TOTAL	811,000	706,000	105,000

Table 2. Projected beach fill quantities and sand sources for the Union Beach project.

3.4 Elberon to Loch Arbour

Elberon to Loch Arbour is one designated reach along the coast of NJ. The project area covers approximately 3.5 miles from Lake Takanassee to Deal Lake. The initial construction of the proposed action is expected to begin in September, 2014, and will last for approximately 12-16 months. Beach nourishment cycles will take place every 6 years for approximately 50 years.

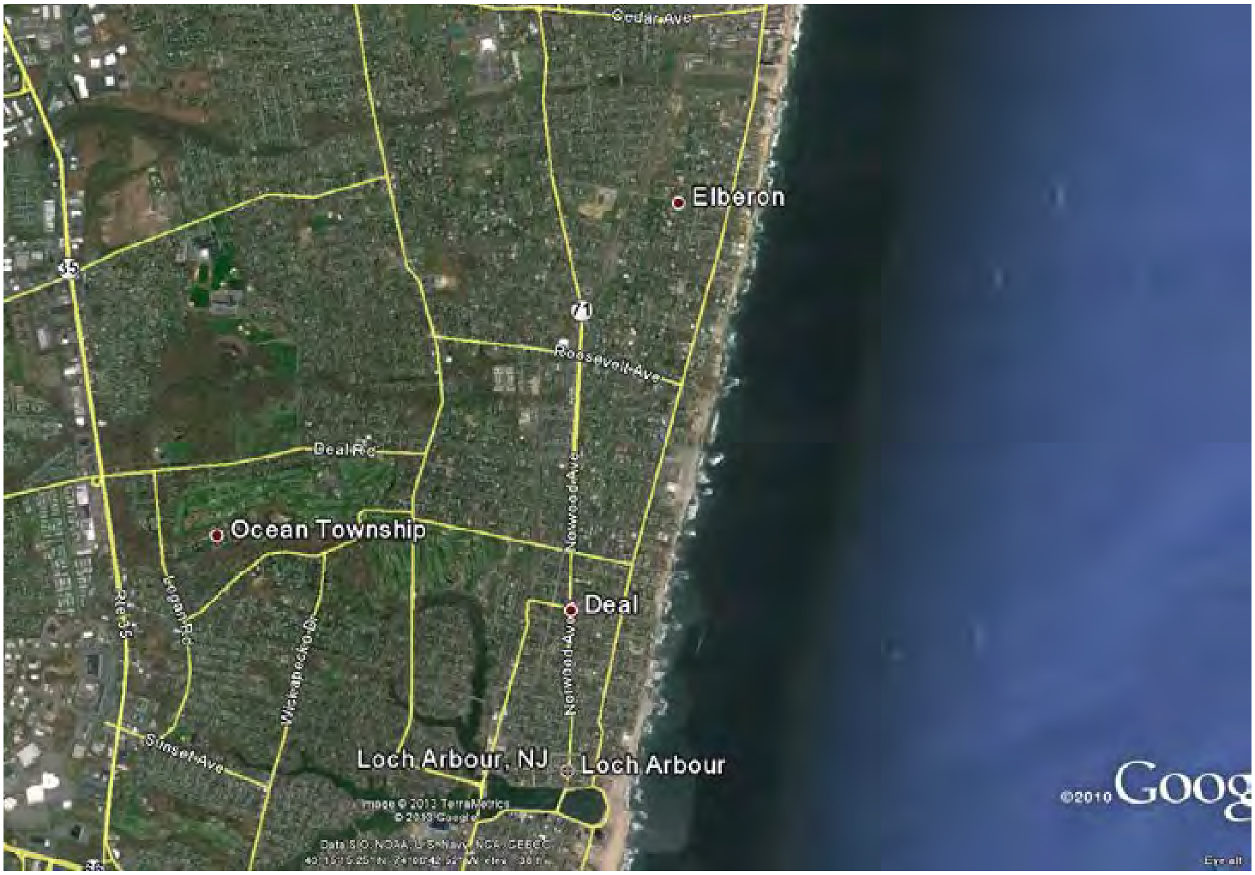


Figure 4. Elberon to Loch Arbour, Monmouth County, NJ.

Approximately 4,450,000 CY of sand will be dredged from SBOBA via a hopper dredge equipped with a UXO screen similar to the previous two projects. The hopper will dredge the material and then sail to a pumpout area. The distance from the SBOBA to the pump out station is anticipated to be approximately 12 miles. The approximate transit speed from the SBOBA to the project area are expected to be: 9.8-10.8 mph (8.5-9.4 knots); and 2-3 mph (1.7-2.6 knots) while dredging. The hopper will then connect to a pumpout barge where it will pump sand from the hopper onto the shoreline via a pipeline along approximately 17,000 linear feet of shoreline and would include construction of a 100 foot wide berm at an elevation of 10 feet above MLW with a 2 foot high storm berm cap.

Beach Renourishment will occur every 6 years for 50 years at an expected volume of 1,298,000 CY of sand per cycle. The sand will be dredged from SBOBA and will follow the same procedure as the initial dredging operation.

Six existing stone groins within this reach of the project area will be notched to allow for sediment transport and to prevent sediment impoundment. Notching involves removing a portion of the landward end of the groin such that water and sediment can follow its natural long shore flow and deposition patterns. It is accomplished by land based heavy equipment, such as front loaders and cranes. Rocks from the groins are simply removed from the line of the groin and placed elsewhere, usually along side of the groin at the beach side of the “notch.”

Approximately 14 storm water outfalls will be extended beyond the construction template. Outfall extensions are to be supported by timber piles (10 to 12 inches in diameter) or a similar composite material pile. The piles and outfall extensions will be constructed after sand fill is placed under the pipe alignment. The piles will be driven via an impact or vibratory hammer. This operation will take place in near shore waters. Effects of increased underwater noise levels will be present within a 30 meter radius surrounding the piles being driven. Construction in the landward (shallowest) sections of the pipe alignment will be done with land based equipment. For the outfall alignments that extend further seaward into subtidal areas barge based equipment may be utilized. All outfalls will not be constructed at once and would be sequenced throughout the overall beach construction schedule.

In summary, the total amount of beach fill required from the SBOBA for the construction and maintenance of Elberon to Loch Arbour is as follows:

Construction Year	Estimated Beach Fill Quantity (CY) from SBOBA
Initial Construction – 2014	4,450,452
6 Years Post Initial Construction	1,298,000
12 Years Post Initial Construction	1,298,000
18 Years Post Initial Construction	1,298,000
24 Years Post Initial Construction	1,298,000
30 Years Post Initial Construction	1,298,000
36 Years Post Initial Construction	1,298,000
42 Years Post Initial Construction	1,298,000
48 Years Post Initial Construction	1,298,000
TOTAL	14,834,452

Table 3. Estimated dredge quantities for Elberon to Loch Arbour beach fill.

4.0 STATUS OF LISTED SPECIES IN THE ACTION AREA

Several species listed under our jurisdiction occur in the action area for this consultation. We have determined that the actions being considered in the Opinion are not likely to adversely affect shortnose sturgeon (*Acipenser brevirostrum*) and hawksbill sea turtles (*Eretmochelys imbricata*), both of which are listed as endangered species under the ESA. These species are not known to occur in the action area. Thus, these species will not be considered further in this Opinion.

We have determined that the actions being considered in this biological opinion may affect the following endangered or threatened species under our jurisdiction:

Sea Turtles

Northwest Atlantic DPS of Loggerhead sea turtle (*Caretta caretta*)	Threatened
Leatherback sea turtle (*Dermochelys coriacea*)	Endangered
Kemp's ridley sea turtle (*Lepidochelys kempi*)	Endangered
Green sea turtle (*Chelonia mydas*)	Endangered/Threatened[1]

Cetaceans

North Atlantic right whale (*Eubalaena glacialis*)	Endangered
Humpback whale (*Megaptera novaeangliae*)	Endangered
Fin whale (*Balaenoptera physalus*)	Endangered

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

This section will focus on the status of the various species within the action area, summarizing information necessary to establish the environmental baseline and to assess the effects of the proposed actions.

4.1 Status of Sea Turtles

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

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

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

following numbers were obtained from http://www.nmfs.noaa.gov/pr/health/oilspill/). To date, 469 of the live recovered sea turtles have been successfully returned to the wild, 25 died during rehabilitation, and 42 are still in care but will hopefully be returned to the wild eventually. During the clean-up period, 613 dead sea turtles were recovered in coastal waters or on beaches in Mississippi, Alabama, Louisiana, and the Florida Panhandle. As of February 2011, 478 of these dead turtles had been examined. Many of the examined sea turtles showed indications that they had died as a result of interactions with trawl gear, most likely used in the shrimp fishery, and not as a result of exposure to or ingestion of oil.

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

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

4.1.1 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 also exposed to a variety of natural and anthropogenic threats in the terrestrial and marine environment.

Listing History

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

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

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

On March 16, 2010, NMFS and USFWS published a proposed rule (75 FR 12598) to divide the worldwide population of loggerhead sea turtles into nine DPSs, as described in the 2009 Status Review. Two of the DPSs were proposed to be listed as threatened and seven of the DPSs, including the Northwest Atlantic Ocean DPS, were proposed to be listed as endangered. NMFS and the USFWS accepted comments on the proposed rule through September 13, 2010 (75 FR 30769, June 2, 2010). On March 22, 2011 (76 FR 15932), NMFS and USFWS extended the date by which a final determination 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 is composed of nine DPSs (as defined in Conant *et al*., 2009) that constitute species that may be listed as threatened or endangered under the ESA. Five DPSs were listed as endangered (North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Northeast Atlantic Ocean, and Mediterranean Sea), and four DPSs were listed as threatened (Northwest Atlantic Ocean, South Atlantic Ocean, Southeast Indo-Pacific Ocean, and Southwest Indian Ocean). Note that the Northwest Atlantic Ocean (NWA) DPS and the Southeast Indo-Pacific Ocean DPS were originally proposed as endangered. The NWA DPS was determined to be threatened based on review of nesting data available after the proposed rule was published, information provided in public comments on the proposed rule, and further discussions within the agencies. The two primary factors considered were population abundance and population trend. NMFS and USFWS found that an endangered status for the NWA DPS was not warranted given the large size of the nesting population, the overall nesting population remains widespread, the trend for the nesting population appears to be stabilizing, and substantial conservation efforts are underway to address threats. This final listing rule became effective on October 24, 2011.

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

Presence of Loggerhead Sea Turtles in the Action Areas

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

Distribution and Life History

Ehrhart *et al.* (2003) provided a summary of the literature identifying known nesting habitats and foraging areas for loggerheads within the Atlantic Ocean. Detailed information is also provided in the 5-year status review for loggerheads (NMFS and USFWS 2007a), the TEWG report (2009), and the final revised recovery plan for loggerheads in the Northwest Atlantic Ocean (NMFS and USFWS 2008).

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

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

Recent studies have established that the loggerhead's life history is more complex than previously believed. Rather than making discrete developmental shifts from oceanic to neritic environments, research is showing that both adults and (presumed) neritic stage juveniles continue to use the oceanic environment and will move back and forth between the two habitats (Witzell 2002; Blumenthal *et al.* 2006; Hawkes *et al.* 2006; McClellan and Read 2007; Mansfield *et al*. 2009). One of the studies tracked the movements of adult post-nesting females and found that differences in habitat use were related to body size with larger adults staying in coastal waters and smaller adults traveling to oceanic waters (Hawkes *et al.* 2006). A tracking study of large juveniles found that the habitat preferences of this life stage were also diverse with some remaining in neritic waters and others moving off into oceanic waters (McClellan and Read 2007). However, unlike the Hawkes *et al.* (2006) study, there was no significant difference in the body size of turtles that remained in neritic waters versus oceanic waters (McClellan and Read 2007).

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

As presented below, Table 3 from the 2008 loggerhead recovery plan (Table 4 in this Opinion) highlights the key life history parameters for loggerheads nesting in the United States.

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

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

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

Table 4. Typical values of life history parameters for loggerheads nesting in the U.S.

Population Dynamics and Status

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

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

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

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

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

From the beginning of standardized index surveys in 1989 until 1998, the PFRU, the largest nesting assemblage in the Northwest Atlantic by an order of magnitude, had a significant increase in the number of nests. However, from 1998 through 2008, there was a 41% decrease in annual nest counts from index beaches, which represent an average of 70% of the statewide nesting activity (NMFS and USFWS 2008). From 1989-2008, the PFRU had an overall declining nesting trend of 26% (95% CI: -42% to -5%; NMFS and USFWS 2008). With the addition of nesting data through 2010, the nesting trend for the PFRU does not show a nesting decline statistically different from zero (76 FR 58868, September 22, 2011).

The NRU, the second largest nesting assemblage of loggerheads in the United States, has been declining at a rate of 1.3% annually since 1983 (NMFS and USFWS 2008). The NRU dataset included 11 beaches with an uninterrupted time series of coverage of at least 20 years; these beaches represent approximately 27% of NRU nesting (in 2008). Through 2008, there was strong statistical data to suggest the NRU has experienced a long-term decline, but with the inclusion of nesting data through 2010, nesting for the NRU is showing possible signs of stabilizing (76 FR 58868, September 22, 2011).

Evaluation of long-term nesting trends for the NGMRU is difficult because of changed and expanded beach coverage. However, the NGMRU has shown a significant declining trend of 4.7% annually since index nesting beach surveys were initiated in 1997 (NMFS and USFWS 2008). 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 (from 1989-2008) with approximately 1,272 females nesting per year; (2) for the PFRU, a mean of 64,513 nests per year (from 1989-2007) with approximately 15,735 females nesting per year; (3) for the DTRU, a mean of 246 nests per year (from 1995-2004, excluding 2002) with approximately 60 females nesting per year; and (4) for the NGMRU, a mean of 906 nests per year (from 1995-2007) with approximately 221 females nesting per year. For the GCRU, the only estimate available for the number of loggerhead nests per year is from Quintana Roo, Yucatán, Mexico, where a range of 903-2,331 nests per year was estimated from 1987-2001 (NMFS and USFWS 2007a). There are no annual nest estimates available for the Yucatán since 2001 or for any other regions in the GCRU, nor are there any estimates of the number of nesting females per year for any nesting assemblage in this recovery unit. Note that the above values for average nesting females per year were based upon 4.1 nests per female per Murphy and Hopkins (1984).

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

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

Maier *et al.* (2004) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the southeast coast of the United States. (Winyah Bay, South Carolina to St. Augustine, Florida) during the period 2000-2003. A comparison of loggerhead catch data from this study with historical values suggested that in-water populations of loggerhead sea turtles along the southeast U.S. coast appear to be larger, possibly an order of magnitude higher than they were 25 years ago, but the authors caution a direct comparison between the two studies given differences in sampling methodology (Maier *et al.* 2004). A comparison of catch rates for sea turtles in pound net gear fished in the Pamlico-Albemarle Estuarine Complex of North Carolina between the years 1995-1997 and 2001-2003 found a significant increase in catch rates for loggerhead sea turtles for the latter period (Epperly *et al.* 2007). A long-term, on-going study of loggerhead abundance in the Indian River Lagoon System of Florida found a significant increase in the relative abundance of loggerheads over the last 4 years of the study (Ehrhart *et al.* 2007). However, there was no discernible trend in loggerhead abundance during the 24-year time period of the study (1982-2006) (Ehrhart *et al.* 2007). At St. Lucie Power Plant, data collected from 1977-2004 show an increasing trend of loggerheads at the power plant intake structures (FPL and Quantum Resources 2005).

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

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

As part of the Atlantic Marine Assessment Program for Protected Species (AMAPPS), line transect aerial abundance surveys and turtle telemetry studies were conducted along the Atlantic coast in the summer of 2010. AMAPPS is a multi-agency initiative to assess marine mammal, sea turtle, and seabird abundance and distribution in the Atlantic. Aerial surveys were conducted from Cape Canaveral, Florida to the Gulf of St. Lawrence, Canada. Satellite tags on juvenile loggerheads were deployed in two locations – off the coasts of northern Florida to South Carolina (n=30) and off the New Jersey and Delaware coasts (n=14). As presented in NMFS NEFSC (2011), the 2010 survey found a preliminary total surface abundance estimate within the entire study area of about 60,000 loggerheads (CV=0.13) or 85,000 if a portion of unidentified hard-shelled sea turtles were included (CV=0.10). Surfacing times were generated from the satellite tag data collected during the aerial survey period, resulting in a 7% (5%-11% inter-quartile range) median surface time in the South Atlantic area and a 67% (57%-77% inter-quartile range) median surface time to the north. The calculated preliminary regional abundance estimate is about 588,000 loggerheads along the U.S. Atlantic coast, with an inter-quartile range of 382,000-817,000 (NMFS NEFSC 2011). The estimate increases to approximately 801,000 (inter-quartile range of 521,000-1,111,000) when based on known loggerheads and a portion of unidentified turtle sightings. The density of loggerheads was generally lower in the north than the south; based on number of turtle groups detected, 64% were seen south of Cape Hatteras, North Carolina, 30% in the southern Mid-Atlantic Bight, and 6% in the northern Mid-Atlantic Bight. Although they have been seen farther north in previous studies (*e.g.*, Shoop and Kenney 1992), no loggerheads were observed during the aerial surveys conducted in the summer of 2010 in the more northern zone encompassing Georges Bank, Cape Cod Bay, and the Gulf of Maine. These estimates of loggerhead abundance over the U.S. Atlantic continental shelf are considered very preliminary. A more thorough analysis will be completed pending the results of further studies related to improving estimates of regional and seasonal variation in loggerhead surface time (by increasing the sample size and geographical area of tagging) and other information needed to improve the biases inherent in aerial surveys of sea turtles (*e.g.*, research on depth of detection and species misidentification rate). This survey effort represents the most comprehensive assessment of sea turtle abundance and distribution in many years. Additional aerial surveys and research to improve the abundance estimates are anticipated in 2011-2014, depending on available funds.

Threats

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

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

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

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

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

Of the many fisheries known to adversely affect loggerheads, the U.S. South Atlantic and Gulf of Mexico shrimp fisheries were considered to pose the greatest threat of mortality to neritic juvenile and adult age classes of loggerheads (NRC 1990, Finkbeiner *et al*. 2011). Significant changes to the South Atlantic and Gulf of Mexico shrimp fisheries have occurred since 1990, and the effects of these shrimp fisheries on ESA-listed species, including loggerhead sea turtles, have been assessed several times through section 7 consultation. There is also a lengthy regulatory history with regard to the use of Turtle Excluder Devices (TEDs) in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries (Epperly and Teas 2002; NMFS 2002a; Lewison *et al.* 2003). 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 designs and TED enforcement, interactions between loggerheads and the shrimp fishery have also been declining because of reductions in fishing effort unrelated to fisheries management actions. The 2002 South Atlantic and GOM Shrimp Opinion (NMFS 2002a) take estimates are based in part on fishery effort levels. In recent years, low shrimp prices, rising fuel costs, competition with imported products, and the impacts of recent hurricanes in the Gulf of Mexico have all impacted the shrimp fleets; in some cases reducing fishing effort by as much as 50% for offshore waters of the Gulf of Mexico (GMFMC 2007). As a result, loggerhead interactions and mortalities in the Gulf of Mexico have been substantially less than projected in the 2002 Opinion. Currently, the estimated annual number of interactions between loggerheads and shrimp trawls in the Gulf of Mexico shrimp fishery is 23,336, with 647 (2.8%) of those interactions resulting in mortality (Memo from Dr. B. Ponwith, Southeast Fisheries Science Center to Dr. R. Crabtree, Southeast Region, PRD, December 2008). 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 5-year status reviews as a priority for the recovery of all sea turtle species. In the threats analysis of the loggerhead recovery plan, trawl bycatch is identified as the greatest source of mortality. While loggerhead bycatch in U.S. Mid-Atlantic bottom otter trawl gear was previously estimated for the period 1996-2004 (Murray 2006, 2008), a recent bycatch analysis estimated the number of loggerhead sea turtle interactions with U.S. Mid-Atlantic bottom trawl gear from 2005-2008 (Warden 2011a). Northeast Fisheries Observer Program data from 1994-2008 were used to develop a model of interaction rates and those predicted rates were applied to 2005-2008 commercial fishing data to estimate the number of interactions for the trawl fleet. The number of predicted average annual loggerhead interactions for 2005-2008 was 292 (CV=0.13, 95% CI=221-369), with an additional 61 loggerheads (CV=0.17, 95% CI=41-83) interacting with trawls but being released through a TED. Of the 292 average annual observable loggerhead interactions, approximately 44 of those were adult equivalents. Warden (2011b) found that latitude, depth and SST were associated with the interaction rate, with the rates being highest south of 37°N latitude in waters < 50 m deep and SST > 15°C. This estimate is a decrease from the average annual loggerhead bycatch in bottom otter trawls during 1996-2004, estimated to be 616 sea turtles (CV=0.23, 95% CI over the 9-year period: 367-890) (Murray 2006, 2008).

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

An estimate of the number of loggerheads taken annually in U.S. Mid-Atlantic gillnet fisheries has also recently been published (Murray 2009a, b). From 1995-2006, the annual bycatch of loggerheads in U.S. Mid-Atlantic gillnet gear was estimated to average 350 turtles (CV=0.20, 95% CI over the 12-year period: 234 to 504). Bycatch rates were correlated with latitude, sea surface temperature, and mesh size. The highest predicted bycatch rates occurred in warm waters of the southern Mid-Atlantic in large-mesh (>7 inch/17.8 cm) gillnets (Murray 2009a).

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

U.S. Atlantic tuna and swordfish longline fisheries as well as others).

Documented takes also occur in other fishery gear types and by non-fishery mortality sources (*e.g.*, hopper dredges, power plants, vessel collisions), although quantitative/qualitative estimates are only available for activities on which NMFS has consulted (See sections 5 below). Past and future impacts of global climate change are considered in Section 6.0 below.

Summary of Status for Loggerhead Sea Turtles

Loggerheads continue to be affected by many factors occurring on nesting beaches and in the water. These include poaching, habitat loss, and nesting predation that affects eggs, hatchlings, and nesting females on land, as well as fishery interactions, vessel interactions, marine pollution, and non-fishery (*e.g*., dredging) operations affecting all sexes and age classes in the water (NRC 1990; NMFS and USFWS 2007a, 2008). As a result, loggerheads still face many of the original threats that were the cause of their listing under the ESA. 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.

4.1.2 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. 2011).

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

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

The location and size classes of dead turtles recovered by the STSSN suggests that benthic immature developmental areas occur along the U.S. coast and that these areas may change given resource quality and quantity (TEWG 2000). Developmental habitats are defined by several characteristics, including coastal areas sheltered from high winds and waves such as embayments and estuaries, and nearshore temperate waters shallower than 50 m (NMFS and USFWS 2007b). 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 2007b).

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

Adult Kemp's ridleys are found in the coastal regions of the Gulf of Mexico and southeastern United States, but are typically rare in the northeastern U.S. waters of the Atlantic (TEWG

2000). Adults are primarily found in nearshore waters of 37 m or less that are rich in crabs and have a sandy or muddy bottom (NMFS and USFWS 2007b).

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 2007b; 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 2007b). 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 2007b; NMFS *et al*. 2011). Conservation efforts by Mexican and U.S. agencies have aided this species by eliminating egg harvest, protecting eggs and hatchlings, and reducing at-sea mortality through fishing regulations (TEWG 2000). Since the mid-1980s, the number of nests observed at Rancho Nuevo and nearby beaches has increased 14-16% per year (Heppell *et al.* 2005), allowing cautious optimism that the population is on its way to recovery. An estimated 5,500 females nested in the State of Tamaulipas over a 3-day period in May 2007 and over 4,000 of those nested at Rancho Nuevo (NMFS and USFWS 2007b). In 2008, 17,882 nests were documented on Mexican nesting beaches (NMFS 2011). There is limited nesting in the United States, most of which is located in South Texas. While six nests were documented in 1996, a record 195 nests were found in 2008 (NMFS 2011).

Threats

Kemp's ridleys face many of the same natural threats as loggerheads, including destruction of nesting habitat from storm events, predators, and oceanographic-related events such as cold-stunning. Although cold-stunning can occur throughout the range of the species, it may be a greater risk for sea turtles that utilize the more northern habitats of Cape Cod Bay and Long Island Sound. In the last five years (2006-2010), the number of cold-stunned turtles on Cape Cod beaches averaged 115 Kemp's ridleys, 7 loggerheads, and 7 greens (NMFS unpublished data). The numbers ranged from a low in 2007 of 27 Kemp's ridleys, 5 loggerheads, and 5 greens to a high in 2010 of 213 Kemp's ridleys, 4 loggerheads, and 14 greens. Annual cold stun events vary in magnitude; the extent of episodic major cold stun events may be associated with numbers of turtles utilizing Northeast U.S. waters in a given year, oceanographic conditions, and/or the occurrence of storm events in the late fall. Although many cold-stunned turtles can survive if they are found early enough, these events represent a significant source of natural mortality for Kemp's ridleys.

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

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

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

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 2007b; NMFS *et al*. 2011). The number of nesting females in the Kemp's ridley population declined dramatically from the late 1940s through the mid-1980s, with an estimated 40,000 nesting females in a single *arribada* in 1947 and fewer than 300 nesting females in the entire 1985 nesting season (TEWG 2000; NMFS *et al*. 2011). However, the total annual number of nests at Rancho Nuevo gradually began to increase in the 1990s (NMFS and USFWS 2007b). 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 2007b). 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 2007b). While there is cautious optimism for recovery, events such as the Deepwater Horizon oil release, and stranding events associated increased skimmer trawl use and poor TED compliance in the northern Gulf of Mexico may dampen recent population growth.

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

Based on this and the current best available information, we believe that the Kemp's ridley sea turtle population is currently stable; as protective measures for sea turtles are currently in place and continue to be implemented, we expect this trend to continue or over the next 2 years. This stable trend is based solely on information we have on nesting trends. The number of sea turtles comprising the neritic and oceanic life stages of the population is currently unknown. As a result, the status and future trend of the population as a whole remains unclear. Therefore, until information and data become available on the numbers of individuals comprising the neritic and oceanic life stages, nesting trends represent the best available information and serve as the best representative of the population's trend.

4.1.3 Leatherback sea turtle

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

The largest, extant leatherback nesting group in the Indo-Pacific lies on the North Vogelkop coast of West Papua, Indonesia, with 3,000-5,000 nests reported annually in the 1990s (Suárez *et al.* 2000). However, in 1999, local villagers started reporting dramatic declines in sea turtles near their villages (Suárez 1999). Declines in nesting groups have been reported throughout the western Pacific region where observers report that nesting groups are well below abundance levels that were 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), but a total of only 120 nests on the four primary index beaches (combined) were counted in the 2003-2004 season (Sarti Martinez *et al.* 2007). Since the early 1980s, the Mexican Pacific population of adult female leatherback turtles has declined to slightly more than 200 during 1998-1999 and 1999-2000 (Sarti *et al.* 2000). Spotila *et al.* (2000) reported the decline of the leatherback nesting at Playa Grande, Costa Rica, which had been the fourth largest nesting group in the world and the most important nesting beach in the Pacific. Between 1988 and 1999, the nesting group declined from 1,367 to 117 female leatherback sea turtles. Based on their models, Spotila *et al.* (2000) estimated that the group could fall to less than 50 females by 2003-2004. Another, more recent, analysis of the Costa Rican nesting beaches indicates a decline in nesting during 15 years of monitoring (1989-2004) with approximately 1,504 females nesting in 1988-1989 to an average of 188 females nesting in 2000-2001 and 2003-2004 (NMFS and USFWS 2007d), indicating that the reductions in nesting females were not as extreme as the reductions predicted by Spotila *et al*. (2000).

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

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

Indian Ocean

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

Mediterranean Sea

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

Atlantic Ocean

Distribution and Life History

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

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

The CETAP aerial survey of the outer Continental Shelf from Cape Hatteras, North Carolina to Cape Sable, Nova Scotia conducted between 1978 and 1982 showed leatherbacks to be present throughout the area with the most numerous sightings made from the Gulf of Maine south to Long Island. Leatherbacks were sighted in water depths ranging from 1 to 4,151 m, but 84.4% of sightings were in waters less than 180 m (Shoop and Kenney 1992). Leatherbacks were sighted in waters within a sea surface temperature range similar to that observed for loggerheads; from 7°-27.2°C (Shoop and Kenney 1992). However, leatherbacks appear to have a greater tolerance for colder waters in comparison to loggerhead sea turtles since more leatherbacks were found at the lower temperatures (Shoop and Kenney 1992). Studies of satellite tagged leatherbacks suggest that they spend 10%-41% of their time at the surface, depending on the phase of their migratory cycle (James *et al.* 2005b). The greatest amount of surface time (up to 41%) was recorded when leatherbacks occurred in continental shelf and slope waters north of 38°N (James *et al.* 2005b).

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

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

Population Dynamics and Status

As described earlier, sea turtle nesting survey data is important in that it provides information on the relative abundance of nesting, and the contribution of each population/subpopulation to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually, and as an indicator of the trend in the number of nesting females in the nesting group. The 5-year review for leatherback sea turtles (NMFS and USFWS 2009 compiled the most recent information on mean number of leatherback nests per year for each of the seven leatherback populations or groups of populations that were identified by the Leatherback TEWG as occurring within the Atlantic. These are: Florida, North Caribbean, Western Caribbean, Southern Caribbean, West Africa, South Africa, and Brazil (TEWG 2007).

In the United States, the Florida Statewide Nesting Beach Survey program has documented an increase in leatherback nesting numbers from 98 nests in 1988 to between 800 and 900 nests in the early 2000s (NMFS and USFWS 2007d). Stewart *et al*. (2011) evaluated nest counts from 68 Florida beaches over 30 years (1979-2008) and found that nesting increased at all beaches with trends ranging from 3.1%-16.3% per year, with an overall increase of 10.2% per year. An analysis of Florida's index nesting beach sites from 1989-2006 shows a substantial increase in leatherback nesting in Florida during this time, with an annual growth rate of approximately 1.17 (TEWG 2007). The TEWG reports an increasing or stable nesting trend for all of the seven populations or groups of populations with the exception of the Western Caribbean and West Africa. The leatherback rookery along the northern coast of South America in French Guiana and Suriname supports the majority of leatherback nesting in the western Atlantic (TEWG 2007), and represents more than half of total nesting by leatherback sea turtles worldwide (Hilterman and Goverse 2004). Nest numbers in Suriname have shown an increase and the long-term trend for the Suriname and French Guiana nesting group seems to show an increase (Hilterman and

Goverse 2004). In 2001, the number of nests for Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). The TEWG (2007) report indicates that using nest numbers from 1967-2005, a positive population growth rate was found over the 39-year period for French Guinea and Suriname, with a 95% probability that the population was growing. Given the magnitude of leatherback nesting in this area compared to other nest sites, negative impacts in leatherback sea turtles in this area could have profound impacts on the entire species.

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

Threats

The 5-year status review (NMFS and USFWS 2007d) and TEWG (2009) report provide summaries of natural as well as anthropogenic threats to leatherback sea turtles. Of the Atlantic sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear, trap/pot gear in particular. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), their diving and foraging behavior, their distributional overlap with the gear, their possible attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, and perhaps to the lightsticks used to attract target species in longline fisheries. Leatherbacks entangled in fishing gear generally have a reduced ability to feed, dive, surface to breathe, or perform any other behavior essential to survival (Balazs 1985). In addition to drowning from forced submergence, they may be more susceptible to boat strikes if forced to remain at the surface, and entangling lines can constrict blood flow resulting in tissue necrosis. The long-term impacts of entanglement on leatherback health remain unclear. Innis *et al*. (2010) conducted a health evaluation of leatherback sea turtles during direct capture (n=12) and disentanglement (n=7). They found no significant difference in many of the measured health parameters between entangled and directly captured turtles. However, blood parameters, including but not limited to sodium, chloride, and blood urea nitrogen, for entangled turtles showed several key differences that were most likely due to reduced foraging and associated seawater ingestion, as well as a general stress response.

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

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

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

Leatherback interactions with the U.S. South Atlantic and Gulf of Mexico shrimp fisheries are also known to occur (NMFS 2002b). Leatherbacks are likely to encounter shrimp trawls working in the coastal waters off the U.S. Atlantic coast (from Cape Canaveral, Florida through North Carolina) as they make their annual spring migration north. For many years, TEDs that

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

were required for use in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries were less effective for leatherbacks as compared to the smaller, hard-shelled turtle species, because the TED openings were too small to allow leatherbacks to escape. To address this problem, NMFS issued a final rule on February 21, 2003, to amend the TED regulations (68 FR 8456, February 21, 2003). Modifications to the design of TEDs are now required in order to exclude leatherbacks as well as large benthic immature and sexually mature loggerhead and green sea turtles. Given those modifications, Epperly *et al.* (2002) anticipated an average of 80 leatherback mortalities a year in shrimp gear interactions, dropping to an estimate of 26 leatherback mortalities in 2009 due to effort reduction in the Southeast shrimp fishery (Memo from Dr. B. Ponwith, SEFSC, to Dr. R. Crabtree, SERO, January 5, 2011).

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

Gillnet fisheries operating in the waters of the Mid-Atlantic states are also known to capture, injure, and/or kill leatherbacks when these fisheries and leatherbacks co-occur. Data collected by the NEFSC Fisheries Observer Program from 1994-1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54%-92%. In North Carolina, six additional leatherbacks were reported captured in gillnet sets in the spring (NMFS SEFSC 2001). In addition to these, in September 1995, two dead leatherbacks were removed from an 11-inch (28.2-cm) monofilament shark gillnet set in the nearshore waters off of Cape Hatteras (STSSN unpublished data reported in NMFS SEFSC 2001). Lastly, Murray (2009a) reports five observed leatherback captures in Mid-Atlantic sink gillnet fisheries between 1994 and 2008.

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

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

Summary of Status for Leatherback Sea Turtles

In the Pacific Ocean, the abundance of leatherback sea turtles on nesting beaches has declined dramatically over the past 10 to 20 years. Nesting groups throughout the eastern and western Pacific Ocean have been reduced to a fraction of their former abundance by the combined effects of human activities that have reduced the number of nesting females and reduced the reproductive success of females that manage to nest (for example, egg poaching) (NMFS and USFWS 2007d). 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 2007d).

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

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

Based on this and the current best available information, we believe that the leatherback sea turtle population is currently stable; as protective measures for sea turtles are currently in place and continue to be implemented, we expect this trend to continue or over the next 2 years. This stable trend is based solely on information we have on nesting trends. The number of sea turtles comprising the neritic and oceanic life stages of the population is currently unknown. As a result, the status and future trend of the population as a whole remains unclear. Therefore, until information and data become available on the numbers of individuals comprising the neritic and oceanic life stages, nesting trends represent the best available information and serve as the best representative of the population's trend.

4.1.4 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, 2007c; Seminoff 2004). In 1978, the Atlantic population of the green sea turtle was listed as threatened under the ESA, except for the breeding populations in Florida and on the Pacific coast of Mexico, which were listed as endangered. As it is difficult to differentiate between breeding populations away from the nesting beaches, all green sea turtles in the water are considered endangered.

Pacific Ocean

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

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

Indian Ocean

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

Mediterranean Sea
There are four nesting concentrations of green sea turtles in the Mediterranean from which data are available – Turkey, Cyprus, Israel, and Syria. Currently, approximately 300-400 females nest each year, about two-thirds of which nest in Turkey and one-third in Cyprus. Although green sea turtles are depleted from historic levels in the Mediterranean Sea (Kasparek *et al.* 2001), nesting data gathered since the early 1990s in Turkey, Cyprus, and Israel show no apparent trend in any direction. However, a declining trend is apparent along the coast of Palestine/Israel, where 300-350 nests were deposited each year in the 1950s (Sella 1982) compared to a mean of 6 nests per year from 1993-2004 (Kuller 1999; Y. Levy, Israeli Sea Turtle Rescue Center, unpublished data). A recent discovery of green sea turtle nesting in Syria adds roughly 100 nests per year to green sea turtle nesting activity in the Mediterranean (Rees *et al.* 2005). That such a major nesting concentration could have gone unnoticed until recently (the Syria coast was surveyed in 1991, but nesting activity was attributed to loggerheads) bodes well for the ongoing speculation that the unsurveyed coast of Libya may also host substantial nesting.

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

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

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

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

Population Dynamics and Status

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

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

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

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

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

Threats

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

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

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

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

As highly migratory, wide-ranging organisms that are biologically tied to temperature regimes, green sea turtles are vulnerable to effects of climate change in aspects of their physiology and behavior (Van Houtan 2011). Analysis on potential effects of climate change on green sea turtles in the action area is included below in section 7.0.

Summary of Status of Green Sea Turtles

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

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

However, the 5-year review also noted that the Tortuguero nesting stock continued to be affected by ongoing directed take at their primary foraging area in Nicaragua (NMFS and USFWS 2007c). 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 account for an unknown level of other mortality. Based on its 5-year status review of the species, NMFS and USFWS (2007c) determined that the listing classification for green sea turtles should not be changed. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPSs should be identified (NMFS and USFWS 2007c).

Based on this and the current best available information, we believe that the green sea turtle population is currently stable; as protective measures for sea turtles are currently in place and continue to be implemented, we expect this trend to continue or over the next 2 years. This stable

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

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

trend is based solely on information we have on nesting trends. The number of sea turtles comprising the neritic and oceanic life stages of the population is currently unknown. As a result, the status and future trend of the population as a whole remains unclear. Therefore, until information and data become available on the numbers of individuals comprising the neritic and oceanic life stages, nesting trends represent the best available information and serve as the best representative of the population's trend.

4.2 Status of Large Whales

All of the cetacean species considered in this Opinion were once the subject of commercial whaling, which likely caused their initial decline. Commercial whaling for right whales along the U.S. Atlantic coast peaked in the 18th century, but right whales continued to be taken opportunistically along the coast and in other areas of the North Atlantic into the early 20th century (Kenney 2002). Worldwide, humpback whales were often the first species to be targeted and frequently hunted to commercial extinction (Clapham *et al.* 1999), meaning that their numbers had been reduced so low by commercial exploitation that it was no longer profitable to target the species. Wide-scale exploitation of the more offshore fin whale occurred later with the introduction of steam-powered vessels and harpoon gun technology (Perry *et al.* 1999). 1999). 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). Sei whales became the target of modern commercial whalers in the late 19th and early 20th centuries after populations of other whales, including right, humpback, fin, and blue, had already been depleted. The species continued to be exploited in Iceland until 1986, even though measures to stop whaling of sei whales had been enacted in the 1970s (Perry *et al.* 1999). 1999). 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. In 2011 and 2012, Iceland temporarily suspended commercial whaling for fin whales due to decreased demand from Japan, but is expected to have resumed in 2013. Today, the greatest known threats to these cetaceans are ship strikes and gear interactions, although the number of each species affected by these activities does vary.

Information on the range-wide status of each species as it is listed under the ESA is included here to provide the reader with information on the status of each species. Additional background information on the range-wide status of these species can be found in a number of published documents, including recovery plans (NMFS 1991a, b; 2005a), the Marine Mammal Stock Assessment Reports (SAR) (*e.g.*, Waring *et al.* 2013), status reviews (*e.g.*, Conant *et al.* 2009), and other publications (*e.g.* Clapham *et al.* 1999; Perry *et al.* 1999; Best *et al.* 2001).

4.2.1 North Atlantic Right Whale

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).

The North Atlantic right whale (*Eubalaena glacialis*) has been listed as endangered under the

ESA since 1973. Originally called the "northern right whale," it was listed as endangered under the Endangered Species Conservation Act, the precursor to the ESA in June 1970. The species is also designated as depleted under the Marine Mammal Protection Act (MMPA).

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

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).

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 better 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 poisioning (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)[5] 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 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

5 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.

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.

4.2.2 Humpback Whale

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 1991a).

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, 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.

4.2.3 Fin Whale

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.

4.3 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. 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[6] (77 FR 5880 and 77 FR 5914). These are: the Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs (see Figure 5.).

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

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

Maine DPS as threatened (77 FR 5880 and 77 FR 5914). The effective date of the listings was April 6, 2012. The DPSs do not include Atlantic sturgeon spawned in Canadian rivers. Therefore, fish that originated in Canada are not included in the listings.

Atlantic Sturgeon Life History

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

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

Age Class	**Size**	**Description**
Egg		Fertilized or unfertilized
Larvae		Negative photo-taxic, nourished by yolk sac
Young of Year (YOY)	**0.3 grams <41 cm TL**	Fish that are > 3 months and < one year; capable of capturing and consuming live food
Non-migrant subadults or juveniles	**>41 cm and <76 cm TL**	Fish that are at least age 1 and are not sexually mature and do not make coastal migrations.
Subadults	**>76cm and <150cm TL**	Fish that are not sexually mature but make coastal migrations
Adults	**>150 cm TL**	Sexually mature fish

Table 5. Descriptions of Atlantic sturgeon life history stages.

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

Atlantic sturgeon can grow to over 14 feet weighing 800 pounds(Pikitch *et al.* 2005). 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).

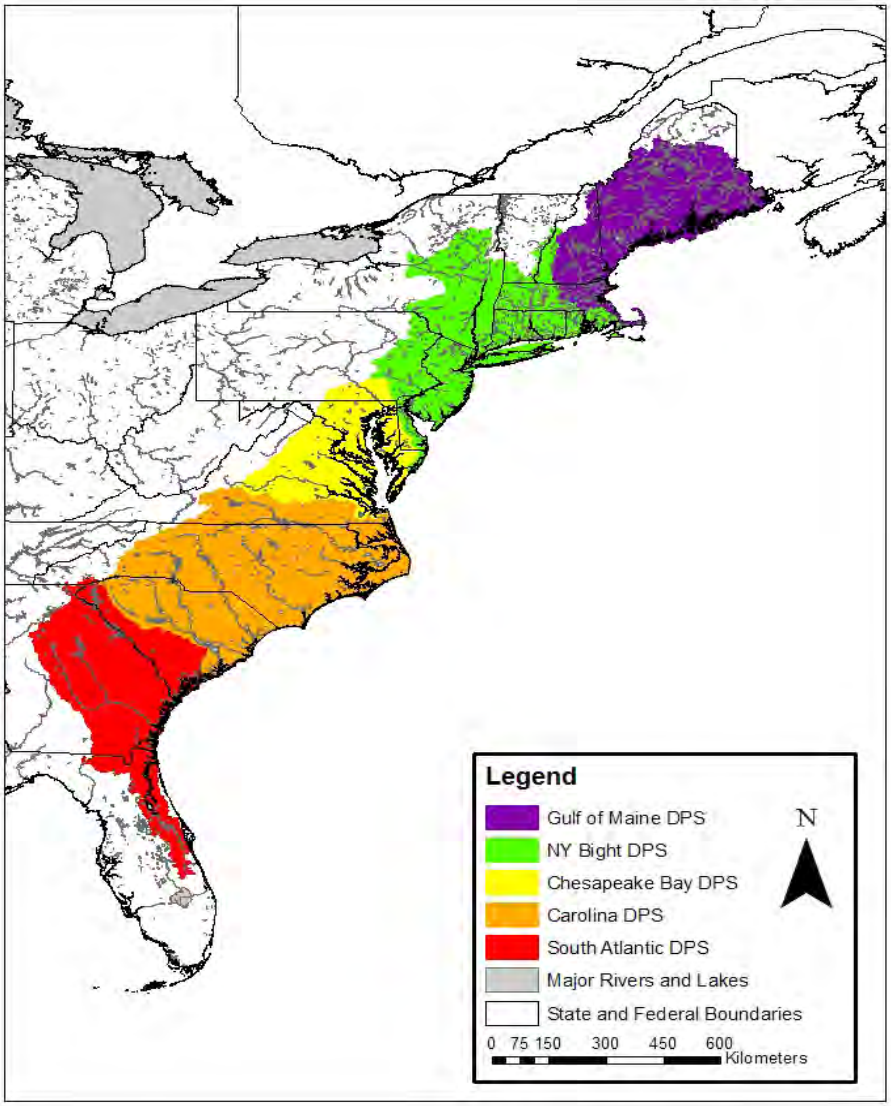


Figure 5. Map Depicting the five Atlantic sturgeon DPSs

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.

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 is approximately half of what it was 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 6). The ASPI provides a general abundance metric to assess risk for actions that may affect Atlantic sturgeon in the ocean. 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 addition to the ASPI, a population estimate was derived from the Northeast Area Monitoring and Assessment Program (NEAMAP) (
Table). 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 them 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 6. 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 7. 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

The information from the NEAMAP 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 7). 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).

Table 8. Annual minimum swept area estimates for Atlantic sturgeon during the spring and fall from the Northeast Area Monitoring and Assessment Program survey. Estimates assume 100% net efficiencies. Estimates provided by Dr. Chris Bonzek, Virginia Institute of Marine Science (VIMS).

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 catchabilty (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. True efficiencies less than 100% are common for most species. 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 where the NEAMAP survey does not sample. Additionally, although the NEAMAP surveys are not conducted in the Gulf of Maine or south of Cape Hatteras, NC, the NEAMAP surveys are conducted from Cape Cod to Cape Hatteras at depths up to 18.3 meters (60 feet), which is within 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 a large portion of the marine range of the five DPSs, 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.

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 (see Table 7). The ASPI model uses 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 does not depend on as many assumptions. For the purposes of this Opinion, we consider the NEAMAP estimate resulting from the 50% catchability rate is the best available information on the number of subadult and adult Atlantic sturgeon in the ocean.

The ocean population abundance of 67,776 fish estimated from the NEAMAP survey assuming 50% efficiency (based on net efficiency and the fraction of the total population exposed to the survey) was subsequently partitioned by DPS based on genetic frequencies of occurrence (Table 9) in the sampled area. 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, which is only a fraction of the total number of subadults.

Table 9. Summary of calculated population estimates based upon the NEAMAP Survey swept area assuming 50% efficiency (based on net efficiency and area sampled) derived from applying the Mixed Stock Analysis to the total estimate of Atlantic sturgeon in the Ocean and the 1:3 ratio of adults to subadults)

DPS	Estimated Ocean Population Abundance	Estimated Ocean Population of Adults	Estimated Ocean Population of Subadults (of size vulnerable to capture in fisheries)
GOM	7,455	1,864	5,591
NYB	34,566	8,642	25,925
CB	8,811	2,203	6,608
Carolina	1,356	339	1,017
SA	14,911	3,728	11,183
Canada	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 that make up the DPS 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 bycatch 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 are variable 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 may result in delayed 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 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%.

Determination of DPS Composition in the Action Area

As explained above, the range of all five DPSs overlaps and extends from Canada through Cape Canaveral, Florida. We have considered the best available information to determine from which DPSs individuals in the action area are likely to have originated. Based on mixed-stock analysis, we have determined that Atlantic sturgeon in the action area likely originate from the five DPSs at the following frequencies: NYB 51%; South Atlantic 22%; Chesapeake Bay 13%; Gulf of Maine 11%; and Carolina 2.0%. These percentages are largely based on genetic sampling of individuals (n=173) sampled in commercial fisheries by the Northeast Fisheries Observers Program (NEFOP). This covers captures from the Gulf of Maine to Cape Hatteras and is generally aligned with the action area for this consultation. Therefore, this represents 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 above, which approximate the mid-point of the range, as a reasonable indication of the likely genetic makeup of Atlantic sturgeon in the action area. These assignments and the data from which they are derived are described in detail in Damon-Randall *et al.* (2012a).

4.3.1 Gulf of Maine DPS of Atlantic sturgeon

The Gulf of Maine DPS includes the following: all anadromous Atlantic sturgeons that are spawned in the watersheds from the Maine/Canadian border and, extending southward, all watersheds draining into the Gulf of Maine as far south as Chatham, MA. Within this range, Atlantic sturgeon historically spawned in the Androscoggin, Kennebec, Merrimack, Penobscot, and Sheepscot Rivers (ASSRT, 2007). Spawning still occurs in the Kennebec River, and it is possible that it still occurs in the Penobscot River as well. Recent evidence indicates that spawning may also be occurring in the Androscoggin River. During the 2011 spawning season, the Maine Department of Marine Resources captured a larval Atlantic sturgeon 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 percent of Atlantic sturgeon habitat in the river (Oakley, 2003; ASSRT, 2007). However, the accessible portions of the Merrimack seem to be suitable habitat for Atlantic sturgeon spawning and rearing (i.e., nursery habitat) (Keiffer and Kynard, 1993). Therefore, the availability of spawning habitat does not appear to be the reason for the lack of observed spawning in the Merrimack River. Studies are on-going to determine whether Atlantic sturgeon are spawning in these rivers. Atlantic sturgeons that are spawned elsewhere continue to use habitats within all of these rivers as part of their overall marine range (ASSRT, 2007). The movement of subadult and adult sturgeon between rivers, including to and from the Kennebec River and the Penobscot River, demonstrates that coastal and marine migrations are key elements of Atlantic sturgeon life history for the Gulf of Maine DPS as well as likely throughout the entire range (ASSRT, 2007; Fernandes, *et al.*, 2010).

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

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

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

Connectivity is disrupted by the presence of dams on several rivers in the Gulf of Maine region, including the Penobscot and Merrimack Rivers. While there are also dams on the Kennebec, Androscoggin and Saco Rivers, these dams are near the site of natural falls and likely represent the maximum upstream extent of sturgeon occurrence even if the dams were not present. Because no Atlantic sturgeon are known to occur upstream of any hydroelectric projects in the Gulf of Maine region, passage over hydroelectric dams or through hydroelectric turbines is not a source of injury or mortality in this area. While not expected to be killed or injured during passage at a dam, the extent that Atlantic sturgeon are affected by the existence of dams and their operations in the Gulf of Maine region is currently unknown. The documentation of an Atlantic sturgeon larvae downstream of the Brunswick Dam in the Androscoggin River suggests however, that Atlantic sturgeon spawning may be occurring in the vicinity of at least that project and therefore, may be affected by project operations. Until it was breached in July 2013, the range of Atlantic sturgeon in the Penobscot River was limited by the presence of the Veazie Dam. Since the removal of the Veazie Dam, sturgeon can now travel as far upstream as the Great Works Dam. The Great Works Dam prevents Atlantic sturgeon from accessing the presumed historical spawning habitat located downstream of Milford Falls, the site of the Milford Dam. While removal of the Great Works Dams is anticipated to occur in the near future, the presence of this dam is currently preventing access to significant habitats within the Penobscot River. While Atlantic sturgeon are known to occur in the Penobscot River, it is unknown if spawning is currently occurring or whether the presence of the Great Works 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. Like the Penobscot, it is unknown how the Essex Dam affects the likelihood of spawning occurring in this river.

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

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

Summary of the Gulf of Maine DPS

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

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

Gulf of Maine DPS (Wirgin *et al.*, in draft).

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

4.3.2 New York Bight DPS of Atlantic sturgeon

The New York Bight DPS includes the following: all anadromous Atlantic sturgeon spawned in the watersheds that drain into coastal waters from Chatham, MA to the Delaware-Maryland border on Fenwick Island. Within this range, Atlantic sturgeon historically spawned in the Connecticut, Delaware, Hudson, and Taunton Rivers (Murawski and Pacheco, 1977; Secor, 2002; ASSRT, 2007). Spawning still occurs in the Delaware and Hudson Rivers, but there is no recent evidence (within the last 15 years) of spawning in the Connecticut and Taunton Rivers (ASSRT, 2007). Atlantic sturgeon that are spawned elsewhere continue to use habitats within the Connecticut and Taunton Rivers as part of their overall marine range (ASSRT, 2007; Savoy, 2007; Wirgin and King, 2011).

The abundance of the Hudson River Atlantic sturgeon riverine population prior to the onset of expanded exploitation in the 1800s is unknown but, has been conservatively estimated at 10,000 adult females (Secor, 2002). Current abundance is likely at least one order of magnitude smaller than historical levels (Secor, 2002; ASSRT, 2007; Kahnle *et al.*, 2007). As described above, an estimate of the mean annual number of mature adults (863 total; 596 males and 267 females) was calculated for the Hudson River riverine population based on fishery-dependent data collected from 1985-1995 (Kahnle *et al.*, 2007). Kahnle *et al.* (1998; 2007) also showed that the level of fishing mortality from the Hudson River Atlantic sturgeon fishery during the period of 1985-1995 exceeded the estimated sustainable level of fishing mortality for the riverine population and may have led to reduced recruitment. 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 1970s followed by a secondary drop in the late 1980s (Kahnle *et al.*, 1998; Sweka *et al.*, 2007; ASMFC, 2010). Catch-per-unit-effort 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). In examining the CPUE data from 1985-2007, there are significant fluctuations during this time. There appears to be a decline in the number of juveniles between the late 1980s and early 1990s although the CPUE is generally higher in the 2000s as compared to the 1990s. Given the significant annual fluctuation, it is difficult to discern any trend. Despite the CPUEs from 2000-2007 being generally higher than those from 1990-1999, they are low compared to the late 1980s. In addition to bycatch mortality in Federal waters, bycatch and mortality also occur in state fisheries; however, the primary fishery that impacted juvenile sturgeon (shad), has now been closed and there is no indication that it will reopen soon. In the

Hudson River sources of potential mortality include vessel strikes and entrainment in dredges. Individuals are also exposed to effects of bridge construction (including the ongoing replacement of the Tappan Zee bridge). Impingement at water intakes, including the Danskammer, Roseton and Indian Point power plants also occurs. There is currently not enough information regarding any life stage to establish a trend for the Hudson River population.

There is no 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 mm TL (Fisher, 2009) and the collection of 32 YOY Atlantic sturgeon in a separate study (Brundage and O'Herron in Calvo *et al.*, 2010). Genetics information collected from 33 of the 2009 year class YOY indicates that at least 3 females successfully contributed to the 2009 year class (Fisher, 2011). Therefore, while the capture of YOY in 2009 provides evidence that successful spawning is still occurring in the Delaware River, the relatively low numbers suggest the existing riverine population is limited in size.

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

Summary of the New York Bight DPS

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

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

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

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

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

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

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

4.3.3 Chesapeake Bay DPS of Atlantic sturgeon

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

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

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

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

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

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

Summary of the Chesapeake Bay DPS

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

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

4.3.4 Carolina DPS of Atlantic sturgeon

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

Rivers known to have current spawning populations within the range of the Carolina DPS include the Roanoke, Tar-Pamlico, Cape Fear, Waccamaw, and Pee Dee Rivers. We determined spawning was occurring if young-of-the-year (YOY) were observed, or mature adults were present, in freshwater portions of a system (Table 10). However, in some rivers, spawning by Atlantic sturgeon may not be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development. There may also be spawning populations in the Neuse, Santee and Cooper Rivers, though it is uncertain. Historically, both the Sampit and Ashley Rivers were documented to have spawning populations at one time. However, the spawning population in the Sampit River is believed to be extirpated and the current status of the spawning population in the Ashley River is unknown. Both rivers may be used as nursery habitat by young Atlantic sturgeon originating from other spawning populations. This represents our current knowledge of the river systems utilized by the Carolina DPS for specific life functions, such as spawning, nursery habitat, and foraging. However, fish from the Carolina DPS likely use other river systems than those listed here for their specific life functions.

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

Table 10. Major rivers, tributaries, and sounds within the range of the Carolina DPS and currently available data on the presence of an Atlantic sturgeon spawning population in each

system.

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

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

Threats

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

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

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

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

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

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

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

Summary of the Status of the Carolina DPS of Atlantic Sturgeon
In summary, the Carolina DPS is a small fraction of its historic population size. The ASSRT

estimated to be less than 300 spawning adults per year (total of both sexes) in each of the major river systems occupied by the DPS in which spawning still occurs. Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon. While a long life-span allows multiple opportunities to contribute to future generations, this is hampered within the Carolina DPS by habitat alteration and bycatch. This DPS was severely depleted by past directed commercial fishing, and faces ongoing impacts and threats from habitat alteration or inaccessibility, bycatch, and the inadequacy of existing regulatory mechanisms to address and reduce habitat alterations and bycatch that have prevented river populations from rebounding and will prevent their recovery.

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

4.3.5 South Atlantic DPS of Atlantic sturgeon

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

Rivers known to have current spawning populations within the range of the South Atlantic DPS include the Combahee, Edisto, Savannah, Ogeechee, Altamaha, and Satilla Rivers. We determined spawning was occurring if young-of-the-year (YOY) were observed, or mature adults were present, in freshwater portions of a system (Table 11). However, in some rivers, spawning by Atlantic sturgeon may not be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development. Historically, both the Broad-Coosawatchie and St. Marys Rivers were documented to have spawning populations at one time; there is also evidence that spawning may have occurred in the St. Johns River or one of its tributaries. However, the spawning population in the St. Marys River, as well as any historical spawning population present in the St. Johns, is believed to be extirpated, and the status of the spawning population in the Broad-Coosawatchie is unknown. Both the St. Marys and St. Johns Rivers are used as nursery habitat by young Atlantic sturgeon originating from other spawning populations. The use of the Broad-Coosawatchie by sturgeon from other spawning populations is unknown at this time. The presence of historical and current spawning populations in the Ashepoo River has not been documented; however, this river may currently be used for nursery habitat by young Atlantic sturgeon originating from other spawning populations. This represents our current knowledge of the river systems utilized by the South Atlantic DPS for specific life functions, such as spawning, nursery habitat, and foraging. However, fish from the South Atlantic DPS likely use other river systems than those listed here for their specific life functions.

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

Table 11. Major rivers, tributaries, and sounds within the range of the South Atlantic DPS and currently available data on the presence of an Atlantic sturgeon spawning population in each system.

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

Secor (2002) estimates that 8,000 adult females were present in South Carolina prior to 1890. Prior to the collapse of the fishery in the late 1800s, the sturgeon fishery was the third largest fishery in Georgia. Secor (2002) estimated from U.S. Fish Commission landing reports that approximately 11,000 spawning females were likely present in the state prior to 1890. Reductions from the commercial fishery and ongoing threats have drastically reduced the numbers of Atlantic sturgeon within the South Atlantic DPS. Currently, the Atlantic sturgeon spawning population in at least two river systems within the South Atlantic DPS has been extirpated. The Altamaha River population of Atlantic sturgeon, with an estimated 343 adults spawning annually, is believed to be the largest population in the Southeast, yet is estimated to be only 6 percent of its historical population size. The ASSRT estimated the abundances of the remaining river populations within the DPS, each estimated to have fewer than 300 spawning adults to be less than 1 percent of what they were historically (ASSRT 2007).

Threats

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

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

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

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

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

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

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

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

Dredging is contributing to the status of the South Atlantic DPS by modifying spawning, nursery, and foraging habitat. Habitat modifications through reductions in water quality are also contributing to the status of the South Atlantic DPS through reductions in DO, particularly during times of high water temperatures, which increase the detrimental effects on Atlantic sturgeon habitat. Interbasin water transfers and climate change threaten to exacerbate existing water quality issues. Bycatch is also a current impact to the South Atlantic DPS that is contributing to its status. Fisheries known to incidentally catch Atlantic sturgeon occur throughout the marine range of the species and in some riverine waters as well. Because Atlantic sturgeon mix extensively in marine waters and may utilize multiple river systems for nursery and foraging habitat in addition to their natal spawning river, they are subject to being caught in multiple fisheries throughout their range. In addition to direct mortality, stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (e.g., exposure to toxins). This may result in reduced ability to perform major life functions, such as foraging and spawning. While many of the threats to the South Atlantic DPS have been ameliorated or reduced due to the existing regulatory mechanisms, such as the moratorium on directed fisheries for Atlantic sturgeon, bycatch is currently not being addressed through existing mechanisms. Further, access to habitat and water quality continues to be a problem even with NMFS' authority under the Federal Power Act to recommend fish passsage 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. 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 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. The activities that shape the environmental baseline in the action area of this consultation generally include: dredging operations, vessel and fishery operations, water quality/pollution, and recovery activities associated with reducing those impacts.

5.1 Federal Actions that have Undergone Formal or Early Section 7 Consultation

NMFS has undertaken several ESA section 7 consultations to address the effects of actions authorized, funded or carried out by Federal agencies. Each of those consultations sought to develop ways of reducing the probability of adverse impacts of the action on listed species. Consultations are detailed below.

5.1.1 New York and New Jersey Harbor Deepening Project (HDP)

An Opinion regarding the HDP was issued by NMFS to the USACE on October 13, 2000. The Opinion included an Incidental Take Statement (ITS) exempting the incidental taking of two (2) loggerhead, one (1) green, one (1), Kemp's ridley, or one (1) leatherback for the duration (i.e., 3 years) of the deepening, via a hopper dredge, of the Ambrose Channel. Consultation was reinitiated in 2012 and an Opinion was issued on October 25, 2012. The Opinion included an ITS exempting the incidental taking of (1), Kemp's ridley, or one (1) leatherback, and (1) Atlantic sturgeon (any DPS) for the duration of the deepening, via a hopper dredge, of the Ambrose Channel. To date, no adverse impacts to listed species have been reported as a result of the HDP.

5.1.2 Emergency Beach Renourishment Along the Shoreline of New Jersey

The USACE, NY District, is undertaking Hurricane Sandy emergency beach renourishment activities along the shorelines of New Jersey. Currently, under the authority of Public Law 84-99, the USACE is renourishing the following coastal areas of New Jersey, which the ACOE had previously authorized and constructed: Sea Bright to Monmouth; Belmar to Manasquan; Long Branch, Asbury to Avon, and Keansburg. All material for renourshing these stretches of NJ coastline have or will be obtained from the Sea Bright Borrow Area. Table 12 provides information on the approximate time frame for renourishment activities and estimated volume of material to be removed from Sea Bright Borrow Area and placed on the designated shoreline.

Location of Project	**Approximate Duration**	**Volume of Material (million CY)**	**Acres Dredged**
Sea Bright to Monmouth	July 2013 – January 2014	2.2	138
Belmar to Manasquan	October 2013 – March 2014	1.5	133
Keansburg (Raritan Bay)	November 2013 – May 2014	1.1	120
Long Beach	November 2013 – June 2013	3.3	181
Asbury to Avon	December 2013 – May 2014	1.2	115

Table 12. Approximate time frame for renourishment activities and estimated volume of material to be removed

5.1.3 Amboy Aggregate Mining of Ambrose Channel

On October 11, 2002 NMFS issued an Opinion that considered the effects of the USACE's proposed issuance of a permit to Amboy Aggregates, Inc. for sand mining activities in the Ambrose Channel, New Jersey. The permit authorizes sand mining activities every year for a period of ten years. NMFS concluded that the proposed action may adversely affect, but would not likely jeopardize the continued existence of listed species of sea turtles. The 2002 Opinion included an ITS which exempted the take, via injury or mortality, of two (2) loggerhead, one (1) green, one (1) Kemp's ridley, or one (1) leatherback sea for the ten year duration of the permit. To date, no takes of listed species have been recorded.

5.1.4 Federal Vessel Operations

Potential adverse effects from federal vessel operations in the action area of this consultation include operations of the US Navy (USN) and the US Coast Guard (USCG), which maintain the largest federal vessel fleets, the EPA, the National Oceanic and Atmospheric Administration

(NOAA), and the USACE. NMFS has conducted formal consultations with the USCG, the USN, EPA and NOAA on their vessel operations. In addition to operation of USACE vessels, NMFS has consulted with the USACE to provide recommended permit restrictions for operations of contract or private vessels around whales. Through the section 7 process, where applicable, NMFS has and will continue to establish conservation measures for all these agency vessel operations to avoid adverse effects to listed species. Refer to the biological opinions for the USCG (September 15, 1995; July 22, 1996; and June 8, 1998) and the USN (May 15, 1997) for details on the scope of vessel operations for these agencies and conservation measures being implemented as standard operating procedures.

5.1.5 Federally Authorized Fisheries

NMFS authorizes the operation of several fisheries in the action area under the authority of the Magnuson-Stevens Fishery Conservation and Management Act and through Fishery Management Plans (FMPs) and their implementing regulations. The action area includes a portion of NOAA Statistical Area 612. Fisheries that operate in the action area that may affect ESA-listed species include: American lobster, Atlantic bluefish, Atlantic herring, Atlantic mackerel/squid/ butterfish, Atlantic sea scallop, monkfish, Northeast multispecies, spiny dogfish, surf clam/ocean quahog and summer flounder/scup/black sea bass. Section 7 consultations have been completed on these fisheries to consider effects to listed whales, sea turtles and sturgeon. Of the fisheries noted above, we expect that interactions may occur in all except Atlantic herring and surf clam/ocean quahog.

Batched Fisheries

On December 16, 2013, NMFS issued an Opinion on the continued implementation of management measures for the Northeast multispecies, monkfish, spiny dogfish, Atlantic bluefish, Northeast skate complex, mackerel/squid/butterfish, and summer flounder/scup/black sea bass fisheries. NMFS concluded that the proposed actions may adversely affect, but would not likely jeopardize the continued existence of listed whales, sea turtles and Atlantic sturgeon. The Opinion included an ITS which exempted the following take, via injury or mortality:

- Loggerhead sea turtles: 269 over a five-year average in gillnet gear, 213 loggerheads over a four-year average in bottom trawl gear, and one loggerhead in trap/pot gear
- Leatherback sea turtles: the annual take of 4 leatherbacks in gillnet gear, 4 in bottom trawl, and 4 in trap/pot gear
- Kemp's ridley sea turtles: the annual take of 4 in gillnet gear and 3 in bottom trawl gear.
- Green sea turtles: annual take of 4 in gillnet gear, and 3 in bottom trawl
- Atlantic sturgeon from the GOM DPS, annual take of up to 137 individuals over a five-year average in gillnet gear, the annual take of up to 148 individuals over a five-year average in bottom trawl gear
- Atlantic sturgeon from the NYB DPS, annual take of up to 632 individuals over a five-year average in gillnet gear, the annual take of up to 685 individuals over a five-year average in bottom trawl gear
- Atlantic sturgeon from the CB DPS, annual take of up to 162 individuals over a five-year average in gillnet gear, the annual take of up to 175 individuals over a five-year average in bottom trawl gear

- Atlantic sturgeon from the Carolina DPS, annual take of up to 162 individuals over a five-year average in gillnet gear, the annual take of up to 175 individuals over a five-year average in bottom trawl gear
- Atlantic sturgeon from the SA DPS, annual take of up to 273 individuals over a five-year average in gillnet gear, the annual take of up to 296 individuals over a five-year average in bottom trawl gear
- GOM DPS Atlantic Salmon, 5 over a five-year average in gillnet gear and 5 over a five-year average in trawl gear

American Lobster Fishery
The American lobster fishery has been identified as causing injuries to and mortality of loggerhead and leatherback sea turtles as a result of entanglement in buoy lines of the pot/trap gear. Pot/trap gear has also been identified as a gear type causing injuries and mortality of right, Humpback, and fin whales. The most recent Opinion for this fishery, completed on August 3, 2012, concluded that operation of the federally regulated portion of the lobster trap fishery may adversely affect loggerhead and leatherback sea turtles as a result of entanglement in the groundlines and/or buoy lines associated with this type of gear. An ITS was issued with the 2012 Opinion that exempted the take of 1 loggerhead sea turtle and 5 leatherback sea turtles.

Atlantic Sea Scallop Fishery
Loggerhead, Kemp's ridley, and green sea turtles have been reported by NMFS observers as being captured in scallop dredge and or trawl gear. The average number of annual observable interactions of hard-shelled sea turtles in the Mid-Atlantic dredge fishery prior to the implementation of chain mats (January 1, 2001, through September 25, 2006) was estimated to be 288 turtles, of which 218 could be confirmed as loggerheads (Murray 2011). After the implementation of chain mats (September 26, 2006, through December 31, 2008), the average annual number of observable plus unobservable, quantifiable interactions in the Mid-Atlantic dredge fishery was estimated to be 125 turtles, of which 95 could be confirmed as loggerheads (Murray 2011). An estimate of loggerhead bycatch in Mid-Atlantic scallop trawl gear from 2005-2008 averaged 95 turtles annually (Warden 2011a).

Formal section 7 consultation on the continued authorization of the scallop fishery was last reinitiated on February 28, 2012, with an Opinion issued by NMFS on July 12, 2012. In this Opinion, NMFS determined that the continued authorization of the Scallop FMP (including the seasonal use of turtle deflector dredges [TDDs] in Mid-Atlantic waters starting in 2013) may adversely affect but was not likely to jeopardize the continued existence of loggerhead, leatherback, Kemp's ridley, and green sea turtles, or the five DPSs of Atlantic sturgeon, and issued an ITS. In the ITS, the scallop fishery is estimated to interact annually with up to 301 loggerhead, two leatherback, three Kemp's ridley, and two green sea turtles, as well as one Atlantic sturgeon from any of the five DPSs. Of the loggerhead interactions, up to 112 per year are anticipated to be lethal from 2013 going forward.

5.1.6 Research Activities
We have completed ESA section 7 consultation on two research projects that occur in the action area. The US Fish and Wildlife Service funds an ocean trawl survey carried out by the State of

New Jersey; the project is currently funded through May 3, 2014. This federal action was the subject of a consultation completed in May, 2012. In the Opinion, we concluded that the action may adversely affect, but was not likely to jeopardize the continued existence of any DPS of Atlantic sturgeon. The ITS exempts the take of 109 Atlantic sturgeon through May 2014. All captured Atlantic sturgeon are expected to be released alive and no lethal take is anticipated.

We provide funding to the Virginia Institute of Marine Science (VIMS) to carry out the Northeast Area Monitoring and Assessment Program (NEAMAP) Near Shore Trawl Program. In an April 2012 Opinion, we concluded that the 2012 spring and fall surveys may adversely affect, but were not likely to jeopardize the continued existence of any DPS of Atlantic sturgeon. The ITS exempted the take of 32 Atlantic sturgeon through 2012. All captured Atlantic sturgeon were expected to be released alive and no lethal take was anticipated.

5.2 Non-Federal Regulatory Actions

Private and Commercial Vessel Operations

The New York/New Jersey Harbor complex is a major shipping port and center of commerce, there are numerous private and commercial vessels (e.g., container ships, commuter ferries) that operate in the action area that have the potential to interact with listed species. On an annual basis more than 5,124 commercial vessels and approximately 5,292,020 container vessels, as well as numerous recreational vessels transit the New York Harbor complex.

Ship strikes have been identified as a significant source of mortality to the North Atlantic right whale population (Kraus 1990) and are also known to impact all other endangered whales. Data also shows that vessel traffic is a substantial cause of sea turtle mortality. Fifty to 500 loggerheads and 5 to 50 Kemp's ridley turtles are estimated to be killed by vessel traffic per year in the U.S. (National Research Council 1990). In ceratin geographic areas, vessel strikes have also been identified as a threat to Atlantic sturgeon. Although the exact number of Atlantic sturgeon killed as a result of being stuck by vessels is unknown, records of these interactions have been documented (e.g., Brown and Murphy, 2010). These commercial and private activities therefore, have the potential to result in lethal (boat strike) or non-lethal (through harassment) takes of listed species that could prevent or slow a species' recovery. As whales, Atlantic sturgeon, and turtles may be in the area where high vessel traffic occurs, the potential exists for collisions with vessels transiting from within and out of the action area.

An unknown number of private recreational boaters frequent coastal waters; some of these are engaged in whale watching or sport fishing activities. These activities have the potential to result in lethal (through entanglement or boat strike) or non-lethal (through harassment) takes of listed species. Effects of harassment or disturbance which may be caused by such vessel activities are currently unknown; however, no conclusive detrimental effects have been demonstrated. Recent federal efforts regarding mitigating impacts of the whale watch and shipping industries on endangered whales are discussed below.

Non-Federally Regulated Fishery Operations

State fisheries do operate in the state waters of New Jersey. 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. Impacts of state fisheries on endangered whales are addressed as appropriate through the MMPA take reduction planning process. NMFS is 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.

5.3 Other Potential Sources of Impacts to Listed Species

Pollution and Water Quality

Dredging and point source discharges (e.g., municipal wastewater, industrial or power plant cooling water or waste water) and the compounds either associated with discharges or released from the sediments during dredging operations (e.g., metals, dioxins, dissolved solids, phenols, and hydrocarbons) contribute to poor water quality and may also impact the health of sturgeon populations. The compounds associated with discharges can alter the pH or dissolved oxygen levels of receiving waters, which may lead to mortality, changes in fish behavior, deformations, and reduced egg production and survival. Additionally, concentrated amounts of suspended solids discharged into a river system may lead to smothering of fish eggs and larvae and may result in a reduction in the amount of available dissolved oxygen.

Sources of contamination in the action area include atmospheric loading of pollutants, stormwater runoff from coastal development, groundwater discharges, and industrial development. Chemical contaminants may also have an effect on sea turtle reproduction and survival. Although the effects of contaminants on turtles is relatively unclear, pollution may be linked to the fibropapilloma virus that kills many turtles each year (NMFS 1997). If pollution is not the causal agent, it may make sea turtles more susceptible to disease by weakening their immune systems.

Excessive turbidity due to coastal development and/or construction sites could influence Atlantic sturgeon, sea turtle, and whale foraging ability; however, based on the best available information, whales, Atlantic sturgeon, and turtle foraging ability is not very easily affected by changes in increased suspended sediments unless these alterations make habitat less suitable for listed species and hinder their capability to forage and/or for their foraging items to exist. If the latter occurs, eventually these species will tend to leave or avoid these less desirable areas (Ruben and Morreale 1999).

Marine debris (*e.g.*, discarded fishing line or lines from boats) can entangle turtles and whales causing serious injuries or mortalities to these species. Turtles commonly ingest plastic or mistake debris for food (Magnuson et al. 1990). Sources of contamination in the action area include atmospheric loading of pollutants, stormwater runoff from coastal development, groundwater discharges, industrial development, and debris. While the effects of contaminants on Atlantic sturgeon, whales, and turtles are relatively unclear, pollutants may make Atlantic sturgeon, sea turtles and whales more susceptible to disease by weakening their immune systems or may have an effect on Atlantic sturgeon, sea turtle, and whale reproduction and survival. For instance, pollution may be linked to the fibropapilloma virus that kills many turtles each year (NMFS 1997).

Noise pollution has been raised as a concern primarily for marine mammals. The potential effects of noise pollution on marine mammals range from minor behavioral disturbance to injury to death. The noise level in the ocean is thought to be increasing at a substantial rate due to increases in shipping and other activities, including seismic exploration, offshore drilling and sonar used by military and research vessels (NMFS 2007b). Because under some conditions, low frequency sound travels very well through water, few oceans are free of the threat of human noise. While there is no hard evidence of a whale population being adversely impacted by noise, scientists think it is possible that masking, the covering up of one sound by another, could interfere with marine mammals ability to feed and to communicate for mating (NMFS 2007b). Masking is a major concern with shipping, but only a few species of marine mammals have been observed to demonstrate behavioral changes to low level sounds. Concerns about noise in the action area of this consultation include increasing noise due to increasing commercial shipping and recreational vessels. Although noise pollution has been identified as a concern for marine mammals, these elevated levels of underwater noise may also be of concern for sea turtles and Atlantic sturgeon. Until additional studies are undertaken, it is difficult to determine the effects these elevated levels of noise will have on sea turtles and Atlantic sturgeon and to what degree these levels of noise may be altering the behavior or physiology of these species.

It should be noted, NMFS and the US Navy have been working cooperatively to establish a policy for monitoring and managing acoustic impacts from anthropogenic sound sources in the marine environment. Acoustic impacts can include temporary or permanent injury, habitat exclusion, habituation, and disruption of other normal behavior patterns. It is expected that the policy on managing anthropogenic sound in the oceans will provide guidance for programs such as the use of acoustic deterrent devices in reducing marine mammal-fishery interactions and review of federal activities and permits for research involving acoustic activities.

As noted above, private and commercial vessels, including fishing vessels, operating in the action area of this consultation also have the potential to interact with sea turtles, Atlantic sturgeon, or whales. The effects of fishing vessels, recreational vessels, or other types of commercial vessels on 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 entanglements. Listed species may also be affected by fuel oil spills resulting from vessel accidents. Fuel oil spills could affect animals directly or indirectly through the food chain. Fuel spills involving fishing vessels are common events. However, these spills typically involve small amounts of material that are unlikely to adversely affect listed species.

5.4 Conservation and Recovery Actions Reducing Threats to Listed Species

A number of activities are in progress that may ameliorate some of the threat that activities summarized in the *Environmental Baseline* pose to threatened and endangered species in the action area of this consultation. These include education/outreach activities; specific measures to reduce the adverse effects of entanglement in fishing gear, including: gear modifications; fishing gear time area closures; and whale disentanglement. In addition there are measures to reduce ship and other vessel impacts to protected species. Many of these measures have been implemented to reduce risk to critically endangered right whales. Despite the focus on right whales, other cetaceans and some sea turtles will likely benefit from the measures as well.

5.4.1 Reducing Threats to Listed Whales

5.4.1.1 Atlantic Large Whale Take Reduction Plan

The Atlantic Large Whale Take Reduction Plan (ALWTRP) reduces the risk of serious injury or mortality to 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 Marine Mammal Protection Act (MMPA) and has been developed by NOAA's National Marine Fisheries Service (NMFS). The ALWTRP covers the U.S. Atlantic Exclusive Economic Zone (EEZ) from Maine through Florida. The requirements are year-round in the Northeast, and seasonal in the Mid and South Atlantic.

Regulatory actions are directed at reducing serious entanglement injuries and mortality of right, humpback, and fin whales from fixed gear fisheries (*i.e.*, trap 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. The first ALWTRP went into effect in 1997. For more information, see the ALWTRP (available online at http://www.nero.noaa.gov/whaletrp/)

5.4.1.2 Ship Strike Reduction Program

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

5.4.1.3 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 an interim final rule in February 1997 that prohibits, except in limited circumstances, both boats and aircraft from approaching any right whale closer than 500 yards.

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 US, 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 and submission 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 U.S. Atlantic in areas and seasons where right whales predictably occur in high concentrations. We published regulations on October 10, 2008 to implement a 10-knot speed restriction for all vessels 65 feet (19.8 m) or longer in Seasonal Management Areas (SMAs) along the east coast of the U.S. Atlantic seaboard at certain times of the year (73 FR 60173; October 10, 2008).

SMAs are supplemented by Dynamic Management Areas (DMAs) that are implemented for 15 day periods in areas in which right whales are sighted outside of SMA boundaries. When NOAA aerial surveys or other reliable sources report aggregations of 3 or more right whales in a density that indicates the whales are likely to persist in the area, NOAA calculates a buffer zone around the aggregation and announces the boundaries of the zone to mariners via various mariner communication outlets, including NOAA Weather Radio, USCG Broadcast Notice to Mariners, MSR return messages, email distribution lists, and the Right Whale Sighting Advisory System (SAS). NOAA requests mariners to route around these zones or transit through them at 10 knots or less. Compliance with DMAs is voluntary.

The rule was set to expire five years from the date of effectiveness. NOAA has analyzed data on compliance with the rule and the effectiveness of the rule since its implementation and published a final rule (78 FR 73726: December 9, 2013) to eliminate the planned December 2013 expiration date of the 2008 rule.

Vessel Routing Measures to Reduce the Co-occurrence of Ships and Whales

Another critical, non-regulatory component of NOAA's right whale ship strike reduction program involves the development and implementation of routing measures that reduce the co-occurrence of vessels and right whales, thus reducing the risk of vessel collisions. Recommended routes were developed for the Cape Cod Bay feeding grounds and Southeast calving grounds by overlaying right whale sightings data on existing vessel tracks, and plotting alternative routes where vessels could expect to encounter fewer right whales. Full implementation of these routes was completed at the end of November 2006. The routes are now charted on all NOAA electronic and printed charts, published in U.S. Coast Pilots, and mariners have been notified through USCG Notices to Mariners.

Through a joint effort between NOAA and the USCG, the U.S. also submitted a proposal to the IMO to shift the northern leg of the existing Boston Traffic Separation Scheme (TSS) 12 degrees to the north to reduce vessel strikes. In 2009 this TSS was modified by narrowing the width of the north-south portion by one mile to further reduce the threat of ship collisions with endangered right whales and other whale species.

In 2009, NOAA and the USCG established the Great South Channel as an Area To Be Avoided (ATBA). This is a voluntary seasonal ATBA for ships weighing 300 gross tons or more. The ATBA will be in effect each year from April 1 to July 31, when right whales are known to congregate around the Great South Channel. Implementing this ATBA coupled with narrowing the TSS by one nautical mile will reduce the relative risk of right whale ship strikes by an estimated 74% during April-July (63% from the ATBA and 11% from the narrowing of the TSS).

Sighting Advisory System (SAS)

The right whale Sighting Advisory System (SAS) was initiated in early 1997 as a partnership among several federal and state agencies and other organizations to conduct aerial and ship board surveys to locate right whales and to alert mariners to right whale sighting locations in a near real time manner. The SAS surveys and opportunistic sightings reports document the presence of right whales and are provided to mariners via fax, email, NAVTEX, Broadcast Notice to Mariners, NOAA Weather Radio, several web sites, 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.

5.4.2 Reducing Threats to Listed Sea Turtles

NMFS has implemented multiple measures to reduce the capture and mortality of sea turtles in fishing gear, and other measures to contribute to the recovery of these species. While some of these actions occur outside of the action area for this consultation, the measures affect sea turtles that do occur within the action area.

5.4.2.1 Education and Outreach Activities

Education and outreach activities are considered one of the primary tools to reduce the threats to all protected species. For example, NMFS has been active in public outreach to educate fishermen regarding sea turtle handling and resuscitation techniques, as well as guidelines for recreational fishermen and boaters to avoid the likelihood of interactions with marine mammals. NMFS intends to continue these outreach efforts in an attempt to reduce interactions with protected species, and to reduce the likelihood of injury to protected species when interactions do occur.

5.4.2.2 Sea Turtle Stranding and Salvage Network (STSSN)

The Sea Turtle Stranding and Salvage Network (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, reducing mortality of injured or sick animals. NMFS manages the activities of the STSSN. Data collected by the STSSN are used to monitor stranding levels, to identify areas where unusual or elevated mortality is occurring, and to identify sources of mortality. 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 improve our understanding of sea turtle movements, longevity, and reproductive patterns, all of which contribute to our ability to reach recovery goals for the species.

5.4.2.3 Sea Turtle Disentanglement Network (STDN)

The Sea Turtle Disentanglement Network (STDN) is considered a component of the larger STSSN program, and it operates in all states in the region. The STDN responds to entangled sea turtles and disentangles and releases live animals, thereby reducing serious injury and mortality. In addition, the STDN collects data on live and dead sea turtle entanglement events, providing valuable information for management purposes. The NMFS Northeast Regional Office oversees the STDN program and manages the STDN database.

5.4.2.4 Regulatory Measures for Sea Turtles

Large-Mesh Gillnet Requirements in the Mid-Atlantic

Since 2002, NMFS has regulated the use of large mesh gillnets in Federal waters off North Carolina and Virginia (67 FR 13098, March 21, 2002) to reduce the impact of these fisheries on ESA-listed sea turtles. Currently, gillnets with stretched mesh size 7-inches (17.8 cm) or larger are prohibited in the Exclusive Economic Zone during the following times and in the following areas: (1) north of the NC/SC border to Oregon Inlet, NC at all times, (2) north of Oregon Inlet to Currituck Beach Light, NC from March 16 through January 14, (3) north of Currituck Beach Light, NC to Wachapreague Inlet, VA from April 1 through January 14, and (4) north of Wachapreague Inlet, VA to Chincoteague, VA from April 16 through January 14.

NMFS has also issued regulations to address the take of sea turtles in gillnet gear fished in Pamlico Sound, NC. Waters of Pamlico Sound are closed to fishing with gillnets with a stretched mesh size larger than 4 ¼ inch (10.8 cm) from September 1 through December 15 each year to protect sea turtles. The closed area includes all inshore waters of Pamlico Sound, and all contiguous tidal waters, south of 35°46.3' N. lat., north of 35 ° 00' N. lat., and east of 76 ° 30' W. long.

TED Requirements in Trawl Fisheries

Turtle Excluder Devices (TEDs) are required in the shrimp and summer flounder fisheries. TEDs allow sea turtles to escape the trawl net, reducing injury and mortality resulting from capture in the net. Approved TEDs are required in the shrimp trawl fishery operating in the Atlantic and Gulf Areas unless the trawler is fishing under one of the exemptions (*e.g*., skimmer trawl, try net) and all requirements of the exemption (50 CFR§ 223.206) are met. On February 21, 2003, NMFS issued a final rule to amend the TED regulations to enhance their effectiveness in reducing sea turtle mortality resulting from shrimp trawling in the Atlantic and Gulf Areas of the southeastern United States by requiring an escape opening designed to exclude leatherbacks as well as large loggerhead and green turtles (68 FR 8456; February 21, 2003). In 2011, NMFS published a Notice of Intent to prepare an Environmental Impact Statement (EIS) and to conduct scoping meetings. NMFS is considering a variety of regulatory measures to reduce the bycatch of threatened and endangered sea turtles in the shrimp fishery of the southeastern United States in light of new concerns regarding the effectiveness of existing TED regulations in protecting sea turtles (76 FR 37050, June 24, 2011). TEDs are also required for summer flounder trawlers in the summer flounder fishery-sea turtle protection area (50 CFR §223.206).

5.4.3 Reducing Threats to Atlantic Sturgeon

Atlantic Sturgeon Recovery Planning

Several conservation actions aimed at reducing threats to Atlantic sturgeon are currently ongoing. We will be convening a recovery team and drafting a recovery plan to outline recovery goals and criteria, as well as steps necessary to recover all Atlantic sturgeon DPSs. Numerous research activities are underway involving NMFS and other federal, state, and academic partners to obtain more information on the distribution and abundance of Atlantic sturgeon throughout their range, including in the action area, and to develop population estimates for each DPS. We will be working closely with ASMFC and NEFSC on the new stock assessment process described above. Efforts are also underway to better understand threats faced by the DPSs and to find ways to minimize these threats, including bycatch and water quality. Fishing gear researchers are working on designing 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.

Education and Outreach Activities

NMFS has a program called "SCUTES" (Student Collaborating to Undertake Tracking Efforts for Sturgeon), which offers educational programs and activities about the movements, behaviors, and threats to Atlantic sturgeon. NMFS intends to continue these outreach efforts in an attempt to reduce interactions with protected species, and to reduce the likelihood of injury to protected species when interactions do occur.

Stranding and Salvage Programs

A salvage program is now in place for Atlantic sturgeon. Atlantic sturgeon carcasses can provide pertinent life history data and information on new or evolving threats to Atlantic sturgeon. Their use in scientific research studies can reduce the need to collect live Atlantic sturgeon. The NMFS Sturgeon Salvage Program is a network of individuals qualified to retrieve and/or use Atlantic and shortnose sturgeon carcasses and parts for scientific research and education. All carcasses and parts are retrieved opportunistically and participation in the network is voluntary.

6.0 CLIMATE CHANGE

The discussion below presents background information on 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 sea turtles, whales, and sturgeon may be affected by those predicted environmental changes over the life of the proposed actions (i.e., between now and 2064). Generally speaking, climate change may be relevant to the Status of the Species, Environmental Baseline, and Cumulative Effects sections of an Opinion; rather than include partial discussion in several sections of this Opinion, we are synthesizing this information into one discussion. Effects of the proposed actions that are relevant to climate change are included in the Effects of the Action section below (section 8.0 below).

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

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

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

While predictions are available regarding potential effects of climate change globally, it is more difficult to assess the potential effects of climate change over the next few decades on coastal and marine resources on smaller geographic scales, such as the shoreline of Elberon to Loch Arbour or Raritan Bay, especially as climate variability is a dominant factor in shaping coastal and marine systems. The effects of future change will vary greatly in diverse coastal regions for the U.S. Warming is very likely to continue in the U.S. over the next 25 to 50 years regardless of reduction in GHGs, due to emissions that have already occurred (NAST 2000). It is very likely that the magnitude and frequency of ecosystem changes will continue to increase in the next 25 to 50 years, and it is possible that the rate of change will accelerate. Climate change can cause or exacerbate direct stress on ecosystems through high temperatures, a reduction in water availability, and altered frequency of extreme events and severe storms. Water temperatures in streams and rivers are likely to increase as the climate warms and are very likely to have both direct and indirect effects on aquatic ecosystems. Changes in temperature will be most evident during low flow periods when they are of greatest concern (NAST 2000). In some marine and freshwater systems, shifts in geographic ranges and changes in algal, plankton, and fish abundance are associated with high confidence with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels and circulation (IPCC 2007).

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

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

6.2 Species Specific Information on Climate Change Effects

6.2.1 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 (Intergovernmental Panel on Climate Change (IPCC) 2007). Climate change related increasing temperatures, sea level rise, changes in ocean productivity, and increased frequency of storm events may affect loggerhead sea turtles.

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

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

Following the publication of the 2009 Status Review, Van Houtan and Halley (2011) developed climate forcing 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.

6.2.2 Kemp's Ridley Sea Turtles

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

Considering that the Kemp's ridley has temperature-dependent sex determination (Wibbels

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

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

6.2.3 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 2007d). 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 2007d).

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 2007d). 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 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).

6.2.4 Green Sea Turtles

The five year status review for green sea turtles (NMFS and USFWS 2007c) 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 2007c). 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).

6.2.5 *Right, Humpback, and Fin Whales*

Whales have persisted for millions of years and throughout this time have experienced wide variations in global climate conditions and have successfully adapted to these changes. Climate change at historical rates (thousands of years) is not thought to have been a problem for 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). Depending on habitat preferences, changes in water temperature due to climate change may affect the distribution of certain species of cetaceans. For instance, fin and 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 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 pole-ward.

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. For example, Greene *et al.* (2003) described the potential oceanographic processes linking climate variability to the reproduction of North Atlantic right whales. Climate-driven changes in ocean circulation have had a significant impact on the plankton ecology of the Gulf of Maine, including effects on *Calanus finmarchicus*, a primary prey resource for right whales.

More information is needed in order to determine the potential impacts global climate change will have on the timing and extent of population movements, abundance, recruitment, distribution and species composition of prey (Learmonth *et al.* 2006). Changes in climate patterns, ocean currents, storm frequency, rainfall, salinity, melting ice, and an increase in river inputs/runoff (nutrients and pollutants) will all directly affect the distribution, abundance and migration of prey species (Waluda *et al.* 2001; Tynan 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 that will also indirectly affect marine mammals (Learmonth *et al.* 2006).

6.2.6 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 affect 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 for salinity and remain in waters with little to no salinity. If the salt wedge moves further upstream, Atlantic sturgeon spawning and rearing habitat could be restricted. In river systems with dams or natural falls that are impassable by sturgeon, the extent that spawning or rearing may be shifted upstream to compensate for the shift in the movement of the saltwedge would be limited. While there is an indication that an increase in sea level rise would result in a shift in the location of the salt wedge, at this time there are no predictions on the timing or extent of any shifts that may occur; thus, it is not possible to predict any future loss in spawning or rearing habitat. However, in all river systems, spawning occurs miles upstream of the saltwedge. It is unlikely that shifts in the location of the saltwedge would eliminate freshwater spawning or rearing habitat. If habitat was severely restricted, productivity or survivability may decrease.

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

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

6.3 Effects of Climate Change in the Action Area

Information on how climate change will impact the action area is limited. According to the New York State Energy Research and Development Authority's 2011 ClimAid Synthesis Report, temperatures across New York State are expected to rise by 1.5 to 3°F by the 2020s, 3 to 5.5°F by the 2050s, and 4 to 9°F by the 2080s (ClimAid 2011). In addition, data from the Office of the New Jersey State Climatologist has shown a statistically significant rise in average statewide temperature (approximately 2 degrees Fahrenheit) over the last 113 years. It is predicted that in the Northeastern US, precipitation, particularly in the form of rainfall, and runoff are expected to increase in future years (NECIA 2007). NOAA tide gauge data reported by the State indicates that the sea level within the Battery of New York Harbor has risen at a rate of approximately 2.77 mm/yr since recordings began in 1856, while at the New Jersey coast site of Sandy Hook, sea level has risen at a rate of approximately 3.9 mm/y since recording began in the early- to mid-1900s.

Sea surface temperatures have fluctuated around a mean for much of the past century, as measured by continuous 100+ year records at Woods Hole (Mass.), and Boothbay Harbor (Maine) and shorter records from Boston Harbor and other bays. Periods of higher than average temperatures (in the 1950s) and cooler periods (1960s) have been associated with changes in the North Atlantic Oscillation (NAO), which affects current patterns. Over the past 30 years however, records indicate that ocean temperatures in the Northeast have been increasing; for example, Boothbay Harbor's temperature has increased by about 1°C since 1970. While we are not able to find predictive models for New Jersey, given the geographic proximity of these waters to the Northeast, we assume that predictions would be similar. The model projections are for an increase of somewhere between 3-4°C by 2100 and a pH drop of 0.3-0.4 units by 2100 (Frumhoff *et al.* 2007). Assuming that these predictions also apply to the action area, one could anticipate similar conditions in the action area over that same time period.

Assuming that there is a linear trend in increasing water temperatures, and that a predicted 3-4°C increase in water temperature by 2100 for the waters to the Northeast would also be experienced in the action area, one could anticipate a 0.03 - 0.05°C increase each year. Because the action considered here will be complete in 50 years, we expect an increase in temperature of no more than 2.5°C in the action area over the duration of the proposed action.

6.4 Effects of Climate Change in the Action Area to Listed Species Sea Turtles

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 sea turtles; however, we have considered the available information to analyze likely impacts to these species in the action area. The proposed actions under consideration are the three beach nourishment projects through 2064. Thus, we consider here likely effects of climate change during the period from now until 2064.

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

Over the time period considered in this Opinion, sea surface temperatures are expected to rise up to 2.5°C in the action area. It is unknown if that is enough of a change to contribute to shifts in the range or distribution of sea turtles. Theoretically we expect that as waters in the action area warm, more sea turtles could be present or sea turtles could be present for longer periods of time. However, if temperature affected the distribution of sea turtle forage in a way that decreased forage in the action area, sea turtles may be less likely to occur in the action area. It has been speculated that the nesting range of some sea turtle species may shift northward. Nesting in the mid-Atlantic generally is extremely rare and no nesting has been documented at any beach in the action area. In 2010, one green sea turtle came up on the beach in Sea Isle City, New Jersey; however, it did not lay any eggs. In August 2011, a loggerhead came up on the beach in Stone Harbor, New Jersey but did not lay any eggs. On August 18, 2011, a green sea turtle laid one nest at Cape Henlopen Beach in Lewes Delaware near the entrance to Delaware Bay. The nest contained 190 eggs and was transported indoors to an incubation facility on October 7. A total of twelve eggs hatched, with eight hatchlings surviving. In December, seven of the hatchlings were released in Cape Hatteras, North Carolina. It is important to consider that in order for nesting to be successful in New Jersey, 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 2064 are not great enough to allow successful rearing of sea turtle eggs in the action area. Therefore, it is unlikely that over the time period considered here, that there would be an increase in nesting activity in the action area or that hatchlings would be present in the action area.

We have considered whether the placement of sand at Port Monmouth, Union Beach, and Elberon to Loch Arbour would impact sea turtles. As noted above, there is the potential for a northward shift in nesting by sea turtles. Given existing nesting locations and the duration of time considered in this Opinion (50 years), it seems extremely unlikely that the range of sea turtle nesting would shift enough so that nesting would occur on beaches in New Jersey. The furthest north that leatherbacks nest is southeastern Florida. Kemp's ridleys only nest in Mexico. It is more likely that any shift in nesting to New Jersey beaches would be from loggerheads (which nest as far north as Virginia) and/or green sea turtles (which normally nest as far north as North Carolina. The placement of sand in the proposed actions is meant to stabilize and restore eroding habitats and maintain existing beach. None of the activity is likely to reduce the suitability of these beaches for potential future nesting.

6.5 Effects of Climate Change in the Action Area to Listed Species Whales

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 whales; however, we have considered the available information to analyzer likely impacts to these species in the action area. The proposed actions under consideration are the three beach nourishment projects through 2064; thus, we consider here, likely effects of climate change during the period from now until 2064.

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 in the portion of the action area located in the waters off the coast of New Jersey (i.e., SBOBA site). 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 affect 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 remaining life of the action (through 2064) it is unlikely that this possible shift in range will be observed due the relatively small increase in water temperature predicted to occur during the lifetime of the project (i.e., approximately 2.5^{o}C); if any shift does occur, it is likely to be minimal and thus, it seems unlikely that this small increase in temperature will cause a significant effect to right whales or a significant modification to the number of whales likely to be present in the action area through 2064.

6.6 Effects of Climate Change in the Action Area to Atlantic Sturgeon

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 Atlantic sturgeon; however, we have considered the available information to analyze likely impacts to sturgeon in the action area. We consider here, likely effects of climate change during the period from now until 2064.

Over time, the most likely effect to Atlantic sturgeon would be if sea level rise was great enough to consistently shift the salt wedge far enough north in a spawning river which would restrict the range of juvenile sturgeon and may affect the development of these life stages. However, there are no spawning rivers in the action area.

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

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

Limited information on the thermal tolerances of Atlantic sturgeon is available. Atlantic sturgeon have been observed in water temperatures above 30°C in the south (see Damon-Randall *et al.* 2010). In the laboratory, juvenile Atlantic sturgeon showed negative behavioral and bioenergetics responses (related to food consumption and metabolism) after prolonged exposure to temperatures greater than 28°C (82.4°F) (Niklitschek 2001). Tolerance to temperatures is thought to increase with age and body size (Ziegweid *et al.* 2008 and Jenkins *et al.* 1993), however, no information on the lethal thermal maximum or stressful temperatures for subadult or adult Atlantic sturgeon is available.

Mean monthly ambient temperatures in the Sandy Hook NJ, range from 2.2 - 22.2°C[8]. As explained above, available predictions estimate an increase in ambient water temperature in the area of up to 2.5°C over the duration of the proposed actions. This would result in the ambient temperatures in Sandy NJ, to range from 4.7 – 24.7°C. Warming temperatures predicted to occur over the next 50 years would likely result in a northward shift/extension of their range (i.e. into the St. Lawrence River, Canada) while truncating the southern distribution, thus effecting the recruitment and distribution of sturgeon rangewide. However, Atlantic sturgeon are known to currently occur at temperatures consistent with the predicted range over the next 50 years (4.7 – 24.7°C). If any shift does occur, 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.

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

7.0 EFFECTS OF THE ACTION

This section of an Opinion assesses the direct and indirect effects of the proposed action on threatened and endangered species or critical habitat, together with the effects of other activities that are interrelated or interdependent (50 CFR § 402.02). Indirect effects are those that are caused later in time, but are still reasonably certain to occur. Interrelated actions are those that are part of a larger action and depend upon the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration (50 CFR § 402.02). We have not identified any interdependent or interrelated actions. Because there is no critical habitat in the action areas, there are no effects to critical habitat to consider in this Opinion.

This Opinion examines the likely effects (direct and indirect) of the proposed actions on whales, sea turtles, and five DPSs of Atlantic sturgeon in the action areas and their habitat within the context of the species current status now and projected over the course of the action, the environmental baseline and cumulative effects. As explained in the "Description of the Action" section, the action under consideration in this Opinion includes the initial dredging cycles needed to aquire sand for three beach nourishment projects (i.e., Port Monmouth, Union Beach, and Elberon to Loch Arbour) as well as proposed actions the USACE may undertake for shore

[8] Information obtained from www.nodc.noaa.gov/dsdt/cwtg/satl.html; last accessed 7-25-12.

protection and flood risk management (i.e., placement of fill, groin construction, and pile driving). We also consider effects of dredging in the SBOBA for beach renourishment cycles through 2064.

7.1 Effects of Dredging Operations

As explained in the "Description of the Action" section above, the USACE plans on dredging within the SBOBA. Below, the effects of dredging, via the use of a hopper dredge, on threatened and endangered species will be considered. Effects of dredging include (1) entrainment and impingement of Atlantic sturgeon and sea turtles; (2) alteration of sea turtle and Atlantic sturgeon prey items and foraging behavior due to dredging; (3) suspended sediment associated with dredging operations; and (4) the potential for interactions between project vessels and individual Atlantic sturgeon, whales, or sea turtles.

As noted above, sea turtles are likely to occur in the action area from May-November of any year. The primary concern for loggerhead, Kemp's ridley, and green sea turtles is entrainment and the potential for effects to foraging, while the primary concern for leatherbacks is vessel collision. Right whales are likely to be present from November 1 – April 30 of any year; fin and humpback whales are most likely to be present in the spring, summer and fall; however, individual transient right, humpback, and fin whales could be present in the action area outside of these time frames as this area (SBOBA) is used by whales moving between calving/mating grounds and foraging grounds. Due to their large size, whales are not vulnerable to entrainment in dredges; as such, the primary concern for listed species of whales is the potential for vessel collisions. Atlantic sturgeon are likely to be present in the action area year round. The primary concern for Atlantic sturgeon is entrainment, loss of forage, and vessel collision.

Hopper dredges are self-propelled seagoing vessels that are equipped with propulsion machinery, sediment containers (hoppers), dredge pumps, and trailing suction drag-heads required to perform their essential function of excavating sediments from the ocean bottom. Hopper dredges have propulsion power adequate for required free-running speed and dredge against strong currents. They also have excellent maneuverability. This allows hopper dredges to provide a safe working environment for crew and equipment dredging bar channels or other areas subject to rough seas. Hopper dredges also are more practicable when interference with vessel traffic must be minimized.

A hopper dredge removes material from the bottom of the channel in relatively thin layers, usually 2-12 inches, depending upon the density and cohesiveness of the dredged material. Pumps located within the hull, but sometimes mounted on the drag arm, create a region of low pressure around the dragheads and force water and sediment up the drag arm and into the hopper. The more closely the draghead is maintained in contact with the sediment, the more efficient the dredging, provided sufficient water is available to slurry the sediments. Hopper dredges can efficiently dredge non-cohesive sands and cohesive silts and low density clay. Draghead types may consist of IHC and California type dragheads.

California type dragheads sit flatter in the sediment than the IHC configuration which is more upright. Individual draghead designs (i.e. dimensions, structural reinforcing/configuration) vary between dredging contractors and hopper vessels. Port openings on the bottom of dragheads also vary between contractors and draghead design. Generally speaking, the port geometry is typically rectangular or square with minimum openings of ten inch by ten inch or twelve inch by twelve inch or some rectangular variation.

Industry and government hopper dredges are equipped with various power and pump configurations and may differ in hopper capacity with different dredging capabilities. An engineering analysis of the known hydraulic characteristics of the pump and pipeline system on the USACE hopper dredge “Essayons” (a 6,423 cy hopper dredge) indicates an operational flow rate of forty cubic feet per second with a flow velocity of eleven feet per second at the draghead port openings. The estimated force exerted on a one-foot diameter turtle (i.e., one foot diameter disc shaped object) at the pump operational point in this system was estimated to be twenty-eight pounds of suction or drag force on the object at the port opening of the draghead.

Dredging is typically parallel to the centerline or axis of the channel. Under certain conditions, a waffle or crisscross pattern may be utilized to minimize trenching or during clean-up dredging operations to remove ridges and produce a more level channel bottom. This movement up and down the channel while dredging is called trailing and may be accomplished at speeds of 1-3 knots, depending on the shoaling, sediment characteristics, sea conditions, and numerous other factors. In the hopper, the slurry mixture of the sediment and water is managed by a weir system to settle out the dredged material solids and overflow the supernatant water. When an economic load is achieved, the vessel suspends dredging, the drag arms are raised, and the dredge travels to the designated placement site. Because dredging stops during the trip to the placement site, the overall efficiency of the hopper dredge is dependent on the distance between the dredging location and placement sites; the more distance to the placement site, the less efficient the dredging operation resulting in longer contract periods to accomplish the work.

Sea turtle deflectors utilized on hopper dredges are rigid V-shaped attachments on the front of the dragheads and are designed and intended to plow the sediment in front of the draghead. The plowing action creates a sand wave that rolls in front of the deflector. The propagated sand wave is intended to shed a turtle away from the deflector and out of the path of the draghead. The effectiveness of the rigid deflector design and its ability to reduce entrainment was studied by the USACE through model and field testing during the 1980s and early 1990s. The deflectors are most effective when operating on a uniform or flat bottom. The deflector effectiveness may be diminished when significant ridges and troughs are present that prevent the deflector from plowing and maintaining the sand wave and the dragheads from maintaining firm contact with the channel bottom.

There has been evidence of UXO mined along with the sand at the SBOBA, and because of the danger to human safety posed by these objects if taken directly into a hopper dredge, the hopper dredges used in the proposed actions will utilize UXO screens. UXO screens are comprised of longitudinal bars with openings/spacings of 1.25/ 1.5-inches by 6 inches. These dimensions will prevent any UXO from being brought on-board the hopper dredge. The screens will also prevent any whole ESA-listed species from being entrained by the dredge, instead, small pieces of the animal may be entrained. Animals impinged on the UXO screen may free or dislodge themselves from the screen once the suction of the dredge has been turned off. Animals that free themselves may suffer severe injuries that may result in death.

7.1.1 Alteration of foraging habitat
As discussed above, listed species of whales may be present within the action area year round as this area is used by whales moving between southern calving/mating grounds and northern foraging grounds. Whales forage upon pelagic prey items (e.g., krill, copepods, sand lance) and as such, dredging and its impacts on the benthic environment will not have any direct or indirect effects on whale prey/foraging items. As such, the remainder of this section will discuss the effects of dredging and the alteration of sea turtle and Atlantic sturgeon foraging habitat.

Atlantic sturgeon
Subadult (less than 150cm in total length, not sexually mature, but have left their natal rivers) and adult Atlantic sturgeon undertake seasonal, nearshore (i.e., typically depths less than 50 meters), coastal marine migrations along the United States eastern coastline (Erickson *et al*. 2011; Dunton *et al*. 2010). Based on tagging data, it is believed that beginning in the fall, Atlantic sturgeon undergo large scale migrations to more southerly waters (e.g., off the coast North Carolina, the mouth of the Chesapeake Bay) and primarily remain in these waters throughout the winter (i.e., approximately December through March), while in the spring, it appears that migrations begin to shift to more northerly waters (e.g., waters off New Jersey and New York) (Dovel and Berggren 1983; Dunton *et al*. 2010; Erikson *et al*. 2011). 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). Areas between these sites serve as migration corridors to and from these areas, as well as to spawning grounds found within natal rivers.

The SBOBA is approximately 1-3 miles from the nearest identified aggregation areas (i.e., off Sandy Hook, New Jersey). Atlantic sturgeon have been captured near the SBOBA. Based on this information, as well as information on the habitat characteristics of the SBOBA and the distribution of Atlantic sturgeon, opportunistic foraging may occur at this site. While opportunistic foraging may occur at these sites, it is more likely that the SBOBA is used by migrating individuals as they move from foraging, overwintering, and spawning grounds. As the foraging may occur in the SBOBA, foraging impacts to Atlantic sturgeon, as a result of dredging the SBOBA, will be considered below.

Sea Turtles
As outlined above, sea turtles may occur in the waters of New Jersey from May to the first week

in November each year when water temperatures are above 15°C, with the largest numbers present from June through October of any year. The sea turtles present in these waters are typically small juveniles with the most abundant being the threatened loggerhead (*Caretta caretta*) followed by the endangered Kemp's ridley (*Lepidochelys kempi*). Endangered green sea turtles (*Chelonia mydas*) also occur in these waters from June through October. Endangered leatherback sea turtles (*Dermochelys coriacea*) are typically found further offshore but may occur in nearshore waters while pursuing jellyfish, their preferred prey.

During the warmer months, most turtles in the Northeast appear to spend the majority of the time in waters between 16 and 49 feet. 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). As the SBOBA has a mean water depth of 50 feet (USACE-NYD 2013), the SBOBA is likely too deep to be considered suitable for sea turtle foraging. However, it is possible for foraging sea turtles to be present in the SBOBA. Therefore, effects to foraging sea turtles may occur within this portion of the action area and are considered below.

Alteration of Foraging Habitat

Dredging can cause indirect effects on Atlantic sturgeon and sea turtles by reducing prey species through the alteration of the existing biotic assemblages. As noted above, the SBOBA is not believed to be an area where Atlantic sturgeon concentrate to forage. However, opportunistic foraging may occur at this site. Since dredging involves removing the bottom material down to a specific depth, dredging is likely to entrain and kill some of these forage items that may be consumed by Atlantic sturgeon during their migrations.

Similar to Atlantic sturgeon, the SBOBA is not known to be an area where sea turtles concentrate to forage; however, based on surveys conducted in the area, potential sea turtle foraging items appear to be present, including crabs and mollusks. Of the listed sea turtle species found in the action area, loggerhead and Kemp's ridley sea turtles are the most likely to utilize these areas for feeding, foraging mainly on benthic species, such as crabs and mollusks (Morreale and Standora 1992; Bjorndal 1997). As no seagrass beds exist within the SBOBA, green sea turtles will not use the area as foraging areas and as such, dredging activities are not likely to disrupt normal feeding behaviors of green sea turtles. Additionally, jellyfish, the primary foraging item of leatherback sea turtles, are not likely to be affected by dredging activities as jellyfish occur within the upper portions of the water column and away from the sediment surface where dredging will occur. As jellyfish are not likely to be entrained during dredging, there is not likely to be any reduction in available forage for leatherback sea turtles due to the dredging operations. However, as suitable loggerhead and Kemp's ridley sea turtle foraging items may occur on the benthos of the areas, some loggerhead and Kemp's ridley sea turtle foraging may occur at the SBOBA and therefore, may be affected by dredging activities within this portion of the action area.

While some areas may be more desirable to certain sturgeon and turtles due to prey availability, there is no information to indicate that the SBOBA has more abundant turtle prey or better foraging habitat than other surrounding areas. The assumption can be made that sturgeon and sea turtles are not likely to be more attracted to the SBOBA than to other foraging areas and should be able to find sufficient prey in alternate areas. Depending on the species, recolonization of a dredged area can begin within as short as a month (Guerra-Garcia and Garcia-Gomez 2006). The dredged area is expected to be completely recolonized by benthic organisms within approximately 12 months after the dredging is complete. These conclusions are supported by a benthic habitat study which examined an area of Sandbridge Shoals following dredging, which concluded that recolonization of the dredged area was rapid, with macrobenthic organisms abundant on the first sampling date following cessation of dredging activities (less than a month later), and that there was no significant difference in macrofaunal abundance or biomass/production between areas that had and had not been dredged (Diaz *et al*. 2006); suggesting that dredging had no long term impact on prey availability. Based on this information, sturgeon and sea turtles should only be exposed to a reduction in forage in the areas where dredging occurs for one to two seasons immediately following dredging. Additionally, suitable foraging items should continue to be available within other portions of the Atlantic Ocean at all times.

Based on this and the best available information, NMFS anticipates that while the dredging activities may temporarily disrupt normal feeding behaviors for sturgeon and sea turtles by causing them to move to alternate areas, the action is not likely to remove critical amounts of prey resources from the portion of the action area located in SBOBA and any disruption to normal foraging is likely to be insignificant. In addition, the dredging activities are not likely to alter the habitat in any way that prevents sturgeon and sea turtles from using the action area as a migratory pathway to other near-by areas that may be more suitable for foraging.

7.1.2 Entrainment

7.1.2.1 Sea Turtles

Entrainment is defined as the direct uptake of aquatic organisms by the suction field generated at the draghead. Dredging operations within the SBOBA will involve the use of a hopper dredge. Given their large size, leatherback sea turtles are not vulnerable to entrainment in hopper dredges. To date, no leatherback sea turtles have been documented entrained in any dredge operation along the U.S. Atlantic coast (USACE Sea Turtle Warehouse, 2012). Therefore, this section of the Opinion will only consider the effects of entrainment on loggerhead, Kemp's ridley and green sea turtles. Sea turtles are likely to be feeding on or near the bottom of the water column during the warmer months, with loggerhead and Kemp's ridley sea turtles being the most common species in these waters. Although not expected to be as numerous as loggerheads and Kemp's ridleys, green sea turtles are also likely to occur seasonally in the SBOBA.

Sea turtles become entrained in hopper dredges as the draghead moves along the bottom. Entrainment occurs when sea turtles do not or cannot escape from the suction of the dredge. Sea turtles can also be crushed on the bottom by the moving draghead. Mortality most often occurs when turtles are sucked into the dredge draghead, pumped through the intake pipe and then killed as they cycle through the centrifugal pump and into the hopper. Because entrainment is believed to occur primarily while the draghead is operating on the bottom, it is likely that only those species feeding or resting on or near the bottom would be vulnerable to entrainment. Turtles can also be entrained if suction is created in the draghead by current flow while the device is being placed or removed, or if the dredge is operating on an uneven or rocky substrate and rises off the bottom. Recent information from the USACE suggests that the risk of entrainment is highest when the bottom terrain is uneven or when the dredge is conducting “clean up” operations at the end of a dredge cycle when the bottom is trenched and the dredge is working to level out the bottom. In these instances, it is difficult for the dredge operator to keep the draghead buried in the sand and sea turtles near the bottom may be more vulnerable to entrainment.

Sea turtles have been found resting in deeper waters, which could increase the likelihood of interactions with dredging activities. In 1981, observers documented the take of 71 loggerheads by a hopper dredge at the Port Canaveral Ship Channel, Florida (Slay and Richardson 1988). This channel is a deep, low productivity environment in the Southeast Atlantic where sea turtles are known to rest on the bottom, making them extremely vulnerable to entrainment. The large number of turtle mortalities at the Port Canaveral Ship Channel in the early 1980s resulted in part from turtles being buried in the soft bottom mud, a behavior known as brumation. Since 1981, 77 loggerhead sea turtles have been taken by hopper dredge operations in the Port Canaveral Ship Channel, Florida. Chelonid turtles have been found to make use of deeper, less productive channels as resting areas that afford protection from predators because of the low energy, deep water conditions. Habitat conditions in the SBOBA are not consistent with the areas where brumation has been documented; therefore, we do not anticipate that bromating sea turtles would be present in the action area.

Background Information on Entrainment of Sea Turtles in Hopper Dredges

Sea turtles have been killed in hopper dredge operations along the East and Gulf coasts of the US. Documented turtle mortalities during dredging operations in the USACE South Atlantic Division (SAD; i.e., south of the Virginia/North Carolina border) are more common than in the USACE North Atlantic Division (NAD; Virginia-Maine) probably due to the greater abundance of turtles in these waters and the greater frequency of hopper dredge operations. For example, in the USACE SAD, over 400 sea turtles have been entrained in hopper dredges since 1980 and in the Gulf Region over 160 sea turtles have been killed since 1995. Records of sea turtle entrainment in the USACE NAD begin in 1994. Through December 2013, 76 sea turtles deaths (see Table 13) related to hopper dredge activities have been recorded in waters north of the North Carolina/Virginia border (USACE Sea Turtle Database[9]); the majority of these turtles have been entrained in hopper dredges operating in Chesapeake Bay.

Table 13. Sea Turtle Takes in USACE NAD Dredging Operations

Project Location	Year of Operation	Cubic Yardage Removed	Observed Takes
Sandbridge Shoal	2013	Not Available (NA)	1 loggerhead
Cape Henry Channel	2012	NA	1 loggerhead
York Spit	2012	NA	1 Loggerhead

[9] The USACE Sea Turtle Data Warehouse is maintained by the USACE’s Environmental Laboratory and contains information on USACE dredging projects conducted since 1980 with a focus on information on interactions with sea turtles.

Thimble Shoal Channel	2009	NA	3 Loggerheads
York Spit	2007	608,000	1 Kemp's Ridley
Cape Henry	2006	NA	3 Loggerheads
Thimble Shoal Channel	2006	300,000	1 loggerhead
Delaware Bay	2005	50,000	2 Loggerheads
Thimble Shoal Channel	2003	1,828,312	7 Loggerheads 1 Kemp's ridley 1 unknown
Cape Henry	2002	1,407,814	6 Loggerheads 1 Kemp's ridley 1 green
VA Beach Hurricane Protection Project (Cape Henry)	2002	NA	1 Loggerhead
York Spit Channel	2002	911,406	8 Loggerheads 1 Kemp's ridley
Cape Henry	2001	1,641,140	2 loggerheads 1 Kemp's ridley
VA Beach Hurricane Protection Project (Thimble Shoals)	2001	NA	5 loggerheads 1 unknown
Thimble Shoal Channel	2000	831,761	2 loggerheads 1 unknown
York River Entrance Channel	1998	672,536	6 loggerheads
Atlantic Coast of NJ	1997	1,000,000	1 Loggerhead
Thimble Shoal Channel	1996	529,301	1 loggerhead
Delaware Bay	1995	218,151	1 Loggerhead
Cape Henry	1994	552,671	4 loggerheads 1 unknown
York Spit Channel	1994	61,299	4 loggerheads
Delaware Bay	1994	NA	1 Loggerhead
Delaware Bay	1993	NA	2 Loggerheads
Off Ocean City MD	1992	1,592,262	3 Loggerheads
			TOTAL = 76 Turtles

Before 1994, endangered species observers were not required on board hopper dredges and dredge baskets were not inspected for sea turtles or sea turtle parts. The majority of sea turtle takes in the NAD have occurred in the Norfolk District. This is largely a function of the large number of loggerhead and Kemp's ridley sea turtles that occur in the Chesapeake Bay each summer and the intense dredging operations that are conducted to maintain the Chesapeake Bay entrance channels and for beach nourishment projects at Virginia Beach. Since 1992, the take of 10 sea turtles (all loggerheads) has been recorded during hopper dredge operations in the Philadelphia, Baltimore, and New York Districts. Hopper dredging is relatively rare in New England waters where sea turtles are known to occur, with most hopper dredge operations being completed by the specialized Government owned dredge Currituck which operates at low suction and has been demonstrated to have a very low likelihood of entraining or impinging sea turtles.

It should be noted that the observed takes may not be representative of all the turtles killed during dredge operations. Typically, endangered species observers are required to observe a total of 50% of the dredge activity (i.e., 6 hours on watch, 6 hours off watch). As such, if the observer was off watch or the cage was emptied and not inspected or the dredge company either did not report or was unable to identify the turtle incident, there is the possibility that a turtle could be taken by the dredge and go unnoticed. Additionally, in older Opinions (i.e., prior to 1995), NMFS frequently only required 25% observer coverage and monitoring of the overflows which has since been determined to not be as effective as monitoring of the intakes. These conditions may have led to sea turtle takes going undetected.

NMFS raised this issue to the USACE during the 2002 season, after several turtles were taken in the Cape Henry and York Spit Channels, and expressed the need for 100% observer coverage. On September 30, 2002, the USACE informed the dredge contractor that when the observer was not present, the cage should not be opened unless it is clogged. This modification was to ensure that any sea turtles that were taken and on the intake screen (or in the cage area) would remain there until the observer evaluated the load. The USACE's letter further stated "Crew members will only go into the cage and remove wood, rocks, and man-made debris; any aquatic biological material is left in the cage for the observer to document and clear out when they return on duty. In addition, the observer is the only one allowed to clean off the overflow screen. This practice provides us with 100% observation coverage and shall continue." Theoretically, all sea turtle parts were observed under this scheme, but the frequency of clogging in the cage is unknown at this time. Obviously, the most effective way to ensure that 100% observer coverage is attained is to have a NMFS-approved endangered species observer monitoring all loads at all times. This level of observer coverage would document all turtle interactions and better quantify the impact of dredging on turtle populations. More recently issued Opinions have required 100% observer coverage which increases the likelihood of takes being detected and reported. However, some actions require the use of UXO screens on the dragheads. If there is an interaction with a draghead equipped with a UXO screen and a listed species, it would likely occur entirely underwater and all interactions would not be observed by an on-board observer. Due to the limited ability to observe an interaction from on deck, requiring the presence of an ESA observer on hopper dredges operating with a UXO screen is an impractical means to monitor interactions. Therefore, some hopper dredging projects (involving UXO screens) are not required to have an observer on board.

It is likely that not all sea turtles killed by dredges are observed onboard the hopper dredge. Several sea turtles stranded on Virginia shores with crushing type injuries from May 25 to October 15, 2002. The Virginia Marine Science Museum (VMSM) found 10 loggerheads, 2 Kemp's ridleys, and 1 leatherback exhibiting injuries and structural damage consistent with what they have seen in animals that were known dredge takes. While it cannot be conclusively determined that these strandings were the result of dredge interactions, the link is possible given the location of the strandings (e.g., in the southern Chesapeake Bay near ongoing dredging activity), the time of the documented strandings in relation to dredge operations, the lack of other ongoing activities which may have caused such damage, and the nature of the injuries (e.g., crushed or shattered carapaces and/or flipper bones, black mud in mouth). Additionally, in 1992, three dead sea turtles were found on an Ocean City, Maryland beach while dredging operations were ongoing at a borrow area located 3 miles offshore. Necropsy results indicate that the deaths of all three turtles were dredge related. It is unknown if turtles observed on the beach with these types of injuries were crushed by the dredge and subsequently stranded on shore or whether they were entrained in the dredge, entered the hopper and then were discharged onto the beach with the dredge spoils.

A dredge could crush an animal as it was setting the draghead on the bottom, or if the draghead was lifting on and off the bottom due to uneven terrain, but the actual cause of these crushing injuries cannot be determined at this time. Further analyses need to be conducted to better understand the link between stranded sea turtles with evidence of injury from crushing and dredging activities, and if those strandings need to be factored into an incidental take level. Regardless, it is possible that dredges are taking animals that are not observed on the dredge which may result in strandings on nearby beaches.

Due to the nature of interactions between listed species and dredge operations, it is difficult to predict the number of interactions that are likely to occur from a particular dredging operation. Projects that occur in an identical location with the same equipment year after year may result in interactions in some years and none in other years as noted above in the examples of sea turtle takes. Dredging operations may go on for months, with sea turtle takes occurring intermittently throughout the duration of the action. For example, dredging occurred at Cape Henry over 160 days in 2002 with 8 sea turtle takes occurring over 3 separate weeks while dredging at York Spit in 1994 resulted in 4 sea turtle takes in one week. In Delaware Bay, dredge cycles have been conducted during the May-November period with no observed entrainment and as many as two sea turtles have been entrained in as little as three weeks. Even in locations where thousands of sea turtles are known to be present (e.g., Chesapeake Bay) and where dredges are operating in areas with preferred sea turtle depths and forage items (as evidenced by entrainment of these species in the dredge), the numbers of sea turtles entrained is an extremely small percentage of the likely number of sea turtles in the action area. This is likely due to the distribution of individuals throughout the action area, the relatively small area which is affected at any given moment and the ability of some sea turtles to avoid the dredge even if they are in the immediate area.

The number of interactions between dredge equipment and sea turtles seems to be best associated with the volume of material removed, which is closely correlated to the length of time dredging takes, with a greater number of interactions associated with a greater volume of material removed and a longer duration of dredging. The number of interactions is also heavily influenced by the time of year dredging occurs (with more interactions correlated to times of year when more sea turtles are present in the action area) and the type of dredge plant used (sea turtles are apparently capable of avoiding pipeline and mechanical dredges as no takes of sea turtles have been reported with these types of dredges). The number of interactions may also be influenced by the terrain in the area being dredged, with interactions more likely when the draghead is moving up and off the bottom frequently. Interactions are also more likely at times and in areas when sea turtle forage items are concentrated in the area being dredged, as sea turtles are more likely to be spending time on the bottom while foraging.

Estimating Sea Turtle Entrainment During Deepening of the Ambrose Channel

As noted above, sea turtles are likely to be less concentrated in the action area for this consultation than they are in areas under the jurisdiction of the Norfolk District (e.g., Chesapeake Bay). Based on this information, NMFS believes that hopper dredges operating in the SBOBA are less likely to interact with sea turtles than hopper dredges operating in areas under the jurisdiction of the Norfolk District (e.g., Chesapeake Bay). As a result, all Norfolk District hopper dredging projects will not be considered further in our analysis as they do not accurately reflect the potential rate of entrainment for projects that occur in areas where sea turtles are not as concentrated.

It is most appropriate to look at other hopper dredging projects that have been undertaken in similar environments or with similar geographic characteristics as the SBOBA to determine a comparable level of potential sea turtle entrainment. Some operations in similar environments have, and still are, operated with a UXO screen on the draghead of the hopper. Large pieces of a sea turtle were recently observed entrained within a dredge equipped with a UXO screen at Sandbridge Shoal, VA. The dredge was inspected after the incident and it was determined that the UXO screen was not damaged. Upon closer examination of the engineering design of the draghead and dredge assembly, it is possible that the sea turtle may have entered through ports or "trunions" that surround the draghead itself. The USACE is beginning to discuss a demo or pilot project, sometime during the summer of 2014, off NY. The project will apply different width screens over the ports and trunions, coupled with observers, to investigate the different efficiencies and effectiveness of the screens and any impacts on performance on the draghead itself. This investigation is currently ongoing.

Despite this information, we still believe that UXO screens are likely to preclude an observer from detecting all entrained sea turtle or sea turtle parts (see section 11.0 for further information and clarification). Accordingly past observer records from these projects are not appropriate to use in our assessment as they may not reliably and accurately reflect entrainment in relation to the cubic yards of material removed.

As the SBOBA is located in an “offshore” / nearshore environment in the waters of the Atlantic Ocean, we looked at all hopper dredging projects in the NAD, excluding the Norfolk District, that had comparable environmental or geographic characteristics of this area to use as baseline information on the levels of sea turtle entrainment that have occurred in these areas/environments. The most appropriate projects to consider were those undertaken in offshore/nearshore (i.e., within 10 miles off the U.S. Eastern coastline) environments or open estuarine environments (see Table 14). We did not consider riverine or enclosed to semi-enclosed bays or estuaries in our assessment as we do not feel the environmental characteristics of these areas are comparable to open estuarine or offshore environments and thus, the level of entrainment in these areas would not be comparable to the level of entrainment that may occur in the SBOBA.

We have compiled records for 21 projects occurring during "sea turtle season" (i.e., May – November 15th) in the Baltimore, Philadelphia and New York District. As noted above, all projects listed in Table 14 are located in environments that are comparable to that of the SBOBA and report the cubic yardage removed during a project; however an important caveat is that observer coverage for some of these projects has ranged from 0 to 50% (see Table 14).

As explained above, for projects prior to 1995, observers were only present on the dredge for every other week of dredging. For projects in 1995 to the present, observers were present on board the dredge full time and worked a 6-hour on, 6-hour off shift. The only time that cages (where sea turtle parts are typically observed) were cleaned by anyone other than the observer was when there was a clog. If a turtle or turtle part was observed in such an instance, crew were instructed to inform the observer, even if off-duty. As such, it is reasonable to expect that even though there was only 50% observer coverage, an extremely small amount of biological material went unobserved. To make the data from the 1993 and 1994 dredge events when observers were only on board every other week, comparable to the 1995-2006 data when observers were on board full time, NMFS has assumed that an equal number of turtles were entrained when observers were not present. This calculation is reflected in Table 14 as "adjusted entrainment number."

Table 14. Projects in USACE NAD (with recorded cubic yardage; all Norfolk District projects removed).[10]

Project Location	Year of Operation	Cubic Yards Removed	Observed Entrainment	Adjusted Entrainment Number
Dewey and Bethany Beach (DE)	2009	397,956	0	0
Sandy Hook Channel	2008	23,500	0	0
Dewey Beach/Cape Henlopen (DE Bay)	2005	1,134,329	0	0
Delaware Bay	2005	50,000	2 Loggerhead	2 Loggerhead

[10] All projects were operating during "sea turtle season" (i.e., May to November 15). Additionally, only dredges operating without a UXO screen were included, as these screens, are likely to preclude an observer from detecting entrained sea turtles or sea turtle parts (see section 11.0 for further information and clarification) and thus, do not accurately reflect observed entrainment in relation to the cubic yards of material removed.

Cape May Point, NJ	2005	2,425,268	0	0
Off Ocean City MD	2002	744,827	0	0
East Rockaway Inlet, NY	2002	140,000	0	0
Westhampton, NY (offshore borrow site)	1997	884571	0	0
Offshore New Jersey	1997	3,700,000	1 Loggerhead	1 Loggerhead
Off Ocean City MD	1998	1,289,817	0	0
East Rockaway Inlet, NY	1996	2,685,000	0	0
Westhampton, NY (offshore borrow site)	1996	2518592	0	0
Delaware Bay	1995	218,151	1 Loggerhead	1 Loggerhead
East Rockaway Inlet, NY	1995	412,000	0	0
Bethany Beach (DE Bay)	1994	184,451	0	0
Dewey Beach (DE Bay)	1994	624,869	0	0
Off Ocean City MD	1994	1,245,125	0	0
Westhampton, NY (offshore borrow site)	1993	1455071	0	0
Off Ocean City MD	1992	1,592,262	3 Loggerheads	6 Loggerheads
Off Ocean City MD	1991	1,622,776	0	0
Off Ocean City MD	1990	2,198,987	0	0
	TOTAL	***25,547,552 cy***	***7 Loggerheads***	***10 Loggerheads***

Based on the data presented in Table 14, NMFS has made calculations which indicate that an average of one sea turtle is killed for approximately every 2.6 million cubic yards of material removed by a hopper dredge in environments similar to, or like, the SBOBA. This calculation is based on a number of assumptions including the following: that sea turtles are evenly distributed throughout all open estuarine or "offshore" areas, that all hopper dredges have a similar entrainment rate, and that sea turtles are equally likely to be encountered throughout the April to November time frame.

Sea turtle species likely to be entrained

With the exception of one green turtle entrained in a hopper dredge operating in Chesapeake Bay, all other sea turtles entrained in dredges operating in the USACE NAD have been loggerheads and Kemp's ridley. Of these 76 sea turtles, 66 have been loggerhead, 5 have been Kemp's ridleys, 1 green and 4 unknown. No Kemp's ridleys or greens have been entrained in dredge operations outside of the Chesapeake Bay area. The high percentage of loggerheads is likely due to several factors including their tendency to forage on the bottom where the dredge is operating and the fact that this species is the most numerous of the sea turtle species in Northeast and Mid-Atlantic waters. It is likely that the documentation of only one green sea turtle entrainment in Virginia dredging operations is a reflection of the low numbers of green sea turtles that occur in waters north of North Carolina. The low number of green sea turtles in the action area makes an interaction with a green sea turtle extremely unlikely to occur.

Based on the above information, it is reasonable to expect that 1 sea turtle is likely to be injured or killed for approximately every 2.6 million cy of material removed from the SBOBA. Based on the information outlined above, NMFS anticipates that no more than 1 sea turtle is likely to be entrained during the dredging for the Port Monmouth project (i.e., 391,000 cy of material is removed), no more than 1 sea turtle for the Union Beach project (i.e., 688,000 cy of material is removed), and no more than 6 sea turtles during the dredging for the Elberon to Loch Arbour project (i.e., 14,834,452 cy of material is removed). Due to the nature of the injuries expected to result from entrainment, these turtles are expected to die.

We expect that nearly all of the sea turtles will be loggerheads and that the entrainment of a Kemp's ridley during a dredge cycle will be rare; however, as Kemp's ridleys have been documented in the action area and have been entrained in hopper dredges, it is likely that this species will interact with the dredge over the course of the project life. As explained above, approximately 91% of the sea turtles taken in dredges operating in the USACE North Atlantic Division have been loggerheads.

Based on the ratio of sea turtle entrainment in the USACE NAD, it is likely that entrainments in all projects will involve loggerhead sea turtles. However, given that the data suggests there is a 9% chance that a sea turtle interaction with a hopper dredge will be a Kemp's ridley, it's possible that up to 1 Kemp's ridley sea turtle interaction, per project, will occur. As noted above, interactions with green sea turtles are extremely unlikely. The anticipated number of sea turtle entrainments by project is presented in Table 15.

Table 15. Anticipated number of loggerhead and green sea turtle entrainments by project

Project Name	Total Sea Turle Entrainments	Loggerhead	Kemp's Ridley
Port Monmouth	1	1*	1*
Union Beach	1	1*	1*
Elberon to Loch Arbour	6	5 (up to 6)	1*
Total	8	Up to 8	Up to 3

*Loggerhead or Kemp's ridley

7.1.2.2 Atlantic Sturgeon

Entrainment Risk: Hopper Dredge

Atlantic sturgeon are vulnerable to entrainment in hopper dredges; however, given the large size of adults (greater than 150cm) and the size of the openings on the dragheads, adult Atlantic sturgeon are unlikely to be vulnerable to entrainment. From 1990-2012, the USACE has documented a total of 36 confirmed incidences of entrainment or capture of sturgeon species on monitored projects for all types of dredge plants (mechanical, hydraulic pipeline, and hopper dredge). Of these, 23 were reported as Atlantic sturgeon, with 21 of these entrained in hopper dredges. Of the entrained Atlantic sturgeon for which size is available, all were subadults (larger than 50cm but less than 150cm). Information on these interactions is presented in Table 16. Most of these interactions occurred within harbors; however, to date, few records exist for interactions between hopper dredges and Atlantic sturgeon within offshore environments similar to the SBOBA (see Table 17).

Table 16. USACE Atlantic Sturgeon Entrainment Records from Hopper Dredge Operations 1990-2012

Project Location	Corps Division/District *	Month/Year of Operation	Cubic Yards Removed	Observed** Entrainment
Winyah Bay, Georgetown (SC)	SAD/SAC	Oct-90	517,032	1
Savannah Harbor (GA)	SAD/SAS	Jan-94	2,202,800	1
Savannah Harbor	SAD/SAS	Dec-94	2,239,800	2
Wilmington Harbor, Cape Fear River (NC)	SAD/SAW	Sep-98	196,400	1
Charleston Harbor (SC)	SAD/SAC	Mar-00	5,627,386	1
Brunswick Harbor (GA)	SAD/SAS	Feb-01	1,459,630	1
Charleston Harbor	SAD/SAC	Jan-04	1,449,234	1
Brunswick Harbor	SAD/SAS	Mar-05	966,000	1
Brunswick Harbor	SAD/SAS	Dec-06	1,198,571	1
Savannah Entrance Channel	SAD/SAS	Nov-07	973,463	1
Sandy Hook Channel (NJ)	NAD/NANY	Aug-Nov-08	23,500	1
Savannah Entrance Channel	SAD/SAS	Mar-09	261,780	1
Brunswick Entrance Channel	SAD/SAS	Feb-10	1,728,339	3
Wilmington Harbor	SAD/SAW	Dec-10	857,726	1
York Spit (VA)	NAD/NAN	Apr-11	1,630,713	2
Charleston Harbor	SAD/SAC	**Mar-12**	**1,100,000**	**1**

Ambrose Channel-Contract B	NAD/NANY	**Oct-12**	**1,510,000**	**1**
		Total	**23,942,374**	**21**

* SAD= South Atlantic Division; NAD= North Atlantic Division; SAC=Charleston District; SAS=Savannah District; SAW=Wilmington District; NANY=New York District; NAN=Norfolk District.
** Records based on sea turtle observer reports which record listed species entrained as well as all other organisms entrained during dredge operations.

Table 17: Open Estuarine Channel Deepening projects in USACE NAD since 1998 with recorded cubic yardage[11]

*a: Observed entrainment of Atlantic sturgeon believed to be a result of a damaged UXO screen.
*b: 14 Atlantic sturgeon removed during pre-dredge trawl/relocation trawling (September and November, 2003).
*c: 1 Atlantic sturgeon removed during pre-dredge trawl/relocation trawling on 10/26/02.
*d: 1 Atlantic sturgeon removed during pre-dredge trawl/relocation trawling on 11/02/02.

Project Location	**Year of Operation**	**Cubic Yards Removed**	**Observed Entrainment**	Observed Entrainment
Ambrose Channel-Contract Area B*a	2012	1,510,000	1	1
York Spit Channel, VA	2011	1,630,713	2	2
Cape Henry Channel, VA	2011	2,472,000	0	0
York Spit Channel, VA	2009	372,533	0	0
Sandy Hook Channel, NJ	2008	23,500	1	1
York Spit Channel, VA	2007	608,000	0	0
Atlantic Ocean Channel, VA	2006	1,118,749	0	0
Thimble Shoal	2006	300,000	0	0

[11] Only dredges operating without a UXO screen were included, as these screens, are likely to preclude an observer from detecting entrained sturgeon or sturgeon parts (see section 12.0 for further information and clarification) and thus, may not accurately reflect observed entrainment in relation to the cubic yards of material removed.

Channel, VA				
Thimble Shoal Channel, VA	2004	139,200	0	0
VA Beach Hurricane Protection Project	2004	844,968	0	0
Thimble Shoal Channel (*b)	2003	1,828,312	0	0
Cape Henry Channel, VA (*c)	2002	1,407,814	0	0
York Spit Channel, VA (*d)	2002	911,406	0	0
East Rockaway Inlet, NY	2002	140,000	0	0
Cape Henry Channel, VA	2001	1,641,140	0	0
Thimble Shoal Channel, VA	2000	831,761	0	0
Cape Henry Channel, VA	2000	759,986	0	0
York Spit Channel, VA	1998	296,140	0	0
Cape Henry Channel, VA	1998	740,674	0	0
Thimble Shoal Channel, VA	1996	529,301	0	0
East Rockaway Inlet, NY	1996	2,685,000		
Cape Henry Channel, VA	1995	485,885	0	0
East Rockaway Inlet, NY	1995	412,000	0	0
York Spit Channel, VA	1994	61,299	0	0
Cape Henry Channel , VA	1994	552,671	0	0
	TOTAL	22,303,052	4	4

** Records based on sea turtle observer reports which record listed species entrained as well as all other organisms entrained during dredge operations.*

*** On September 16, 2012, the New York District USACE informed us that the anterior portion of an Atlantic sturgeon was found within the inflow screening of the hopper dredge operating within the Ambrose Channel-Contract B. The sturgeon part was moderately decomposed. It is believed that the animal had died by some other cause(s) and thus, was not attributed as an entrainment incident related to or as a result of the Ambrose Channel deepening, and thus, was not considered in the table above.*

As described above, dredging operations within the SBOBA will be conducted with a UXO screen on the draghead of the hopper. Although an Atlantic sturgeon was recently observed entrained within a dredge operating in the Ambrose Channel, it was concluded that this entrainment was likely due to damage to the screen which permitted the entrainment of the sturgeon. However, without this damage, an interaction with the sturgeon may have still occurred, but would have likely gone unobserved. As some dredges have been operating with a UXO screen since 2006, we cannot discount the possibility that, so long as the screen was undamaged, unobservable interactions may have still occurred with Atlantic sturgeon. As a result, we strongly believe that UXO screens, in undamaged states, are likely to preclude an observer from detecting entrained sturgeon or sturgeon parts (see section 11.0 for further information and clarification). Accordingly, it is not appropriate to use data from dredging operations in which a UXO screen was used in our assessment of Atlantic sturgeon entrainment. In the absence of sufficient information specific to the SBOBA that we can rely on to make our assessment, it is most appropriate to consider other projects that have been conducted in a comparable environment to that of the SBOBA (see Table 17). The most appropriate projects to consider were those in "offshore"/ nearshore (i.e., within 10 miles off the U.S. Eastern coastline) environments or open estuarine environments. We did not consider riverine or enclosed to semi-enclosed bays or estuaries in our assessment as the environmental characteristics of these areas are not comparable to open estuarine or offshore environments. As such, the level of entrainment in these areas would not be comparable to the level of entrainment that may occur in the SBOBA.

As explained above, in the Northeast Region (Maine through Virginia), endangered species observers have been present on all hopper dredges operating between April 1 and November 30 since 1994. While the primary responsibility of observers is to document sea turtle interactions, observers document all biological material entrained in the dredges. As such, they record any observed interactions with sturgeon. Sturgeon interactions have routinely been reported to NMFS. Therefore, we expect that the "observed entrainment" numbers noted above are comprehensive and that any interactions with Atlantic sturgeon would be recorded. While observers have not operated on dredges working from December – March, in the Northeast Region dredging during this time of year is rare (due to weather conditions) and we do not anticipate that there are many undocumented interactions between Atlantic sturgeon and hopper dredges.

In general, entrainment of large mobile animals, such as sturgeon or sea turtles, is relatively rare. Several factors are thought to contribute to the likelihood of entrainment. In areas where animals are present in high density, the risk of an interaction is greater because more animals are exposed to the potential for entrainment. It has also been suggested that the risk of entrainment is highest in areas where the movements of animals are restricted (e.g., in river channels) where there is limited opportunity for animals to move away from the dredge. Because hopper dredging will occur in an offshore environment (i.e., the SBOBA), the movements of Atlantic sturgeon will not be restricted and we anticipate that most Atlantic sturgeon will be able to avoid the dredge. Further, because Atlantic sturgeon are likely to be using the action area as a migration corridor, the density of Atlantic sturgeon in any portion of the action area is likely to be low. In addition, the hopper dredge draghead operates on the bottom and is typically at least partially buried in the sediment. Sturgeon are benthic feeders and are often found at or near the bottom while foraging or while moving within rivers. Information suggests that Atlantic sturgeon migrating in the marine environment do not move along the bottom, but move further up in the water column. If Atlantic sturgeon are up off the bottom while in offshore areas, such as the SBOBA, the potential for interactions with the dredge are further reduced. Based on this information, the likelihood of an interaction of an Atlantic sturgeon with a hopper dredge operating in the SBOBA is expected to be low.

However, because we know that entrainment is possible and that not all mobile animals will be able to escape from the dredge (as evidenced by past entrainment of sea turtles and sturgeon), we anticipate that entrainment is still possible and as such, effects of these interactions on Atlantic sturgeon must be assessed. As noted above, outside of rivers/harbors, only 4 Atlantic sturgeon have been observed entrained in a hopper dredge (see Table 17). The low level of interactions may be due to the use of pre-trawl/dredge relocation trawling (see Table 17. Although no Atlantic sturgeon were entrained in some locations, they were documented in the area prior to dredging operations. Another explanation for the low levels of interactions may be that some interactions were not reported to NMFS; however, based on information that has been provided to NMFS and discussions with observers, under-reporting is likely to be very rare.

As noted above, based on what we know about Atlantic sturgeon behavior in environments comparable to the SBOBA, it is reasonable to consider that the risk of entrainment at this site is similar to that of sites located within open estuarine environments (i.e., see Table 17). Some of the areas considered in this analysis (see Table 17) are closer to shore than the area being dredged with a hopper dredge in the SBOBA and may be more heavily used than this area. Thus, an estimate of interactions derived from this information is likely an overestimate; however, at this time, this is the best available information on the potential for interactions with Atlantic sturgeon.

It is important to note that because observer coverage has been variable, observed interactions may not be representative of all Atlantic sturgeon injured or killed during dredge. As such, we have adjusted the entrainment numbers to account for any instances where observer coverage was less than 100%.

Past experience calculating the likelihood of interactions between hopper dredges and other species (i.e., sea turtles) indicates that there is a relationship between the number of animals entrained and the volume of material removed. The volume of material removed is correlated to the amount of time spent dredging but is a more accurate measure of effort because reports often provide the total days of a project but may not provide information on the actual hours of dredging vs. the number of hours steaming to the disposal site or in port for weather or other delays. Thus, we will use information available for all dredging projects that have been undertaken in open estuarine or offshore environments in the mid-Atlantic for which cubic yards of material removed are available to calculate the number of Atlantic sturgeon likely to be entrained during dredging operations (see Table 17). Using this method, and using the dataset presented in Table 17, we have calculated an entrainment rate of 1 Atlantic sturgeon is likely to be injured or killed for approximately every 5.6 million cy of material removed during hopper dredging operations undertaken at the SBOBA. This calculation is based on a number of assumptions including the following: that adult and subadult Atlantic sturgeon are evenly distributed throughout the action area, that all hopper dredges will have the same entrainment rate, and that Atlantic sturgeon are equally likely to be encountered throughout the time period when dredging will occur. While this estimate is based on several assumptions, it is reasonable because it uses the best available information on entrainment of Atlantic sturgeon from past dredging operations, including dredging operations in the vicinity of the action area, it includes multiple projects over several years, and all of the projects have had observers present which we expect would have documented any entrainment of Atlantic sturgeon.

Based on the information outlined above, NMFS anticipates that while dredging at the SBOBA, no more than 1 Atlantic sturgeon is likely to be entrained during the Port Monmouth project, no more than 1 Atlantic sturgeon is likely to be entrained during the Union Beach project, and no more than 3 Atlantic sturgeon are likely to be entrained during the Elberon to Loch Arbour project. Because we expect that adult Atlantic sturgeon are too large to be vulnerable to entrainment and given the size of other sturgeon that have been entrained in other hopper dredging operations, we expect that these sturgeon will be subadult.

There is evidence that some Atlantic sturgeon, particularly juveniles and small subadults, could be entrained in the dredge and survive. However, as the extent of internal injuries and the likelihood of survival is unknown, and the size of the fish likely to be entrained is impossible to predict, it is reasonable to conclude that any Atlantic sturgeon entrained in the hopper dredge are likely to be killed. Based on the NEFOP mixed-stock analysis, we have determined that Atlantic sturgeon in the action area likely originate from the five DPSs at the following frequencies: NYB 51%; South Atlantic 22%; Chesapeake Bay 13%; Gulf of Maine 11%; and Carolina 2%; we anticipate that entrained Atlantic sturgeon will occur at similar frequencies.

Table 18. Anticipated number of Atlantic sturgeon interactions by project

Project Name	Total Atlantic Sturgeon	NYB DPS	SA DPS	CB DPS	GOM DPS	Carolina DPS
Port Monmouth	1	1*	1*	1*	1*	1*
Union Beach	1	1*	1*	1*	1*	1*
Elberon to Loch Arbour	3	2	1**	1**	1**	1**
Possible Total	5	Up to 4	Up to 3	Up to 3	Up to 3	Up to 3

* NYB, SA, CB, GOM or Carolina DPS Atlantic sturgeon
** SA, CB, GOM or Carolina DPS Atlantic sturgeon

7.1.3 Interactions with the Sediment Plume

7.1.3.1 Hopper Dredge

Dredging operations cause sediment to be suspended in the water column. This results in a sediment plume in the water, typically present from the dredge site and decreasing in concentration as sediment falls out of the water column as distance increases from the dredge site. The nature, degree, and extent of sediment suspension around a dredging operation are controlled by many factors including: the particle size distribution, solids concentration, and composition of the dredged material; the dredge type and size, discharge/cutter configuration, discharge rate, and solids concentration of the slurry; operational procedures used; and the characteristics of the hydraulic regime in the vicinity of the operation, including water composition, temperature and hydrodynamic forces (i.e., waves, currents, etc.) causing vertical and horizontal mixing (USACE 1983).

Resuspension of fine-grained dredged material during hopper dredging operations is caused by the dragheads as they are pulled through the sediment, turbulence generated by the vessel and its prop wash, and overflow of turbid water during hopper filling operations. During the filling operation, dredged material slurry is often pumped into the hoppers after they have been filled with slurry in order to maximize the amount of solid material in the hopper. The lower density turbid water at the surface of the filled hoppers overflows and is usually discharged through ports located near the waterline of the dredge. In the vicinity of hopper dredge operations, a near-bottom turbidity plume of resuspended bottom material may extend 2,300 to 2,400 ft down current from the dredge (USACE 1983). In the immediate vicinity of the dredge, a well-defined upper plume is generated by the overflow process. Approximately 1,000 ft behind the dredge, the two plumes merge into a single plume (USACE 1983). Suspended solid concentrations may be as high as several tens of parts per thousand (ppt; grams per liter) near the discharge port and as high as a few parts per thousand near the draghead. In a study done by Anchor Environmental

(2003), nearfield concentrations ranged from 80.0-475.0 mg/l. Turbidity levels in the near-surface plume appear to decrease exponentially with increasing distance from the dredge due to settling and dispersion, quickly reaching concentrations less than 1 ppt. By a distance of 4,000 feet from the dredge, plume concentrations are expected to return to background levels (USACE 1983). Studies also indicate that in almost all cases, the vast majority of resuspended sediments resettle close to the dredge within one hour, and only a small fraction takes longer to resettle (Anchor Environmental 2003).

Total suspended sediment (TSS) is most likely to affect sea turtles, subadult and adult Atlantic sturgeon, or whales if a plume causes a barrier to normal behaviors or if elevated levels of suspended sediment affects prey species. As whales, sturgeon, and sea turtles are highly mobile, individuals are likely to be able to avoid any sediment plume that is present and any effect on their movements or behavior is likely to be insignificant. In addition, the total suspended sediment levels expected (80 – 475 mg/L) 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)). TSS may reach levels that can have an adverse effect on benthic communities (390.0 mg/L (EPA 1986)); however, McCauley et al. (1977) observed that while infauna populations declined significantly after dredging, infauna at dredging and placement areas recovered to pre-dredging conditions within 28 and 14 days, respectively. Therefore, the direct and indirect impacts to benthic communities are anticipated to be minimal. Rapid recovery and resettlement of benthic species is expected. Given this information, effects to whales, sturgeon, and sea turtle prey from increased turbidity is extremely unlikely; effects to listed whales, sturgeon and sea turtles will be discountable.

7.1.4 Collisions with vessels

There have not been any reports of dredge vessels colliding with listed species but contact injuries resulting from dredge movements could occur at or near the water surface and could therefore involve any of the listed species present in the area. Because the dredge is unlikely to be moving at speeds greater than three knots during dredging operations, blunt trauma injuries resulting from contact with the hull are unlikely during dredging. It is more likely that contact injuries during actual dredging would involve the propeller of the vessel. Contact injuries with the dredge are more likely to occur when the dredge is moving from the dredging area to port, or between dredge locations. While the distance between these areas is relatively short (12 – 16 miles), the dredge in transit would be moving at faster speeds (9.8 – 10.8 mph) than during dredging operations (2 – 3 mph), particularly when empty while returning to the borrow area.

The dredge vessel may collide with marine mammals and sea turtles when they are at the surface or, in the case of Atlantic sturgeon, in the water column when migrating. These species have been documented with injuries consistent with vessel interactions and it is reasonable to believe that the dredge vessels considered in this Opinion could inflict such injuries on Atlantic sturgeon, marine mammals and sea turtles, should they collide. As mentioned, sea turtles are found distributed throughout the action area in the warmer months, generally from May through November; Right whales primarily from November 1 through April 30; humpback and fin whales, spring, summer, and fall; and, Atlantic sturgeon throughout the year.

Effects of Vessel Collisions on Sea Turtles

Interactions between vessels and sea turtles occur and can take many forms, from the most severe (death or bisection of an animal or penetration to the viscera), to severed limbs or cracks to the carapace which can also lead to mortality directly or indirectly. Sea turtle stranding data for the U.S. Gulf of Mexico and Atlantic coasts, Puerto Rico, and the U.S. Virgin Islands show that between 1986 and 1993, about 9% of living and dead stranded sea turtles had propeller or other boat strike injuries (Lutcavage *et al*. 1997). According to 2001 STSSN stranding data, at least 33 sea turtles (loggerhead, green, Kemp's ridley and leatherbacks) that stranded on beaches within the northeast (Maine through North Carolina) were struck by a boat. This number underestimates the actual number of boat strikes that occur since not every boat struck turtle will strand, every stranded turtle will not be found, and many stranded turtles are too decomposed to determine whether the turtle was struck by a boat. It should be noted, however, that it is not known whether all boat strikes were the cause of death or whether they occurred post-mortem (NMFS SEFSC 2001).

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. The speed of the dredge is not expected to exceed 2.6 knots while dredging or 10 knots while transiting to the pump out site with a full load and it is expected to operate at a maximum speed of 11 knots while empty. In addition, the risk of ship strike will be influenced by the amount of time the animal remains near the surface of the water. For the proposed action, the greatest risk of vessel collision will occur during transit between shore and the areas to be dredged. The presence of an experienced endangered species observer who can advise the vessel operator to slow the vessel or maneuver safely when sea turtles are spotted will further reduce the potential risk for interaction with vessels. The addition of one to two slow moving vessels in the action area have an insignificant effect on the risk of interactions between sea turtles and vessels in the action area.

Effects of Vessel Collisions on Atlantic Sturgeon

Information regarding the risk of vessel strikes to Atlantic sturgeon is discussed in the Status of the Species and Environmental Baseline sections above. As explained there, we have limited information on vessel strikes and many variables likely affect the potential for vessel strikes in a given area. Assuming that the risk of vessel strike increases with an increase in vessel traffic, we have considered whether an increase in vessel traffic in the action area during dredging (one to two slow moving vessels per day) would increase the risk of vessel strike for Atlantic sturgeon in this area. Although little is known about a sturgeon's reaction to vessel traffic, it is generally assumed that sturgeon are more likely to avoid injury from slower-moving vessels since the sturgeon has more time to maneuver and avoid the vessel. The speed of the dredge is not expected to exceed 2.6 knots while dredging or 10 knots while transiting to the pump out site with a full load and it is expected to operate at a maximum speed of 11 knots while empty. In addition, the risk of ship strike will be influenced by the amount of time the animal remains near the surface of the water. For the proposed action, the greatest risk of vessel collision will occur during transit between shore and the areas to be dredged. The presence of an experienced endangered species observer who can advise the vessel operator to slow the vessel or maneuver safely when sturgeon are spotted will further reduce the potential risk for interaction with vessels. Given the large volume of traffic in the area and the wide variability in traffic in any given day, the increase in traffic of one to two vessels per day is negligible and the increased risk to Atlantic sturgeon is insignificant.

Effects of Vessel Collisions on Whales

Large whales, particularly right whales, are vulnerable to injury and mortality from ship strikes. Ship strike injuries to whales take two forms: (1) propeller wounds characterized by external gashes or severed tail stocks; and (2) blunt trauma injuries indicated by fractured skulls, jaws, and vertebrae, and massive bruises that sometimes lack external expression (Laist *et al*. 2001). Collisions with smaller vessels may result in propeller wounds or no apparent injury, depending on the severity of the incident. Laist *et al*. (2001) reports that of 41 ship strike accounts that reported vessel speed, no lethal or severe injuries occurred at speeds below ten knots, and no collisions have been reported for vessels traveling less than six knots. A majority of whale ship strikes seem to occur over or near the continental shelf, probably reflecting the concentration of vessel traffic and whales in these areas (Laist *et al*. 2001). As discussed in the Status of the Species section, all whales are potentially subject to collisions with ships. However, due to their critical population status, slow speed, and behavioral characteristics that cause them to remain at the surface, vessel collisions pose the greatest threat to right whales. From 2003-2007, NMFS confirmed that 7 female right whales have been killed by ship collisions, one of which was carrying a near-term fetus. Because females are more critical to a population's ability to replace its numbers and grow, the premature loss of even one reproductively mature female could hinder the species' likelihood of recovering.

Most ship strikes have occurred at vessel speeds of 13-15 knots or greater (Jensen and Silber 2003; Laist *et al*. 2001). An analysis by Vanderlaan and Taggart (2006) showed that at speeds greater than 15 knots, the probability of a ship strike resulting in death increases asymptotically to 100%. At speeds below 11.8 knots, the probability decreases to less than 50%, and at ten knots or less, the probability is further reduced to approximately 30%. As noted above, the speed of the dredge is not expected to exceed 2.6 knots while dredging, 10 knots while transiting to the disposal sites, and no more than 11 knots while empty. In addition, all vessels will have lookouts on board and operators will receive training on prudent vessel operating procedures to avoid vessel strikes with all protected species. Based on this information, the potential interaction of a dredge/vessel and a listed species of whale is likely to be discountable.

7.2 Effects of Beach Nourishment

Dredged material will be used for beach nourishment or shoreline restoration work. As these sites are generally located within shallow, nearshore, waters, listed species of whales are not expected to occur within the vicinity of these sites, and thus, any effects of these operations on whales are expected to be discountable. However, as Atlantic sturgeon and sea turtles could potentially be present in the vicinity of such sites, effects to these species are possible. These effects include alteration of habitat and increases in turbidity.

7.2.1 *Alteration of foraging habitat*

Placement of material at beach nourishment sites, such as the Port Monmouth, Union Beach, and Elberon to Loch Arbour, can affect Atlantic sturgeon and sea turtles by reducing prey species through the alteration of the existing biotic assemblages (i.e., burying existing subtidal benthic organisms (e.g., crabs, clams, mussels)). As the purpose of placing dredge material at these sites is to restore or replenish the affected area, in general, the environment in which the material is to be placed can be characterized as an area exposed to high wave energy and thus, erosion, and one devoid of high densities or colonies of benthic organisms (e.g., shellfish beds, mollusks, crabs, SAV). Instead, these sites consist primarily of benthic infaunal communities (e.g., polycheates) that can withstand the variable and continually changing environment. Preferred prey items or habitat for Atlantic sturgeon and sea turtles (e.g., shellfish beds, crabs, mollusks, areas of SAV) are therefore, rarely established in these areas. Thus, it is extremely unlikely that the placement of dredged material in the nearshore waters of New Jersey, will result in the removal of critical amounts of prey resources from the area. Should any prey items be removed from the area in which dredged material is to be placed, depending on the species, recolonization of a newly renourished beach can begin in as short as 2-6 months (Burlas *et al*. 2001) when there is a good match between the fill material and the natural beach sediment. As the sand being placed along shorelines is similar in grain size to the indigenous beach sand, it is expected that recolonization of the nearshore benthos will occur within 2-6 months after initial beach renourishment or shoreline restoration cycles are complete. As such, no long term impacts on the numbers of species or community composition of the beach infauna is expected (USACE 1994; Burlas *et al*. 2001). In addition, beach nourishment or shoreline restoration operations in the proposed projects 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 areas with more suitable foraging habitat. As such, the effects of these operations on foraging or migrating sea turtles or Atlantic sturgeon will be insignificant.

7.2.2 Interactions with the Sediment Plume

The placement of dredged material along beaches or shorelines will cause an increase in localized turbidity in the nearshore environment. Nearshore turbidity impacts from fill placement are directly related to the quantity of fines (silt and clay) in the nourishment material. As the material from the SBOBA to be placed at sites is comprised of medium sized grains of sand, and consists of beach quality sand of similar grain size and composition as indigenous beach sands, short suspension time and containment of sediment during and after placement activities is expected. As such, turbidity impacts are expected to be short-term (i.e., within several hours of the cessation of operations (Greene 2002)) and spatially limited to the vicinity of the dredge outfall pipe, the pump-out station, and dredge anchor points.

The Atlantic States Marine Fisheries Commission (Greene 2002) review of the biological and physical impacts of beach nourishment cites several studies that report that the turbidity plume and elevated total suspended sediment levels drop off rapidly seaward of the sand placement operations. Wilber *et al*. (2006) evaluated the effects of a beach nourishment project along the coast of northern New Jersey and reported that maximum bottom surf zone and nearshore total suspended sediment concentrations related to nourishment activities were 64 mg/L and 34 mg/L, which were only slightly higher than background maximum bottom total suspended sediment concentrations in the surf and nearshore zones on unnourished portions of the beach (i.e., less than 20 mg/L). Additionally, Wilber *et al*. (2006) reported that elevated total suspended sediment concentrations associated with the active beach nourishment site were limited to within 400 m (1,310 feet) of the discharge pipe in the swash zone (defined as the area of the nearshore that is intermittently covered and uncovered by waves), while other studies found that the turbidity plume and elevated total suspended sediment levels are expected to be limited to a narrow area of the swash zone up to 500 m (1,640 feet) down current from the discharge pipe (Schubel *et al*. 1978; Burlas *et al*. 2001). Based on this and the best available information, turbidity levels created by the beach fill operations along the shoreline are expected to be between 34-64 mg/l; limited to an area approximately 500 meters down current from the discharge pipe, with dissipation occurring within several hundred meters along the shore; and, are expected to be short term, only lasting several hours.

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). Total suspended sediment concentrations are most likely to affect Atlantic sturgeon and sea turtles if a plume causes a barrier to normal behaviors or if sediment settles on the bottom affecting sea turtle prey. As Atlantic sturgeon and sea turtles are highly mobile they are likely to be able to avoid any sediment plume and any effect on Atlantic sturgeon or sea turtle movements is likely to be insignificant. Additionally, the total suspended sediment levels expected (i.e., 34-64 mg/l) 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 benthic resources that sturgeon and sea turtles may eat are extremely unlikely. While the increase in suspended sediments may cause Atlantic sturgeon and sea turtles to alter their normal movements, any change in behavior is likely to be insignificant as it will only involve movements to alter course out of the sediment plume and is not likely to affect the migration ability of Atlantic sturgeon and sea turtles. Based on this information, it is likely that the effect of the suspension of sediment resulting from beach nourishment or shoreline restoration operations, such as those to occur at Port Monmouth, Union Beach, and Elberon to Loch Arbour, on sea turtles and Atlantic sturgeon will be insignificant. As listed species of whales will not be present in the shallow, nearshore environments where beach nourishment or shoreline restoration activities will be undertaken, listed species of whales will not be exposed to any elevated levels of suspended sediment that may be produced from these activities.

7.3 Groin Construction

Groins will be constructed for shoreline stabilazation work. As these sites are generally located within shallow, nearshore, waters, listed species of whales are not expected to occur within the vicinity of these sites, and thus, no effects of these operations on whales are expected. However, as Atlantic sturgeon and sea turtles could potentially be present in the vicinity of such sites, effects to these species are possible. These effects include alteration of habitat and increases in turbidity.

7.3.1 Alteration of foraging habitat

The placement of stone can cause effects on sea turtles and sturgeon by reducing prey species through the alteration of the existing biotic assemblages and habitat. The footprint of the groin being constructed at Port Monmouth will affect approximately 0.57 acres of seafloor. At Union Beach, the construction of the groins will affect approximately .09 acres of seafloor. The groin construction at Elberon to Loch Arbour will not affect any new area of seafloor as it involves removing rocks from existing groins and placing them along different areas of the groins.

Shallow waters (<10 feet) where the groins will be located are not known to provide optimal foraging for sea turtles (16-49 feet is preferred), and may or may not provide adequate opportunistic foraging for Atlantic sturgeon. In general, minor disruptions or removal of small proportions of benthic habitat associated with these projects that may provide opportunistic foraging habitat will have minimal impacts on the overall availability of suitable foraging habitat for both Atlantic sturgeon and sea turtles throughout Raritan Bay and the Atlantic Ocean off of New Jersey. These structures will take up well less than one acre in size. Less than 1 acre is minor in comparison to the size of the surrounding area of Sandy Hook Bay and Raritan Bay (more than 50,000 acres). As such, ample habitat will remain available for both sea turtles and Atlantic sturgeon to opportunistically forage. Additionally, the proposed stone placement operations are not likely to alter the habitat in any way that prevents sturgeon and sea turtles from using any portion of the action area as a migratory pathway and therefore, would not disrupt any essential behaviors such as migrating or foraging. Based on this information, the effects of stone placement on Atlantic sturgeon and sea turtle migration and foraging are expected to be insignificant and discountable.

7.3.2 *Interactions with the Sediment Plume*

The placement of stone (bedding, armor, and underlayer) during the construction of the groins will disturb shoreline sediments and may cause a temporary increase in suspended sediment in the nearshore area. If any sediment plume does occur, it is expected to be small, and is expected to settle out of the water column within a few hours. Turbidity levels associated with any sediment plume are expected to be only slightly elevated above background levels (< 5mg/L). Based on this information, it is likely that effects of stone placement to sea turtles and sturgeon will be discountable.

7.4 Installation of Piles

7.4.1 Installation of Timber Piles via Jetting

Approximately 40 timber piles, one foot in diameter will be installed to modify an existing timber pier. The method for placing the wood piles supporting the pier will be to water jet/push the piles into place. As the site is located in shallow, nearshore, waters, listed species of whales are not expected to occur within the vicinity of these sites, and thus, any effects of this operation on whales are expected to be discountable. However, as Atlantic sturgeon and sea turtles could potentially be present in the vicinity of such sites, effects to these species are possible. These effects include alteration of habitat and increases in turbidity.

Jetting is a method of forcing water around and under a pile to loosen and displace the surrounding soils resulting in the disturbance of bottom sediments. The operation does have the potential to result in an increase in suspended sediment levels in the area immediately surrounding the pile; however, suspended sediment is expected to settle out of the water column rapidly with both lateral and vertical distance from the operating jet plow. Within 100 meters of the jet plow, both maximum and mean suspended sediment concentrations are predicted to be less than 200.0 mg/L and after 24 hours, the suspended sediment concentration level above ambient is predicted to be below 50.0 mg/L, with the concentration dropping to less than 20.0 mg/L above ambient after 48 hours (ESS Group, Inc., 2008). In addition, under all tidal conditions, suspended sediment concentrations >100.0 mg/L are predicted to remain in the bottom 2 to 3 meters of the water column and concentrations are predicted to decrease rapidly to approximately 10.0 mg/L or less 5 to 7 meters above the bottom (ESS Group, Inc., 2008).

7.4.1.1 Alteration of foraging habitat
Some disturbance or removal of benthic invertebrates, which may serve as Atlantic sturgeon or sea turtle prey, may occur in the area where the piles will be installed via jetting. Depending on the species, recolonization of a dredged/jetted channel can begin in as short as a month (Guerra-Garcia *et al.* 2003), with the area expected to be completely recolonized by benthic organisms within approximately 12 months (USACE, 2001; US DOI, 2000). Some reduction in the amount of potential Atlantic sturgeon and sea turtle prey in the area to be jetted is likely; however, the action will not result in the permanent removal of forage items, as prey species will continually recolonize the area following a disturbance. In summary, as the area affected by jetting is small and recolonization of the benthic community will be rapid, we have determined that any effects of jetting to foraging Atlantic sturgeon and sea turtles will be insignificant.

7.4.1.2 Interactions with sediment plume
No information is available on the effects of total suspended solids (TSS) on juvenile and adult sea turtles. 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 sturgeon and sea turtles 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 and sea turtles are highly mobile, they are likely to be able to avoid any sediment plume and any effect on sturgeon and sea turtle movements is likely to be insignificant. Additionally, the TSS levels expected for jetting (20.0 to 200.0 mg/L) 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; see summary of scientific literature in Burton 1993) and benthic communities (390.0 mg/L (EPA 1986)); therefore, effects to benthic resources that sturgeon or sea turtles may eat are unlikely. Additionally, while the increase in suspended sediments may cause Atlantic sturgeon or sea turtles to alter their normal movements, any change in behavior is likely to be insignificant as it will only involve movements to alter course out of the sediment plume and is not likely to affect the overall movement or migration ability of sturgeon and sea turtles. Based on this information, the effect of suspended sediment resulting from jetting on Atlantic sturgeon or sea turtles will be insignificant.

7.4.2 Pile Driving
Steel sheet piles will be driven in Flat Creek, East Creek, and Pews Creek. No ESA-listed species are present in any of these creeks; therefore, there will be no effect to ESA-listed species as a result of driving steel sheet piles in these creeks.

The Elberon to Loch Arbour project will involve the installation of timber piles to support the outfall extensions. Piles will be installed with an impact or vibratory hammer depending on substrate conditions in the area. In general vibratory hammers are quieter than impact hammers, and the larger the pile, the greater the noise level (Illingworth and Rodkin Inc. and Jones and Stoke 2009). Therefore, for the purpose of this consultation, we will analyze the sound levels from 12 inch timber piles driven via an impact hammer.

As the site is located in shallow, nearshore waters, listed species of whales are not expected to occur within the vicinity of these sites, and thus, any effects of this operation on whales are expected to be discountable. However, as Atlantic sturgeon and sea turtles could potentially be present in the vicinity of such sites, effects to these species are possible.

The installation of piles can produce underwater sound pressure waves that can affect aquatic species. The available literature indicates that the the driving of 12 inch timber piles via an impact hammer produces underwater noise levels of approximately 170 dB_{RMS} within 10 meters of the pile being driven (Illingworth and Rodkin Inc. and Jones and Stoke 2009).

As the distance from the source increases, underwater sound levels produced by pile driving are known to attenuate rapidly. Using data from Illingworth and Rodkin, Inc. and Jones and Stoke (2009), underwater noise levels produced from the driving of timber piles will attenuate approximately 10 dB every 10 meters. This is based on a conservative estimate of attenuation rates for the driving of piles (Illingworth and Rodkin, Inc. 2007, 2009).

As a source of underwater noise, pile driving produces underwater sound pressure waves of varying intensity (*i.e*., sound attenuates over distance so noise levels are greater closer to the source) that can cause behavioral and/or physiological effects to aquatic species, such as whales, sea turtles, and Atlantic sturgeon. The intensity of the underwater noise and the ability of the animal to detect the sound may result in behavioral modification of the animal (*e.g*., temporary avoidance of an area; Richardson *et al*. 1995). The physical nature of the sound (*i.e.,* pressure waves and particle motion) produced by pile driving; however, may result in physiological effects to an animal. Pressure waves, generated from particle motion, cause fields of compression and rarefaction to move through the water, as well as through any object that contains air or gas filled chambers (*e.g*., swim bladders of fish), thereby causing injury to internal organs of the organism. The latter can result in a range of physiological effects on fish ranging from those that are not likely to affect the survival of the species (*e.g.,* small ruptures of capillaries in fins) to those that result in mortality (*e.g*., rupturing of the swimbladder) (Reyff 2003; Abbott and Bing-Sawyer 2002; Caltrans 2001; Longmuir and Lively 2001; Stotz and Colby 2001; Stephensen *et al.* 2010). These characteristics, as well as many other factors (*e.g*., the type and size of pile; installation method; type and size of fish (smaller fish are more often impacted); fish hearing sensitivity; received distance), contribute to the likelihood of behavioral and physiological effects to an individual fish.

Sea Turtles

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} 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 other studies have been done to assess the effects of 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 information, we believe that any underwater noise level at or above 166 re 1μPa_{RMS} has the potential to adversely affect sea turtles (*e.g.,* injury, temporary threshold shifts).

As described in above, sound levels may be as high as 170 dB re 1μPa_{RMS} within 10 meters of the timber pile being driven with an impact hammer and thus, at a distance beyond approximately 15 meters from the timber piles being driven, noise levels will be below 166 dB re 1μPa_{RMS}. The nearshore area where the timber piles will be installed is not known to be a high use area for sea turtles and as such, it is extremely unlikely that sea turtles will occur within 0 to15 meters of the piles being driven and therefore, be exposed to adverse elevated under water noise levels at or above 166 dB re 1μ$Pa_{RMS.}$ Based on this information, the noise effects of pile driving on sea turtles is discountable.

Atlantic sturgeon
An interagency work group, including the U.S. Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS), has reviewed the best available scientific information and developed criteria for assessing the potential of pile driving activities to cause injury to fish (Fisheries Hydroacoustic Working Group (FHWG) 2008). The workgroup established dual sound criteria for injury, measured 10 meters away from the pile, of 206 dB re 1 μPa_{Peak} and 187 dB accumulated sound exposure level (dBcSEL; re: 1μPa^2•sec) (183 dB accumulated SEL for fish less than 2 grams). While this work group is based on the US West coast, species similar to shortnose and Atlantic sturgeon were considered in developing this guidance (green sturgeon). As these species are biologically similar to the species being considered herein, it is reasonable to use the criteria developed by the FHWG to assess the potential for injury to Atlantic sturgeon from pile driving operations.

No studies have been undertaken to determine the noise levels that would result in behavioral disturbance to Atlantic sturgeon. Given the available information from studies on other fish species (*i.e*., Anderson *et al.* 2007; Purser and Radford 2011; Wysocki *et al.* 2007), we consider 150 dB re 1 μPa_{RMS} to be a reasonable estimate of the noise level at which exposure may result in behavioral modifications. These behaviors could range from a temporary startle to avoidance of the noisy area.

Based on the best available information, the driving of timber piles, via an impact hammer, will produce underwater noise levels below 206 dB re 1 μPa_{Peak} and 187cSEL. As such, even if sturgeon were present in the area where piles were being installed, no injury would occur.

Based on attenuation rates and the information presented above, underwater noise levels are expected to be below 150 dB re 1 μPa $_{RMS}$ at a distance beyond 30 meters from the timber pile being driven. In the worst case, sturgeon would avoid the area where noise levels are above 150 dB re 1 μPa $_{RMS}$. Given the small size of the area where noise levels will be elevated at any one time, (*i.e.,* an area with a radius of no more than 30 meters), and the open ocean that will provide a large area for a zone of passage, temporary avoidance of the noisy area would involve small changes in the movements of individual sturgeon. These small behavioral changes are not expected to result in any increased energy expenditure or cause any disruption to essential behaviors such as foraging, migrating or resting. As such, all effects to Atlantic sturgeon from pile driving will be insignificant and discountable.

7.4.2.1 Alteration of foraging habitat
The installation of piles will disturb bottom sediments. However, little increase in sedimentation or turbidity is expected to result from these construction activities. If any sediment plume does occur, it is expected to be small and suspended sediment is expected to settle out of the water column within a few hours and any increase in turbidity will be short term. Additionally, sea turtles and Atlantic sturgeon are expected to be able to temporarily avoid the area and continue normal behaviors in nearby portions of the bay. Therefore, there would not be any disruption of essential behaviors such as migrating or foraging. As such, any effects of pile driving are expected to be discountable.

8.0 CUMULATIVE EFFECTS
Cumulative effects as defined in 50 CFR § 402.02 to include the effects of future State or private activities, which are reasonably certain to occur within the action area. Future Federal actions are not considered in the definition of "cumulative effects."

Sources of human-induced mortality, injury, and/or harassment of Atlantic sturgeon, whales, or sea turtles resulting from future State, tribal, local or private actions in the action area that are reasonably certain to occur in the future include incidental takes in state-regulated fishing activities, pollution, global climate change, and vessel collision. While the combination of these activities may affect Atlantic sturgeon, whales, or sea turtles, preventing or slowing the species' recovery, the magnitude of these effects in the action area is currently unknown. However, this Opinion assumes effects in the future, with the exception of climate change, 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.

State Water Fisheries- Fishing activities are considered one of the most significant causes of death and serious injury for sea turtles. A 1990 National Research Council report estimated that 550 to 5,500 sea turtles (juvenile and adult loggerheads and Kemp's ridleys) die each year from all other fishing activities besides shrimp fishing. Fishing gear in state waters, such as bottom trawls, gillnets, trap/pot gear, and pound nets, take sea turtles each year. NMFS is working with state agencies to address the take of sea turtles in state-water fisheries within the action area of this consultation where information exists to show that these fisheries take sea turtles. Action has been taken by some states to reduce or remove the likelihood of sea turtle takes in one or more gear types. However, given that state managed commercial and recreational fisheries along the Atlantic coast are reasonably certain to occur within the action area in the foreseeable future, additional takes of sea turtles in these fisheries are anticipated. There is insufficient information by which to quantify the number of sea turtle takes presently occurring as a result of state water fisheries as well as the number of sea turtles injured or killed as a result of such takes. While actions have been taken to reduce sea turtle takes in some state water fisheries, the overall effect of these actions on reducing the take of sea turtles in state water fisheries is unknown, and the future effects of state water fisheries on sea turtles cannot be quantified.

Right and humpback whale entanglements in gear set for state fisheries are also known to have occurred (e.g., Waring *et al.* 2007; Glass *et al.* 2008). Actions have been taken to reduce the risk of entanglement to large whales, although more information is needed on the effectiveness of these actions. State water fisheries continue to pose a risk of entanglement to large whales to a level that cannot be quantified.

Information on interactions with Atlantic sturgeon with state fisheries operating in the action area is not available, and it is not clear to what extent these future activities will affect listed species differently than the current 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/environmental baseline section.

State PDES Permits – The states of New Jersey, Delaware and Pennsylvania have been delegated authority to issue NPDES permits by the EPA. These permits authorize the discharge of pollutants in the action area. Permitees include municipalities for sewage treatment plants and other industrial users. The states will continue to authorize the discharge of pollutants through the SPDES permits. 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.

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 with 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

NMFS has estimated that over the life of the 3 projects (i.e., through 2064), up to 8 sea turtles will be entrained in hopper dredging operations; these sea turtles could either be Kemp's ridley or loggerhead sea turtles. Additionally, NMFS has estimated that over the life of the 3 projects, up to 5 subadult Atlantic sturgeon will be entrained in hopper dredging operations. As explained in the "Effects of the Action" section, effects of habitat alteration, suspended sediment, increased underwater noise, and vessel interactions on sea turtles, whales, or Atlantic sturgeon as a result of the projects will be insignificant and/or discountable. In addition, as explained above, no whales or green or leatherback sea turtles are likely to be entrained in any dredge operating within the SBOBA, and thus, NMFS has determined that the likelihood of an interaction (i.e., entrainment) between a green or leatherback sea turtle or a whale and a hopper dredge is discountable.

In the discussion below, we consider whether the effects of the proposed actions 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 the listed species that will be adversely affected by the actions. The purpose of this analysis is to determine whether the proposed actions, in the context established by the status of the species, environmental baseline, and cumulative effects, would jeopardize the continued existence of any listed species. In the NMFS/USFWS Section 7 Handbook, for the purposes of determining jeopardy, survival is defined as, "the species' persistence as listed or as a recovery unit, beyond the conditions leading to its endangerment, with sufficient resilience to allow for the potential recovery from endangerment. Said in another way, survival is the condition in which a species continues to exist into the future while retaining the potential for recovery. This condition is characterized by a species with a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, which exists in an environment providing all requirements for completion of the species' entire life cycle, including reproduction, sustenance, and shelter."

Recovery is defined as, "Improvement in the status of listed species to the point at which listing is no longer appropriate under the criteria set out in Section 4(a)(1) of the Act." Below, for the listed species that may be affected by the proposed actions, we summarize the status of the species and consider whether the proposed action will result in reductions in reproduction, numbers or distribution of these species and then consider whether any reductions in reproduction, numbers or distribution resulting from the proposed actions would reduce appreciably the likelihood of both the survival and recovery of these species, as those terms are defined for purposes of the federal Endangered Species Act.

9.1 Atlantic sturgeon

As explained above, the proposed actions are likely to result in the mortality of a total of five Atlantic sturgeon from the Gulf of Maine, New York Bight, Chesapeake Bay, Carolina and/or South Atlantic DPSs through 2064 during the dredging at SBOBA. We expect that the Atlantic sturgeon killed will be all be subadults. No mortality of any adults is anticipated. All other effects to Atlantic sturgeon, including effects to habitat and prey due to dredging and fill placement, and elevated underwater noise, will be insignificant and discountable.

9.1.1 Determination of DPS Composition

We have considered the best available information to determine from which DPSs individuals that will be killed are likely to have originated. Using mixed stock analysis explained above, we have determined that Atlantic sturgeon in the action area likely originate from the five DPSs at the following frequencies: NYB 51%; South Atlantic 22%; Chesapeake Bay 13%; Gulf of Maine 11%; and Carolina 2%. Given these percentages, of the five sturgeon likely to be killed during the dredging operations, up to 4 will originate from the NYB DPS and up to 3 from the South Atlantic, Cheasapeake Bay, Carolina, and the Gulf of Maine DPSs.

9.2.2 Gulf of Maine DPS

We expect that 11% of the Atlantic sturgeon in the action area will originate from the GOM DPS. Most of these fish are expected to be subadults, with few adults from the GOM DPS expected to be present in the action area. No mortality of adult Atlantic sturgeon is anticipated to result from the proposed actions. We expect that no more than three GOM DPS Atlantic sturgeon will be killed during dredging. These mortalities will occur between now and the end of 2064.

While Atlantic sturgeon occur in several rivers in the GOM DPS, recent spawning has only been documented in the Kennebec and Androscoggin rivers. No total population estimates are available for any river population or the DPS as a whole. As discussed in section 4.3, we have estimated a total of 7,544 GOM DPS adults and subadults in the ocean (1,864 adults and 5,591 subadults). This estimate is the best available at this time and represents only a percentage of the total GOM DPS population as it does not include young of the year or juveniles and does not include all adults and subadults. GOM origin Atlantic sturgeon are affected by numerous sources of human induced mortality and habitat disturbance throughout the riverine and marine portions of their range. While there are some indications that the status of the GOM DPS may be improving, there is currently not enough information to establish a trend for any life stage or for the DPS as a whole.

The number of subadult GOM DPS Atlantic sturgeon we expect to be killed due to the dredging of the SBOBA represents an extremely small percentage of the GOM DPS. While the death of three subadult GOM DPS Atlantic sturgeon over the next 50 years will reduce the number of GOM DPS Atlantic sturgeon compared to the number that would have been present absent the proposed actions, it is not likely that this reduction in numbers will change the status of this species. Even if there were only 5,591 subadults in the GOM DPS, this loss would represent only 0.05% of the subadults in the DPS. The percentage would be much less if we also considered the number of young of the year, juveniles, adults, and other subadults not included in the NEAMAP-based oceanic population estimate.

Because there will be no loss of adults, the reproductive potential of the GOM DPS will not be affected in any way other than through a reduction in numbers of individual future spawners as opposed to current spawners. The loss of three female subadults would have the effect of reducing the amount of potential reproduction as any dead GOM DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in an extremely small reduction in the number of eggs laid or larvae produced in future years and similarly, an extremely small effect on the strength of subsequent year classes. Even considering the potential future spawners that would be produced by the individuals that would be killed as a result of the proposed action, any effect to future year classes is anticipated to be extremely small and would not change the status of this species. The loss of male subadults may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in a particular year. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning. The proposed actions will also not affect the spawning grounds within the rivers where GOM DPS fish spawn.

The proposed actions are not likely to reduce distribution because the action will not impede GOM DPS Atlantic sturgeon from accessing any seasonal concentration areas, including foraging, spawning or overwintering grounds. Any effects to distribution will be minor and temporary and limited to the temporary avoidance of the area of increased sediment around the working dredge.

Based on the information provided above, the death of no more than three subadult GOM DPS Atlantic sturgeon over 50 years, will not appreciably reduce the likelihood of survival of the GOM DPS (*i.e.*, it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The actions will not affect GOM DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is the case because: (1) the death of three subadult GOM DPS Atlantic sturgeon represents an extremely small percentage of the species; (2) the death of these GOM DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these GOM DPS Atlantic sturgeon is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of these subadult GOM DPS Atlantic sturgeon is likely to have such a small effect on reproductive output that the loss of these individuals will not change the status or trends of the species; (5) the actions will have only a minor and temporary effect on the distribution of GOM DPS Atlantic sturgeon in the action area and no effect on the distribution of the species throughout its range; and, (6) the actions will have only an insignificant effect on individual foraging or sheltering GOM DPS Atlantic sturgeon.

In rare instances, an action that does not appreciably reduce the likelihood of a species' survival might appreciably reduce its likelihood of recovery. As explained above, we have determined that the proposed actions will not appreciably reduce the likelihood that the GOM DPS of Atlantic sturgeon will survive in the wild. Here, we consider whether the action will appreciably reduce the likelihood of recovery from the perspective of ESA Section 4. As noted above, recovery is defined as the improvement in status such that listing under Section 4(a) as "in danger of extinction throughout all or a significant portion of its range" (endangered) or "likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range…" (threatened) is no longer appropriate. Thus, we have considered whether the proposed actions will appreciably reduce the likelihood that the GOM DPS of Atlantic sturgeon can rebuild to a point where it is no longer in danger of becoming endangered within the foreseeable future throughout all or a significant portion of its range.

No Recovery Plan for the GOM DPS has been published. The Recovery Plan will outline the steps necessary for recovery and the demographic criteria which once attained would allow the species to be delisted. We know that in general, to recover, a listed species must have a sustained positive trend of increasing population over time. To allow that to happen for GOM Atlantic sturgeon, individuals must have access to enough habitat in suitable condition for foraging, resting and spawning. Conditions must be suitable for the successful development of early life stages. Mortality rates must be low enough to allow for recruitment to all age classes so that successful spawning can continue over time and over generations. There must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. For Atlantic sturgeon, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness. Here, we consider whether these proposed actions will affect the GOM DPS likelihood of recovery.

These actions will not change the status or trend of the GOM DPS as a whole. The proposed actions will result in a small amount of mortality (three subadults from a population estimated to have at least 5,000 subadults) and a subsequent small reduction in future reproductive output. This reduction in numbers will be small and the impact on reproduction and future year classes will also be small enough not to affect the trend of the population. The proposed actions will have only insignificant effects on habitat and forage and will not impact the area in a way that makes additional growth of the population less likely. This is because the area that sturgeon may avoid is small and any avoidance will be temporary and limited to the period of time when increased suspended sediment is experienced or increased underwater noise. The proposed actions will not affect GOM DPS of Atlantic sturgeon outside of the action area or affect habitats outside of the action area. Therefore, it will not affect estuarine or oceanic habitats that are important for sturgeon. For these reasons, the actions will not reduce the likelihood that the GOM DPS can recover. Therefore, the proposed actions will not appreciably reduce the likelihood that the GOM DPS of Atlantic sturgeon can be brought to the point at which they are no longer listed as threatened. Based on the analysis presented herein, the proposed actions, are not likely to appreciably reduce the survival and recovery of this species.

9.2.3 New York Bight DPS
The NYB DPS is listed as endangered. We expect that 51% of the Atlantic sturgeon in the action area will originate from the NYB DPS. No mortality of adult Atlantic sturgeon is anticipated. We anticipate the mortality of up to four NYB DPS Atlantic sturgeon as a result of entrainment in a hopper dredge. These fish are expected to be subadults originating from the

Delaware or Hudson River. While it is possible that entrained fish could survive, we assume here that these fish will be killed.

While Atlantic sturgeon occur in several rivers in the NYB DPS, recent spawning has only been documented in the Hudson and Delaware rivers. No total population estimates are available for any river population or the DPS as a whole. As discussed in section 4.3, we have estimated a total of 34,566 NYB DPS adults and subadults in the ocean (8,642 adults and 25,925 subadults). This estimate is the best available at this time and represents only a percentage of the total NYB DPS population as it does not include young of the year or juveniles and does not include all adults and subadults. NYB origin Atlantic sturgeon are affected by numerous sources of human induced mortality and habitat disturbance throughout the riverine and marine portions of their range. There is currently not enough information to establish a trend for any life stage or for the DPS as a whole.

We have limited information from which to determine the percentage of NYB DPS fish in the SBOBA that are likely to originate from the Delaware vs. the Hudson River. The overall ratio of Delaware River to Hudson River fish in the DPS as a whole is unknown. Some Delaware River fish have a unique genetic haplotype (the A5 haplotype); however, whether there is any evolutionary significance or fitness benefit provided by this genetic makeup is unknown. Genetic evidence indicates that while spawning continued to occur in the Delaware River and in some cases Delaware River origin fish can be distinguished genetically from Hudson River origin fish, there is free interchange between the two rivers. This relationship is recognized by the listing of the New York Bight DPS as a whole and not separate listings of a theoretical Hudson River DPS and Delaware River DPS. Thus, while we can consider the loss of Delaware River fish on the Delaware River population and the loss of Hudson River fish on the Hudson River population, it is more appropriate, because of the interchange of individuals between these two populations, to consider the effects of this mortality on the New York Bight DPS as a whole.

The mortality of up to four subadult Atlantic sturgeon from the NYB DPS over a 50-year period represents a very small percentage of the subadult population. While the death of four subadult Atlantic sturgeon will reduce the number of NYB DPS Atlantic sturgeon compared to the number that would have been present absent the proposed action, it is not likely that this reduction in numbers will change the status of this species as these losses represents a very small percentage of the subadult population and an even smaller percentage of the overall population of the DPS (juveniles, subadults and adults combined).

The reproductive potential of the NYB DPS will not be affected in any way other than through a reduction in numbers of individuals. The loss of four female subadults over a 50 year period (average of one per 12.5 years) would have the effect of reducing the amount of potential reproduction as any dead NYB DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in an extremely small reduction in the number of eggs laid or larvae produced in future years and similarly, an extremely small effect on the strength of subsequent year classes. Even considering the potential future spawners that would be produced by the individuals that would be killed as a result of the proposed action, any effect to future year classes is anticipated to be extremely small and would not change the status of this species. The loss of four male subadult sturgeon may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in a particular year. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning. The proposed actions will also not affect the spawning grounds within the Delaware River or the Hudson River where most NYB DPS fish spawn. There will be no effects to spawning adults and therefore no reduction in individual fitness or any future reduction in spawning by these individuals.

The proposed actions are not likely to reduce distribution because the actions will not impede NYB DPS Atlantic sturgeon from accessing any seasonal concentration areas, including foraging, spawning or overwintering grounds. Any effects to distribution will be minor and temporary and limited to the temporary avoidance of a small ensonified area and sediment plumes. Further, the action is not expected to reduce the river by river distribution of Atlantic sturgeon.

Based on the information provided above, the death of four NYB DPS Atlantic sturgeon over a 50-year period, will not appreciably reduce the likelihood of survival of the New York Bight DPS (*i.e.*, it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The actions will not affect NYB DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is the case because: (1) the death of these subadult NYB DPS Atlantic sturgeon represents an extremely small percentage of the species; (2) the death of these NYB DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these NYB DPS Atlantic sturgeon is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of these subadults will not result in the loss of any age class; (5) the loss of these NYB DPS Atlantic sturgeon is likely to have such a small effect on reproductive output that the loss of these individuals will not change the status or trends of the species; (6) the actions will have only a minor and temporary effect on the distribution of NYB DPS Atlantic sturgeon in the action area and no effect on the distribution of the species throughout its range; and, (7) the actions will have no effect on the ability of NYB DPS Atlantic sturgeon to shelter and only an insignificant effect on individual foraging NYB DPS Atlantic sturgeon.

In rare instances, an action that does not appreciably reduce the likelihood of a species' survival might appreciably reduce its likelihood of recovery. As explained above, we have determined that the proposed actions will not appreciably reduce the likelihood that the NYB DPS of Atlantic sturgeon will survive in the wild,. Here, we consider whether the action will appreciably reduce the likelihood of recovery from the perspective of ESA Section 4. As noted above, recovery is defined as the improvement in status such that listing under Section 4(a) as "in danger of extinction throughout all or a significant portion of its range" (endangered) or "likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range…" (threatened) is no longer appropriate. Thus, we have considered whether the proposed action will appreciably reduce the likelihood that the NYB DPS of Atlantic sturgeon can rebuild to a point where it is no longer in danger of extinction throughout all or a significant part of its range.

No Recovery Plan for the NYB DPS has been published. The Recovery Plan will outline the steps necessary for recovery and the demographic criteria which once attained would allow the species to be delisted. We know that in general, to recover, a listed species must have a sustained positive trend of increasing population over time. To allow that to happen for sturgeon, individuals must have access to enough habitat in suitable condition for foraging, resting and spawning. Conditions must be suitable for the successful development of early life stages. Mortality rates must be low enough to allow for recruitment to all age classes so that successful spawning can continue over time and over generations. There must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. For Atlantic sturgeon, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness. Here, we consider whether these proposed actions will affect the NYB DPS likelihood of recovery.

These actions will not change the status or trend of the Hudson or Delaware River population of Atlantic sturgeon or the status and trend of the NYB DPS as a whole. The proposed actions will result in a small amount of mortality (no more than four individuals over a 50 year period) and a subsequent small reduction in future reproductive output. This reduction in numbers will be small and the impact on reproduction and future year classes will also be small enough not to affect the trend of the population. Any effects to habitat will be insignificant and discountable and will not affect the ability of Atlantic sturgeon to carry out any necessary behaviors or functions. Any impacts to available forage will also be insignificant. The proposed projects will result in a small reduction in future reproductive output. For these reasons, it is not expected to affect the persistence of the NYB DPS of Atlantic sturgeon. These actions will not change the status or trend of the NYB DPS of Atlantic sturgeon. The very small reduction in numbers and future reproduction resulting from the proposed projects will not reduce the likelihood of improvement in the status of the NYB DPS of Atlantic sturgeon. The effects of the proposed projects will not delay the recovery timeline or otherwise decrease the likelihood of recovery. The effects of the proposed actions will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed projects will not appreciably reduce the likelihood that the NYB DPS of Atlantic sturgeon can be brought to the point at which they are no longer listed as endangered. Based on the analysis presented herein, the proposed actions, are not likely to appreciably reduce the survival and recovery of this species.

9.2.4 Chesapeake Bay DPS
Individuals originating from the CB DPS are likely to occur in the action area. The CB DPS has been listed as endangered. We expect that 13% of the Atlantic sturgeon in the action area will

originate from the CB DPS. No mortality of adult Atlantic sturgeon is anticipated. We anticipate the mortality of up to three subadult CB DPS Atlantic sturgeon as a result of entrainment in a hopper dredge. While it is possible that entrained fish could survive, we assume here that these fish will be killed.

While Atlantic sturgeon occur in several rivers in the CB DPS, recent spawning has only been documented in the James River. No total population estimates are available for any river population or the DPS as a whole. As discussed in section 4.3, we have estimated a total of 8,811 CB DPS adults and subadults in the ocean (2,203 adults and 6,608 subadults). This estimate is the best available at this time and represents only a percentage of the total CB DPS population as it does not include young of the year or juveniles and does not include all adults and subadults. CB origin Atlantic sturgeon are affected by numerous sources of human induced mortality and habitat disturbance throughout the riverine and marine portions of their range. There is currently not enough information to establish a trend for any life stage or for the DPS as a whole.

The number of subadult CB DPS Atlantic sturgeon that may be killed due to the proposed projects (three over a 50-year period) represents an extremely small percentage of the CB DPS. While the death of three subadult CB DPS Atlantic sturgeon over the next 50 years will reduce the number of CB DPS Atlantic sturgeon compared to the number that would have been present absent the proposed action, it is not likely that this reduction in numbers will change the status of this species Even if there were only 6,608 subadults in the CB DPS, this loss would represent only 0.04% of the subadults in the DPS. The percentage would be much less if we also considered the number of young of the year, juveniles, adults, and other subadults not included in the NEAMAP-based oceanic population estimate.

Because there will be no loss of adults, the reproductive potential of the CB DPS will not be affected in any way other than through a reduction in numbers of individual future spawners as opposed to current spawners. The loss of three female subadults would have the effect of reducing the amount of potential reproduction as any dead CB DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in an extremely small reduction in the number of eggs laid or larvae produced in future years and similarly, an extremely small effect on the strength of subsequent year classes. Even considering the potential future spawners that would be produced by the individuals that would be killed as a result of the proposed action, any effect to future year classes is anticipated to be extremely small and would not change the status of this species. The loss of three male subadults may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in a particular year. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals with the exception of three individual and their progeny. The proposed actions will also not affect the spawning grounds within the rivers where CB DPS fish spawn.

The proposed actions are not likely to reduce distribution because the action will not impede CB

DPS Atlantic sturgeon from accessing any seasonal concentration areas, including foraging, spawning or overwintering grounds. Any effects to distribution will be minor and temporary and limited to the temporary avoidance of the area of increased sediment and increased underwater noise levels.

Based on the information provided above, the death of no more than three subadult CB DPS Atlantic sturgeon over 50 years, will not appreciably reduce the likelihood of survival of the CB DPS (*i.e.*, it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The actions will not affect CB DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is the case because: (1) the death of these subadult CB DPS Atlantic sturgeon represents an extremely small percentage of the species; (2) the death of these CB DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these CB DPS Atlantic sturgeon is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of these subadult CB DPS Atlantic sturgeon is likely to have such a small effect on reproductive output that the loss of these individuals will not change the status or trends of the species; (5) the actions will have only a minor and temporary effect on the distribution of CB DPS Atlantic sturgeon in the action area and no effect on the distribution of the species throughout its range; and, (6) the actions will have only an insignificant effect on individual foraging or sheltering CB DPS Atlantic sturgeon.

In rare instances, an action that does not appreciably reduce the likelihood of a species' survival might appreciably reduce its likelihood of recovery. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the CB DPS of Atlantic sturgeon will survive in the wild. Here, we consider whether the action will appreciably reduce the likelihood of recovery from the perspective of ESA Section 4. As noted above, recovery is defined as the improvement in status such that listing under Section 4(a) as "in danger of extinction throughout all or a significant portion of its range" (endangered) or "likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range…" (threatened) is no longer appropriate. Thus, we have considered whether the proposed actions will appreciably reduce the likelihood that the CB DPS of Atlantic sturgeon can rebuild to a point where it is no longer in danger of extinction through all or a significant part of its range.

No Recovery Plan for the CB DPS has been published. The Recovery Plan will outline the steps necessary for recovery and the demographic criteria which once attained would allow the species to be delisted. We know that in general, to recover, a listed species must have a sustained positive trend of increasing population over time. To allow that to happen for sturgeon, individuals must have access to enough habitat in suitable condition for foraging, resting and spawning. Conditions must be suitable for the successful development of early life stages. Mortality rates must be low enough to allow for recruitment to all age classes so that successful spawning can continue over time and over generations. There must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. For Atlantic sturgeon, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness. Here, we consider whether these proposed actions will affect the CB DPS likelihood of recovery.

These actions will not change the status or trend of the CB DPS as a whole. The proposed actions will result in a small amount of mortality (up to three subadults from a population estimated to have at least 6,000 subadults) and a subsequent small reduction in future reproductive output. This reduction in numbers will be small and the impact on reproduction and future year classes will also be small enough not to affect the trend of the population. The proposed action will have only insignificant effects on habitat and forage. This is because the area that sturgeon may avoid is small and any avoidance will be temporary and limited to the period of time when increased suspended sediment is experienced or increased underwater noise. The proposed actions will not affect CB DPS of Atlantic sturgeon outside of the action area or affect habitats outside of the action area. Therefore, it will not affect estuarine or oceanic habitats that are important for sturgeon. For these reasons, the actions will not reduce the likelihood that the CB DPS can recover. Therefore, the proposed actions will not appreciably reduce the likelihood that the CB DPS of Atlantic sturgeon can be brought to the point at which they are no longer listed as endangered. Based on the analysis presented herein, the proposed actions, are not likely to appreciably reduce the survival and recovery of this species.

9.2.6 South Atlantic DPS

Individuals originating from the SA DPS are likely to occur in the action area. The SA DPS has been listed as endangered. We expect that 22% of the Atlantic sturgeon in the action area will originate from the SA DPS. No mortality of adult Atlantic sturgeon is anticipated. We expect that no more than three subadult SA DPS Atlantic sturgeon will be killed during the hopper dredging at SBOBA. While it is possible that the entrained fish could survive, we assume here that these fish will be killed.

No total population estimates are available for any river population or the SA DPS as a whole. As discussed in section 4.3, NMFS has estimated a total of 14,911 SA DPS adults and subadults in the ocean (3,728 adults and 11,183 subadults). This estimate is the best available at this time and represents only a percentage of the total SA DPS population as it does not include young of the year or juveniles and does not include all adults and subadults. SA origin Atlantic sturgeon are affected by numerous sources of human induced mortality and habitat disturbance throughout the riverine and marine portions of their range. There is currently not enough information to establish a trend for any life stage or for the DPS as a whole.

The number of subadult SA DPS Atlantic sturgeon that may be killed due to the proposed projects (up to three over a 50-year period) represents an extremely small percentage of the SA DPS. While the death of three subadult SA DPS Atlantic sturgeon over the next 50 years will reduce the number of SA DPS Atlantic sturgeon compared to the number that would have been present absent the proposed action, it is not likely that this reduction in numbers will change the status of this species. Even if there were only 11,183 subadults in the SA DPS, this loss would represent 0.02% of the subadults in the DPS. The percentage would be much less if we also considered the number of young of the year, juveniles, adults, and other subadults not included in the NEAMAP-based oceanic population estimate.

Because there will be no loss of adults, the reproductive potential of the SA DPS will not be affected in any way other than through a reduction in numbers of individual future spawners as opposed to current spawners. The loss of three female subadult would have the effect of reducing the amount of potential reproduction as any dead SA DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in an extremely small reduction in the number of eggs laid or larvae produced in future years and similarly, an extremely small effect on the strength of subsequent year classes. Even considering the potential future spawners that would be produced by the individual that would be killed as a result of the proposed action, any effect to future year classes is anticipated to be extremely small and would not change the status of this species. The loss of male subadults may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in a particular year. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals with the exception of three individuals and their progeny. The proposed actions will also not affect the spawning grounds within the rivers where SA DPS fish spawn.

The proposed actions are not likely to reduce distribution because the actions will not impede SA DPS Atlantic sturgeon from accessing any seasonal concentration areas, including foraging, spawning or overwintering grounds. Any effects to distribution will be minor and temporary and limited to the temporary avoidance of the area of increased sediment around the working dredge.

Based on the information provided above, the death of three subadult SA DPS Atlantic sturgeon over 50 years, will not appreciably reduce the likelihood of survival of the SA DPS (*i.e.*, it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The actions will not affect SA DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is the case because: (1) the death of these subadult SA DPS Atlantic sturgeon represents an extremely small percentage of the species; (2) the death of these SA DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these SA DPS Atlantic sturgeon is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of these subadult SA DPS Atlantic sturgeon is likely to have such a small effect on reproductive output that the loss of these individuals will not change the status or trends of the species; (5) the actions will have only a minor and temporary effect on the distribution of SA DPS Atlantic sturgeon in the action area and no effect on the distribution of the species throughout its range; and, (6) the actions will have only an insignificant effect on individual foraging or sheltering SA DPS Atlantic sturgeon.

In rare instances, an action that does not appreciably reduce the likelihood of a species' survival might appreciably reduce its likelihood of recovery. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the SA DPS of Atlantic sturgeon will survive in the wild. Here, we consider whether the actions will appreciably reduce the likelihood of recovery from the perspective of ESA Section 4. As noted above, recovery is defined as the improvement in status such that listing under Section 4(a) as "in danger of extinction throughout all or a significant portion of its range" (endangered) or "likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range…" (threatened) is no longer appropriate. Thus, we have considered whether the proposed actions will appreciably reduce the likelihood that SA DPS of Atlantic sturgeon can rebuild to a point where it is no longer in danger of extinction through all or a significant part of its range.

No Recovery Plan for the SA DPS has been published. The Recovery Plan will outline the steps necessary for recovery and the demographic criteria which once attained would allow the species to be delisted. We know that in general, to recover, a listed species must have a sustained positive trend of increasing population over time. To allow that to happen for sturgeon, individuals must have access to enough habitat in suitable condition for foraging, resting and spawning. Conditions must be suitable for the successful development of early life stages. Mortality rates must be low enough to allow for recruitment to all age classes so that successful spawning can continue over time and over generations. There must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. For Atlantic sturgeon, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness. Here, we consider whether these proposed actions will affect the SA DPS likelihood of recovery.

This action will not change the status or trend of the SA DPS as a whole. The proposed actions will result in a small amount of mortality (up to three subadults from a population estimated to have at least 11,000 subadults) and a subsequent small reduction in future reproductive output. This reduction in numbers will be small and the impact on reproduction and future year classes will also be small enough not to affect the trend of the population. The proposed actions will have only insignificant effects on habitat and forage. This is because the area that sturgeon may avoid is small and any avoidance will be temporary and limited to the period of time when increased suspended sediment is experienced or increased underwater noise. The proposed actions will not affect SA DPS of Atlantic sturgeon outside of the action area or affect habitats outside of the action area. Therefore, it will not affect estuarine or oceanic habitats that are important for sturgeon. For these reasons, the action will not reduce the likelihood that the SA DPS can recover. Therefore, the proposed action will not appreciably reduce the likelihood that the SA DPS of Atlantic sturgeon can be brought to the point at which they are no longer listed as endangered. Based on the analysis presented herein, the proposed actions, are not likely to appreciably reduce the survival and recovery of this species.

9.2.5 Carolina DPS

Individuals originating from the CA DPS are likely to occur in the action area. The CA DPS has been listed as endangered. We expect that 2% of the Atlantic sturgeon in the action area will originate from the CA DPS. No mortality of adult Atlantic sturgeon is anticipated. We expect that no more than three subadult CA DPS Atlantic sturgeon will be killed during the hopper dredging at SBOBA. While it is possible that the entrained fish could survive, we assume here that these fish will be killed.

No total population estimates are available for any river population or the CA DPS as a whole. As discussed in section 4.3, NMFS has estimated a total of 1,356 CA DPS adults and subadults in the ocean (339 adults and 1,017 subadults). This estimate is the best available at this time and represents only a percentage of the total CA DPS population as it does not include young of the year or juveniles and does not include all adults and subadults. CA origin Atlantic sturgeon are affected by numerous sources of human induced mortality and habitat disturbance throughout the riverine and marine portions of their range. There is currently not enough information to establish a trend for any life stage or for the DPS as a whole.

The number of subadult CA DPS Atlantic sturgeon that may be killed due to the proposed projects (up to three over a 50-year period) represents an extremely small percentage of the CA DPS. While the death of three subadult CA DPS Atlantic sturgeon over the next 50 years will reduce the number of CA DPS Atlantic sturgeon compared to the number that would have been present absent the proposed action, it is not likely that this reduction in numbers will change the status of this species. Even if there were only 1,017 subadults in the CA DPS, this loss would represent 0.02% of the subadults in the DPS. The percentage would be much less if we also considered the number of young of the year, juveniles, adults, and other subadults not included in the NEAMAP-based oceanic population estimate.

Because there will be no loss of adults, the reproductive potential of the CA DPS will not be affected in any way other than through a reduction in numbers of individual future spawners as opposed to current spawners. The loss of three female subadult would have the effect of reducing the amount of potential reproduction as any dead CA DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in an extremely small reduction in the number of eggs laid or larvae produced in future years and similarly, an extremely small effect on the strength of subsequent year classes. Even considering the potential future spawners that would be produced by the individual that would be killed as a result of the proposed actions, any effect to future year classes is anticipated to be extremely small and would not change the status of this species. The loss of male subadults may have less of an impact on future reproduction as other males are expected to be available to fertilize eggs in a particular year. Additionally, we have determined that any impacts to behavior will be minor and temporary and that there will not be any delay or disruption of any normal behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals with the exception of three individuals and their progeny. The proposed actions will also not affect the spawning grounds within the rivers where CA DPS fish spawn.

The proposed actions are not likely to reduce distribution because the action will not impede CA DPS Atlantic sturgeon from accessing any seasonal concentration areas, including foraging, spawning or overwintering grounds. Any effects to distribution will be minor and temporary and limited to the temporary avoidance of the area of increased sediment is experienced or increased underwater noise.

Based on the information provided above, the death of three subadult CA DPS Atlantic sturgeon over 50 years, will not appreciably reduce the likelihood of survival of the CA DPS (*i.e.*, it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The actions will not affect CA DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle or completing essential behaviors including reproducing, foraging and sheltering. This is the case because: (1) the death of these subadult CA DPS Atlantic sturgeon represents an extremely small percentage of the species; (2) the death of these CA DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these CA DPS Atlantic sturgeon is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of these subadult CA DPS Atlantic sturgeon is likely to have such a small effect on reproductive output that the loss of these individuals will not change the status or trends of the species; (5) the actions will have only a minor and temporary effect on the distribution of CA DPS Atlantic sturgeon in the action area and no effect on the distribution of the species throughout its range; and, (6) the actions will have only an insignificant effect on individual foraging or sheltering CA DPS Atlantic sturgeon.

In rare instances, an action that does not appreciably reduce the likelihood of a species' survival might appreciably reduce its likelihood of recovery. As explained above, we have determined that the proposed actions will not appreciably reduce the likelihood that the CA DPS of Atlantic sturgeon will survive in the wild. Here, we consider whether the actions will appreciably reduce the likelihood of recovery from the perspective of ESA Section 4. As noted above, recovery is defined as the improvement in status such that listing under Section 4(a) as "in danger of extinction throughout all or a significant portion of its range" (endangered) or "likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range…" (threatened) is no longer appropriate. Thus, we have considered whether the proposed actions will appreciably reduce the likelihood that CA DPS of Atlantic sturgeon can rebuild to a point where it is no longer in danger of extinction through all or a significant part of its range.

No Recovery Plan for the CA DPS has been published. The Recovery Plan will outline the steps necessary for recovery and the demographic criteria which once attained would allow the species to be delisted. We know that in general, to recover, a listed species must have a sustained positive trend of increasing population over time. To allow that to happen for sturgeon, individuals must have access to enough habitat in suitable condition for foraging, resting and spawning. Conditions must be suitable for the successful development of early life stages. Mortality rates must be low enough to allow for recruitment to all age classes so that successful spawning can continue over time and over generations. There must be enough suitable habitat for spawning, foraging, resting and migrations of all individuals. For Atlantic sturgeon, habitat conditions must be suitable both in the natal river and in other rivers and estuaries where foraging by subadults and adults will occur and in the ocean where subadults and adults migrate, overwinter and forage. Habitat connectivity must also be maintained so that individuals can migrate between important habitats without delays that impact their fitness. Here, we consider whether this proposed actions will affect the CA DPS likelihood of recovery.

These actions will not change the status or trend of the CA DPS as a whole. The proposed actions will result in a small amount of mortality (up to three subadults from a population estimated to have at least 1,017 subadults) and a subsequent small reduction in future reproductive output. This reduction in numbers will be small and the impact on reproduction and future year classes will also be small enough not to affect the trend of the population. The proposed actions will have only insignificant effects on habitat and forage. This is because the area that sturgeon may avoid is small and any avoidance will be temporary and limited to the period of time when increased suspended sediment is experienced or increased underwater noise. The proposed actions will not affect CA DPS of Atlantic sturgeon outside of the action area or affect habitats outside of the action area. Therefore, it will not affect estuarine or oceanic habitats that are important for sturgeon. For these reasons, the action will not reduce the likelihood that the CA DPS can recover. Therefore, the proposed actions will not appreciably reduce the likelihood that the CA DPS of Atlantic sturgeon can be brought to the point at which they are no longer listed as endangered. Based on the analysis presented herein, the proposed actions, are not likely to appreciably reduce the survival and recovery of this species.

9.2 Kemp's Ridley Sea Turtle

In the "Effects of the Action" section above, we determined that Kemp's ridleys could be entrained in a hopper dredge working in the SBOBA. Based on a calculated entrainment rate of sea turtles for projects using hopper dredges in areas comparable to the SBOBA, we estimate that 1 sea turtle is likely to be entrained for every 2.6 million cy of material removed with a hopper dredge. Also, based on the ratio of loggerhead and Kemp's ridleys entrained in other hopper dredge operations in the USACE North Atlantic Division, we estimate that no more than 10% of the sea turtles entrained during each project operation were likely to be Kemp's ridleys with the remainder loggerheads. As it is possible that each project may take a Kemp's ridley, we determined that one Kemp's ridley may be entrained by each dredge project, resulting in up to 3 Kemp's ridley entrainments due to the proposed actions.

Kemp's Ridley sea turtles are listed as a single species classified as "endangered" under the ESA. Kemp's ridleys occur in the Atlantic Ocean and Gulf of Mexico. The only major nesting site for Kemp's ridleys is a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; USFWS and NMFS 1992; NMFS and USFWS 2007c).

Nest count data provides the best available information on the number of adult females nesting each year. As is the case with the other sea turtle species discussed above, nest count data must be interpreted with caution given that these estimates provide a minimum count of the number of nesting Kemp's ridley sea turtles. In addition, the estimates do not account for adult males or juveniles of either sex. Without information on the proportion of adult males to females, and the age structure of the Kemp's ridley population, nest counts cannot be used to estimate the total population size (Meylan 1982; Ross 1996; Zurita *et al.* 2003; Hawkes *et al.* 2005; letter to J. Lecky, NMFS Office of Protected Resources, from N. Thompson, NMFS Northeast Fisheries Science Center, December 4, 2007). Nevertheless, the nesting data does provide valuable information on the extent of Kemp's ridley nesting and the trend in the number of nests laid. Estimates of the adult female nesting population reached a low of approximately 250-300 in 1985 (USFWS and NMFS 1992; TEWG 2000). From 1985 to 1999, the number of nests observed at Rancho Nuevo and nearby beaches increased at a mean rate of 11.3% per year (TEWG 2000). Current estimates suggest an adult female population of 7,000-8,000 Kemp's ridleys (NMFS and USFWS 2007c).

The most recent review of the Kemp's ridleys suggests that this species is in the early stages of recovery (NMFS and USFWS 2007b). Nest count data indicate increased nesting and increased numbers of nesting females in the population. NMFS also takes into account a number of recent conservation actions including the protection of females, nests, and hatchlings on nesting beaches since the 1960s and the enhancement of survival in marine habitats through the implementation of TEDs in the early 1990s and a decrease in the amount of shrimping off the coast of Tamaulipas and in the Gulf of Mexico in general (NMFS and USFWS 2007b). We expect this increasing trend to continue over the time period considered in this Opinion.

The mortality of up to three Kemp's ridley sea turtles over a 50 year time period represents a very small percentage of the Kemp's ridleys worldwide. Even taking into account just nesting females, the death of three Kemp's ridleys represents less than 0.01% of the population. While the death of a Kemp's ridley will reduce the number of Kemp's ridleys compared to the number that would have been present absent the proposed actions, it is not likely that this reduction in numbers will change the status of this species or its stable to increasing trend as this loss represents a very small percentage of the population (less than 0.01%). Reproductive potential of Kemp's ridleys is not expected to be affected in any other way other than through a reduction in numbers of individuals. A reduction in the number of Kemp's ridleys would have the effect of reducing the amount of potential reproduction as any dead Kemp's ridleys would have no potential for future reproduction. In 2006, the most recent year for which data is available, there were an estimated 7-8,000 nesting females. While the species is thought to be female biased, there are likely to be several thousand adult males as well. Given the number of nesting adults, it is unlikely that the loss of three Kemp's ridleys would affect the success of nesting in any year. Additionally, this small reduction in potential nesters is expected to result in a small reduction in the number of eggs laid or hatchlings produced in future years and similarly, a very small effect on the strength of subsequent year classes. Even considering the potential future nesters that would be produced by the individuals that would be killed as a result of the proposed actions, any effect to future year classes is anticipated to be very small and would not change the stable to increasing trend of this species. Additionally, the proposed actions will not affect nesting beaches in any way or disrupt migratory movements in a way that hinders access to nesting beaches or otherwise delays nesting.

The proposed actions are not likely to reduce distribution because the actions will not impede Kemp's ridleys from accessing foraging grounds or cause more than a temporary disruption to other migratory behaviors. Additionally, given the small percentage of the species that will be killed as a result of the dredging, there is not likely to be any loss of unique genetic haplotypes and no loss of genetic diversity.

While generally speaking, the loss of a small number of individuals from a subpopulation or species may have an appreciable reduction on the numbers, reproduction and distribution of the species, this is likely to occur only when there are very few individuals in a population, the individuals occur in a very limited geographic range or the species has extremely low levels of genetic diversity. This situation is not likely in the case of Kemp's ridleys because: the species is widely geographically distributed, it is not known to have low levels of genetic diversity, there are several thousand individuals in the population and the number of Kemp's ridleys is likely to be increasing and, at worst, is stable.

Based on the information provided above, the death of three Kemp's ridley sea turtles between now and 2064 will not appreciably reduce the likelihood of survival (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect Kemp's ridleys in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring and it will not result in effects to the environment which would prevent Kemp's ridleys from completing their entire life cycle, including reproduction, sustenance, and shelter. This is the case because: (1) the species' nesting trend is increasing; (2) the death of three Kemp's ridleys represents an extremely small percentage of the species as a whole; (3) the death of three Kemp's ridleys will not change the status or trends of the species as a whole; (4) the loss of these Kemp's ridleys is not likely to have an effect on the levels of genetic heterogeneity in the population; (5) the loss of these Kemp's ridleys is likely to have such a small effect on reproductive output that the loss of these individuals will not change the status or trends of the species; (6) the actions will have only a minor and temporary effect on the distribution of Kemp's ridleys in the action area and no effect on the distribution of the species throughout its range; and, (7) the actions will have no effect on the ability of Kemp's ridleys to shelter and only an insignificant effect on individual foraging Kemp's ridleys.

In rare instances, an action may not appreciably reduce the likelihood of a species survival (persistence) but may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed actions will not appreciably reduce the likelihood that Kemp's ridley sea turtles will survive in the wild. Here, we consider the potential for the actions to reduce the likelihood of recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed actions will affect the likelihood that Kemp's ridleys can rebuild to a point where listing is no longer appropriate. In 2011, NMFS and the USFWS issued a recovery plan for Kemp's ridleys (NMFS and USFWS 2011). The plan includes a list of criteria necessary for recovery. These include:

1. An increase in the population size, specifically in relation to nesting females[12];

[12] A population of at least 10,000 nesting females in a season (as measured by clutch frequency per female per

2. An increase in the recruitment of hatchlings[13];
3. An increase in the number of nests at the nesting beaches;
4. Preservation and maintenance of nesting beaches (i.e. Rancho Nuevo, Tepehuajes, and Playa Dos); and,
5. Maintenance of sufficient foraging, migratory, and inter-nesting habitat.

Kemp's ridleys have an increasing trend; as explained above, the loss of three Kemp's ridleys during the duration of the proposed actions (50 years) will not affect the population trend. The number of Kemp's ridleys likely to die as a result of the proposed actions is an extremely small percentage of the species. This loss will not affect the likelihood that the population will reach the size necessary for recovery or the rate at which recovery will occur. As such, the proposed actions will not affect the likelihood that criteria one, two or three will be achieved or the timeline on which they will be achieved. The action area does not include nesting beaches; therefore, the proposed actions will have no effect on the likelihood that recovery criteria four will be met. All effects to habitat will be insignificant and discountable; therefore, the proposed actions will have no effect on the likelihood that criteria five will be met.

The effects of the proposed actions will not hasten the extinction timeline or otherwise increase the danger of extinction. Further, the actions will not prevent the species from growing in a way that leads to recovery and the actions will not change the rate at which recovery can occur. This is the case because while the actions may result in a small reduction in the number of Kemp's ridleys and a small reduction in the amount of potential reproduction due to the loss of three individuals, the actions are not expected to have long term impacts on the future growth of the population or its potential for recovery. Therefore, based on the analysis presented above, the proposed actions will not appreciably reduce the likelihood that Kemp's ridley sea turtles can be brought to the point at which they are no longer listed as endangered or threatened.

Despite the threats faced by individual Kemp's ridley sea turtles inside and outside of the action area, the proposed actions will not increase the vulnerability of individual sea turtles to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the proposed actions. We have considered the effects of the proposed actions in light of cumulative effects explained above, including climate change, and have concluded that even in light of the ongoing impacts of these activities and conditions; the conclusions reached above do not change. Based on the analysis presented herein, the proposed actions, resulting in the mortality of up to three Kemp's ridley sea turtles between now and 2064, is not likely to appreciably reduce the survival and recovery of this species.

9.3 Northwest Atlantic DPS of Loggerhead Sea Turtles

In the "Effects of the Action" section above, we determined that loggerheads could be entrained in a hopper dredge working in the SBOBA. Based on a calculated entrainment rate of sea turtles

---

season) distributed at the primary nesting beaches in Mexico (Rancho Nuevo, Tepehuajes, and Playa Dos) is attained in order for downlisting to occur; an average of 40,000 nesting females per season over a 6-year period by 2024 for delisting to occur.

[13] Recruitment of at least 300,000 hatchlings to the marine environment per season at the three primary nesting beaches in Mexico (Rancho Nuevo, Tepehuajes, and Playa Dos).

for projects using hopper dredges in areas comparable to the SBOBA, we estimate that 1 sea turtle is likely to be entrained for every 2.6 million cy of material removed with a hopper dredge. Also, based on the ratio of loggerhead and Kemp's ridleys entrained in other hopper dredge operations in the USACE North Atlantic Division, we estimate that 90% of the sea turtles entrained during project operations were likely to be loggerheads. Based on this, we determined that of the eight sea turtles likely to be entrained during lifetime of the projects (through 2064), all eight may be loggerheads. All entrained loggerheads are expected to be juveniles. We determined that all other effects of the actions on this species will be insignificant and discountable.

The Northwest Atlantic DPS of loggerhead sea turtles is listed as "threatened" under the ESA. It takes decades for loggerhead sea turtles to reach maturity. Once they have reached maturity, females typically lay multiple clutches of eggs within a season, but do not typically lay eggs every season (NMFS and USFWS 2008). There are many natural and anthropogenic factors affecting the survival of loggerheads prior to their reaching maturity as well as for those adults who have reached maturity. As described in the Status of the Species, Environmental Baseline and Cumulative Effects sections above, loggerhead sea turtles in the action area continue to be affected by multiple anthropogenic impacts including bycatch in commercial and recreational fisheries, habitat alteration, dredging, power plant intakes and other factors that result in mortality of individuals at all life stages. Negative impacts causing death of various age classes occur both on land and in the water. Many actions have been taken to address known negative impacts to loggerhead sea turtles. However, many remain unaddressed, have not been sufficiently addressed, or have been addressed in some manner but whose success cannot be quantified.

The SEFSC (2009) estimated the number of adult females in the NWA DPS at 30,000, and if a 1:1 adult sex ratio is assumed, the result is 60,000 adults in this DPS. Based on the reviews of nesting data, as well as information on population abundance and trends, NMFS and USFWS determined in the September 2011 listing rule that the NWA DPS should be listed as threatened. They found that an endangered status for the NWA DPS was not warranted given the large size of the nesting population, the overall nesting population remains widespread, the trend for the nesting population appears to be stabilizing, and substantial conservation efforts are underway to address threats. This stable trend is expected to continue over the time period considered in this Opinion.

As stated above, we expect the lethal entrainment of up to eight loggerheads over the 50 year time period. The lethal removal of up to eight loggerhead sea turtles from the action area over this time period would be expected to reduce the number of loggerhead sea turtles from the recovery unit of which they originated as compared to the number of loggerheads that would have been present in the absence of the proposed actions (assuming all other variables remained the same). However, this does not necessarily mean that these recovery units will experience reductions in reproduction, numbers or distribution in response to these effects to the extent that survival and recovery would be appreciably reduced. The final revised recovery plan for loggerheads compiled the most recent 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 with approximately 1,272 females nesting per year; (2) for the PFRU, a mean of 64,513 nests per year with approximately 15,735 females nesting per year; (3) for the DTRU, a mean of 246 nests per year with approximately 60 females nesting per year; and (4) for the NGMRU, a mean of 906 nests per year with approximately 221 females nesting per year. For the GCRU, the only estimate available for the number of loggerhead nests per year is from Quintana Roo, Yucatán, Mexico, where a range of 903-2,331 nests per year was estimated from 1987-2001 (NMFS and USFWS 2007a). There are no annual nest estimates available for the Yucatán since 2001 or for any other regions in the GCRU, nor are there any estimates of the number of nesting females per year for any nesting assemblage in this recovery unit.

It is likely that the loggerhead sea turtles in the action area originate from several of the recovery units. Limited information is available on the genetic makeup of sea turtles in the mid-Atlantic, where the majority of sea turtle interactions are expected to occur. Cohorts from each of the five western Atlantic subpopulations are expected to occur in the action area. Genetic analysis of samples collected from immature loggerhead sea turtles captured in pound nets in the Pamlico-Albemarle Estuarine Complex in North Carolina from September-December of 1995-1997 indicated that cohorts from all five western Atlantic subpopulations were present (Bass *et al.* 2004). In a separate study, genetic analysis of samples collected from loggerhead sea turtles from Massachusetts to Florida found that all five western Atlantic loggerhead subpopulations were represented (Bowen *et al.* 2004). Bass *et al.* (2004) found that 80 percent of the juveniles and sub-adults utilizing the foraging habitat originated from the south Florida nesting population, 12 percent from the northern subpopulation, 6 percent from the Yucatan subpopulation, and 2 percent from other rookeries. The previously defined loggerhead subpopulations do not share the exact delineations of the recovery units identified in the 2008 recovery plan. However, the PFRU encompasses both the south Florida and Florida panhandle subpopulations, the NRU is roughly equivalent to the northern nesting group, the Dry Tortugas subpopulation is equivalent to the DTRU, and the Yucatan subpopulation is included in the GCRU.

Based on the genetic analysis presented in Bass *et al.* (2004) and the small number of loggerheads from the DTRU or the NGMRU likely to occur in the action area it is extremely unlikely that the loggerheads likely to be killed during dredging projects will originate from either of these recovery units. The majority, at least 80% of the loggerheads killed, are likely to have originated from the PFRU, with the remainder from the NRU and GCRU. As such, of the eight loggerheads likely to be killed, seven are expected to be from the PFRU, with the other one from the NRU or from the GCRU. Below, we consider the effects of these mortalities on these three recovery units and the species as a whole.

As noted above, the most recent population estimates indicate that there are approximately 15,735 females nesting annually in the PFRU and approximately 1,272 females nesting per year in the NRU. For the GCRU, the only estimate available for the number of loggerhead nests per year is from Quintana Roo, Yucatán, Mexico, where a range of 903-2,331 nests per year was estimated from 1987-2001 (NMFS and USFWS 2007a). There are no annual nest estimates available for the Yucatán since 2001 or for any other regions in the GCRU, nor are there any estimates of the number of nesting females per year for any nesting assemblage in this recovery unit; however, the 2008 recovery plan indicates that the Yucatan nesting aggregation has at least

1,000 nesting females annually. As the numbers outlined here are only for nesting females, the total number of loggerhead sea turtles in each recovery unit is likely significantly higher.

The loss of eight loggerheads over a 50 year period represents an extremely small percentage of the number of sea turtles in the PFRU. Even if the total population was limited to 15,735 loggerheads, the loss of seven individuals would represent approximately 0.04% of the population. Similarly, the loss of one loggerhead from the NRU represents an extremely small percentage of the recovery unit. Even if the total population was limited to 1,272 sea turtles, the loss of one individual would represent approximately 0.1% of the population. The loss of one loggerhead from the GCRU, which is expected to support at least 1,000 nesting females, represents less than 0.1% of the population. The loss of such a small percentage of the individuals from any of these recovery units represents an even smaller percentage of the species as a whole. The impact of these losses is even less when considering that these losses will occur over a span of 50 years. Considering the extremely small percentage of the populations that will be killed, it is unlikely that these deaths will have a detectable effect on the numbers and population trends of loggerheads in these recovery units or the number of loggerheads in the population as a whole.

All of the loggerheads that are expected to be killed will be juveniles. Thus, any effects on reproduction are limited to the loss of these individuals on their year class and the loss of future reproductive potential. Given the number of nesting adults in each of these populations, it is unlikely that the expected loss of loggerheads would affect the success of nesting in any year. Additionally, this small reduction in potential nesters is expected to result in a small reduction in the number of eggs laid or hatchlings produced in future years and similarly, a very small effect on the strength of subsequent year classes. Even considering the potential future nesters that would be produced by the individuals that would be killed as a result of the proposed actions, any effect to future year classes is anticipated to be very small and would not change the stable trend of this species. Additionally, the proposed actions will not affect nesting beaches in any way or disrupt migratory movements in a way that hinders access to nesting beaches or otherwise delays nesting.

The proposed actions are not likely to reduce distribution because the actions will not impede loggerheads from accessing foraging grounds or cause more than a temporary disruption to other migratory behaviors. Additionally, given the small percentage of the species that will be killed as a result of the dredging, there is not likely to be any loss of unique genetic haplotypes and no loss of genetic diversity.

While generally speaking, the loss of a small number of individuals from a subpopulation or species may have an appreciable reduction on the numbers, reproduction and distribution of the species this is likely to occur only when there are very few individuals in a population, the individuals occur in a very limited geographic range or the species has extremely low levels of genetic diversity. This situation is not likely in the case of loggerheads because: the species is widely geographically distributed, it is not known to have low levels of genetic diversity, there are several thousand individuals in the population and the number of loggerheads is likely to be stable or increasing over the time period considered here.

Based on the information provided above, the death of up to eight loggerheads between now and 2064 will not appreciably reduce the likelihood of survival (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The actions will not affect loggerheads in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring and it will not result in effects to the environment which would prevent loggerheads from completing their entire life cycle, including reproduction, sustenance, and shelter. This is the case because: (1) the species' nesting trend is stabilizing; (2) the death of eight loggerheads represents an extremely small percentage of the species as a whole; (3) the death of eight loggerheads will not change the status or trends of the species as a whole; (4) the loss of these loggerheads is not likely to have an effect on the levels of genetic heterogeneity in the population; (5) the loss of these loggerheads is likely to have such a small effect on reproductive output that the loss of these individuals will not change the status or trends of the species; (5) the actions will have only a minor and temporary effect on the distribution of loggerheads in the action area and no effect on the distribution of the species throughout its range; and, (6) the actions will have no effect on the ability of loggerheads to shelter and only an insignificant effect on individual foraging loggerheads.

In rare instances, an action may not appreciably reduce the likelihood of a species survival (persistence) but may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed actions will not appreciably reduce the likelihood that loggerhead sea turtles will survive in the wild. Here, we consider the potential for the actions to reduce the likelihood of recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Thus, we have considered whether the proposed actions will affect the likelihood that the NWA DPS of loggerheads can rebuild to a point where listing is no longer appropriate. In 2008, NMFS and the USFWS issued a recovery plan for the Northwest Atlantic population of loggerheads (NMFS and USFWS 2008). The plan includes demographic recovery criteria as well as a list of tasks that must be accomplished. Demographic recovery criteria are included for each of the five recovery units. These criteria focus on sustained increases in the number of nests laid and the number of nesting females in each recovery unit, an increase in abundance on foraging grounds, and ensuring that trends in neritic strandings are not increasing at a rate greater than trends in in-water abundance. The recovery tasks focus on protecting habitats, minimizing and managing predation and disease, and minimizing anthropogenic mortalities.

Loggerheads have an increasing trend; as explained above, the loss of eight loggerheads over 50-years as a result of the proposed actions will not affect the population trend. The number of loggerheads likely to die as a result of the proposed actions is an extremely small percentage of any recovery unit or the DPS as a whole. This loss will not affect the likelihood that the population will reach the size necessary for recovery or the rate at which recovery will occur. As such, the proposed actions will not affect the likelihood that the demographic criteria will be achieved or the timeline on which they will be achieved. The action area does not include nesting beaches; all effects to habitat will be insignificant and discountable; therefore, the proposed actions will have no effect on the likelihood that habitat based recovery criteria will be achieved. The proposed actions will also not affect the ability of any of the recovery tasks to be accomplished.

In summary, the effects of the proposed actions will not hasten the extinction timeline or otherwise increase the danger of extinction; further, the actions will not prevent the species from growing in a way that leads to recovery and the actions will not change the rate at which recovery can occur.

This is the case because while the actions may result in a small reduction in the number of loggerheads and a small reduction in the amount of potential reproduction due to the loss of these individuals, the actions are not expected to have long term impacts on the future growth of the population or its potential for recovery. Therefore, based on the analysis presented above, the proposed actions will not appreciably reduce the likelihood that loggerhead sea turtles can be brought to the point at which they are no longer listed as endangered or threatened.

Despite the threats faced by individual loggerhead sea turtles inside and outside of the action area, the proposed actions will not increase the vulnerability of individual sea turtles to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the proposed actions. We have considered the effects of the proposed actions in light of other threats, including climate change, and have concluded that even in light of the ongoing impacts of these activities and conditions, the conclusions reached above do not change. Based on the analysis presented herein, the proposed actions are not likely to appreciably reduce the survival and recovery of the NWA DPS of loggerhead sea turtles.

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 action, and the cumulative effects, it is NMFS' biological opinion that the proposed actions may adversely affect but are not likely to jeopardize the continued existence of any DPS of Atlantic sturgeon, Kemp's ridley and loggerhead sea turtles and is not likely to adversely affect leatherback or green sea turtles or right, humpback or fin whales. Because no critical habitat is designated in the action area, none will be affected by the action.

11.0 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA prohibits the take of endangered species of fish and wildlife. "Fish and wildlife" is defined in the ESA "as any member of the animal kingdom, including without limitation any mammal, fish, bird (including any migratory, non-migratory, or endangered bird for which protection is also afforded by treaty or other international agreement), amphibian, reptile, mollusk, crustacean, arthropod or other invertebrate, and includes any part, product, egg, or offspring thereof, or the dead body or parts thereof." 16 U.S.C. § 1532(8). "Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by NMFS to include any act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation that actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns including breeding, spawning, rearing, migrating, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. "Otherwise lawful activities" are those actions that meet all State and Federal legal requirements except for the prohibition against taking in ESA Section 9 (51 FR 19936, June 3, 1986). 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). See also 16 U.S.C. § 1532(13)(definition of "person"). Under the terms of ESA section 7(b)(4) and section 7(o)(2), taking that is incidental to, and 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 (ITS). In issuing ITSs, NMFS takes no position on whether an action is an "otherwise lawful activity."

The measures described below are non-discretionary, and must be undertaken by USACE so that they become binding conditions for the exemption in section 7(o)(2) to apply. USACE has a continuing duty to regulate the activity covered by this Incidental Take Statement. If USACE (1) fails to assume and implement the terms and conditions or (2) fails to require any contractors to adhere to the terms and conditions of the Incidental Take Statement through enforceable terms that are added to contracts or other documents as appropriate, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, USACE must report the progress of the action and its 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).

Amount or Extent of Take

The proposed actions have the potential to result in the mortality of loggerhead and Kemp's ridley sea turtles and individuals from the New York Bight, Gulf of Maine, Chesapeake Bay, Carolina and South Atlantic DPSs of Atlantic sturgeon due to entrainment in hopper dredges. These interactions are likely to cause injury and/or mortality to the affected sea turtles and sturgeon. This level of take is expected to occur over the entire 50 year period and is not likely to jeopardize the continued existence of listed species. While we have completed one Biological Opinion, the actions considered here consist of three independent actions carried out by the USACE and their contractors. As such, we have organized the ITS for dredging by project. This ITS exempts the following take:

Port Monmouth
Lethal or non-lethal take of up to 1 loggerhead or Kemp's ridley sea turtle
Lethal or non-lethal take of up to 1 Atlantic sturgeon from the NYB, CB, GOM, CA or SA DPS

Union Beach
Lethal or non-lethal take of up to 1 loggerhead or Kemp's ridley sea turtle
Lethal or non-lethal take of up to 1 Atlantic sturgeon from the NYB, CB, GOM, CA or SA DPS

Elberon to Loch Arbour
Lethal or non-lethal take of up to 6 sea turtles

- ***5 loggerhead sea turtles***
- ***1 loggerhead or Kemp's ridley sea turtle***

Lethal or non-lethal take of up to 3 Atlantic sturgeon

- ***2 from the NYB***
- ***1 from the CB, GOM, CA or SA DPS***

While collecting decomposed animals or parts there of in federal operations is considered to be a take, based on the definition of "take" in Section 3 of the ESA and "wildlife" at 50CFR§222.102, NMFS recognizes that decomposed sea turtles or Atlantic sturgeon may be taken in dredging operations that may not necessarily be related to the dredging activity itself. Theoretically, if dredging operations are conducted properly, no takes of sea turtles or Atlantic sturgeon should occur as the turtle draghead defector should push the turtles and Atlantic sturgeon to the side and the suction pumps should be turned off whenever the dredge draghead is away from the substrate. However, due to certain environmental conditions (e.g., rocky bottom, uneven substrate), the dredge draghead may periodically lift off the bottom and entrain, through the high level of suction, previously dead sea turtle or Atlantic sturgeon parts (as well as live turtles or Atlantic sturgeon) that may be on the bottom.

Thus, the aforementioned anticipated level of take refers to those turtles or sturgeon which NMFS confirms as freshly dead. While this definition is subject to some interpretation by the observer, a fresh dead animal may exhibit the following characteristics: little to no odor; fresh blood present; fresh (not necrotic, pink/healthy color) tissue, muscle, or skin; no bloating; color consistent with live animal; and live barnacles. A previously (non-fresh) dead animal may exhibit the following characteristics: foul odor; necrotic, dark or decaying tissues; sloughing of scutes; pooling of old blood; atypical coloration; and opaque eyes. NMFS recognizes that decomposed sea turtles or Atlantic sturgeon may be taken in dredging operations that may not necessarily be related to the dredging activity itself. NMFS expects that dredging may take an additional unquantifiable number of previously dead sea turtle or Atlantic sturgeon parts.

NMFS believes this level of incidental take is reasonable given the seasonal distribution and abundance of these species in the action area and the historic level of take recorded during other dredging operations in the USACE NAD. In the accompanying Opinion, NMFS determined that this level of anticipated take is not likely to result in jeopardy to loggerhead or Kemp's ridley sea turtles or to any DPS of Atlantic sturgeon.

Measures have been undertaken by the USACE to reduce the takes of sea turtles in dredging activities; however, no measures have been undertaken to date for Atlantic sturgeon as the species wasn't listed until April 6, 2012. Measures developed to reduce the take of sea turtles that have been successful in other dredging operations included reevaluating all dredging procedures to assure that the operation of the dragheads and turtle deflectors were in accordance with the project specifications; modifying dredging operations per the recommendation of Mr. Glynn Banks of the USACE Engineering Research and Development Center; training the dredge crew and all inspectors in proper operation of the dragpipe and turtle deflector systems; and, initiating sea turtle relocation trawling. Proper use of draghead deflectors prevent a substantial number of sea turtles from being entrained and killed in dredging operations. Tests conducted by the USACE's Jacksonville District using fake turtles and draghead deflectors showed convincingly that the sea turtle deflecting draghead is useful in reducing entrainments. Based on a discussion with Dana Dickerson and Jenine Gallo of the USACE on January 30, 2014, it was concluded that two new measures be put in place for dredges operating with UXO screens to possibly reduce the take of Atlantic sturgeon and sea turtles. These measures will require the use of a checklist for proper deployment of UXO screens and will require a field validation to ensure the UXO screens are properly in place. As the use of draghead deflectors and other modifications to hopper dredge operations have been demonstrated to be effective at minimizing the number of sea turtles taken in dredging operations and we expect the UXO screen measures may also be effective, NMFS has determined that the use of draghead deflectors, the UXO measures, and certain operating guidelines (as outlined below) are necessary and appropriate to minimize the take of sea turtles and Atlantic sturgeon during the dredging of the SBOBA. In addition to these measures, NMFS has determined that the following reasonable and prudent measures are necessary and appropriate to minimize impacts of incidental take of sea turtles or Atlantic sturgeon.

Reasonable and Prudent Measures (RPMs)

As described in the Opinion, we are able to estimate the likely number of sea turtles and Atlantic sturgeon taken as a result of the proposed actions. However, it is unlikely that all (or even most) interactions would be observed by on-board ESA observers. Hopper dredges used in the proposed action are outfitted with UXO screens, comprised of longitudinal bars with openings/spacings of 1.25/ 1.5-inches by 6 inches (see section 3.0). These dimensions will prevent the whole animal and large parts from being brought on-board the hopper dredge. Rather, it is likely that only internal soft tissue (e.g., intestine) or small, fragmented, external parts (e.g., pieces of shell) of the crushed/impinged animal would be entrained. These parts are extremely unlikely to be detected by ESA observers, and if detected, are likely to be too small to be identifiable as a particular species (pers. comm. Chris Slay, Coast Wise Consulting, Inc.; Trish Bargo, East Coast Observers, Inc.; April 4, 2012). Additionally, animals may impinge on the UXO screens. Animals impinged on the UXO screen may free or dislodge themselves from the screen once the suction of the dredge has been turned off. Animals that free themselves may suffer severe injuries that may result in death. As the entire interaction occurs underwater, it would not be observed by an on-board observer. Due to the limited ability to observe an interaction from on deck, requiring the presence of an ESA observer on all hopper dredges operating under the proposed actions is an ineffective means to monitor take. As there is no practical way to monitor the impingement/entrainment of listed species during hopper dredging operations under the proposed action through ESA observers, we explored several alternatives, including proxies, for monitoring the interactions as described below.

In 2012, the USACE and NMFS considered the following alternatives to monitor take of listed species during hopper dredge operations with a UXO screen in place:

1. Install a camera near the draghead: A camera installed on a draghead would allow users at the surface to observe underwater interactions. However, there are technical challenges to using video, including visibility due to water clarity and available light, improper focus, inappropriate camera angle, and the range of the viewing field. The use of video would require additional resources, and it is unlikely that it would be effective for monitoring this type of dredge work. For these dredges, turbidity levels (i.e., up to 450 mg/l) near the draghead while dredging operations are underway are too high to visually detect any animal impinged on or within the vicinity of the draghead. Therefore, we concluded this would be an ineffective means of monitoring take.

2. Use of sonar/fish finder: Sonar can be used to detect animals within the water and within the vicinity of the dredge. We concluded that sonar alone could not indicate the take of an individual animal or identify the species potentially being taken. As such, we concluded that the use of such devices would be ineffective in monitoring for take.

3. Placement of observers on the shoreline: Observers placed on the shoreline may be able to detect stranded animals either in the water or on the shore. However, animals may not strand in the direct vicinity of the operation, and injured or deceased animal may not float to the surface immediately (i.e., it may take days for this to occur) or may drift far from the incident where the injury occurred. Therefore, an injured or deceased stranded animal often cannot be definitively attributed to a specific action. As such, we concluded that this is not a reasonable way to monitor take.

4. Relocation trawling: Relocation trawling is a method to remove sea turtles from an area before an activity such as dredging occurs. In considering relocation trawling, you must also consider that animals can be injured/entrained in the trawl, and animals can return to the site depending on the length of time between dredging and trawling. While relocation trawling may potentially reduce take it does not provide a means for monitoring take. Therefore, we concluded that this is not a reasonable alternative.

5. Time of year restriction: In dredging operations, time of year restrictions may be used to reduce or eliminate take. Moving the dredge operations outside an area when the animals are present reduces the likelihood of interaction. Time of year restrictions have been suggested for sea turtles in New Jersey waters, based on the best available information. However, Atlantic sturgeon may be in the project area year round. In addition, time of year restrictions do not provide a method for monitoring take, but rather reducing the take level. As sturgeon are present year-round, we did not think this was a reasonable alternative.

Both agencies agreed that none of these monitoring methods were reasonable or appropriate for this action. In situations where individual takes cannot be observed, a proxy must be considered. This proxy must be rationally connected to the taking and provide an obvious threshold of exempted take that, if exceeded, provides a basis for reinitiating consultation. In considering an appropriate proxy for these actions, we evaluated USACE records from 1990 to 2011 of hopper dredging operations occurring in similar habitats to the SBOBA. These records show that one sea turtle is entrained during dredging of 2.6 million cubic yards, and one Atlantic sturgeon in dredging of 5.6 million cubic yards (see section 7.1.2). This estimate provides a proxy for monitoring the amount of incidental take during hopper dredging and will be used as the primary method of determining whether incidental take has occurred. That is, we will consider that one sea turtle (Kemp's ridley or loggerhead) has been taken for every 2.6 million cubic yards material removed during hopper dredging operations. Similarly, we will consider that one subadult Atlantic sturgeon has been taken for every 5.6 million cubic yards of material removed during hopper dredging operations. In addition, there is a possibility that a sea turtle or an Atlantic sturgeon may remain impinged on UXO screens after the dredge has been turned off. These animals can be visually observed, via a lookout, when the draghead is lifted above the water. Animals documented by the lookout on the draghead will be considered a take. This monitoring method (i.e., proxy and/or observed) will be used for the proposed hopper dredging projects.

The amount of material the USACE expects to remove from the SBOBA is a total of approximately 391,000 cy for the Port Monmouth project, 688,000 for the Union beach project, and 14,834,452 cy for the Elberon to Loch Arbour project. Based on the information presented above, this may result in the take of 1 sea turtle (Kemp's ridley or loggerhead) and 1 Atlantic sturgeon (one turtle per 2.6 million cubic yards of material removed; one Atlantic sturgeon per 5.6 million cubic yards of material removed) during the Port Monmouth project, 1 sea turtle (Kemp's ridley or loggerhead) and 1 Atlantic sturgeon during the Union beach project, and 6 sea turtles (5 loggerheads plus 1 loggerhead or Kemp's ridley) and 3 Atlantic sturgeon during the Elberon to Loch Arbour project. In addition, observed animals impinged on the draghead will be considered as take.

As soon as the estimated number of sea turtles or Atlantic sturgeon are observed or believed to be taken (e.g., one take via proxy; or one observed impinged), any additional take of a sea turtle species or Atlantic sturgeon will be considered excess of the exempted level.[14] We expect exceedance of this take unlikely given the RMPs and Terms and Conditions described below. Lookouts will be present on the vessel and volumes of material removed will be continuously monitored during hopper dredge operations. Therefore, take levels can be detected and assessed early in the project and, if needed, consultation can be reinitiated.

[14] Please note, under the scenario of take observed via proxy and take physically observed, take will not be counted more than once. That is, should 2.6 million cy of material be removed and no sea turtles were observed impinged on the draghead, then the first take will be considered via the proxy. Alternatively, if during dredging of 2.6 million cy, a sea turtle is observed impinged, this will be considered take and no other take will be attributed to this round of dredging once it is complete (i.e., 2.6 cy of material removed); that is the proxy will not be applied.

NMFS believes the following reasonable and prudent measures are necessary and appropriate to minimize impacts of incidental take of sea turtles and Atlantic sturgeon resulting from the proposed actions.

RPMs

1. NMFS must be contacted prior to the commencement of hopper dredging and again upon the completion of the dredging activity.

2. The USACE shall ensure that all hopper dredges are outfitted with state-of-the-art sea turtle deflectors on the draghead and operated in a manner that will reduce the risk of interactions with sea turtles or Atlantic sturgeon.

3. The USACE shall obtain and implement a checklist to ensure UXO screens are properly in place and in a manner that will reduce the risk of interactions with sea turtles or Atlantic sturgeon.

4. Hopper dredges must undergo a field inspection prior to being used to ensure the checklist has been implemented and UXO screens are appropriately deployed in a manner that will reduce the risk of interactions with sea turtles or Atlantic sturgeon.

5. A lookout/bridge watch, knowledgeable in listed species identification, will be present on board the hopper dredge at all times to inspect the draghead/UXO screen each time it is removed from the water.

6. The USACE shall provide monthly reports to NMFS regarding the status of dredging and interactions or observations of listed species of sea turtles and Atlantic sturgeon.

7. The USACE shall ensure that all measures are taken to protect any turtles that survive impingement on the hopper dredge. All sea turtles captured must be retained until further coordination with NMFS.

8. The USACE shall ensure that all measures are taken to protect any sturgeon that survive impingement on the hopper dredge.

9. Any dead sturgeon must be transferred to NMFS or an appropriately permitted research facility identified by NMFS so that fin clips and a necropsy can be undertaken to attempt to determine the cause of death. Sturgeon should be held in cold storage.

10. Any dead sea turtles must be held until proper disposal procedures can be discussed with NMFS. Turtles should be held in cold storage.

11. All sturgeon and turtle captures, injuries or mortalities associated with any dredging activities or any other aspect of the project must be reported to NMFS within 24 hours.

Terms and conditions
In order to be exempt from prohibitions of section 9 of the ESA, USACE must comply with the following terms and conditions, which implement the reasonable and prudent measures described above and outline required reporting/monitoring requirements. These terms and conditions are non-discretionary.

To implement RPM #1, the USACE must contact NMFS (Dan Marrone: by email (Daniel.Marrone@noaa.gov) or phone (978)-282-8465) within 3 days of commencement of dredging and again within 3 days of completion of dredging activity. This correspondence will serve both to alert NMFS of the commencement and cessation of dredging activities, to give NMFS an opportunity to provide the USACE with any updated contact information or reporting forms, and to provide NMFS with information of any incidences with listed species.

To implement RPM #2, hopper dredges must be equipped with the rigid deflector draghead as designed by the USACE Engineering Research and Development Center, formerly the Waterways Experimental Station (WES), or if that is unavailable, a rigid sea turtle deflector attached to the draghead. Deflectors must be checked and/or adjusted by a designated expert prior to a dredge operation to insure proper installment and operation during dredging. The deflector must be checked after every load throughout the dredge operation to ensure that proper installation is maintained. Since operator skill is important to the effectiveness of the WES-developed draghead, operators must be properly instructed in its use. Dredge inspectors must ensure that all measures to protect sea turtles are being followed during dredge operations.

To implement RPM #3 the USACE will develop a checklist that decribes in detail the process that must be followed and the equipment that must be checked to ensure that the UXO screen is properly in place. Should the screen not be able to be properly placed, the necessary steps should be taken to resolve any problems with the UXO screen before any dredging begins.

To implement RPM #4 UXO screens must be inspected and/or adjusted by a designated expert (someone with experience deploying and operating the draghead) prior to a dredge operation to ensure proper installment and operation during the dredging. The UXO screen must be checked after every load throughout the dredge operation to ensure that proper installation is maintained. Dredge inspectors must ensure that all measures to protect sea turtles and Atlantic sturgeon are being followed during dredge operations.

To implement RPM #5, the Corps will require the lookout to inspect the draghead/UXO screen for impinged sea turtles or Atlantic sturgeon each time it is brought up from completing a dredge cycle. Should a sea turtle or Atlantic sturgeon be found impinged on the draghead, the incident should be recorded (Appendix D and/or E and F) and NMFS contacted.

To implement RPM #5, the Corps will require the lookout to inspect the UXO screen each time the draghead is lifted from the water to inspect for damages on the screen. Condition of the UXO screen should be recorded on the “Dredge Observer Form” (See Appendix C). Should the screen be damaged, prior to continuing dredging, the Corps will ensure that repairs to the screen are made as soon as possible to avoid possible unintentional entrainment of large objects.

To implement RPM #6, the Corps will provide NMFS reports every 45 days, via email (Daniel.Marrone@noaa.gov) or mail (Protected Resources Division, 55 Great Republic Drive, Gloucester, MA 01930), recording the days that dredging occurred, and summarizing the lookout/bridge watch reports on draghead inspection, the volume of material removed during the previous month for a 30 day period, and any observations of listed species of sea turtles and Atlantic sturgeon. This information will be used in our assessment of take of sea turtles and/or Atlantic sturgeon. Only those monthly reports that occur within "sea turtle" season in New Jersey waters (i.e., May 1-November 15) will be considered in our assessment of sea turtle take. As Atlantic sturgeon may be present in New Jersey waters throughout the year, it is necessary we receive monthly reports for every month dredging operations will be undertaken.

To implement RPM #7, the procedures for handling live sea turtles must be followed in the unlikely event that a sea turtle survives impingement on the dredge (Appendix B). NMFS should be contacted immediately to discuss the transfer of the animal to an appropriate permitted rehabilitation facility.

To implement RPM #8, any live sturgeon impinged on the draghead of a hopper dredge must be photographed and measured (if possible), and released immediately overboard while the dredge is not operating.

To implement RPM #9, in the event of any lethal takes of Atlantic sturgeon, any dead specimens or body parts must be photographed, measured, and preserved (refrigerate or freeze) until disposal procedures are discussed with NMFS. The form included as Appendix F (sturgeon salvage form) must be completed and submitted to NMFS.

To implement RPM #9, if a decomposed Atlantic sturgeon or Atlantic sturgeon body part is entrained/impinged during any dredging operations, the USACE must ensure that an incident report is completed and the specimen is photographed. Any sturgeon or sturgeon body parts that are considered "not fresh" (i.e., they were obviously dead prior to the dredge take (e.g., foul odor; necrotic dark or decaying tissue; sloughing of scutes; atypical coloration; and/or opaque eyes) and that the USACE anticipates that will not be counted towards the ITS, must be frozen. The USACE must submit an incident report for the decomposed sturgeon part, as well as photographs, to NMFS within 24 hours of the take (see Appendix E and Appendix F) and request concurrence that this take should not be attributed to the Incidental Take Statement. NMFS has sole discretion in determining if the take should count towards the Incidental Take Statement.

To implement RPM #10, in the event of any lethal takes of sea turtles, any dead specimens or body parts must be photographed, measured, and preserved (refrigerate or freeze) until disposal procedures are discussed with NMFS. The form included as Appendix D must be completed and submitted to NMFS.

To implement RPM #10, if a decomposed turtle or turtle part is impinged or entrained during any dredging operations, an incident report must be completed and the specimen must be photographed. Any turtle parts that are considered "not fresh" (i.e., they were obviously dead prior to the dredge take and the USACE anticipates that they will not be counted towards the ITS) must be frozen and transported to a nearby stranding or rehabilitation facility for review. The USACE must ensure that the observer or lookout submits the incident report for the decomposed turtle or turtle part, as well as photographs, to NMFS within 24 hours of the take (see Appendix D) and request concurrence that this take should not be attributed to the Incidental Take Statement. NMFS shall have sole discretion in determining if the take should count towards the Incidental Take Statement.

To implement RPM #11, the USACE must contact NMFS within 24 hours of any interactions with Atlantic sturgeon or sea turtles, including non-lethal and lethal takes. NMFS will provide contact information annually when alerted of the start of dredging activity. Until alerted otherwise, the USACE should contact Dan Marrone: by email (Daniel.Marrone@noaa.gov) or phone (978) 282-8465 or the Section 7 Coordinator by phone (978) 281-9328 or fax 978-281-9394). Take information should also be reported by e-mail to: incidental.take@noaa.gov.

To implement RPM #11, the USACE must ensure that any Atlantic sturgeon or sea turtles observed during project operations (including whole sturgeon or sea turtles or body parts observed at the disposal location or on board the hopper) are photographed and measured and the corresponding form (Appendix D and/or E and F) must completed and submitted to NMFS within 24 hours by fax (978-281-9394) or e-mail (incidental.take@noaa.gov).

To implement RPM #11, any time a take occurs, the USACE must immediately contact NMFS to review the situation. At that time, the USACE must provide NMFS with information on the amount of material dredged thus far and the amount remaining to be dredged during that cycle. Also at that time, the USACE should discuss with NMFS whether any new management measures could be implemented to prevent the total incidental take level from being exceeded, with emphasis on determing whether this take represents new information revealing effects of the action that may not have been previously considered.

The reasonable and prudent measures, with their implementing terms and conditions, are designed to minimize and monitor the impact of incidental take that might otherwise result from the action. Specifically, these RPMs and Terms and Conditions will keep NMFS informed of when and where dredging activities are taking place and will require the USACE to report any take in a reasonable amount of time, as well as implement measures to monitor for impingement/entrainment during dredging. The USACE has reviewed the RPMs and Terms and Conditions outlined above and has agreed to implement all of these measures as described herein and in the referenced Appendices. The discussion below explains why each of these RPMs and Terms and Conditions are necessary and appropriate to minimize or monitor the level of incidental take associated with the action and how they represent only a minor change to the action as proposed by the USACE.

RPM #1, #6, and #11 and Term and Condition #1, #7, and #14-16, are necessary and appropriate because they will serve to ensure that NMFS is aware of the dates and locations of all dredging activities as well as serve to monitor take via the proxy or via other incidences of interactions with listed species. This will also allow NMFS to monitor the duration and seasonality of dredging activities as well as give NMFS an opportunity to provide the USACE with any updated contact information for NMFS staff. These RPMs and Terms and Conditions will help us determine whether and when reinitiation may be required due to changes in the action, or exceedances of incidental take. This is only a minor change because it is not expected to result in any delay to the project and will merely involve an occasional telephone call or e-mail between the USACE and NMFS staff.

RPM #2 and Terms and Conditions #2, are necessary and appropriate as the use of draghead deflectors is accepted standard practice for hopper dredges operating in places and at times of year when sea turtles are known to be present and has been documented to reduce the risk of entrainment for sea turtles, thereby minimizing the potential for take of these species. It is believed that this holds true for Atlantic sturgeon as well. This represents only a minor change as all of the hopper dredges likely to be used for this project already have draghead deflectors, dredge operators are already familiar with their use, and the use will not affect the efficiency of the dredging operation. Additionally, maintenance of the existing channel is conducted with draghead deflectors in place.

RPMs #3 and #4 and Terms and Conditions #3 and #4 are necessary and appropriate to ensure that the UXO screen is placed properly on the dredge, thereby minimizing the potential risk of entrainment to Atlantic sturgeon and sea turtles. This represents only a minor change as it will require an inspection of the UXO screens on hopper dredges that will already be equipped with the screens. These procedures will not result in an increase in cost or any delays to the project.

RPM #5 and Terms and Conditions #5 and #6, are necessary and appropriate to ensure the proper monitoring of listed species that may be taken via impingement on the draghead, as well as to ensure the proper monitoring of listed species that may occur in the vicinity of the project areas and thus, the proper operation of the vessel in the presence of these species. This RPM and its Terms and Conditions will also ensure proper documentation of any interactions with listed species as well as requiring that these interactions are reported to NMFS in a timely manner with all of the necessary information. This is essential for monitoring the level of incidental take associated with the actions. In addition, this RPM and its Terms and Conditions are also necessary and appropriate to ensure that any damage to the UXO screen are repaired to prevent the entrainment of listed species. The inclusion of these RPMs and Terms and Conditions is only a minor change as the lookout can be a member of the vessel crew that is knowledgeable in listed species identification and will not result in any delays. These also represent only a minor change as in many instances, they serve to clarify the duties of the inspectors or lookouts.

RPM #7, #8 and Terms and Conditions #8, and #9, are necessary and appropriate to ensure that any sea turtles or Atlantic sturgeon that survive impingement or entrainment in dredging operations are given the maximum probability of remaining alive and not suffering additional injury or subsequent mortality through inappropriate handling. This represents only a minor change as following these procedures will not result in an increase in cost or any delays to the project.

RPM #9 and Term and Condition #10-11, are necessary and appropriate to determine the cause of death of any dead sturgeon observed during the proposed actions. This is necessary for the monitoring of the level of take associated with the action. This represents only a minor change, as following these procedures will have an insignificant impact on the cost of the project and will not result in any delays.

RPM #10 and Terms and Condition #12-13, are necessary and appropriate as future analysis may be needed on the dead sea turtle. Additional analysis will be dependent on available freezer space, availability of organizations capable of conducting the analysis, and the size/condition of the sample. NMFS will provide guidance on this matter upon the USACE's notification of take. If NMFS determines that the animal is not necessary to save for future analysis, dead sea turtle species (loggerhead, leatherback, Kemp's ridley, or green turtles) taken either whole or in parts should be disposed of (after a photograph is taken and a reporting form has been completed) by attaching a weight to the animal and dumping the specimen away from the areas being dredged (e.g., between the shore and the site of dredging operations). This represents only a minor change as following these procedures will have an insignificant impact on the cost of the project and will not result in any delays.

12.0 CONSERVATION RECOMMENDATIONS

In addition to Section 7(a)(2), which requires agencies to ensure that proposed projects will not 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 this Act by carrying out programs for the conservation of endangered 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. As such, NMFS recommends that the USACE consider the following Conservation Recommendations:

1. To the extent practicable, the USACE should avoid dredging during times of year when listed species are likely to be present.

2. To facilitate future management decisions on listed species occurring in the action area, the USACE should maintain a database mapping system to: a) create a history of use of the geographic areas affected; and, b) document endangered/threatened species presence/interactions with project operations.

3. The USACE should support ongoing and/or future research to determine the abundance and distribution of sea turtles and Atlantic sturgeon in New Jersey waters.

4. The USACE should investigate, support, and/or develop additional technological solutions to further reduce the potential for sea turtle or Atlantic sturgeon takes in hopper dredges as well to monitor for take of listed species when a UXO screen is placed on a dredge. For instance, NMFS recommends that the USACE coordinate with other Southeast Districts, the Association of Dredge Contractors of America, and dredge operators regarding additional reasonable measures they may take to further reduce the likelihood of sea turtle or strugeon

takes. The diamond-shaped pre-deflector, or other potentially promising pre-deflector designs such as tickler chains, water jets, sound generators, etc., should be developed and tested and used where conditions permit as a means of alerting sea turtles and sturgeon of approaching equipment. New technology or operational measures that would minimize the amount of time the dredge is spent off the bottom in conditions of uneven terrain should be explored. Pre-deflector use should be noted on observer daily log sheets, and annual reports to NMFS should note what progress has been made on deflector or pre-deflector technology and the benefits of, or problems associated with, their usage.

5. New approaches to sampling for turtle or sturgeon parts should be investigated. Project proponents should seek continuous improvements in detecting takes and should determine, through research and development, a better method for monitoring and estimating sea turtle or Atlantic sturgeon takes by hopper dredges. Observation of overflow and inflow screening appears to be only partially effective and may provide only minimum estimates of total sea turtle or Atlantic sturgeon mortality; however, if a UXO screen is used, this method is ineffective and as such, appropriate methods for observing take in these cases needs to be developed. NMFS believes that some listed species taken by hopper dredges may go undetected because body parts are forced through the sampling screens by the water pressure (as seen in 2002 Cape Henry dredging) and are buried in the dredged material, or animals are crushed or killed, but not entrained by the suction and consequently, the takes may go unnoticed (or may subsequently strand on nearby beaches). The only mortalities that are documented are those where body parts float, are large enough to be caught in the screens, or can be identified to species.

6. NMFS recommends that all sea turtles and Atlantic sturgeon impinged/entrained in hopper dredge dragheads be sampled for genetic analysis by a NMFS laboratory. Any genetic samples from live sea turtles or Atlantic sturgeon must be taken by trained and permitted personnel.

7. The USACE should consider devising and implementing some method of significant economic incentives to hopper dredge operators, such as financial reimbursement based on their satisfactory completion of dredging operations, or a certain number of cubic yards of material removed, or hours of dredging performed, *without taking turtles or sturgeon*. This may encourage dredging companies to research and develop “turtle or sturgeon friendly” dredging methods, more effective deflector dragheads, pre-deflectors, top-located water ports on dragarms, etc.

13.0 REINITIATION OF CONSULTATION

This concludes formal consultation on the USACE’s beach nourishment projects utilizing the SBOBA. 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) a new species is listed or critical habitat designated that may be affected by the action; (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) 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. If the amount or extent of incidental take is exceeded, the USACE must immediately request reinitiation of formal consultation.

14.0 LITERATURE CITED

Agler, B.A., R.L., Schooley, S.E. Frohock, S.K. Katona, and I.E. Seipt. 1993. Reproduction of photographically identified fin whales, *Balaenoptera physalus*, from the Gulf of Maine. J. Mamm. 74:577-587.

Aguilar, A. and C. Lockyer. 1987. Growth, physical maturity and mortality of fin whales (*Balaenoptera physalus*) inhabiting the temperate waters of the northeast Atlantic. Can. J. Zool. 65:253-264.

Allen, B.M., and R.P. Angliss. 2011. Alaska Marine Mammal Stock Assessments, 2010. U.S. Dep. Commer., NOAA Technical Memorandum NOAA-AFSC-223. 292 p.

Anchor Environmental. 2003. Literature review of effects of resuspended sediments due to dredging. June. 140pp.

Andrews, H.V., and K. Shanker. 2002. A significant population of leatherback turtles in the Indian Ocean. Kachhapa. 6:19.

Andrews, H.V., S. Krishnan, and P. Biswas. 2002. Leatherback nesting in the Andaman and Nicobar Islands. Kachhapa. 6:15-18.

Angliss, R.P., D.P. DeMaster, and A.L. Lopez. 2001. Alaska marine mammal stock assessments, 2001. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-124, 203 p.

Angliss, R.P. and R.B. Outlaw. 2007. Alaska Marine Mammal Stock Assessments, 2006. NOAA Technical Memorandum NOAA-TM-AFSC-168. 244 p.

Angliss, R. P., and B.M. Allen. 2009. Alaska Marine Mammal Stock Assessments, 2008. NOAA Technical Memorandum NOAA-TM-AFSC-193. 258 p.

Antonelis, G.A., J.D. Baker, T.C. Johanos, R.C. Braun, and A.L. Harting. 2006. Hawaiian monk seal (*Monachus schauinslandi*): Status and conservation issues. Atoll Res Bull 543:75–101.

Armstrong, J. L., and J. E. Hightower. 2002. Potential for restoration of the Roanoke River population of Atlantic sturgeon. Journal of Applied Ichthyology 18: 475-480.

ASMFC (Atlantic States Marine Fisheries Commission). 1998. Amendment 1 to the interstate fishery management plan for Atlantic sturgeon. Management Report No. 31,43 pp.

ASMFC TC (Technical Committee). 2007. Special Report to the Atlantic Sturgeon Management Board: Estimation of Atlantic sturgeon bycatch in coastal Atlantic commercial fisheries of New England and the Mid-Atlantic. August 2007. 95 pp.

ASMFC (Atlantic States Marine Fisheries Commission). 2009. Atlantic Sturgeon. In: Atlantic Coast Diadromous Fish Habitat: A review of utilization, threats, recommendations for conservation and research needs. Habitat Management Series No. 9. Pp. 195-253.

ASMFC. 2010. Atlantic Sturgeon. Pages 19-20. In Atlantic States Marine Fisheries Commission 2010 Annual Report. 68 pp.

Atlantic Sturgeon Status Review (ASSRT). 2007. http://www.nero.noaa.gov/prot_res/CandidateSpeciesProgram/AtlSturgeonStatusReviewReport.pdf

Attrill, M.J., J. Wright, and M. Edwards. 2007. Climate-related increases in jellyfish frequency suggest a more gelatinous future for the North Sea. Limnology and Oceanography 52:480-485.

Avens, L., J.C. Taylor, L.R. Goshe, T.T. Jones, and M. Hastings. 2009. Use of skeletochronological analysis to estimate the age of leatherback sea turtles *Dermochelys coriacea* in the western North Atlantic. Endangered Species Research 8:165-177.

Bain, M. B. 1997. Atlantic and shortnose sturgeons of the Hudson River: Common and Divergent Life History Attributes. Environmental Biology of Fishes 48: 347-358.

Bain, M.B., N. Haley, D. Peterson, J. R. Waldman, and K. Arend. 2000. Harvest and habitats of Atlantic sturgeon *Acipenser oxyrinchus* Mitchill, 1815, in the Hudson River Estuary: Lessons for Sturgeon Conservation. Instituto Espanol de Oceanografia. Boletin 16: 43-53.

Baker, C.S., L.M. Herman, B.G. Bays and W.F. Stifel. 1982. The impact of vessel traffic on the behavior of humpback whales in southeast Alaska. Report submitted to the National Marine Mammal Laboratory, Seattle, Washington.

Baker, C.S. L.M. Herman, B.G. Bays and G.B. Bauer. 1983. The impact of vessel traffic on the behavior of humpback whales in southeast Alaska: 1982 season. Report submitted to the National Marine Mammal Laboratory, Seattle, Washington.

Baker, C. S. and Herman, L. M. 1989. Behavioral responses of summering humpback whales to vessel traffic: experimental and opportunistic observations. Final Report to the National Park Service, U. S. Department of the Interior, Anchorage, AK.

Baker J.D., C.L. Littnan, D.W. Johnston. 2006. Potential effects of sea level rise on the terrestrial habitats of endangered and endemic megafauna in the Northwestern Hawaiian Islands. Endang Species Res 2:21–30.

Balazs, G.H. 1982. Growth rates of immature green turtles in the Hawaiian Archipelago, p. 117-125. In K.A. Bjorndal (ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington, D.C.

Balazs, G.H. 1985. Impact of ocean debris on marine turtles: entanglement and ingestion. U.S. Department of Commerce, NOAA Tech. Memo. NMFS-SWFSC-54:387-429.

Baldwin, R., G.R. Hughes, and R.T. Prince. 2003. Loggerhead turtles in the Indian Ocean. Pages 218-232. In: A.B. Bolten and B.E. Witherington (eds.) Loggerhead Sea Turtles. Smithsonian Books, Washington, D.C. 319 pp.

Baumgartner, M.F., and B.R. Mate. 2005. Summer and fall habitat of North Atlantic right whales (*Eubalaena glacialis*) inferred from satellite telemetry. Can. J. Fish. Aquat. Sci. 62:527-543.

Barlow, J., and P. J. Clapham. 1997. A new birth-interval approach to estimating demographic parameters of humpback whales. Ecology 78: 535-546.

Bass, A.L., S.P. Epperly, J.Braun-McNeill. 2004. Multi-year analysis of stock composition of a loggerhead sea turtle (*Caretta caretta*) foraging habitat using maximum likelihood and Bayesian methods. Conservation Genetics 5: 784-796.

Bauer, G.B., and L.M. Herman. 1985. Effects of vessel traffic on the behavior of humpback whales in Hawaii. Report submitted to the National Marine Fisheries Service, Honolulu, Hawaii.

Berry, R. J. 1971. Conservation aspects of the genetical constitution of populations. Pages 177-206 in E. D. Duffey and A. S. Watt, eds. The Scientific Management of Animal and Plant Communities for Conservation. Blackwell, Oxford.

Berube, M., A. Aguilar, D. Dendanto, F. Larsen, G. Notarbatolo di Sciara, R. Sears, J. Sigurjonsson, J. Urban-R, and P. Palsboll. 1998. Population genetic structure of North Atlantic, Mediterranean Sea and Sea of Cortez fin whales: analysis of mitochondrial and nuclear loci. Molecular ecology 7: 585-599.

Best, P.B., J. L. Bannister, R.L. Brownell, Jr., and G.P. Donovan (eds.). 2001. Right whales: worldwide status. J. Cetacean Res. Manage. (Special Issue) 2: 309pp.

Bjork, M., F. Short, E. McLeod, and S. Beers. 2008. Managing seagrasses for resilience to climate change. IUCN, Gland.

Bigelow, H.B. and W.C. Schroeder. 1953. Sea Sturgeon. *In*: Fishes of the Gulf of Maine. Fishery Bulletin 74. Fishery Bulletin of the Fish and Wildlife Service, vol. 53.

Bjorndal, K.A. 1997. Foraging ecology and nutrition of sea turtles. Pages 199-233 *In*: Lutz, P.L. and J.A. Musick, eds., The Biology of Sea Turtles. CRC Press, New York. 432 pp.

Blumenthal, J.M., J.L. Solomon, C.D. Bell, T.J. Austin, G. Ebanks-Petrie, M.S. Coyne, A.C. Broderick, and B.J. Godley. 2006. Satellite tracking highlights the need for international cooperation in marine turtle management. Endangered Species Research 2:51-61.

Bolten, A.B., J.A. Wetherall, G.H. Balazs, and S.G. Pooley (compilers). 1996. Status of marine turtles in the Pacific Ocean relevant to incidental take in the Hawaii-based pelagic longline fishery. U.S. Dept. of Commerce, NOAA Technical Memorandum, NOAA-TM-NOAA Fisheries SWFSC-230.

Bolten, A.B., K.A. Bjorndal, H.R. Martins, T. Dellinger, M.J. Biscoito, S.E. Encalada, and B.W. Bowen. 1998. Transatlantic developmental migrations of loggerhead sea turtles demonstrated by mtDNA sequence analysis. Ecological Applications 8(1):1-7.

Boreman, J. 1997. Sensitivity of North American sturgeons and paddlefish to fishing mortality. Environmental Biology of Fishes 48: 399-405.

Borodin, N. 1925. Biological observations on the Atlantic sturgeon, *Acipenser sturio.* Transactions of the American Fisheries Society 55: 184-190.

Boulon, R., Jr. 2000. Trends in sea turtle strandings, U.S. Virgin Islands: 1982 to 1997. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-436:261-263.

Bowman R., EG.. Lyman, D. Mattila, C.A. Mayo, M. Brown. 2003. Habitat Management Lessons from a Satellite-Tracked Right Whale, Presented to the ARGOS Animal Tracking Symposium , March 2003. Annapolis, Maryland.

Bowen, B.W. 2003. What is a loggerhead turtle? The genetic perspective. Pages 7-27 in A.B. Bolten and B.E. Witherington, (eds). Loggerhead Sea Turtles. Washington, D.C.: Smithsonian Press.

Bowen, B.W. , A. L. Bass, S. Chow, M. Bostrom, K. A.Bjorndal, A. B. Bolten, T. Okuyama, B. M. Bolker, S.Epperley, E. Lacasella, D. Shaver, M. Dodd, S. R. Hopkins-Murphy, J. A. Musick, M. Swingle, K. Rankin-Baransky, W. Teas, W. N. Witzell, and P. H. Dutton. 1992. Natal homing in juvenile loggerhead turtles *(Caretta caretta).* Molecular Ecology (2004) 13: 3797-3808.

Bowen, B.W., A.L. Bass, S. Chow, M. Bostrom, K.A. Bjorndal, A.B. Bolten, T. Okuyama, B.M. Bolker, S. Epperly, E. LaCasella, D. Shaver, M. Dodd, S.R. Hopkins-Murphy, J.A. Musick, M. Swingle, K. Rankin-Baransky, W. Teas, W.N. Witzell, and P.H. Dutton. 2004. Natal homing in juvenile loggerhead turtles (*Caretta caretta*). Molecular Ecology 13:3797-3808.

Bowen, B.W., A.L. Bass, L. Soares, and R.J. Toonen. 2005. Conservation implications of complex population structure: lessons from the loggerhead turtle (*Caretta caretta*). Molecular Ecology 14:2389-2402.

Bowen, B.W., and S.A. Karl. 2007. Population genetics and phylogeography of sea turtles. Molecular Ecology 16:4886-4907.

Braun, J., and S.P. Epperly. 1996. Aerial surveys for sea turtles in southern Georgia waters, June 1991. Gulf of Mexico Science. 1996(1): 39-44.

Braun-McNeill, J., and S.P. Epperly. 2004. Spatial and temporal distribution of sea turtles in the western North Atlantic and the U.S. Gulf of Mexico from Marine Recreational Fishery Statistics Survey (MRFSS). Mar. Fish. Rev. 64(4):50-56.

Braun-McNeill, J. , C.R. Sasso, S.P.Epperly, C. Rivero. 2008. Feasibility of using sea surface temperature imagery to mitigate cheloniid sea turtle–fishery interactions off the coast of northeastern USA. Endangered Species Research: Vol. 5: 257–266, 2008.

Brewer, K., M. Gallagher, P. Regos, P. Isert, and J. Hall. 1993. Kuvlum #1 Exploration Prospect: Site Specific Monitoring Program, Final Report. Prepared by Coastal Offshore Pacific Corporation, Walnut Creek, CA, for ARCO Alaska, Inc., Anchorage, AK. 80 pp.

Brodeur, R.D., C.E. Mills, J.E. Overland, G.E. Walters, and J.D. Schumacher. 1999. Evidence for a substantial increase in gelatinous zooplankton in the Bering Sea, with possible links to climate change. Fisheries Oceanography 8(4): 296-306.

Brown, S.G. 1986. Twentieth-century records of right whales (*Eubalaena glacialis*) in the Northeast Atlantic Ocean. In: R.L. Brownell Jr., P.B. Best, and J.H. Prescott (eds.) Right whales: Past and Present Status. IWC Special Issue No. 10. p. 121-128.

Brown, M. W., and M.K. Marx. 2000. Surveillance, Monitoring and Management of North Atlantic Right Whales, Eubalaena glacialis, in Cape Cod Bay, Massachusetts: January to Mid-May, 2000. Final report.

Brown, M.W., O.C. Nichols, M.K. Marx, and J.N. Ciano. 2002. Surveillance, Monitoring, and Management of North Atlantic Right Whales in Cape Cod Bay and Adjacent Waters – 2002. Final report to the Division of Marine Fisheries, Commonwealth of Massachusetts. Center for Coastal Studies.

Brown, J.J. and G.W. Murphy. 2010. Atlantic sturgeon vessel strike mortalities in the Delaware Estuary. Fisheries 35 (2): 72-83.

Brundage, H.M. and J. C. O'Herron. 2009. Investigations of juvenile shortnose and Atlantic sturgeons in the lower tidal Delaware River. Bull. N.J. Acad. Sci. 54(2), pp1-8.

Bryant, L.P. 2008. Governor's Commission on Climate Change. Final Report: A Climate

Change Action Plan. Virginia Department of Environmental Quality.

Burlas, M., G. L Ray, & D. Clarke. 2001. The New York District's Biological Monitoring Program for the Atlantic Coast of New Jersey, Asbury Park to Manasquan Section Beach Erosion Control Project. Final Report. U.S. Army Engineer District, New York and U.S. Army Engineer Research and Development Center, Waterways Experiment Station.

Burton, W. 1993. Effects of bucket dredging on water quality in the Delaware River and the potential for effects on fisheries resources. Prepared by Versar, Inc. for the Delaware Basin Fish and Wildlife Management Cooperative, unpublished report. 30 pp.

Bushnoe, T.M., J.A. Musick, and D.S. Ha. 2005. Essential Spawning and Nursery Habitat of Atlantic Sturgeon (Acipenser oxyrinchus oxyrinchus) in Virginia. VIMS Special Scientific Report #145: 44 pp.

Caillouet, C., C.T. Fontaine, S.A. Manzella-Tirpak, and T.D. Williams. 1995. Growth of head-started Kemp's ridley sea turtles (*Lepidochelys kempi*) following release. Chelonian Conservation and Biology. 1(3): 231-234.

Calambokidis, J., E. A. Falcone, T.J. Quinn., A.M. Burdin, P.J. Clapham, J.K.B. Ford, C.M. Cabriele, R. LeDuc, D. Matilla, L. Rojas-Bracho, J.M. Straley, B.L. Taylor, J. Urban, D. Weller, B.H. Witteveen, M.Yamaguchi, A. Bendlin, D. Camacho, K. Flynn, A. Havron, J. Huggins, and N. Maloney. 2008. SPLASH: Structure of Populations, Levels of Abundance and Status of Humpback Whales in the North Pacific. Final Report for Contract AB133F-03-RP-00078; 57 pp.

Calvo, L., H.M. Brundage, D. Haivogel, D. Kreeger, R. Thomas, J.C. O'Herron, and E. Powell. 2010. Effects of flow dynamics, salinity, and water quality on the Eastern oyster, the Atlantic sturgeon, and the shortnose sturgeon in the oligohaline zone of the Delaware Estuary. Prepared for the U.S. Army Corps of Engineers, Philadelphia District. 108 p.

Cameron, S. 2010. "Assessing the Impacts of Channel Dredging on Atlantic Sturgeon Movement and Behavior". Presented to the Virginia Atlantic Sturgeon Partnership Meeting. Charles City, Virginia. March 19, 2010.

Caron, F., D. Hatin, and R. Fortin. 2002. Biological characteristics of adult Atlantic sturgeon (*Acipenser oxyrinchus*) in the Saint Lawrence River estuary and the effectiveness of management rules. Journal of Applied Ichthyology 18: 580-585. Dadswell, M.J. 1984. Status of the Shortnose Sturgeon, *Acipenser brevirostrum,* in Canada. The Canadian Field-Naturalist 98 (1): 75-79.

Carr, A.R. 1963. Pan specific reproductive convergence in *Lepidochelys kempi*. Ergebn. Biol. 26: 298-303.

Carreras, C., S. Pont, F. Maffucci, M. Pascual, A. Barceló, F. Bentivegna, L. Cardona, F. Alegre,

M. SanFélix, G. Fernández, and A. Aguilar. 2006. Genetic structuring of immature loggerhead sea turtles (*Caretta caretta*) in the Mediterranean Sea reflects water circulation patterns. Marine Biology 149:1269-1279.

Carretta, J.V., K.A. Forney, E. Oleson, K. Martien, M.M. Muto, M. Lowry, J. Barlow, J. Baker, B. Hanson, D. Lynch, L. Carswell, R.L. Brownell Jr., J. Robbins, D.K. Mattila, K. Ralls, and M.C. Hill. 2011. U.S. Pacific Marine Mammal Stock Assessments: 2010. U.S. Department of Commerce, NOAA Technical Memorandum NMFS SWFSC. 352 p.

Casale, P., P. Nicolosi, D. Freggi, M. Turchetto, and R. Argano. 2003. Leatherback turtles (*Dermochelys coriacea*) in Italy and in the Mediterranean basin. Herpetological Journal 13: 135-139.

Castroviejo, J., J.B. Juste, J.P. Del Val, R. Castelo, and R. Gil. 1994. Diversity and status of sea turtle species in the Gulf of Guinea islands. Biodiversity and Conservation 3: 828-836.

Caswell, H., M. Fujiwara, and S. Brault. 1999. Declining survival probability threatens the North Atlantic right whale. Proc. Nat. Acad. Sci. 96: 3308-3313.

Caulfield, R.A. 1993. Aboriginal subsistence whaling in Greenland: the case of Qeqertarsuaq municipality in West Greenland. Arctic 46: 144-155.

Cetacean and Turtle Assessment Program (CeTAP). 1982. Final report of the cetacean and turtle assessment program, University of Rhode Island, to Bureau of Land Management, U.S. Department of the Interior. Ref. No. AA551-CT8-48. 568 pp.

Chan, E.H., and H.C. Liew. 1996. Decline of the leatherback population in Terengganu, Malaysia, 1956-1995. Chelonian Conservation and Biology 2(2): 192-203.

Chevalier, J., X. Desbois, and M. Girondot. 1999. The reason for the decline of leatherback turtles *(Dermochelys coriace*a) in French Guiana: a hypothesis p.79-88. In Miaud, C. and R. Guyétant (eds.), Current Studies in Herpetology, Proceedings of the ninth ordinary general meeting of the Societas Europea Herpetologica, 25-29 August 1998 Le Bourget du Lac, France.

Church, J., J.M. Gregory, P. Huybrechts, M. Kuhn, K. Lambeck, M.T. Nhuan, D. Qin, P.L. Woodworth. 2001. Changes in sea level. In: Houghton, J.T., Y. Ding, D.J. Griggs, M. Noguer, P.J. Vander Linden, X. Dai, K. Maskell, C.A. Johnson CA (eds.) Climate change 2001: the scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, p 639–694

Clapham, P.J. 1992. Age at attainment of sexual maturity in humpback whales, *Megaptera novaeangliae*. Can. J. Zool. 70: 1470-1472.

Clapham, P.J. and C.A. Mayo. 1990. Reproduction of humpback whales (*Megaptera* *novaengliae*) observed in the Gulf of Maine. Rep. Int. Whal. Commn. Special Issue 12: 171-175.

Clapham, P.J., S.B. Young, and R.L. Brownell. 1999. Baleen whales: Conservation issues and the status of the most endangered populations. Mammal Rev. 29(1): 35-60.

Clapham, P., S. Brault, H. Caswell, M. Fujiwara, S. Kraus, R. Pace and P. Wade. 2002. Report of the Working Group on Survival Estimation for North Atlantic Right Whales.

Clark, C.W. 1995. Application of U.S. Navy underwater hydrophone arrays for scientific research on whales. Rep. Int. Whal. Commn. 45: 210-212.

Cliffton, K., D.O. Cornejo, and R.S. Felger. 1982. Sea turtles of the Pacific coast of Mexico. Pages 199-209 in K.A. Bjorndal, ed. Biology and Conservation of Sea Turtles. Washington, D.C.: Smithsonian Institution Press.

Cole, T.V.N., D.L. Hartly, and R.L. Merrick. 2005. Mortality and serious injury determinations for large whale stocks along the eastern seaboard of the United States, 1999-2003. U. S. Dep. Commer., Northeast Fish. Sci. Cent. Ref. Doc. 05-08. 20 pp.

Collins, M. R. and T. I. J. Smith. 1997. Distribution of shortnose and Atlantic sturgeons in South Carolina. North American Journal of Fisheries Management. 17: 995-1000.

Collins, M.R., T.I.J. Smith, W.C. Post, and O. Pashuk. 2000. Habitat Utilization and Biological Characteristics of Adult Atlantic Sturgeon in Two South Carolina Rivers. Transactions of the American Fisheries Society 129: 982–988.

Conant, T.A., P.H. Dutton, T. Eguchi, S.P. Epperly, C.C. Fahy, M.H. Godfrey, S.L. MacPherson, E.E. Possardt, B.A. Schroeder, J.A. Seminoff, M.L. Snover, C.M. Upite, and B.E. Witherington. 2009. Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the U.S. Endangered Species Act. Report of the Loggerhead Biological Review Team to the National Marine Fisheries Service, August 2009. 222 pp.

Coyne, M.S. 2000. Population Sex Ratio of the Kemp's Ridley Sea Turtle (*Lepidochelys kempii*): Problems in Population Modeling. PhD Thesis, Texas A&M University. 136 pp.

Coyne, M. and A.M. Landry, Jr. 2007. Population sex ratios and its impact on population models. In: Plotkin, P.T. (editor). Biology and Conservation of Ridley Sea Turtles. Johns Hopkins University Press, Baltimore, Maryland. p. 191-211.

Crance, J. H. 1987. Habitat suitability index curves for anadromous fishes. *In*: Common Strategies of Anadromous and Catadromous Fishes, M. J. Dadswell (ed.). Bethesda, Maryland, American Fisheries Society. Symposium 1: 554.

Dadswell, M. J., B. D. Taubert, T. S. Squiers, D. Marchette, and J. Buckley. 1984.

Synopsis of Biological Data on Shortnose Sturgeon, *Acipenser brevirostrum*, LeSuer 1818.

Dadswell, M. 2006. A review of the status of Atlantic sturgeon in Canada, with comparisons to populations in the United States and Europe. Fisheries 31: 218-229.

Damon-Randall, K. *et al.* 2012a. Composition of Atlantic sturgeon in rivers, estuaries and marine waters. March 2012. Report from the August 10-11, 2011 workshop on the distribution of Atlantic sturgeon in the Northeast. US Dept of Commerce. 32pp. NMFS NERO Protected Resources Division.Available from: NMFS NERO PRD, 55 Great Republic Drive, Gloucester, MA 01930.

Damon-Randall, K. 2012b. Memorandum to the Record regarding population estimates for Atlantic sturgeon. March 7, 2012. 8 pp.

Daniels, R.C., T.W. White, and K.K. Chapman. 1993. Sea-level rise: destruction of threatened and endangered species habitat in South Carolina. Environmental Management 17(3): 373-385.

Davenport, J., and G.H. Balazs. 1991. 'Fiery bodies' – Are pyrosomas an important component of the diet of leatherback turtles? British Herpetological Society Bulletin 37: 33-38.

Davenport, J. 1997. Temperature and the life-history strategies of sea turtles. J Therm Biol 22: 479–488.

Dees, L. T. 1961. Sturgeons. United States Department of the Interior Fish and Wildlife Service, Bureau of Commercial Fisheries, Washington, D.C.

Defra. 2003. Preliminary investigation of the sensitivity of fish to sound generated by aggregate dredging and marine construction. Project AE0914 Final Report. Defra / Department for Environment, Food And Rural Affairs, London, UK

DFO (Fisheries and Oceans Canada). 2011. Atlantic sturgeon and shortnose sturgeon. Fisheries and Oceans Canada, Maritimes Region. Summary Report. U.S. Sturgeon Workshop, Alexandria, VA, 8-10 February, 2011. 11 pp.

Diaz, R.J., C.O. Tallent and J.A. Nestlerode. 2006. Benthic Resources and Habitats at the Sandbridge Borrow Area: A Test of Monitoring Protocols. In: MMS OCS Study 2005-056. Field Testing of a Physical/Biological Monitoring Methodology for Offshore Dredging and Mining Operations. Pgs 1 – 49.

Dodd, C.K. 1988. Synopsis of the biological data on the loggerhead sea turtles (*Caretta caretta* (Linnaeus 1758)). U.S. Fish and Wildlife Service, Biological Report 88 (14).

Dodd, M. 2003. Northern Recovery Unit - Nesting Female Abundance and Population

Trends. Presentation to the Atlantic Loggerhead Sea Turtle Recovery Team, April 2003.

Doksaeter, L. *et al*. 2009. Behavioral responses of herring (*Clupea harengus*) to 1-2 and 6-7 kHz sonar signals and killer whale feeding sounds. Journal of the Acoustical Society of America, 125(1): 554-564.

Donovan, G.P. 1991. A review of IWC stock boundaries. Rep. Int. Whal. Comm., Spec. Iss. 13: 39-63.

Doughty, R.W. 1984. Sea turtles in Texas: A forgotten commerce. Southwestern Historical Quarterly. pp. 43-70.

Dovel, W. L. and T. J. Berggren. 1983. Atlantic sturgeon of the Hudson River estuary, New York. New York Fish and Game Journal 30: 140-172.

Dunton *et al.* 2010. Abundance and distribution of Atlantic sturgeon (*Acipenser oxyrinchus*) within the Northwest Atlantic Ocean, determined from five fishery-independent surveys. *Fish. Bull. 108(4): 450–465.*

Durbin, E, G. Teegarden, R. Campbell, A. Cembella, M.F. Baumgartner, B.R. Mate. 2002. North Atlantic right whales, *Eubalaena glacialis*, exposed to Paralytic Shellfish Poisoning (PSP) toxins via a zooplankton vector, *Calanus finmarchicus*. Harmful Algae 1: 243-251.

Dutton, P.H., B.W. Bowen, D.W. Owens, A. Barragan, and S.K. Davis. 1999. Global phylogeography of the leatherback turtle (Dermochelys coriacea). Journal of Zoology 248: 397-409.

Dutton, P.H., C. Hitipeuw, M. Zein, S.R. Benson, G. Petro, J. Pita, V. Rei, L. Ambio, and J. Bakarbessy. 2007. Status and genetic structure of nesting populations of leatherback turtles (*Dermochelys coriacea*) in the Western Pacific. Chelonian Conservation and Biology 6(1): 47-53.

Dwyer, K.L., C.E. Ryder, and R. Prescott. 2002. Anthropogenic mortality of leatherback sea turtles in Massachusetts waters. Poster presentation for the 2002 Northeast Stranding Network Symposium.

Eckert, S.A. 1999. Global distribution of juvenile leatherback turtles. Hubbs Sea World Research Institute Technical Report 99-294.

Eckert, S.A. and J. Lien. 1999. Recommendations for eliminating incidental capture and mortality of leatherback sea turtles, *Dermochelys coriacea*, by commercial fisheries in Trinidad and Tobago. A report to the Wider Caribbean Sea Turtle Conservation Network (WIDECAST). Hubbs-Sea World Research Institute Technical Report No. 2000-310, 7 pp.

Eckert, S.A., D. Bagley, S. Kubis, L. Ehrhart, C. Johnson, K. Stewart, and D. DeFreese. 2006. Internesting and postnesting movements of foraging habitats of leatherback sea turtles (*Dermochelys coriacea*) nesting in Florida. Chel. Cons. Biol. 5(2): 239-248.

Ehrhart, L.M., D.A. Bagley, and W.E. Redfoot. 2003. Loggerhead turtles in the Atlantic Ocean: geographic distribution, abundance, and population status. Pp. 157-174 In: Bolten, A.B. and B.E. Witherington (eds.). Loggerhead Sea Turtles. Smithsonian Institution Press, Washington D.C.

Ehrhart. L.M., W.E. Redfoot, and D.A. Bagley. 2007. Marine turtles of the central region of the Indian River Lagoon System, Florida. Florida Scientist 70(4): 415-434.

Elliott, W. and M. Simmonds. 2007. Whales in Hot Water? The impact of a changing climate on whales, dolphins and porpoises: A call for action. WWF-International, Gland Switzerland/WDCS, Chippenham, UK. 14 pp.

Encyclopedia Britannica. 2008. Neritic Zone Defined. Retrieved March 8, 2008, from Encyclopedia Britannica Online: http://www.britannica.com/eb/article-9055318.

Environmental Protection Agency (EPA). 1986. Quality Criteria for Water. EPA 440/5-86-001.

Epperly, S.P. 2003. Fisheries-related mortality and turtle excluder devices. In: P.L. Lutz, J.A. Musick, and J. Wyneken (editors). The Biology of Sea Turtles Vol. II, CRC Press, Boca Raton, Florida. p. 339-353.

Epperly, S.P. and W.G. Teas. 2002. Turtle Excluder Devices - Are the escape openings large enough? Fish. Bull. 100: 466-474.

Epperly, S.P., J. Braun, and A.J. Chester. 1995a. Aerial surveys for sea turtles in North Carolina inshore waters. Fishery Bulletin 93: 254-261.

Epperly, S.P., J. Braun, A.J. Chester, F.A. Cross, J.V. Merriner and P.A. Tester. 1995b. Winter distribution of sea turtles in the vicinity of Cape Hatteras and their interactions with the summer flounder trawl fishery. Bull. of Marine Sci. 56(2): 547-568.

Epperly, S.P., J. Braun, and A. Veishlow. 1995c. Sea turtles in North Carolina waters. Cons. Biol. 9(2): 384-394.

Epperly, S., L. Avens, L. Garrison, T. Henwood, W. Hoggard, J. Mitchell, J. Nance, J. Poffenberger, C. Sasso, E. Scott-Denton, and C. Yeung. 2002. Analysis of sea turtle bycatch in the commercial shrimp fisheries if southeast U.S. waters and the Gulf of Mexico. U.S. Department of Commerce, NOAA Tech. Memo. NMFS-SEFSC-490, 88pp.

Epperly, S.P. and J. Braun-McNeill. 2002. The use of AVHRR imagery and the management of sea turtle interactions in the Mid-Atlantic Bight. National Marine Fisheries Service, Southeast Fisheries Science Center, Miami, FL. 8pp.

Epperly, S.P., J. Braun-McNeill, and P.M. Richards. 2007. Trends in catch rates of sea turtles in North Carolina, USA. Endangered Species Research 3: 283-293.

Ehrhart. L.M., W.E. Redfoot, and D.A. Bagley. 2007. Marine turtles of the central region of the Indian River Lagoon System, Florida. Florida Scientist 70(4): 415-434.

Erickson *et al.* 2011. Use of pop-up satellite archival tags to identify oceanic-migratory patterns for adult Atlantic Sturgeon, Acipenser oxyrinchus oxyrinchus Mitchell, 1815. *J. Appl. Ichthyol. 27: 356–365.*

Ernst, C.H. and R.W. Barbour. 1972. Turtles of the United States. Univ. Press of Kentucky, Lexington. 347 pp.

Eyler, S., M. Mangold, and S. Minkkinen. 2004. Atlantic coast sturgeon tagging database. USFWS, Maryland Fishery Resources Office. Summary Report. 60 pp.

Fernandes, S.J., G. B. Zydlewski, J. D. Zydlewski, G. S. Wippelhauser, and M. T. Kinnison. 2010. Seasonal Distribution and Movements of Shortnose Sturgeon and Atlantic Sturgeon in the Penobscot River Estuary, Maine. Transactions of the American Fisheries Society 139: 1,436–1,449.

Ferreira, M.B., M. Garcia, and A. Al-Kiyumi. 2003. Human and natural threats to the green turtles, *Chelonia mydas*, at Ra's al Hadd turtle reserve, Arabian Sea, Sultanate of Oman. Page 142 in J.A. Seminoff, compiler. Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503.

Finkbeiner, E.M., B.P. Wallace, J.E. Moore, R.L. Lewison, L.B. Crowder, and A.J. Read. 2011. Cumulative estimates of sea turtle bycatch and mortality in USA fisheries between 1990 and 2007. Biological Conservation 144(11): 2,719-2,727.

Fish, M.R., I.M. Cote, J.A. Gill, A.P. Jones, S. Renshoff, A.R.Watkinson. 2005. Predicting the impact of sea-level rise on Caribbean sea turtle nesting habitat. Conserv Biol 19: 482-491.

Fisher, M.T. 2009. Atlantic Sturgeon progress report state wildlife grant, Project T-4-1. Delaware

Division of Fish and Wildlife, Department of Natural Resources and Environmental Control. FPL (Florida Power and Light Company) and Quantum Resources. 2005. Florida Power and Light Company, St. Lucie Plant Annual Environmental Operating Report, 2002. 57 pp.

Frasier, T.R., B.A. McLeod, R.M. Gillett, M.W. Brown and B.N. White. 2007. Right Whales Past and Present as Revealed by Their Genes. Pp 200-231. *In*: S.D. Kraus and R.M. Rolland (eds) *The Urban Whale*. Harvard University Press, Cambridge, Massachusetts, London, England. vii-xv + 543 pp.

Frazer, N.B., and L.M. Ehrhart. 1985. Preliminary growth models for green, *Chelonia mydas*, and loggerhead, *Caretta caretta*, turtles in the wild. Copeia 1985: 73-79.

Fritts, T.H. 1982. Plastic bags in the intestinal tracts of leatherback marine turtles. Herpetological Review 13(3): 72-73.

Fujiwara, M. and H. Caswell. 2001. Demography of the endangered North Atlantic right whale. Nature 414: 537-541.

Gagosian, R.B. 2003. Abrupt climate change: should we be worried? Prepared for a panel on abrupt climate change at the World Economic Forum, Davos, Switzerland, January 27, 2003. 9 pp.

Gambell, R. 1993. International management of whales and whaling: an historical review of the regulation of commercial and aboriginal subsistence whaling. Arctic 46: 97-107.

Garner, J.A, and S.A. Garner. 2007. Tagging and nesting research of leatherback sea turtles (*Dermochelys coriacea*) on Sandy Point St. Croix, U.S. Virgin Islands. Annual Report to U.S. Fish and Wildlife Service. WIMARCS Publication.

Garrison, L.P., L. Stokes, and C. Fairfield. 2009. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2008. NOAA Technical Memorandum NMFS-SEFSC-591: 1-58.

Garrison, L.P., L. Stokes, and C. Fairfield. 2010. Estimated bycatch of marine mammals and sea turtles in the U.S. Atlantic pelagic longline fleet during 2009. NOAA Technical Memorandum NMFS-SEFSC-607: 1-57.

Garrison, L.P. and Stokes, L. 2011a. Preliminary estimates of protected species bycatch rates in the U.S. Atlantic pelagic longline fishery from 1 January to 30 June, 2010. National Marine Fisheries Service, Southeast Fisheries Science Center, Miami, FL, SEFSC Contribution 124 #PRD-2010-10, Revised April 2011, 20 p.

Garrison, L.P. and Stokes, L. 2011b. Preliminary estimates of protected species bycatch rates in the U.S. Atlantic pelagic longline fishery from 1 July to 31 December, 2010. National Marine Fisheries Service, Southeast Fisheries Science Center, Miami, FL, SEFSC Contribution # PRD-2011-03, May 2011, 22 p.

George, R.H. 1997. Health Problems and Diseases of Sea Turtles. Pages 363-386 in P.L. Lutz and J.A. Musick, eds. The Biology of Sea Turtles. Boca Raton, Florida: CRC Press.

Geraci, Joseph R., Daniel M. Anderson, R.J. Timperi, David J. St. Aubin, Gregory A. Early, John H.Prescott, and Charles A. Mayo. 1989. Humpback Whales (*Megaptera novaeangliae*) Fatally Poisoned by Dinoflagellate Toxin. Can. J. Fish. and Aquat. Sci. 46(11): 1,895-1,898.

GHD. (2005). Port of Hay Point Apron Areas and Departure Path Capital Dredging: Draft EIS. GHD Pty Ltd.

Gilbert, C.R. 1989. Atlantic and shortnose sturgeons. United States Department of Interior Biological Report 82, 28 pp.

Girondot, M. and J. Fretey. 1996. Leatherback turtles, *Dermochelys coriacea*, nesting in French Guiana 1978-1995. Chelonian Conserv Biol 2: 204–208.

Girondot, M., M.H. Godfrey, L. Ponge, and P. Rivalan. 2007. Modeling approaches to quantify leatherback nesting trends in French Guiana and Suriname. Chelonian Conservation and Biology 6(1): 37-46.

Glass, A. H., T. V. N. Cole, M. Garron, R. L. Merrick, and R. M. Pace III. 2008. Mortality and Serious Injury Determinations for Baleen Whale Stocks Along the United States Eastern Seaboard and Adjacent Canadian Maritimes, 2002-2006. Northeast Fisheries Science Center Document 08-04; 18 pp.

Glass AH, Cole TVN, Garron M. 2009. Mortality and serious injury determinations for baleen whale stocks along the United States eastern seaboard and adjacent Canadian Maritimes, 2003-2007 (2nd Edition). US Dep Commer, Northeast Fish Sci Cent Ref Doc. 09-04; 19 p.

Glass A, Cole TVN, Garron M. 2010. Mortality and Serious Injury Determinations for Baleen Whale Stocks along the United States and Canadian Eastern Seaboards, 2004-2008. NOAA Technical Memorandum NMFS NE 214 19 p. Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026, or online at http://www.nefsc.noaa.gov/nefsc/publications/

Glen, F., A.C. Broderick, B.J. Godley, and G.C. Hays. 2003. Incubation environment affects phenotype of naturally incubated green turtle hatchlings. Journal of the Marine Biological Association of the United Kingdom 83(5): 1,183-1,186.

Glen, F. and N. Mrosovsky. 2004. Antigua revisited: the impact of climate change on sand and nest temperatures at a hawksbill turtle (*Eretmochelys imbricata*) nesting beach. Global Change Biology 10: 2,036-2,045.

GMFMC (Gulf of Mexico Fishery Management Council). 2007. Amendment 27 to the Reef

Fish FMP and Amendment 14 to the Shrimp FMP to end overfishing and rebuild the red snapper stock. Tampa, Florida: Gulf of Mexico Fishery Management Council. 490 pp. with appendices.

Goff, G.P. and J.Lien. 1988. Atlantic leatherback turtle, *Dermochelys coriacea*, in cold water off Newfoundland and Labrador. Can. Field Nat. 102(1): 1-5.

Goldenberg, S.B., C.W. Landsea, A.M. Mestas-Nunez, W.M. Gray. 2001. The recent increase in Atlantic hurricane activity: causes and implications. Science 293: 474–479

Graff, D. 1995. Nesting and hunting survey of the turtles of the island of S□o Tomé. Progr Report July 1995, ECOFAC Componente de S□o Tomé e Príncipe, 33 pp.

Greene, C.R. 1985a. Characteristics of waterborne industrial noise, 1980-1984. p. 197-253 In: W.J. Richardson (ed.), Behavior, disturbance responses and distribution of bowhead whales *Balaena mysticetus* in the eastern Beaufort Sea, 1980-1984. OCS Study MMS 85-0034. Rep. from LGL Ecol. Res. Assoc. Inc., Bryan, TX, for U.S. Minerals Management Service, Reston, Virgina. 306 p. NTIS PB87-124376.

Greene, R.J. Jr. 1987. Characteristics of oil industry dredge and drilling sounds in the Beaufort Sea. Journal of Acoustical Society of America 82: 1,315-1,324.

Greene, K. 2002. Beach Nourishment: A Review of the Biological and Physical Impacts. Atlantic States Marine Fisheries Commission (ASMFC) Habitat Management Series #7. 179 pp.

Greene, C.H., A.J. Pershing, R.D. Kenney, and J.W. Jossi. 2003. Impact of climate variability on the recover of endangered North Atlantic right whales. Oceanography 16: 96-101.

Greene, C.H and A.J. Pershing. 2004. Climate and the conservation biology of North Atlantic right whales: the right whale at the wrong time? Frontiers in Ecology and the Environment 2(1): 29-34.

Greene, C.H., A.J. Pershing, T. M. Cronin, and N. Ceci1. 2008. ARCTIC CLIMATE CHANGE AND ITS IMPACTS ON THE ECOLOGY OF THE NORTH ATLANTIC. Ecology 89(11) Supplement: S24-S38

Guerra-Garcia, J.M. and J. C. Garcia-Gomez. 2006. Recolonization of defaunated sediments: Fine versus gross sand and dredging versus experimental trays. Estuarine Coastal and Shelf Science 68 (1-2): 328-342

Guilbard, F., J. Munro, P. Dumont, D. Hatin, and R. Fortin. 2007. Feeding ecology of Atlantic sturgeon and Lake sturgeon co-occurring in the St. Lawrence Estuarine Transition Zone. American Fisheries Society Symposium. 56: 85-104.

Hain, J.H.W., M.J. Ratnaswamy, R.D. Kenney, and H.E. Winn. 1992. The fin whale, *Balaenoptera physalus*, in waters of the northeastern United States continental shelf. Rep. Int. Whal. Comm. 42: 653-669.

Hall, J.D., M. Gallagher, K. Brewer, P. Regos, and P. Isert. 1994. 1993 Kuvlum Exploration Area Site Specific Monitoring Program. Prepared for ARCO Alaska, Inc., Anchorage, AK, by Coastal and Offshore Pacific Corporation, Walnut Creek, CA.

Hamann, M., C.J. Limpus, and M.A. Read. 2007. Chapter 15 Vulnerability of marine reptiles in the Great Barrier Reef to climate change. *In:* Johnson JE, Marshall PA (eds) Climate change and the Great Barrier Reef: a vulnerability assessment, Great Barrier Reef Marine Park Authority and Australia Greenhouse Office, Hobart, p 465–496.

Hamilton, P.K., and C.A. Mayo. 1990. Population characteristics of right whales (*Eubalaena glacialis*) observed in Cape Cod and Massachusetts Bays, 1978-1986. Reports of the International Whaling Commission, Special Issue No. 12: 203-208.

Hamilton, P.K., M.K. Marx, and S.D. Kraus. 1998. Scarification analysis of North Atlantic right whales (*Eubalaena glacialis*) as a method of assessing human impacts. Final report to the Northeast Fisheries Science Center, NMFS, Contract No. 4EANF-6-0004.

Hamilton, P.K., A. R. Knowlton, and S. D. Kraus. 2008. Maintenance of the North Atlantic right whale catalog: 1 January-31 December 2007. Final report to the Northeast Fisheries Science Center, NMFS, Contract No. EA133F-05-CN-1231. Edgerton Research Laboratory, New England Aquarium: 27 pp.

Hammond, P.S., K. Macleod, L. Burt, A. Canadas, S. Lens, B. Mikkelsen, E. Rogan, B. Santos, A. Uriarte, O. Van Canneyt, and J.A. Vazquez. 2011. Abundance of baleen whales in the European Atlantic. IWC SC/63/RMP24.

Hatase, H., M. Kinoshita, T. Bando, N. Kamezaki, K. Sato, Y. Matsuzawa, K. Goto, K . Omuta, Y. Nakashima, H. Takeshita, and W. Sakamoto. 2002. Population structure of loggerhead turtles, *Caretta caretta*, nesting in Japan: Bottlenecks on the Pacific population. Marine Biology 141: 299-305.

Hatin, D., R. Fortin, and F. Caron. 2002. Movements and aggregation areas of adult Atlantic sturgeon (*Acipenser oxyrinchus*) in the St. Lawrence River estuary, Québec, Canada. Journal of Applied Ichthyology 18: 586-594.

Hawkes, L. A. Broderick, M. Godfrey and B. Godley. 2005. Status of nesting loggerhead turtles, *Caretta caretta,* at Bald Head Island (North Carolina, USA) after 24 years of intensive monitoring and conservation. Oryx. 39(1): 65-72.

Hawkes, L.A., A.C. Broderick, M.S. Coyne, M.H. Godfrey, L.-F. Lopez-Jurado, P. Lopez-Suarez, S.E. Merino, N. Varo-Cruz, and B.J. Godley. 2006. Phenotypically linked dichotomy in sea turtle foraging requires multiple conservation approaches. Current Biology 16: 990-995.

Hawkes, L.A., A.C. Broderick, M.H. Godfrey, and B.J. Godley. 2007. Investigating the potential impacts of climate change on a marine turtle population. Global Change Biology 13: 1-10.

Hawkes, L.A., A.C. Broderick, M.H. Godfrey, and B.J. Godley. 2009. Climate change and marine turtles. Endangered Species Research 7: 137-159.

Hays, G.C., A.C. Broderick, F. Glen, B.J. Godley, J.D.R. Houghton, and J.D. Metcalfe. 2002. Water temperature and internesting intervals for loggerhead (*Caretta caretta*) and green (*Chelonia mydas*) sea turtles. Journal of Thermal Biology 27: 429-432.

Hays, G.C., S. Akesson, A.C. Broderick, F. Glen, B.J. Godley, F. Papi, P. Luschi. 2003. Island finding ability of marine turtles. Proc. R. Soc. B. Suppl. Biol. Lett. 270: S5–S7.

Heppell, S.S., D.T. Crouse, L.B. Crowder, S.P. Epperly, W. Gabriel, T. Henwood, R. Marquez, and N.B. Thompson. 2005. A population model to estimate recovery time, population size, and management impacts on Kemp's ridley sea turtles. Chelonian Conservation and Biology 4(4): 767-773.

Hildebrand, H. 1982. A historical review of the status of sea turtle populations in the western Gulf of Mexico, P. 447-453. In K.A. Bjorndal (ed.), Biology and conservation of sea turtles. Smithsonian Institution Press, Washington, D.C.

Hildebrand, S.F. and W.C. Schroder. 1928. Fishes of Chesapeake Bay. U.S. Bureau of Fisheries Bulletin 43.

Hilterman, M.L. and E. Goverse. 2004. Annual report of the 2003 leatherback turtle research and monitoring project in Suriname. World Wildlife Fund - Guianas Forests and Environmental Conservation Project (WWF-GFECP) Technical Report of the Netherlands Committee for IUCN (NC-IUCN), Amsterdam, the Netherlands, 21p.

Hirth, H.F. 1971. Synopsis of biological data on the green sea turtle, *Chelonia mydas.* FAO Fisheries Synopsis No. 85: 1-77.

Hirth, H.F. 1997. Synopsis of the biological data of the green turtle, Chelonia mydas (Linnaeus 1758). USFWS Biological Report 97(1): 1-120.

Holland, B.F., Jr. and G.F. Yelverton. 1973. Distribution and biological studies of anadromous fishes offshore North Carolina. North Carolina Department of Natural and Economic Resources, Division of Commercial and Sports Fisheries, Morehead City. Special Scientific Report 24: 1-132.

Holton, J. W., Jr. and J. B. Walsh. 1995. Long-term dredged material management plan for the upper James River, Virginia. Virginia Beach, Waterway Surveys and Engineering, Ltd. 94 pp.

Hulin, V. and J.M. Guillon. 2007. Female philopatry in a heterogenous environment: ordinary conditions leading to extraordinary ESS sex ratios. BMC Evol. Biol. 7:13.

Hulme, P.E. 2005. Adapting to climate change: is there scope for ecological management in the face of a global threat? Journal of Applied Ecology 42(5): 784-794.

IPCC (Intergovernmental Panel on Climate Change). 2007. IPCC (2007) Climate Change 2007: synthesis report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4_syr.pdf.

International Whaling Commission [IWC]. 1979. Report of the sub-committee on protected species. Annex G., Appendix I. Rep. Int. Whal. Comm. 29: 84-86.

International Whaling Commission [IWC]. 1986. Right whales: past and present status. Reports of the International Whaling Commission, Special Issue No. 10; Cambridge, England.

International Whaling Commission (IWC). 1992. Report of the comprehensive assessment special meeting on North Atlantic fin whales. Reports of the International Whaling Commission 42: 595-644.

International Whaling Commission [IWC]. 1995. Report of the Scientific Committee, Annex E. Rep. Int. Whal. Comm. 45: 121-138.

International Whaling Commission [IWC]. 1997. Report of the IWC workshop on climate change and cetaceans. *Report ofthe International Whaling Commission* 47: 293-313.

International Whaling Commission [IWC]. 2001a. Report of the workshop on the comprehensive assessment of right whales: A worldwide comparison. Reports of the International Whaling Commission. Special Issue 2.

International Whaling Commission [IWC]. 2001b. The IWC, Scientific Permits and Japan. Posted at http://www.iwcoffice.org/sciperms.htm.

Jahoda, M., C. L. Lafortuna, N. Biassoni, C. Almirante, A. Azzelino, S. Panigada, M. Zanardelli. 2003. Mediterranean fin whale's (*Balaenoptera physalus*) response to small vessels and biopsy sampling assessed through passive tracking and timing of respiration. Marine Mammal Science 19: 15.

James, M.C., R.A. Myers, and C.A. Ottenmeyer. 2005a. Behavior of leatherback sea turtles, *Dermochelys coriacea*, during the migratory cycle. Proc. R. Soc. B, 272: 1547-1555.

James, M.C., C.A. Ottensmeyer, and R.A. Myers. 2005b. Identification of high-use habitat and threats to leatherback sea turtles in northern waters: new directions for conservation. Ecol. Lett. 8: 195-201.

Jefferson, T.A., M.A. Webber, and R.L. Pitman. 2008. *Marine Mammals of the World, A Comprehensive Guide to their Identification*. Amsterdam, Elsevier. Pp. 47-50.

Jensen, A.S. and G.K. Silber. 2003. Large whale ship strike database. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-F/OPR 25, 37 p.

Johnson, J. H., D. S. Dropkin, B. E. Warkentine, J. W. Rachlin, and W. D. Andrews. 1997. Food habits of Atlantic sturgeon off the central New Jersey coast. Transactions of the American Fisheries Society 126: 166-170.

Johnson, M. P. & P.L. Tyack. 2003. A digital acoustic recording tag for measuring the response of wild marine mammals to sound. IEEE J. Oceanic Engng 28: 3–12.

Johnson, J.H.and A.A. Wolman. 1984. The humpback whale, *Megaptera novaengliae*. Mar. Fish. Rev. 46(4): 30-37.

Jones A.R., W. Gladstone, N.J. Hacking. 2007. Australian sandy beach ecosystems and climate change: ecology and management. Aust Zool 34: 190–202

Kahnle, A. W., K. A. Hattala, K. A. McKown, C. A. Shirey, M. R. Collins, T. S. Squiers, Jr., and T. Savoy. 1998. Stock status of Atlantic sturgeon of Atlantic Coast estuaries. Report for the Atlantic States Marine Fisheries Commission. Draft III.

Kahnle, A.W., K.A. Hattala, K.A. McKown. 2007. Status of Atlantic sturgeon of the Hudson River Estuary, New York, USA. American Fisheries Society Symposium. 56: 347-363.

Kasparek, M., B.J. Godley, and A.C. Broderick. 2001. Nesting of the green turtle, *Chelonia mydas*, in the Mediterranean: a review of status and conservation needs. Zoology in the Middle East 24: 45-74.

Kieffer, M. C. and B. Kynard. 1993. Annual movements of shortnose and Atlantic sturgeons in the Merrimack River, Massachusetts. Transactions of the American Fisheries Society 122: 1,088-1,103.

Kelle, L., N. Gratiot, I. Nolibos, J. Therese, R. Wongsopawiro, and B. DeThoisy. 2007. Monitoring of nesting leatherback turtles (*Dermochelys coriacea*): contribution of remote-sensing for real time assessment of beach coverage in French Guiana. Chelonian Conserv Biol 6: 142–149.

Kennebec River Resource Management Plan. 1993. Kennebec River resource management plan: balancing hydropower generation and other uses. Final Report to the Maine State Planning Office, Augusta, ME. 196 pp.

Kenney, R.D. 2000. Are right whales starving? Electronic newsletter of the Center for Coastal Studies, posted at www.coastalstudies.org/entanglementupdate/kenney1.html on November 29, 2000. 5 pp.

Kenney, R.D. 2001. Anomalous 1992 spring and summer right whale (*Eubalaena glacialis*) distribution in the Gulf of Maine. Journal of Cetacean Research and Management (special Issue) 2: 209-23.

Kenney, R.D. 2002. North Atlantic, North Pacific and Southern right whales, *Eubalaena glacialis, E. japonica and E. australis*. Pp 806-813 in Perrin et al., editors, Encyclopedia of Marine Mammals.

Kenney, R.D., M.A.M. Hyman, R.E. Owen, G.P. Scott, and H.E. Winn. 1986. Estimation of prey densities required by Western North Atlantic right whales. Mar. Mamm. Sci. 2(1): 1-13.

Kenney, R.D., H.E. Winn, and M.C. Macaulay. 1995. Cetaceans in the Great South Channel, 1979-1989: right whale (*Eubalaena glacialis*). Cont. Shelf. Res. 15: 385-414

Ketten, D.R. 1998. Marine mammal auditory systems: a summary of audiometric and anatomical data and its implications for underwater acoustic impacts. NOAA Technical Memorandum NMFS: NOAA-TM-NMFS-SWFSC-256.

Ketten, D.R. and S.M. Bartol. 2005. Functional Measures of Sea Turtle Hearing. ONR Award No: N00014-02-1-0510.

Khan, C., TVN Cole, P. Duley, A.H. Glass, M. Niemeyer, and C. Christman. 2009. North Atlantic Right Whale Sighting Survey (NARWSS) and Right Whale Sighting Advisory System (RWSAS) 2008 Results Summary. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 09-05; 7 p.

Khan, C., TVN Cole, P. Duley, A.H. Glass, and J. Gatzke. 2010. North Atlantic Right Whale Sighting Survey (NARWSS) and Right Whale Sighting Advisory System (RWSAS). Northeast Fish Sci Cent Ref Doc. 10-07; 6 p.

Khan, C., TVN Cole, P. Duley, A. Henry, J. Gatzke. 2011. North Atlantic Right Whale Sighting Survey (NARWSS) and Right Whale Sighting Advisory System (RWSAS) 2010 Results Summary. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 11-05; 6 p.

Knowlton, A. R., J. Sigurjonsson, J.N. Ciano, and S.D. Kraus. 1992. Long-distance movements of North Atlantic right whales (*Eubalaena glacialis*). Mar. Mamm. Sci. 8(4): 397-405.

Knowlton, A.R., S.D. Kraus, and R.D. Kenney. 1994. Reproduction in North Atlantic right whales (*Eubalaena glacialis*). Can. J. Zool. 72: 1297-1305.

Kraus, S.D., M.J. Crone, and A.R. Knowlton. 1988. The North Atlantic right whale. Pages 684-98 *in* W.J. Chandler, ed. Audbon wildlife report 1988/1989. Academic Press, San Diego, CA.

Kraus, S.D. 1990. Rates and potential causes of mortality in North Atlantic right whales (*Eubaleana glacialis*). Mar. Mamm. Sci. 6(4): 278-291.

Kraus, S.D., J. H. Prescott, and A. R. Knowlton. 1986. Wintering right whales (*Eubalaena glacialis*) along the Southeastern coast of the United, 1984-1986. New England Aquarium: 15pp. Kraus, S.D., M.J. Crone, and A.R. Knowlton. 1988. The North Atlantic right whale. Pages 684-98 *in* W.J. Chandler, ed. Audbon wildlife report 1988/1989. Academic Press, San Diego, CA.

Kraus, S.D., P.K. Hamilton, R.D. Kenney, A.R. Knowlton, and C.K. Slay. 2001. Reproductive parameters of the North Atlantic right whale. J. Cetacean Res. Manage. 2: 231-236.

Kraus, S.D., M.W. Brown, H. Caswell, C.W. Clark, M. Fujiwara, P.K. Hamilton, R.D. Kenney, A.R. Knowlton, S. Landry, C.A. Mayo, W.A. McLellan, M.J. Moore, D.P. Nowacek, D.A. Pabst, A.J. Read, R.M. Rolland. 2005. North Atlantic Right Whales in Crisis. *Science*, 309: 561-562.

Kraus S.D., R. M. Pace III and T.R. Frasier. 2007. High Investment, Low Return: The Strange Case of Reproduction in *Eubalaena Glacialis*. Pp 172-199. *In*: S.D. Kraus and R.M. Rolland (eds) *The Urban Whale*. Harvard University Press, Cambridge, Massachusetts, London, England. vii-xv + 543pp.

Kuller, Z. 1999. Current status and conservation of marine turtles on the Mediterranean coast of Israel. Marine Turtle Newsletter 86: 3-5.

Kynard, B., M. Horgan, M. Kieffer, and D. Seibel. 2000. Habitat used by shortnose sturgeon in two Massachusetts rivers, with notes on estuarine Atlantic sturgeon: A hierarchical approach. Transactions of the American Fisheries Society 129: 487-503.

Kynard, B. and M. Horgan. 2002. Ontogenetic behavior and migration of Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*, and shortnose sturgeon, *A. brevirostrum*, with notes on social behavior. Environmental Behavior of Fishes 63: 137-150.

LaCasella, E.L., P.H. Dutton, and S.P. Epperly. 2005. Genetic stock composition of loggerheads (*Caretta caretta*) encountered in the Atlantic northeast distant (NED) longline fishery using additional mtDNA analysis. Pages 302-303 *in* Frick M., A. Panagopoulou, A.F. Rees, and K. Williams (compilers). Book of Abstracts of the Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.

Lageux, C.J., C. Campbell, L.H. Herbst, A.R. Knowlton and B. Weigle. 1998. Demography of marine turtles harvested by Miskitu Indians of Atlantic Nicaragua. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-412: 90.

Laist, D.W., A.R. Knowlton, J.G. Mead, A.S. Collet, M. Podesta. 2001. Collisions between ships and whales. Marine Mammal Science 17(1): 35-75.

Lalli, C.M. and T.R. Parsons. 1997. Biological oceanography: An introduction – 2nd Edition. Pages 1-13. Butterworth-Heinemann Publications. 335 pp.

Laney, R.W., J.E. Hightower, B.R. Versak, M.F. Mangold, W.W. Cole Jr., and S.E. Winslow. 2007. Distribution, Habitat Use, and Size of Atlantic Sturgeon Captured during Cooperative Winter Tagging Cruises, 1988-2006. American Fisheries Society Symposium 56: 000-000.

Laurent, L., J. Lescure, L. Excoffier, B. Bowen, M. Domingo, M. Hadjichristophorou, L. Kornaraki, and G. Trabuchet. 1993. Genetic studies of relationships between Mediterranean and Atlantic populations of loggerhead turtle *Caretta caretta* with a mitochondrial marker. Comptes Rendus de l'Academie des Sciences (Paris), Sciences de la Vie/Life Sciences 316: 1,233-1,239.

Laurent, L., P. Casale, M.N. Bradai, B.J. Godley, G. Gerosa, A.C. Broderick, W. Schroth, B. Schierwater, A.M. Levy, D. Freggi, E.M. Abd El-Mawla, D.A. Hadoud, H.E. Gomati, M. Domingo, M. Hadjichristophorou, L. Kornaraki, F. Demirayak, and C. Gautier. 1998. Molecular resolution of the marine turtle stock composition in fishery bycatch: A case study in the Mediterranean. Molecular Ecology 7: 1529-1542.

Learmonth, J.A., C.D. MacLeod, M.B. Santos, G.J. Pierce, H.Q.P. Crick, and R.A. Robinson. 2006. Potential effects of climate change on marine mammals. Oceanogr Mar BioI Annu Rev 44: 431-464.

Leland, J. G., III. 1968. A survey of the sturgeon fishery of South Carolina. Bears Bluff Labs. No. 47, 27 pp.

Lenhardt, M.L. 1994. Seismic and very low frequency sound induced behaviors in captive loggerhead marine turtles (*Caretta caretta*). In Bjorndal, K.A., A.B. Bolten, D.A. Johnson, and P.J. Eliazar (Compilers) Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-351, 323 pp.

Lenhardt, M.L., S. Moein, and J. Musick. 1996. A method for determining hearing thresholds in marine turtles. Proceedings of the Fifteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-387, pp 160-162.

Lewison, R.L., L.B. Crowder, and D.J. Shaver. 2003. The impact of turtle excluder devices and fisheries closures on loggerhead and Kemp's ridley strandings in the western Gulf of Mexico. Conservation Biology 17(4): 1,089-1,097.

Lewison, R.L., S.A. Freeman, and L.B. Crowder. 2004. Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles. Ecology Letters. 7: 221-231.

Limpus, C.J. and D.J. Limpus. 2000. Mangroves in the diet of *Chelonia mydas* in Queensland, Australia. Mar Turtle Newsl 89: 13–15.

Limpus, C.J. and D.J. Limpus. 2003. Loggerhead turtles in the equatorial Pacific and southern Pacific Ocean: A species in decline. *In*: Bolten, A.B., and B.E. Witherington (eds.), Loggerhead Sea Turtles. Smithsonian Institution.

Lichter J, Caron H, Pasakarnis TS, Ridgers SL, Squiers TS Jr, Todd CS (2006) The ecological collapse and partial recovery of a freshwater tidal ecosystem. Northeastern Nat 13(2): 153–178.

Lutcavage, M.E. and P.L. Lutz. 1997. Diving Physiology. Pp. 277-296 *in* The Biology of Sea Turtles. P.L. Lutz and J.A. Musick (Eds). CRC Press.

Lutcavage, M.E. and P. Plotkin, B. Witherington, and P.L. Lutz. 1997. Human impacts on sea turtle survival, p.387-409. In P.L. Lutz and J.A. Musick, (eds.), The Biology of Sea Turtles, CRC Press, Boca Raton, Florida. 432 pp.

MacLeod, C.D. 2009. Global c1imate change, range changes and potential implications for the conservation of marine cetaceans: a review and synthesis. Endang Species Res 7: 125-136.

Magnuson, J.J., J.A. Bjorndal, W.D. DuPaul, G.L. Graham, D.W. Owens, C.H. Peterson, P.C.H. Prichard, J.I. Richardson, G.E. Saul, and C.W. West. 1990. Decline of Sea Turtles: Causes and Prevention. Committee on Sea Turtle Conservation , Board of Environmental Studies and Toxicology, Board on Biology, Commission of Life Sciences, National Research Council, National Academy Press, Washington, D.C. 259 pp.

Maier, P. P., A. L. Segars, M. D. Arendt, J. D. Whitaker, B. W. Stender, L. Parker, R. Vendetti, D. W. Owens, J. Quattro, and S. R. Murphy. 2004. Development of an index of sea turtle abundance based on in-water sampling with trawl gear. Final report to the National Marine Fisheries Service. 86 pp.

Malik, S., M. W. Brown, S.D. Kraus and B. N. White. 2000. Analysis of mitochondrial DNA diversity within and between North and South Atlantic right whales. Mar. Mammal Sci. 16: 545-558.

MALSF (Marine Aggregate Levy Sustainability Fund). 2009. A generic investigation into noise profiles of marine dredging in relation to the acoustic sensitivity of the marine fauna in UK waters with particular emphasis on aggregate dredging: Phase I Scoping and Review of key issues. MEPF Ref No.: MEPF 08/21.

Mangin, E. 1964. Croissance en Longueur de Trois Esturgeons d'Amerique du Nord: *Acipenser oxyrhynchus,* Mitchill, *Acipenser fulvescens*, Rafinesque, et *Acipenser brevirostris* LeSueur. Verh. Int. Ver. Limnology 15: 968-974.

Mansfield, K. L. 2006. Sources of mortality, movements, and behavior of sea turtles in Virginia. Chapter 5. Sea turtle population estimates in Virginia. pp.193-240. Ph.D. dissertation. School of Marine Science, College of William and Mary.

Mansfield, K.L., V.S. Saba, J.A. Keinath, and J.A. Musick. 2009. Satellite tracking reveals a dichotomy in migration strategies among juvenile loggerhead turtles in the Northwest Atlantic. Marine Biology 156: 2555–2570.

Marcano, L.A. and J.J. Alio-M. 2000. Incidental capture of sea turtles by the industrial shrimping fleet off northwestern Venezuela. U.S. department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-436: 107.

Marcovaldi, M.A., and M. Chaloupka. 2007. Conservation status of the loggerhead sea turtle in Brazil: an encouraging outlook. Endangered Species Research 3: 133-143.

Margaritoulis, D., R. Argano, I. Baran, F. Bentivegna, M.N. Bradai, J.A. Camiñas, P. Casale, G. De Metrio, A. Demetropoulos, G. Gerosa, B.J. Godley, D.A. Haddoud, J. Houghton, L. Laurent, and B. Lazar. 2003. Loggerhead turtles in the Mediterranean Sea: Present knowledge and conservation perspectives. Pages 175-198. *I*n: A.B. Bolten and B.E. Witherington (eds.) Loggerhead Sea Turtles. Smithsonian Books, Washington, D.C. 319 pp.

Márquez, R. 1990. FAO Species Catalogue, Vol. 11. Sea turtles of the world, an annotated and illustrated catalogue of sea turtle species known to date. FAO Fisheries Synopsis, 125. 81 pp.

Mate, B.M., S.L. Nieukirk, R. Mescar, and T. Martin. 1992. Application of remote sensing methods for tracking large cetaceans: North Atlantic right whales (*Eubalaena glacialis*). Final Report to the Minerals Management Service, Contract No. 14-12-0001-30411, 167 pp.

Mate, B.M., S.L. Nieukirk, and S.D. Kraus. 1997. Satellite monitored movements of the North Atlantic right whale. J. Wildl. Manage. 61: 1,393-1,405.

Martin, R.E. 1996. Storm impacts on loggerhead turtle reproductive success. Mar Turtle Newsl

73: 10–12.

Mazaris A.D., G. Mastinos, J.D. Pantis. 2009. Evaluating the impacts of coastal squeeze on sea turtle nesting. Ocean Coast Manag 52: 139–145.

Meyer, M., R.R. Fay, and A.N. Popper, A. N. 2010. Frequency tuning and intensity coding of sound in the auditory periphery of the lake sturgeon, *Acipenser fulvescens*. Journal of Experimental Biology 213: 1,567-1,578.

McClellan, C.M., and A.J. Read. 2007. Complexity and variation in loggerhead sea turtle life history. Biology Letters 3: 592-594.

McCord, J.W., M.R. Collins, W.C. Post, and T.J. Smith. 2007. Attempts to develop an index of abundance for age-1 Atlantic sturgeon in South Carolina, USA. Am. Fisheries Society Symposium 56: 397-403.

McMahon, C.R., and G.C. Hays. 2006. Thermal niche, large-scale movements and implications of climate change for a critically endangered marine vertebrate. Global Change Biology 12: 1,330-1,338.

Mellinger, D.K. 2004. A comparison of methods for detecting right whale calls. Canadian Acoustics 32: 55-65.

Meylan, A. 1982. Estimation of population size in sea turtles. In: K.A. Bjorndal (ed.) Biology and Conservation of Sea Turtles. Smithsonian Inst. Press, Wash. D.C. p 135-138.

Meylan, A., B.E. Witherington, B. Brost, R. Rivero, and P.S. Kubilis. 2006. Sea turtle nesting in Florida, USA: Assessments of abundance and trends for regionally significant populations of Caretta, Chelonia, and Dermochelys. pp 306-307. *In*: M. Frick, A. Panagopoulou, A. Rees, and K. Williams (compilers). 26th Annual Symposium on Sea Turtle Biology and Conservation Book of Abstracts.

Meylan, A., B. Schroeder, and A. Mosier. 1995. Sea turtle nesting activity in the state of Florida. Fla. Mar. Res. Publ. 52: 1-51.

Meyer M., Popper A.N. (2002) Hearing in "primitive" fish: brainstem responses to pure tone stimuli in the lake sturgeon, *Acipenser fulvescens*. Abst. Assn. Res. Otolaryngol. 25: 11-12.

Minton, G., T. Collins, C. Pomilla, K.P. Findlay, H. Rosenbaum, R. Baldwin, and R.L. Brownell Jr. 2008. *Megaptera novaeangliae* (Arabian Sea subpopulation). In 2010 IUCN Red List of Threatened Species, Version 2010.2. Accessed 23 August 2010. http://www.iucnredlist.org/apps/redlist/details/132835/0.

Mitchell, E., V.M. Kozicki, and R.R. Reeves. 1986. Sightings of right whales, *Eubalaena glacialis*, on the Scotian Shelf, 1966-1972. Reports of the International Whaling

Commission (Special issue) 10: 83-107.

Mitchell, G.H., R.D. Kenney, A.M. Farak, and R.J. Campbell. 2003. Evaluation of occurrence of endangered and threatened marine species in naval ship trial areas and transit lanes in the Gulf of Maine and offshore of Georges Bank. NUWC-NPT Technical Memo 02-121A. March 2003. 113 pp.

Mizroch, S.A. and A.E. York. 1984. Have pregnancy rates of Southern Hemisphere fin whales, *Balaenoptera physalus*, increased? Reports of the International Whaling Commission, Special Issue No. 6: 401-410.

Mohler, J. W. 2003. Culture manual for the Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*. U.S. Fish and Wildlife Service, Hadley, Massachusetts. 70 pp.

Monzón-Argüello, C., A. Marco., C. Rico, C. Carreras, P. Calabuig, and L.F. López-Jurado. 2006. Transatlantic migration of juvenile loggerhead turtles (*Caretta caretta*): magnetic latitudinal influence. Page 106 *in* Frick M., A. Panagopoulou, A.F. Rees, and K. Williams (compilers). Book of Abstracts of the Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.

Moore, JC and E. Clark. 1963. Discovery of Right Whales in the Gulf of Mexico. Science 141: 269.

Moore M.J., A.R., Knowlton, S.D. Kraus, W.A. McLellan, R.K. Bonde. 2004. Morphometry, gross morphology and available histopathology in North Atlantic right whale (*Eubalaena glacialis*) mortalities (1970–2002). Journal of Cetacean Research and Management 6(3): 199-214.

Moore, M.J., W.A. McLellan, P.Daous, R.K. Bonde and A.R. Knowlton. 2007. Right Whale Mortality: A Message from the Dead to the Living. Pp 358-379. *In*: S.D. Kraus and R.M. Rolland (eds) *The Urban Whale*. Harvard University Press, Cambridge, Massachusetts, London, England. vii-xv + 543 pp.

Morreale, S.J. and E.A. Standora. 1990. Occurrence, movement, and behavior of the Kemp's ridley and other sea turtles in New York waters. Annual report for the NYSDEC, Return A Gift To Wildlife Program, April 1989 - April 1990.

Morreale, S.J., and E.A. Standora. 1992. Occurrence, movement, and behavior of the Kemp's ridley and other sea turtles in New York waters. Annual report for the NYSDEC, Return A Gift To Wildlife Program, April 1991 - April 1992.

Morreale, S.J., and E.A. Standora. 1993. Occurrence, movement, and behavior of the Kemp's ridley and other sea turtles in New York waters. Okeanos Ocean Research Foundation Final Report April 1988-March 1993. 70 pp.

Morreale, S.J. and E.A. Standora. 1998. Early life stage ecology of sea turtles in northeastern U.S. waters. U.S. Dep. Commer. NOAA Tech. Mem. NOAA Fisheries-SEFSC-413, 49 pp.

Morreale, S.J., C.F. Smith, K. Durham, R.A. DiGiovanni, Jr., and A.A. Aguirre. 2005. Assessing health, status, and trends in northeastern sea turtle populations. Interim report - Sept. 2002 - Nov. 2004. Gloucester, Massachusetts: National Marine Fisheries Service.

Mrosovsky, N. 1981. Plastic jellyfish. Marine Turtle Newsletter 17: 5-6.

Mrosovsky, N., G.D. Ryan, M.C. James. 2009. Leatherback turtles: The menace of plastic. Marine Pollution Bulletin 58: 287-289.

Mueller-Blenke, C., McGregor, P.K., Gill, A.B., Andersson, M.H., Metcalfe, J., Bendall, V., Sigray, P., Wood, D. and Thomsen, F. (2010). Effects of pile driving noise on the behaviour of marine fish. COWRIE technical report. 31st March 2010. Ref: Fish 06-08.

Munro, J. 2007. Anadromous sturgeons: Habitats, threats, and management - synthesis and summary. Am. Fisheries Society Symposium 56: 1-15.

Murawski, S. A. and A. L. Pacheco. 1977. Biological and fisheries data on Atlantic Sturgeon, *Acipenser oxyrhynchus* (Mitchill). National Marine Fisheries Service Technical Series Report 10: 1-69.

Murdoch, P.S., J.S. Baron, and T.L. Miller. 2000. Potential effects of climate change on surface-water quality in North America. J. Amer. Water Resour. Assoc. 36: 347–366.

Murphy, T.M. and S.R. Hopkins. 1984. Aerial and ground surveys of marine turtle nesting beaches in the southeast region. United States Final Report to NMFS-SEFSC. 73pp.

Murphy, T.M., S.R. Murphy, D.B. Griffin, and C. P. Hope. 2006. Recent occurrence, spatial distribution and temporal variability of leatherback turtles (*Dermochelys coriacea*) in nearshore waters of South Carolina, USA. Chel. Cons. Biol. 5(2): 216-224.

Murray, K.T. 2004. Bycatch of sea turtles in the Mid-Atlantic sea scallop (*Placopecten magellanicus*) dredge fishery during 2003. NEFSC Reference Document 04-11; 25 pp.

Murray, K.T. 2006. Estimated average annual bycatch of loggerhead sea turtles (*Caretta caretta*) in U.S. Mid-Atlantic bottom otter trawl gear, 1996-2004. NEFSC Reference Document 06- 19; 26 pp.

Murray, K.T. 2007. Estimated bycatch of loggerhead sea turtles (*Caretta caretta*) in U.S. Mid-Atlantic scallop trawl gear, 2004-2005, and in sea scallop dredge gear, 2005. NEFSC Reference Document 07-04; 30 pp.

Murray, K.T. 2008. Estimated average annual bycatch of loggerhead sea turtles (*Caretta caretta*) in U.S. Mid-Atlantic bottom otter trawl gear, 1996-2004 (2nd edition). NEFSC Reference Document 08-20; 32 pp.

Murray, K.T. 2009a. Proration of estimated bycatch of loggerhead sea turtles in U.S. Mid-Atlantic sink gillnet gear to vessel trip report landed catch, 2002-2006. NEFSC Reference Document 09-19; 7 pp.

Murray, K.T. 2009b. Characteristics and magnitude of sea turtle bycatch in U.S. mid-Atlantic gillnet gear. Endangered Species Research 8: 211-224.

Murray, K.T. 2011. Sea turtle bycatch in the U.S. sea scallop (*Placopecten magellanicus*) dredge fishery, 2001–2008. Fish Res. 107: 137-146.

Musick, J,4., R.E. Jenkins, and N,B. Burkhead. 1994. Sturgeons, Family Acipenseridae. Pp. 183-190. *In* Freshwater Fishes of Virginia, Jenkins, R.E. and N.B. Burkhead (eds.). American Fisheries Society, Bethesda MD.

Musick, J.A. and C.J. Limpus. 1997. Habitat utilization and migration in juvenile sea turtles. Pp. 137-164 *In*: Lutz, P.L., and J.A. Musick, eds., The Biology of Sea Turtles. CRC Press, New York. 432 pp.

National Aeronautic and Space Administration (NASA). 2010. Draft Programmatic Environmental Impact Statement: Wallops Flight Facility Shoreline Restoration and Infrastructure Protection Program. Volume I of II. February.

NAST (National Assessment Synthesis Team). 2008. Climate Change Impacts on the United States: The Potential Consequences of Climate Variability and Change, US Global Change Research Program, Washington DC.

National Marine Fisheries Service (NMFS). 1991a. Final recovery plan for the humpback whale (*Megaptera novaeangliae*). Prepared by the Humpback Whale Recovery Team for the national Marine Fisheries Service, Silver Spring, Maryland. 105 pp.

NMFS. 1991b. Final recovery plan for the northern right whale (*Eubalaena glacialis*). Prepared by the Right Whale Recovery Team for the National Marine Fisheries Service. 86 pp.

NMFS. 1997. Endangered Species Act – Section 7 Consultation on the Atlantic Pelagic Fishery for Swordfish, Tuna, and Shark in the Exclusive Economic Zone (EEZ). NMFS Northeast Regional Office, Gloucester, Massachusetts.

NMFS. 1998a. Draft recovery plans for the fin whale (*Balaenoptera physalus*) and sei whale (*Balaenoptera borealis*). Prepared by R.R. Reeves, G.K. Silber, and P.M. Payne for the National Marine Fisheries Service, Silver Spring, Maryland. July 1998.

NMFS. 2002. Endangered Species Act – Section 7 Consultation on dredging in the Thimble Shoal Federal Navigation Channel and Atlantic Ocean Channel, Virginia. NMFS Northeast Regional Office, Gloucester, Massachusetts. April 25, 2002. 83 pp.

NMFS. 2002. Endangered Species Act Section 7 Consultation on Shrimp Trawling in the Southeastern United States, under the Sea Turtle Conservation Regulations and as Managed by the Fishery Management Plans for Shrimp in the South Atlantic and Gulf of Mexico. December 2.

NMFS. 2002b. Endangered Species Act Section 7 Consultation on Shrimp Trawling in the Southeastern United States, under the Sea Turtle Conservation Regulations and as Managed by the Fishery Management Plans for Shrimp in the South Atlantic and Gulf of Mexico. December 2, 2002.

NMFS. 2004. Endangered Species Act Section 7 Reinitiated Consultation on the Continued Authorization of the Atlantic Pelagic Longline Fishery under the Fishery Management Plan for Atlantic Tunas, Swordfish, and Sharks (HMS FMP). Biological Opinion. June 1, 2004.

NMFS. 2005. Recovery Plan for the North Atlantic Right Whale (*Eubalaena glacialis*). National Marine Fisheries Service, Silver Spring, MD.

NMFS. 2006. Review of the Status of the Right Whales in the North Atlantic and North Pacific Oceans. National Marine Fisheries Service, Washington, D.C. 62 pp.

NMFS. 2007. Potential Application of Vessel Quieting Technology on Large Commercial Vessels. Final Report of NOAA International Symposium. Silver Spring, MD.

NMFS. 2008b. Summary Report of the Workshop on Interactions Between Sea Turtles and Vertical Lines in Fixed-Gear Fisheries. M.L. Schwartz (ed.), Rhode Island Sea Grant, Narragansett, Rhode Island. 54 pp.

NMFS. 2011. Biennial Report to Congress on the Recovery Program for Threatened and Endangered Species, October 1, 2008 – September 30, 2010. Washington, D.C.: National Marine Fisheries Service. 194 pp.

NMFS *et al*. 2011. Bi-National Recovery Plan for the Kemp's Ridley Sea Turtle (Lepidochelys kempii). http://www.nmfs.noaa.gov/pr/pdfs/recovery/kempsridley_revision2.pdf

NMFS Northeast Fisheries Science Center (NEFSC). 2011. Preliminary summer 2010 regional abundance estimate of loggerhead turtles (*Caretta caretta*) in northwestern Atlantic Ocean continental shelf waters. U.S. Dept Commerce, Northeast Fisheries Science Center Reference Document 11-03; 33 pp.

NMFS Southeast Fisheries Science Center (SEFSC). 2001. Stock assessments of loggerheads and leatherback sea turtles and an assessment of the impact of the pelagic longline fishery on the loggerhead and leatherback sea turtles of the Western North Atlantic. U.S. Department of Commerce, National Marine Fisheries Service, Miami, FL, SEFSC Contribution PRD-00/01-08; Parts I-III and Appendices I-IV. NOAA Tech. Memo NMFS-SEFSC-455, 343 pp.

NMFS NEFSC. 2011. Summary of discard estimates for Atlantic sturgeon. Report prepared by Tim Miller and Gary Shepard, NEFSC Population Dynamics Branch, NMFS Northeast Fisheries Science Center. August 19, 2011.

NMFS SEFSC. 2009. An assessment of loggerhead sea turtles to estimate impacts of mortality reductions on population dynamics. NMFS SEFSC Contribution PRD-08/09-14. 45 pp.

NMFS and U.S. Fish and Wildlife Service (USFWS). 1991a. Recovery plan for U.S. population of loggerhead turtle. National Marine Fisheries Service, Washington, D.C. 64 pp.

NMFS and USFWS. 1991b. Recovery plan for U.S. population of Atlantic green turtle. National Marine Fisheries Service, Washington, D.C. 58 pp.

NMFS and USFWS. 1992. Recovery plan for leatherback turtles in the U.S. Caribbean, Atlantic, and Gulf of Mexico. National Marine Fisheries Service, Washington, D.C. 65 pp.

NMFS and USFWS. 1995. Status reviews for sea turtles listed under the Endangered Species Act of 1973. National Marine Fisheries Service, Silver Spring, Maryland. 139 pp.

NMFS and USFWS. 1998a. Recovery Plan for the U.S. Pacific Population of the Leatherback Turtle (*Dermochelys coriacea*). National Marine Fisheries Service, Silver Spring, Maryland.

NMFS and USFWS. 1998b. Recovery Plan for U.S. Pacific Populations of the Green Turtle (*Chelonia mydas*). Silver Spring, Maryland: National Marine Fisheries Service. 84 pp.

NMFS and USFWS. 2007a. Loggerhead sea turtle (*Caretta caretta*) 5-year review: Summary and Evaluation. National Marine Fisheries Service, Silver Spring, Maryland. 65pp.

NMFS and FWS. 2007b. Kemp's ridley sea turtle (*Lepidochelys kempii*) 5 year review: summary and evaluation. National Marine Fisheries Service, Silver Spring, Maryland. 50 pp.

NMFS and FWS. 2007c. Green sea turtle (*Chelonia mydas*) 5 year review: summary and evaluation. National Marine Fisheries Service, Silver Spring, Maryland. 102 pp.

NMFS and USFWS. 2007d. Leatherback sea turtle (*Dermochelys coriacea*) 5 year review: summary and evaluation. National Marine Fisheries Service, Silver Spring, Maryland. 79 pp.

NMFS and USFWS. 2008. Recovery plan for the Northwest Atlantic population of the loggerhead turtle (*Caretta caretta*), Second revision. Washington, D.C.: National Marine Fisheries Service. 325 pp.

National Oceanic and Atmospheric Administration (NOAA). 2008. High numbers of right whales seen in Gulf of Maine: NOAA researchers identify wintering ground and potential breeding ground. NOAA press release; December 31, 2008.

National Research Council (NRC). 1990. Decline of the Sea Turtles: Causes and Prevention. Committee on Sea Turtle Conservation. Natl. Academy Press, Washington, D.C. 259 pp.

NRC. 2003. Ocean noise and marine mammals. National Academy Press; Washington, D.C.

Nicholls, R.J. 1998. Coastal vulnerability assessment for sea level rise: evaluation and selection of methodologies for implementation. Technical Report R098002, Caribbean Planning for Adaption to Global Climate Change (CPACC) Project. Available at: www.cpacc.org.

Nicholls, R.J., P.P. Wong, V.R. Burkett, J.O. Codignotto, J.E. Hay, R.F. McLean, S. Ragoonaden and C.D. Woodroffe (2007). In: "Coastal systems and low-lying areas." Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden and C.E. Hanson, Eds., Cambridge University Press, Cambridge, UK, and New York, USA, pp. 315-356.

Niklitschek, E.J. and D.H. Secor. 2005. Modeling spatial and temporal variation of suitable nursery habitats for Atlantic sturgeon in the Chesapeake Bay. Estuarine, Coastal and Shelf Science 64: 135-148.

Niklitschek, E.S. and D. H. Secor . 2010. Dissolved oxygen temperature and salinity effects on the ecophysiology and survival of juvenile Atlantic sturgeon in estuarine waters: I. Laboratory Results. J. Exp. Mar. Biol. Ecol. 381, Suppl. 1: 150-160.

Nowacek, D.P., M. P. Johnson and P. L. Tyack. 2004. North Atlantic right whales (*Eubalaena glacialis*) ignore ships but respond to alerting stimuli. Proc. R. Soc. Lond. B **271**: 227–231.

Oakley, N. C. 2003. Status of shortnose sturgeon, *Acipenser brevirostrum*, in the Neuse River, North Carolina. Thesis. Department of Fisheries and Wildlife Science, North Carolina State University, Raleigh, NC.

Pace, R.M. III, S.D. Kraus, P.K. Hamilton and A.R. Knowlton. 2008. Life on the edge:

examining North Atlantic right whale population viability using updated reproduction data and survival estimates. 17th Biennial Meeting of the Society for Marine Mammalogy. South Africa.

Palka, D. 1995. Abundance estimate of the Gulf of Maine harbor porpoise. Rep. Int. Whal. Comm. (Special Issue) 16: 27-50.

Palka, D.L. 2006. Summer abundance estimates of cetaceans in US North Atlantic Navy Operating Areas. Northeast Fish. Sci. Cent. Ref. Doc. 06-03. 41 pp. http://www.nefsc.noaa.gov/nefsc/publications/crd/crd0603/crd0603.pdf

Palka , D. 2000. Abundance and distribution of sea turtles estimated from data collected during cetacean surveys. *In*: Bjorndal, K.A. and A.B. Bolten. Proceedings of a workshop on assessing abundance and trends for in-water sea turtle populations. U.S. Dep. Commer. NOAA Tech. Mem. NMFS-SEFSC-445, 83pp.

Palmer, M.A., C.A.R. Liermann, C. Nilsson, M. Florke, J. Alcamo et al. 2008. Climate change and the world's river basins: Anticipating management options. Frontiers in Ecology and the Environment 6: 81–89.

Parmesan, C., and G. Yohe. 2003. A globally coherent fingerprint of climate change impacts across natural systems. Nature 421: 37-42.

Patrician, M. R., I. S. Biedron, H. C. Esch, F. W. Wenzel, L. A. Cooper, A. H. Glass, and M. F. Baumgartner. 2009. Evidence of a North Atlantic right whale calf (*Eubalaena glacialis*) born in northeastern U.S.waters. Mar. Mamm. Sci. 25(2): 462–477.

Payne, P.M., J.R. Nicolas, L. O'Brien, and K.D. Powers. 1986. The distribution of the humpback whale on Georges Bank and in the Gulf of Maine in relation to densities of the sand eel (*Ammodytes americanus*). Fishery Bulletin 84 (2): 271-277.

Payne, K. and R.S. Payne. 1985. Large-scale changes over 17 years in songs of humpback whales in Bermuda. Z. Tierpsychol. 68: 89-114.

Payne, P.M., D.N. Wiley, S.B. Young, S. Pittman, P.J. Clapham, and J.W.Jossi. 1990. Recent fluctuations in the abundance of baleen whales in the southern Gulf of Maine in relation to changes in selected prey. Fish. Bull. 88 (4): 687-696.

Pearce, A.F. 2001. Contrasting population structure of the loggerhead turtle (*Caretta caretta*) using mitochondrial and nuclear DNA markers. Master's thesis, University of Florida. 71 pp.

Pearce, A.F. and B.W. Bowen. 2001. Final report: Identification of loggerhead (Caretta caretta) stock structure in the southeastern United States and adjacent regions using nuclear DNA markers. Project number T-99-SEC-04. Submitted to the National Marine Fisheries

Service, May 7, 2001. 79 pp.

Perry, S.L., D.P. DeMaster, and G.K. Silber. 1999. The great whales: History and status of six species listed as endangered under the U.S. Endangered Species Act of 1973. Mar. Fish. Rev. Special Edition. 61(1): 59-74.

Pike, D.A., R.L. Antworth, and J.C. Stiner. 2006. Earlier nesting contributes to shorter nesting seasons for the loggerhead sea turtle, *Caretta caretta*. Journal of Herpetology 40(1): 91-94.

Pike, D.A. and J.C. Stiner. 2007. Sea turtle species vary in their susceptibility to tropical cyclones. Oecologia 153: 471–478.

Pike, D.G., T. Gunnlaugsson, G.A. Víkingsson, and B. Mikkelsen. 2008. Estimates of the abundance of fin whales (*Balaenoptera physalus)* from the T-NASS Icelandic and Faroese ship surveys conducted in 2007. IWC SC/60/PFI13-revised.

Pikitch, E.K., P. Doukakis, L. Lauck, P. Chakrabarty, and D.L. Erickson. 2005. Status, trends and management of sturgeon and paddlefish fisheries. Fish and Fisheries 6: 233–265.

Plachta, D.T.T., and Popper, A.N. (2003). Evasive responses of American shad *(Alosa sapidissima)* to ultrasonic stimuli. Acoustical Research Letters Online 4:25-30.

Plaziat, J.C., and P.G.E.F. Augustinius. 2004. Evolution of progradation/ erosion along the French Guiana magrove coast: a comparison of mapped shorelines since the 18th century with Holocene data. Mar Geol 208: 127–143.

Pritchard, P.C.H. 1982. Nesting of the leatherback turtle, *Dermochelys coriacea*, in Pacific, Mexico, with a new estimate of the world population status. Copeia 1982: 741-747.

Pritchard, P.C.H. 2002. Global status of sea turtles: An overview. Document INF-001 prepared for the Inter-American Convention for the Protection and Conservation of Sea Turtles, First Conference of the Parties (COP1IAC), First part August 6-8, 2002.

Prusty, G., S. Dash, and M.P. Singh. 2007. Spatio-temporal analysis of multi-date IRS imageries for turtle habitat dynamics characterisation at Gahirmatha coast, India. Int J Remote Sens 28: 871–883

Pyzik, L., J. Caddiek, and P. Marx. 2004. Chesapeake Bay: Introduction to an ecosystem. EPA 903-R-04-003, CBP/TRS 232100. 35 pp.

Rahmstorf, S. 1997. Risk of sea-change in the Atlantic. Nature 388: 825–826.

Rahmstorf, S. 1999. Shifting seas in the greenhouse? Nature 399: 523–524.

Rankin-Baransky, K., C.J. Williams, A.L. Bass, B.W. Bowen, and J.R. Spotila. 2001. Origin of loggerhead turtles stranded in the northeastern United States as determined by mitochondrial DNA analysis. Journal of Herpetology 35(4):638-646.

Rebel, T.P. 1974. Sea turtles and the turtle industry of the West Indies, Florida and the Gulf of Mexico. Univ. Miami Press, Coral Gables, Florida.

Rees, A.F., A. Saad, and M. Jony. 2005. Marine turtle nesting survey, Syria 2004: discovery of a "major" green turtle nesting area. Page 38 in Book of Abstracts of the Second Mediterranean Conference on Marine Turtles. Antalya, Turkey, 4-7 May 2005.

Revelles, M., C. Carreras, L. Cardona, A. Marco, F. Bentivegna, J.J. Castillo, G. de Martino, J.L. Mons, M.B. Smith, C. Rico, M. Pascual, and A. Aguilar. 2007. Evidence for an asymmetrical size exchange of loggerhead sea turtles between the Mediterranean and the Atlantic through the Straits of Gibraltar. Journal of Experimental Marine Biology and Ecology 349: 261-271.

Richardson, W.J., M.A. Fraker, B. Wursig, and R.S. Wells. 1985. Behavior of bowhead whales *Balaena mysticetus* summering in the Beaufort Sea: Reactions to industrial activities. Biol. Conserv. 32: 195-230.

Richardson, W.J., B. Wuersig, and C.R. Green. 1990. Reactions of bowhead whales, *Balaena mysticetus*, to drilling and dredging noise in the Canadian Beaufort Sea. Marine Environmental Research 29: 135-160.

Richardson, W.J., C.I. Malme, C.R. Green, and D.H. Thomson. 1995. Marine Mammals and noise: Volume 1. Academic Press, San Diego California.

Richardson A.J., A. Bakun, G.C. Hays, and M.J. Gibbons. 2009. The jellyfish joyride: causes, consequences and management responses to a more gelatinous future. Trends in Ecology and Evolution 24: 312-322.

Rivalan, P., P.H. Dutton, E. Baudry, S.E. Roden, and M. Girondot. 2005. Demographic scenario inferred from genetic data in leatherback turtles nesting in French Guiana and Suriname. Biol Conserv 1: 1–9.

Robbins, J., and D. Mattila. 1999. Monitoring entanglement scars on the caudal peduncle of Gulf of Maine humpback whales. Report to the National Marine Fisheries Service. Order No. 40EANF800288. 15 pp.

Robbins, J., and Mattila, D. 2004. Estimating humpback whale (*Megaptera novaeangliae*) entanglement rates on the basis of scar evidence: Report to the Northeast Fisheries Science Center, National Marine Fisheries Service. Order number 43EANF030121. 21 p.

Robinson *et al*. 2008. Travelling through a warming world: climate change and migratory species. Endangered Species Research Preprint: 1-13.

Rochard, E., M. Lepage, and L. Meauzé. Identification et caractérisation de l'aire de répartition marine de l'esturgeon éuropeen *Acipenser sturio* a partir de déclarations de captures. 1997. Aquat. Living. Resour. 10: 101-109.

Rolland, R.M, K.E. Hunt, G.J. Doucette, L.G. Rickard and S. K. Wasser. 2007. The Inner Whale: Hormones, Biotoxins, and Parasites. Pp 232-272. *In*: S.D. Kraus and R.M. Rolland (eds) *The Urban Whale*. Harvard University Press, Cambridge, Massachusetts, London, England. vii-xv + 543 pp.

Ross, J.P. 1996. Caution urged in the interpretation of trends at nesting beaches. Marine Turtle Newsletter 74: 9-10.

Ross. J.P. 2005. Hurricane effects on nesting *Caretta caretta*. Mar Turtle Newsl 108: 13–14.

Ruben, H.J, and S.J. Morreale. 1999. Draft Biological Assessment for Sea Turtles in New York and New Jersey Harbor Complex. Unpublished Biological Assessment submitted to National Marine Fisheries Service.

Sarti, L., S.A. Eckert, N. Garcia, and A.R. Barragan. 1996. Decline of the world's largest nesting assemblage of leatherback turtles. Marine Turtle Newsletter 74: 2-5.

Sarti, L., S. Eckert, P. Dutton, A. Barragán, and N. García. 2000. The current situation of the leatherback population on the Pacific coast of Mexico and central America, abundance and distribution of the nestings: an update. Pages 85-87 *In*: H. Kalb and T. Wibbels, compilers. Proceedings of the Nineteenth Annual Symposium on Sea Turtle Conservation and Biology. NOAA Technical Memorandum NMFS-SEFSC-443.

Sarti Martinez, L., A.R. Barragan, D.G. Munoz, N. Garcia, P. Huerta, and F. Vargas. 2007. Conservation and biology of the leatherback turtle in the Mexican Pacific. Chelonian Conservation and Biology 6(1): 70-78.

Savoy, T. and D. Pacileo. 2003. Movements and important habitats of subadult Atlantic sturgeon in Connecticut waters. Transactions of the American Fisheries Society 132: 1-8.

Savoy, T. 2007. Prey eaten by Atlantic sturgeon in Connecticut waters. Am. Fisheries Society Symposium 56: 157-165.

Schaeff, C.M., Kraus, S.D., Brown, M.W., Perkins, J.S., Payne, R., and White, B.N. 1997. Comparison of genetic variability of North and South Atlantic right whales (*Eubalaena*), using DNA fingerprinting. Can. J. Zool. 75: 1,073-1,080.

Schevill, WE., WA Watkins, and KE Moore. 1986. Status of *Eubalaena glacialis* off Cape Cod. Report of the International Whaling Commission, Special Issue 10: 79-82.

Schick, Robert S., P.N. Halpin, A.J. Read, C.K. Slay, S.D. Kraus, B.R. Mate, M.F. Baumgartner, J.J. Roberts, B.D. Best, C.P. Good, S.R. Loarie, and J.S. Clark. 2009. Striking the right balance in right whale conservation. NRC Research Press Web site at cjfas.nrc.ca. J21103.

Schmid, J.R., and W.N. Witzell. 1997. Age and growth of wild Kemp's ridley turtles (*Lepidochelys kempi*): cumulative results of tagging studies in Florida. Chelonian Conservation and Biology 2(4): 532-537.

Schmidly, D.J., CO Martin, and GF Collins. 1972. First occurrence of a black right whale (*Balaena glacialis*) along the Texas coast. The Southwestern Naturalist.

Schubel, J.R., H.H. Carter, R.E. Wilson, W.M. Wise, M.G. Heaton, and M.G. Gross. 1978. Field investigations of the nature, degree, and extent of turbidity generated by open-water pipeline disposal operations. Technical Report D-78-30; U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss., 245 pp.

Schueller, P. and D.L. Peterson. 2006. Population status and spawning movements of Atlantic sturgeon in the Altamaha River, Georgia. Presentation to the 14th American Fisheries Society Southern Division Meeting, San Antonio, February 8-12th, 2006.

Schultz, J.P. 1975. Sea turtles nesting in Surinam. Zoologische Verhandelingen (Leiden), Number 143: 172 pp.

Scott, W.B. and E.J. Crossman. 1973. Freshwater fishes of Canada. Fisheries Research Board of Canada. Bulletin 184. pp. 80-82.

Scott, W. B., and M. C. Scott. 1988. Atlantic fishes of Canada. Canadian Bulletin of Fisheries and Aquatic Science No. 219. pp. 68-71.

Seaturtle.org. Sea turtle tracking database. Available at http://www.seaturtle.org. Accessed on August 23, 2007.

Secor, D. H. 2002. Atlantic sturgeon fisheries and stock abundances during the late nineteenth century. Pages 89-98 *In*: W. Van Winkle, P. J. Anders, D. H. Secor, and D. A. Dixon, (editors), Biology, management, and protection of North American sturgeon. American Fisheries Society Symposium 28, Bethesda, Maryland.

Secor, D.H. and J.R. Waldman. 1999. Historical abundance of Delaware Bay Atlantic sturgeon and potential rate of recovery. American Fisheries Society Symposium 23: 203-216.

Seipt, I., P.J. Clapham, C.A. Mayo, and M.P. Hawvermale. 1990. Population characteristics of individually identified fin whales, *Balaenoptera physalus*, in Massachusetts Bay. Fish. Bull. 88: 271-278.

Sella, I. 1982. Sea turtles in the EasternMediterranean and Northern Red Sea. Pages 417-423 in Bjorndal, K.A. (editor). Biology and Conservation of Sea Turltes. Smithsonian Institution Press, Washington, D.C.

Seminoff, J.A. 2004. Chelonia mydas. In 2007 IUCN Red List of Threatened Species. Accessed 31 July 2009. http://www.iucnredlist.org/search/details.php/4615/summ.

Shaffer, M. L. 1981. Minimum Population Sizes for Species Conservation. BioScience Vol. 31, No. 2 (Feb., 1981), pp. 131-134.

Shamblin, B.M. 2007. Population structure of loggerhead sea turtles (*Caretta caretta*) nesting in the southeastern United States inferred from mitochondrial DNA sequences and microsatellite loci. Master's thesis, University of Georgia. 59 pp.

Shirey, C., C.C. Martin, and E. D. Stetzar. 1999. Atlantic sturgeon abundance and movement in the lower Delaware River. DE Division of Fish and Wildlife, Dover, DE, USA. Final Report to the National Marine Fisheries Service, Northeast Region, State, Federal & Constituent Programs Office. Project No. AFC-9, Grant No. NA86FA0315. 34 pp.

Shoop, C.R. 1987. The Sea Turtles. Pages 357-358 in R.H. Backus and D.W. Bourne, eds. Georges Bank. Cambridge, Massachusetts: MIT Press.

Shoop, C.R. and R.D. Kenney. 1992. Seasonal distributions and abundances of loggerhead and leatherback sea turtles in waters of the northeastern United States. Herpetological Monographs 6: 43-67.

Short, F.T. and H.A. Neckles. 1999. The effects of global climate change on seagrasses. Aquat Bot 63: 169–196.

Slay, C.K. and J.I. Richardson. 1988. King's Bay, Georgia: Dredging and Turtles. Schroeder, B.A. (compiler). Proceedings of the eighth annual conference on sea turtle biology and conservation. NOAA Technical Memorandum NMFS-SEFC-214, pp. 109-111.

Smith, T. I. J. 1985. The fishery, biology, and management of Atlantic sturgeon, *Acipenser oxyrhynchus*, in North America. Environmental Biology of Fishes 14(1): 61-72.

Smith, T. I. J., E. K. Dingley, and D. E. Marchette. 1980. Induced spawning and culture of Atlantic sturgeon. Progressive Fish-Culturist 42: 147-151.

Smith, T. I. J., D. E. Marchette, and G. F. Ulrich. 1984. The Atlantic sturgeon fishery in South Carolina. North American Journal of Fisheries Management 4: 167-176.

Smith, T.I.J., D.E. Marchette and R.A. Smiley. 1982. Life history, ecology, culture and management of Atlantic sturgeon, *Acipenser oxyrhynchus oxyrhynchus,* Mitchill, in South Carolina. South Carolina Wildlife Marine Resources. Resources Department, Final

Report to U.S. Fish and Wildlife Service Project AFS-9. 75 pp.

Smith, T. I. J. and J. P. Clungston. 1997. Status and management of Atlantic sturgeon, *Acipenser oxyrinchus*, in North America. Environmental Biology of Fishes 48: 335-346.

Snover, M.L., A.A. Hohn, L.B. Crowder, and S.S. Heppell. 2007. Age and growth in Kemp's ridley sea turtles: evidence from mark-recapture and skeletochronology. Pages 89-106 in P.T. Plotkin, ed. Biology and Conservation of Ridley Sea Turtles. Baltimore, Maryland: Johns Hopkins University Press.

Soule, M. E. 1987. Where do we go from here? In M. E. Soule (editor), Viable populations for conservation, p. 175-183. Cambridge Univ. Press, Cambridge.

South Carolina Department of Natural Resources. 2007. Examination of Local Movement and Migratory Behavior of Sea Turtles during spring and summer along the Atlantic coast off the southeastern United States. Unpublished report submitted to NMFS as required by ESA Permit 1540. 45 pp.

Spotila, J.R., A.E. Dunham, A.J. Leslie, A.C. Steyermark, P.T. Plotkin, and F. V. Paladino. 1996. Worldwide Population Decline of *Demochelys coriacea*: Are Leatherback Turtles Going Extinct? Chelonian Conservation and Biology 2(2): 209-222.

Spotila, J.R., R.D. Reina, A.C. Steyermark, P.T. Plotkin, F.V. Paladino. 2000. Nature 405: 529-530.

Squires, T.S., M. Smith, and L. Flagg. 1979. Distribution and abundance of Shortnose and Atlantic sturgeon in the Kennebec River estuary. Maine Department of Marine Resources, Augusta Maine, as cited in Fernandes, S.J., Kinnison, M.T. and G.B Zydlewski. 2006. Investigation into the distribution and abundance of Atlantic sturgeon and other diadromous species in the Penobscot River, Maine. 2006 Annual Report prepared by the University of Maine Sturgeon Research Working Group.

Squiers, T. 2004. State of Maine 2004 Atlantic sturgeon compliance report to the Atlantic States Marine Fisheries Commission. Report submitted to Atlantic States Marine Fisheries Commission, December 22, 2004, Washington, D.C.

Stein, A. B., K. D. Friedland, and M. Sutherland. 2004. Atlantic sturgeon marine distribution and habitat use along the northeastern coast of the United States. Transactions of the American Fisheries Society 133: 527-537.

Stephens, S.H., and J. Alvarado-Bremer. 2003. Preliminary information on the effective population size of the Kemp's ridley (*Lepidochelys kempii*) sea turtle. Page 250 *In*: J.A. Seminoff, compiler. Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503.

Stetzar, E.J. 2002. Population characterization of sea turtles that seasonally inhabit the Delaware Estuary. Masters thesis, Delaware State University, 146pp.

Stevenson, J. T., and D. H. Secor. 1999. Age determination and growth of Hudson River Atlantic sturgeon, *Acipenser oxyrinchus*. Fishery Bulletin 97: 153-166.

Stevick P.T., J. Allen, P.J. Clapham, N. Friday, S.K. Katona, F. Larsen, J. Lien, D.K. Matilla, P.J. Palsboll, J. Sigurjonsson, T.D. Smith, N. Oien, P.S. Hammond. 2003. North Atlantic humpback whale abundance and rate of increase four decades after protection from whaling. Marine Ecology Progress Series. 258: 263-273.

Stevick, P. T., J. Allen, P. J. Clapham, S. K. Katona, F. Larsen, J. Lien, D. K. Mattila, P. J. Palsbøll, R. Sears, J. Sigurjonsson, T. D. Smith, G. Vikingsson, N. Øien and P. S. Hammond. 2006. Population spatial structuring on the feeding grounds in North Atlantic humpback whales (*Megaptera novaeangliae*). Journal of Zoology. 270: 244–255.

Stewart, K., C. Johnson, and M.H. Godfrey. 2007. The minimum size of leatherbacks at reproductive maturity, with a review of sizes for nesting females from the Indian, Atlantic and Pacific Ocean basins. Herp. Journal 17: 123-128.

Stewart, K., M. Sims, A. Meylan, B. Witherington, B. Brost, and L.B. Crowder. 2011. Leatherback nests increasing significantly in Florida, USA; trends assessed over 30 years using multilevel modeling. Ecological Applications, 21(1): 263–273.

Stocker, T.F. and A. Schmittner. 1997. Influence of CO2 emission rates on the stability of the thermohaline circulation. Nature 388: 862–865.

Stone, G.S., L. Flores-Gonzalez, and S. Cotton. 1990. Whale migration record. Nature 346: 705.

Suárez, A. 1999. Preliminary data on sea turtle harvest in the Kai Archipelago, Indonesia. Abstract appears in the 2nd ASEAN Symposium and Workshop on Sea Turtle Biology and Conservation, held from July 15-17, 1999, in Sabah, Malaysia.

Suárez, A., P.H. Dutton and J. Bakarbessy. 2000. Leatherback (*Dermochelys coriacea*) nesting on the North Vogelkop Coast of Irian Jaya, Indonesia. *In*: Kalb, H.J. and T. Wibbels, compilers. 2000. Proceedings of the Nineteenth Annual Symposium on Sea Turtle Biology and Conservation. U.S. Dept. Commerce. NOAA Tech. Memo. NMFS-SEFSC-443, 291 p.

Sweka, J. A., J. Mohler, M.J. Millard, T. Kehler, A. Kahnle, K. Hattala, G. Kenny, and A. Higgs. 2007. Juvenile Atlantic sturgeon habitat use in Newburgh and Hayerstraw Bays of the Hudson River: implications for population monitoring. Transactions of the American Fisheries Society 27:1058-1067.

Swingle, W.M., S.G. Barco, T.D. Pitchford, W.A. McLellan, and D.A. Pabst. 1993. Appearance of juvenile humpback whales feeding in the nearshore waters of Virginia. Mar. Mamm. Sci. 9: 309-315.

Taub, S.H. 1990. Interstate fishery management plan for Atlantic sturgeon. Fisheries Management Report No. 17. Atlantic States Marine Fisheries Commission, Washington, D.C. 73 pp.

Turtle Expert Working Group (TEWG). 1998. An assessment of the Kemp's ridley (*Lepidochelys kempii*) and loggerhead (*Caretta caretta*) sea turtle populations in the Western North Atlantic. NOAA Technical Memorandum NOAA Fisheries-SEFSC-409. 96 pp.

TEWG. 2000. Assessment update for the Kemp's ridley and loggerhead sea turtle populations in the western North Atlantic. U.S. Dep. Commer. NOAA Tech. Mem. NMFS-SEFSC-444, 115 pp.

TEWG. 2007. An assessment of the leatherback turtle population in the Atlantic Ocean. NOAA Technical Memorandum NMFS-SEFSC-555, 116 pp.

TEWG. 2009. An assessment of the loggerhead turtle population in the Western North Atlantic Ocean. NOAA Technical Memorandum NMFS-SEFSC-575: 1-131.

TNC. 2002a. Priority Areas for Freshwater Conservation Action: A Biodiversity Assessment of the Southeastern United States.

Tynan, C.T. and D.P. DeMaster. 1997. Observations and predictions of Arctic climatic change: potential effects on marine mammals. *Arctic* 50: 308-322.

Van Den Avyle, M. J. 1984. Species profile: Life histories and environmental requirements of coastal fishes and invertebrates (South Atlantic): Atlantic sturgeon. U.S. Fish and Wildlife Service Report No. FWS/OBS-82/11.25, and U. S. Army Corps of Engineers Report No. TR EL-82-4, Washington, D.C.

Van Eenennaam, J.P., S.I. Doroshov, G.P. Moberg, J.G. Watson, D.S. Moore and J. Linares. 1996. Reproductive conditions of the Atlantic sturgeon (*Acipenser oxyrhynchus*) in the Hudson River. Estuaries 19: 769-777.

Van Eenennaam, J.P., and S.I. Doroshov. 1998. Effects of age and body size on gonadal development of Atlantic sturgeon. Journal of Fish Biology 53: 624-637.

Vladykov, V.D. and J.R. Greeley. 1963. Order Acipenseroidea. Pages 24-60 in Fishes of the Western North Atlantic. Memoir Sears Foundation for Marine Research 1(Part III). xxi + 630 pp.

U.S. Army Corps of Engineers (USACE). 1983. Dredging and Dredged Material Disposal, EM 1110-2-5025, Washington, D.C. March.

USACE. 1994. Beach Erosion Control and Hurricane Protection Study, Virginia Beach, Virginia- General Reevaluation Report, Main Report, Environmental Assessment , and Appendices. Norfolk District.

USACE. 2007. Winthrop Shores Reservation Restoration Program Endangered Species Biological Assessment. Prepared by Normandeau Associates. Submitted to NMFS Northeast Regional Office on February 7, 2007. 46 p.

USACE. 2009b. North End Sand Borrow Site, NASA Wallops Flight Facility, Wallops Island, Virginia. Report of Subsurface Exploration and Laboratory Testing. Prepared by USACE Norfolk District, GEO Environmental Section, Fort Norfolk, 803 Front Street, Norfolk,Virginia 23510. November.

USACE. 2010a. Storm Damage Reduction Project Design for Wallops Island. USACE, Engineer Research and Development Center. Prepared by King D.B.Jr., D.L. Ward, M.H. Hudgins, and G.G. Williams. ERDC/LAB TR-0X-X. November.

USACE Environmental Laboratory. Sea Turtle Data Warehouse. Available at http://el.erdc.usace.army.mil/seaturtles/index.cfm. Accessed on August 6, 2007.

U.S. Army Corps of Engineers (USACE), New York District. 2000. Raritan Bay and Sandy Hook Bay, New Jersey Feasibility Report for Hurricane and Storm Damage Reduction Port Monmouth, NJ: Final Feasibility Report and Environmental Impact Statement.

U.S. Army Corps of Engineers (USACE), New York District (NYD). 2006. Y-Long Branch beach renourishment project market survey. Solicitation number W912DS-06-SSAPL-174-01.

U.S. Fish and Wildlife Service (USFWS). 1997. Synopsis of the biological data on the green turtle, *Chelonia mydas* (Linnaeus 1758). Biological Report 97(1). U.S. Fish and Wildlife Service, Washington, D.C. 120 pp.

USFWS and NMFS. 1992. Recovery plan for the Kemp's ridley sea turtle (*Lepidochelys kempii*). NMFS, St. Petersburg, Florida.

Van Houtan, K.S. and J.M. Halley. 2011. Long-Term Climate Forcing in Loggerhead Sea Turtle Nesting. PLoS ONE 6(4): e19043. doi:10.1371/journal.pone.0019043.

Vanderlaan, A.S.M. and C.T. Taggart. 2006. Vessel collisions with whales: The probability of lethal injury based on vessel speed. Mar. Mamm. Sci. 22(3).

Van Houton, K.S. and O.L. Bass. 2007. Stormy oceans are associated with declines in sea turtle hatching. Curr Biol 17: R590.

Waldman, J.R., J.T. Hart, and I.I. Wirgin. 1996. Stock composition of the New York Bight Atlantic sturgeon fishery based on analysis of mitochondrial DNA. Transactions of the American Fisheries Society 125: 364-371.

Wallace, B.P., S.S. Heppell, R.L. Lewison, S. Kelez, and L.B. Crowder. 2008. Impacts of fisheries bycatch on loggerhead turtles worldwide inferred from reproductive value analyses. J. Appl. Ecol. 45:1076-1085.

Waluda, C.M., P.G. Rodhouse, G.P. Podesta, P.N. Trathan, and G.J. Pierce. 2001. Surface oceanography of the inferred hatching grounds of *Illex argentinus* (Cephalopoda: Ommastrephidae) and influences on recruitment variability. *Marine Biology* 139: 671-679.

Warden, M.L. 2011a. Modeling loggerhead sea turtle (*Caretta caretta*) interactions with US Mid-Atlantic bottom trawl gear for fish and scallops, 2005-2008. Biological Conservation 144: 2,202-2,212.

Warden, M.L. 2011b. Proration of loggerhead sea turtle (*Caretta caretta*) interactions in U.S. Mid-Atlantic bottom otter trawls for fish and scallops, 2005-2008, by managed species landed. U.S. Department of Commerce, Northeast Fisheries Science Centter Reference Document 11-04. 8 p.

Warden, M. and K. Bisack. 2010. Analysis of Loggerhead Sea Turtle Bycatch in Mid-Atlantic Bottom Trawl Fisheries to Support the Draft Environmental Impact Statement for Sea Turtle Conservation and Recovery in Relation to Atlantic and Gulf of Mexico Bottom Trawl Fisheries. NOAA NMFS NEFSC Ref. Doc.010. 13 pp.

Waring, G.T., E. Josephson, C.P. Fairfield-Walsh, and K. Maze-Foley. 2007. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments – 2006, 2nd edition, US Department of Commerce, NOAA Technical Memorandum NMFS -NE -201.

Waring, G.T., E. Josephson, C.P. Fairfield-Walsh, and K. Maze-Foley. 2008. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments – 2007. NOAA Technical Memorandum NMFS NE 205; 415pp.

Waring GT, Josephson E, Fairfield-Walsh CP, Maze-Foley K, editors. 2009. Final U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments -- 2009. Available from: http://www.nmfs.noaa.gov/pr/sars/draft.htm

Waring, G.T., E. Josephson, K. Maze-Foley and P.E. Rosel. 2010. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments – 2010. NOAA Technical Memorandum NMFS-NE-219; 606 pp.

Watkins, W.A. 1981. Activities and underwater sounds of fin whales. Scientific Reports of the International Whaling Commission 33: 83-117.

Watkins, W.A. 1986. Whale reactions to human activities in Cape Cod waters. Marine Mammal Science 2(4): 251-262.

Watkins, W.A., and W.E. Schevill. 1982. Observations of right whales (*Eubalaena glacialis*) in Cape Cod waters. Fish. Bull. 80(4): 875-880.

Watkins, W.A., K.E. Moore, J. Sigurjonsson, D. Wartzok, and G. Notarbartolo di Sciara. 1984. Fin whale (*Balaenoptera physalus*) tracked by radio in the Irminger Sea. Rit Fiskideildar 8(1): 1-14.

Webster, P.J., G.J. Holland, J.A. Curry, H.R. Chang. 2005. Changes in tropical cyclone number, duration, and intensity in a warming environment. Science 309:1844–1846.

Wehrell, S. 2005. A survey of the groundfish caught by the summer trawl fishery in Minas Basin and Scots Bay. Honours Thesis. Department of Biology, Acadia University, Wolfville, Canada.

Weisbrod, A.V., D. Shea, M.J. Moore, and J.J. Stegeman. 2000. Organochlorine exposure and bioaccumulation in the endangered Northwest Atlantic right whale (*Eubalaena glacialis*) population. Environmental Toxicology and Chemistry, 19(3): 654-666.

Weishampel, J.F., D.A. Bagley, and L.M. Ehrhart. 2004. Earlier nesting by loggerhead sea turtles following sea surface warming. Global Change Biology 10: 1424-1427.

Welsh, Stuart A., Michael F. Mangold, Jorgen E. Skjeveland, and Albert J. Spells. 2002. Distribution and Movement of Shortnose Sturgeon (*Acipenser brevirostrum*) in the Chesapeake Bay. Estuaries Vol. 25 No. 1: 101-104.

Wibbels, T. 2003. Critical approaches to sex determination in sea turtle biology and conservation. In: P. Lutz *et al.* (editors), Biology of Sea Turtles, Vol 2. CRC Press Boca Raton. p. 103-134.

Wilber, D.H., D.G. Clarke & M.H. Burlas. (2006). Suspended sediment concentrations associated with a beach nourishment project on the northern coast of New Jersey. Journal of Coastal Research 22(5): 1,035 – 1,042.

Wiley, D.N., R.A. Asmutis, T.D. Pitchford, and D.P. Gannon. 1995. Stranding and mortality of humpback whales, *Megaptera novaengliae*, in the mid-Atlantic and southeast United States, 1985-1992. Fish. Bull. 93: 196-205.

Winn, H.E., C.A. Price, and P.W. Sorensen. 1986. The distributional biology of the right whale

(*Eubalaena glacialis*) in the western North Atlantic. Reports of the International Whaling Commission (Special issue) 10: 129-138.

Wirgin, I. and T.L. King. 2011. Mixed stock analysis of Atlantic sturgeon from coastal locales and a non-spawning river. Presentation of the 2011 Sturgeon Workshop, Alexandria, VA, February 8-10.

Wise, J.P, S.S. Wise, S. Kraus, R. Shaffley, M. Grau, T.L. Chen, C. Perkins, W.D. Thompson, T. Zhang, Y. Zhang, T. Romano and T. O'Hara. 2008. Hexavalent chromium is cytotoxic and genotoxic to the North Atlantic right whale (*Eubalaena glacialis*) lung and testes fibroblasts. Mutation Research - Genetic Toxicology and Environmental Mutagenesis. 650(1): 30-38.

Witherington, B., P. Kubilis, B. Brost, and A. Meylan. 2009. Decreasing annual nest counts in a globally important loggerhead sea turtle population. Ecological Applications 19: 30-54.

Witt, M.J., A.C. Broderick, D.J. Johns, C. Martin, R. Penrose, M.S. Hoogmoed, and B.J. Godley. 2007. Prey landscapes help identify potential foraging habitats for leatherback turtles in the NE Atlantic. Marine Ecology Progress Series 337: 231-243.

Witt, M.J., A.C. Broderick, M. Coyne, A. Formia and others. 2008. Satellite tracking highlights difficulties in the design of effective protected areas for critically endangered leatherback turtles *Dermochelys coriacea* during the inter-nesting period. Oryx 42: 296–300.

Witzell, W.N. 2002. Immature Atlantic loggerhead turtles (*Caretta caretta*): suggested changes to the life history model. Herpetological Review 33(4): 266-269.

Woodley, T.H., M.W. Brown, S.D. Kraus, and D.E. Gaskin. 1991. Organochlorine levels in North Atlantic right whale *(Eubalaena glacialis)* blubber. Arch. Environ. Contam. Toxicol. 21 (1): 141-145.

Wynne, K. and M. Schwartz. 1999. Guide to marine mammals and turtles of the U.S. Atlantic and Gulf of Mexico. Rhode Island Sea Grant, Narragansett, Rhode Island. 114 pp.

Wysocki, L.E., J.W. Davidson III, M.E. Smith, *et al*. (2007). Effects of aquaculture production noise on hearing, growth, and disease resistance of rainbow trout, *Oncorhynchus mykiss*. Aquaculture 272: 687–97.

Young, J. R., T. B. Hoff, W. P. Dey, and J. G. Hoff. 1998. Management recommendations for a Hudson River Atlantic sturgeon fishery based on an age-structured population model. Fisheries Research in the Hudson River. State of University of New York Press, Albany, New York. pp. 353.

Zemsky, V., A.A. Berzin, Y.A. Mikhaliev, and D.D. Tormosov. 1995. Soviet Antarctic pelagic whaling after WWII: review of actual catch data. Report of the Sub-committee on Southern Hemisphere baleen whales. Rep. Int. Whal. Comm. 45: 131-135.

Zervas, C., 2004. North Carolina Bathymetry/Topography Sea Level Rise Project: Determination of Sea Level Trends. NOAA Technical Report NOS CO-OPS 041

Zug, G.R., and J.F. Parham. 1996. Age and growth in leatherback turtles, *Dermochelys coriacea*: a skeletochronological analysis. Chelonian Conservation and Biology 2(2): 244-249.

Zurita, J.C., R. Herrera, A. Arenas, M.E. Torres, C. Calderon, L. Gomez, J.C. Alvarado, and R. Villavicencio. 2003. Nesting loggerhead and green sea turtles in Quintana Roo, Mexico. Pp. 125-127. I*n:* J.A. Seminoff (compiler). Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Tech. Memo. NMFS-SEFSC-503, 308 p.

APPENDIX A

MONITORING SPECIFICATIONS FOR HOPPER DREDGES

I. EQUIPMENT SPECIFICATIONS

A. Draghead

The draghead of the dredge shall remain on the bottom **at all times** during a pumping operation, except when:

1) the dredge is not in a pumping operation, and the suction pumps are turned completely off;
2) the dredge is being re-oriented to the next dredge line during borrow activities; or
3) the vessel's safety is at risk (i.e., the dragarm is trailing too far under the ship's hull).

At initiation of dredging, the draghead shall be placed on the bottom during priming of the suction pump. If the draghead and/or dragarm become clogged during dredging activity, the pump shall be shut down, the dragarms raised, whereby the draghead and/or dragarm can be flushed out by trailing the dragarm along side the ship. If plugging conditions persist, the draghead shall be placed on deck, whereby sufficient numbers of water ports can be opened on the draghead to prevent future plugging.

Upon completion of a dredge track line, the drag tender shall:

1) throttle back on the RPMs of the suction pump engine to an idling speed (e.g., generally less than 100 RPMs) **prior to** raising the draghead off the bottom, so that no flow of material is coming through the pipe into the dredge hopper. Before the draghead is raised, the vacuum gauge on the pipe should read zero, so that no suction exists both in the dragarm and draghead, and no suction force exists that can impinge a turtle on the draghead grate;

2) hold the draghead firmly on the bottom with no flow conditions for approximately 10 to 15 seconds before raising the draghead; then, raise the draghead quickly off the bottom and up to a mid-water column level, to further reduce the potential for any adverse interaction with nearby turtles;

3) re-orient the dredge quickly to the next dredge line; and

4) re-position the draghead firmly on the bottom prior to bringing the dredge pump to normal pumping speed, and re-starting dredging activity.

II. LOOKOUT PROTOCOL

A. Basic Requirement

A lookout with the ability to identify sea turtles and Atlantic sturgeon must be placed aboard the dredge(s) being used, starting immediately upon project commencement to monitor for the presence of listed species impinged on the draghead or present in the vicinity of dredge operations.

B. Information to be Collected

For each sighting of any endangered or threatened marine species, record the following information on the Dredge Observation Form (Appendix C):

1) Date, time, coordinates of vessel
2) Visibility, weather, sea state
3) Vector of sighting (distance, bearing)
4) Duration of sighting
5) Species and number of animals
6) Observed behaviors (feeding, diving, breaching, etc.)
7) Description of interaction with the operation

4.2.4 For any listed species observed impinged on the draghead, an incident report needs to be filled out and submitted to NMFS (fax (978-281-9394) or e-mail (incidental.take@noaa.gov) within 24 hours of the incident.

C. Disposition of Parts

If any whole sea turtles or Atlantic sturgeon (alive or dead, decomposed or fresh) or turtle or sturgeon parts are taken incidental to the project(s), Danielle Palmer (978) 282-8468 must be contacted within 24 hours of the take. All whole dead sea turtles or Atlantic sturgeon, or turtle or sturgeon parts, must be photographed and described in detail on the Incident Report of Sea Turtle or Atlantic Sturgeon Mortality (Appendix D (sea turtles) or Appendix E and F (Atlantic sturgeon)). The photographs and reports should be submitted to Danielle Palmer, NMFS, Protected Resources Division, 55 Great Republic Drive, Gloucester, MA 01930-2298. After NMFS is notified of the take, observers may be required to retain turtles for future analysis. Additional analysis will dependent on available freezer space, availability of organizations capable of conducting the analysis, and the size/condition of the sample. NMFS will provide guidance on this matter upon the USACEs notification of take. If NMFS determines that the animal is not necessary to save for future analysis, disposition of dead sea turtle species (loggerhead, leatherback, Kemp's ridley, or green turtles) taken either whole or in parts, or any Atlantic sturgeon should be disposed of (after a photograph is taken and a reporting form has been completed) by attaching a weight to the animal and dumping the specimen away from the areas being dredged (e.g., between the shore and the site of dredging operations). If possible, a mark or tag (e.g., Inconel tag) should be placed on the carcass or part in the event that the animal is recaptured or stranded. If the species is unidentifiable or if there are entrails that may have come from a turtle, the subject should be photographed, placed in plastic bags, labeled with location, load number, date and time taken, and placed in cold storage. Unidentifiable species or parts will be collected by NMFS or NMFS-approved personnel (contact Danielle Palmer at (978) 282-8468). Live turtles (both injured and uninjured) should be held onboard the dredge until transported as soon as possible to the appropriate stranding network personnel for rehabilitation (Appendix B). No live turtles should be released back into the water without first being checked by a qualified veterinarian or a rehabilitation facility.

APPENDIX B

Sea Turtle Handling and Resuscitation

It is unlikely that sea turtles will survive impingment in a hopper dredge, as the turtles found in the dragheads are usually dead, dying, or dismantled. However, the procedures for handling live sea turtles follow in case the unlikely event should occur.

Please photograph all turtles (alive or dead) and turtle parts found during dredging activities and complete the Incident Report of Sea Turtle Take (Appendix D).

Handling:
Do not assume that an inactive turtle is dead. The onset of rigor mortis and/or rotting flesh are often the only definite indications that a turtle is dead. Releasing a comatose turtle into any amount of water will drown it, and a turtle may recover once its lungs have had a chance to drain. There are three methods that may elicit a reflex response from an inactive animal:

- Nose reflex. Press the soft tissue around the nose which may cause a retraction of the head or neck region or an eye reflex response.

- Cloaca or tail reflex. Stimulate the tail with a light touch. This may cause a retraction or side movement of the tail.

- Eye reflex. Lightly touch the upper eyelid. This may cause an inward pulling of the eyes, flinching or blinking response.

General handling guidelines:

- Keep clear of the head.

- Adult male sea turtles of all species other than leatherbacks have claws on their fore flippers. Keep clear of slashing fore flippers.

- Pick up sea turtles by the front and back of the top shell (carapace). Do not pick up sea turtles by flippers, the head or the tail.

- If the sea turtle is actively moving, it should be retained at the OCNGS until transported by stranding/rehabilitation personnel to the nearest designated stranding/rehabilitation facility. The rehabilitation facility should eventually release the animal in the appropriate location and habitat for the species and size class of the turtle.

Live sea turtles within dredge gear
When a sea turtle is found in the dredge gear, observe it for activity and potential injuries.

If the turtle is actively moving, it should be retained onboard until evaluated for injuries by a permitted rehabilitation facility. Due to the potential for internal injuries associated with hopper entrainment, it is necessary to transport the live turtle to the nearest rehabilitation facility as soon as possible, following these steps:

Contact the nearest rehabilitation facility to inform them of the incident. If the rehabilitation personnel cannot be reached immediately, please contact NMFS stranding hotline at 866-755-6622 or NMFS Sea Turtle Stranding Coordinate (Kate Sampson) at 978-282-8470.

Keep the turtle shaded and moist (e.g., with a water-soaked towel over the eyes, carapace, and flippers), and in a confined location free from potential injury.

Contact the crew boat to pick up the turtle as soon as possible from the dredge (within 12 to 24 hours maximum). The crew boat should be aware of the potential for such an incident to occur and should develop an appropriate protocol for transporting live sea turtles.

4) Transport the live turtle to the closest permitted rehabilitation facility able to handle such a case.

Sea Turtle Resuscitation Regulations: (50 CFR 223.206(d)(1))

If a turtle appears to be comatose (unconscious), contact the designated stranding/rehabilitation personnel immediately. Once the rehabilitation personnel has been informed of the incident, attempts should be made to revive the turtle at once. Sea turtles have been known to revive up to 24 hours after resuscitation procedures have been followed.

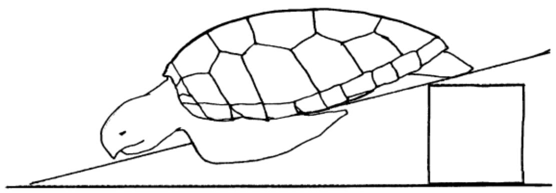
- Place the animal on its bottom shell (plastron) so that the turtle is right side up and elevate the hindquarters at least 6 inches for a period of 4 up to 24 hours. The degree of elevation depends on the size of the turtle; greater elevations are required for larger turtles.

- Periodically, rock the turtle gently left to right and right to left by holding the outer edge of the shell (carapace) and lifting one side about 3 inches then alternate to the other side.

- Periodically, gently conduct one of the above reflex tests to see if there is a response.

- Keep the turtle in a safe, contained place, shaded, and moist (e.g., with a water-soaked towel over the eyes, carapace, and flippers) and observe it for up to 24 hours.

- If the turtle begins actively moving, retain the turtle until the appropriate rehabilitation personnel can evaluate the animal. The rehabilitation facility should eventually release the animal in a manner that minimizes the chances of re-impingement and potential harm to the animal (i.e., from cold stunning).



- Turtles that fail to move within several hours 24) should be transported to a suitable facility for necropsy (if the condition of the sea turtle allows).

Dead sea turtles

The procedures for handling dead sea turtles and parts are described in Appendix A-II-C.

Stranding/rehabilitation contacts

- ***NMFS Stranding Hotline at (866)-755-6622***
- ***New York: Riverhead Foundation for Marine Research and Preservation, hotline:*** 631-369-9829
- ***New Jersey: Marine Mammal Stranding Center, hotline: 609-266-0538***

APPENDIX C

DREDGE OBSERVER FORM

HDP

Daily Report

Date: ________________________________ Time:_______________________________
Geographic Site:____
Location: Lat/Long ____________________ Vessel Name: ___________________________

Weather conditions:____
____
Sea State:____
Water temperature: Surface ____________ Below midwater (if known) ____________

Condition of UXO screening apparatus (e.g., any damages, any changes in screen dimensions, etc..): ____
____
____
Incidents involving endangered or threatened species? (Circle) Yes No
(*If yes, fill out Incident Report of Sea Turtle/Shortnose Sturgeon Mortality*)

Comments (type of material, biological specimens, duration of sighting, observed behaviors, description of interaction, etc:)____
____
____
____

Lookout's Name: ____
Lookout's Signature: ____

Species	# of Sightings	# of Animals	Comments

APPENDIX D

Incident Report of Sea Turtle Take

Species ______________ Date ________________ Time (specimen found) _________________

Geographic Site ____
Location: Lat/Long ____
Vessel Name ___________________________ Load # _______________________________
Begin load time _________________________ End load time__________________________
Begin dump time ________________________ End dump time ________________________

Sampling method ____
Condition of screening ____
Location where specimen recovered____
____
Draghead deflector used? YES NO Rigid deflector draghead? YES NO
Condition of deflector ____
____
Weather conditions____
____
Water temp: Surface _________________ Below midwater (if known) ______________________

Species Information: *(please designate cm/m or inches.)*
Head width ____________________________ Plastron length _____________________________
Straight carapace length _________________ Straight carapace width______________________
Curved carapace length __________________ Curved carapace width ______________________

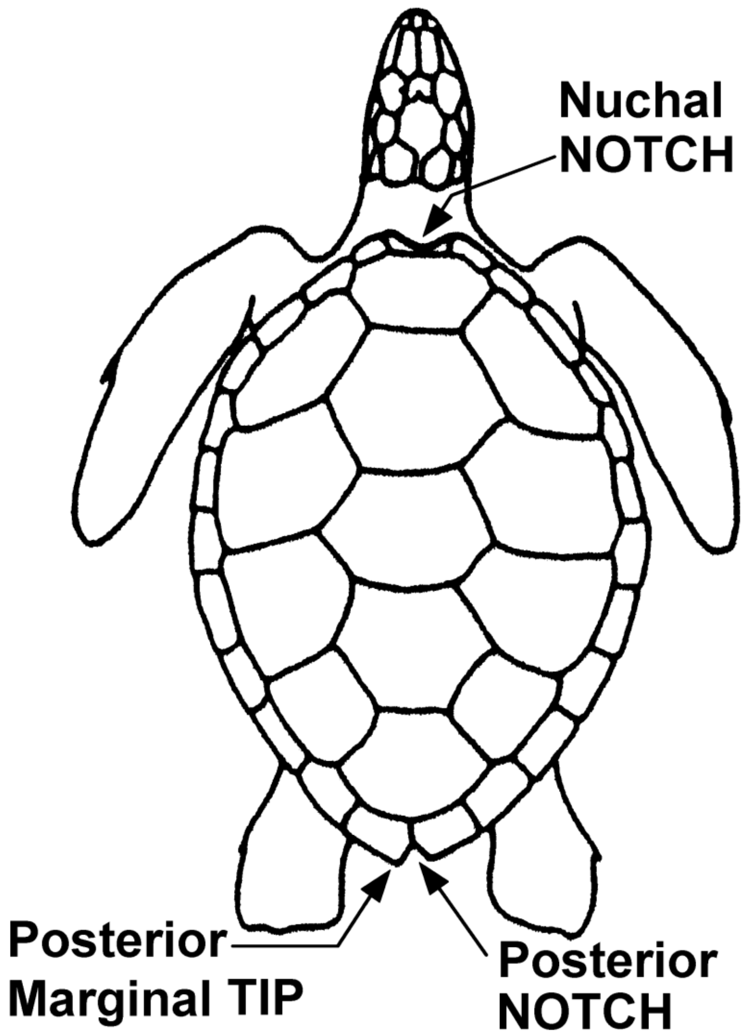
Condition of specimen/description of animal (please complete attached diagram)
____
____
____

Turtle Decomposed: NO SLIGHTLY MODERATELY SEVERELY

Turtle tagged: YES NO *Please record all tag numbers.* Tag # _______________________
Genetic sample taken: YES NO
Photograph attached: YES NO
(please label *species, date, geographic site* and *vessel name* on back of photograph)
Comments/other (include justification on how species was identified) ______________________
____
____
Lookout's/Observer's Name ____
Lookout's/Oberver's Signature _______________________________________

APPENDIX D, Continued
Incident Report of Sea Turtle Take

Draw wounds, abnormalities, tag locations on diagram and briefly describe below.



Description of animal:

APPENDIX E

Incident Report of Atlantic Sturgeon Take

Photographs should be taken and the following information should be collected from all sturgeon (alive and dead) found in association with the HDP.

Date ________________ Time (specimen found) _________________

Geographic Site____
Location: Lat/Long____
Vessel Name ___________________________ Load # _____________________________
Begin load time _________________________ End load time_________________________
Begin dump time ________________________ End dump time ______________________

Sampling method ____
Condition of screening ____
Location where specimen
recovered____
____
Draghead deflector used? YES NO Rigid deflector draghead? YES NO
Condition of deflector ____
____
Weather conditions____
____
Water temp: Surface _________________ Below midwater (if known) ________________

Species Information: *(please designate cm/m or inches.)*
Fork length (or total length) ______________________ Weight ______________________

Condition of specimen/description of animal

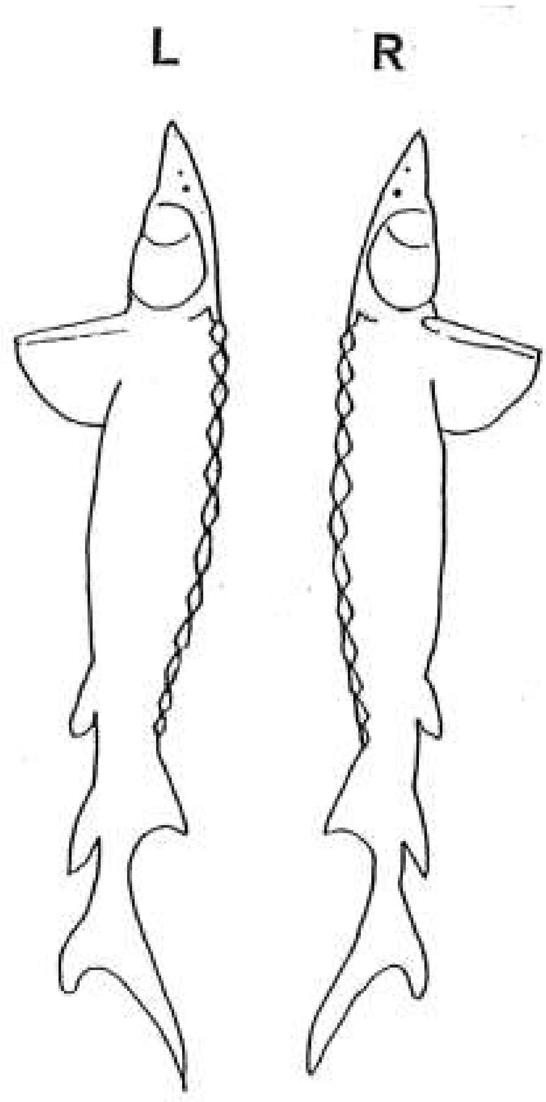
____
____
____
____

Fish Decomposed: NO SLIGHTLY MODERATELY SEVERELY
Fish tagged: YES / NO *Please record all tag numbers.* Tag # _________________
Genetic sample taken: YES NO
Photograph attached: YES / NO
(please label *species, date, geographic site* and *vessel name* on back of photograph)
Comments/other (include justification on how species was identified)

____
____
Lookout's/Observer's Name ____
Lookout's/Observer's Signature____

Appendix E, continued

Draw wounds, abnormalities, tag locations on diagram and briefly describe below



Description of fish condition: