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

Agency: National Aeronautics and Space Administration, U.S. Army Corps of Engineers Norfolk District, and Bureau of Ocean Energy Management

Activity: Wallops Island Shoreline Restoration and Infrastructure Protection Program (Reinitiation) (F/NER/2012/01118)
GARFO-2012-00028

Conducted by: National Marine Fisheries Service
Northeast Regional Office

Date Issued: 8/3/12

Approved by: ____________

1.0 INTRODUCTION

This constitutes the biological opinion (Opinion) of NOAA's National Marine Fisheries Service (NMFS) on the effects of the National Aeronautics and Space Administration's (NASA) Wallops Island Shoreline Restoration and Infrastructure Protection Program (SRIPP) on threatened and endangered species in accordance with section 7 of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531 et seq.). As NASA is funding and carrying out the action, NASA will serve as the lead Federal agency for purposes of this consultation. Other Federal agencies involved in authorizing, funding or carrying out the action include the US Army Corps of Engineers (USACE) and the Bureau of Ocean Energy Management (BOEM). The USACE will be issuing a permit to NASA pursuant to section 10 of the Rivers and Harbors Act. The BOEM will be issuing a non-competitive lease to NASA pursuant to the Outer Continental Shelf Lands Act. These actions will be considered in this consultation.

This Opinion is based on information provided in the 2010 Biological Assessment (BA), the 2010 Final Programmatic Environmental Impact Statement (PEIS), and the 2011 supplemental BA for NASA's Wallops Flight Facility Shoreline Restoration Project; correspondence with NASA; and, other sources of information. A complete administrative record of this consultation will be kept on file at the NMFS Northeast Regional Office. The date April 6, 2012, will be used to mark the start of formal consultation.

2.0 CONSULTATION HISTORY

In October 2006, NASA informed NMFS that it was preparing National Environmental Policy Act (NEPA) documentation for the proposed Wallops Island SRIPP (the project). On a November 13, 2006 conference call, NASA provided an explanation of the proposed project and informed NMFS that while multiple Federal agencies would be involved in the project, NASA would be the lead federal agency for the proposed project.[1] Also during this call, the need for formal consultation pursuant to section 7 of the ESA was discussed. Representatives from NASA and the Norfolk district of the Army Corps of Engineers (USACE) agreed that consultation was necessary and that NASA would be the lead agency for conducting the consultation with NMFS.

In February 2007, NMFS received a draft BA from NASA and NMFS provided comments on the draft BA. In a letter dated May 9, 2007, NASA requested formal consultation on the effects of the proposed project on listed species and submitted the final BA. A Biological Opinion (Opinion) was issued by NMFS to NASA and the USACE on September 25, 2007. In this Opinion, NMFS concluded that the proposed action was likely to adversely affect but was not likely to jeopardize the continued existence of loggerhead and Kemp's ridley sea turtles and was not likely to adversely affect leatherback or green sea turtles or right, humpback, and fin whales. NMFS also concluded that the action would not affect hawksbill turtles as this species is unlikely to occur in the action area. The Opinion included an Incidental Take Statement (ITS) exempting the incidental taking of no more than 1 sea turtle for approximately every 2,000,000 cy of material removed from the borrow areas, which over the life of the project exempted the take of 28 sea turtles, with no more than 3 being Kemp's ridleys and the remainder being loggerheads. The action considered in the September 25, 2007 Opinion was never initiated by NASA, and NASA has now redesigned the Wallop's Island SRIPP.

On February 12, 2010, NMFS received a letter from NASA requesting reinitiation of formal consultation due to the actions previously considered in the September 25, 2007 Opinion being modified in manner that will cause effects to listed species or critical habitat that were not considered in the 2007 Opinion. These modifications included the construction/extension of the existing seawall; the relocation of the borrow site to offshore sites located in Federal waters; and the reduction of the amount of material removed during the initial dredge cycle and subsequent renourishment cycles throughout the 50 year life of the SRIPP. On February 18, 2010, formal consultation was initiated, with an Opinion issued by NMFS to NASA on July 22, 2010. In this Opinion, NMFS concluded that the proposed action may adversely affect, but was not likely to jeopardize the continued existence of loggerhead and Kemp's ridley sea turtles and was not likely to adversely affect leatherback or green sea turtles or right, humpback, and fin whales. NMFS also concluded that the action would not affect hawksbill turtles as this species is unlikely to occur in the action area. The Opinion included an Incidental Take Statement (ITS) exempting the incidental taking of no more than 1 sea turtle for approximately every 1.5 million cy of material removed from the borrow areas, which over the life of the project exempted the take of

[1] The U.S. Army Corps of Engineers Norfolk District will be issuing a permit, pursuant to Section 10 of the Rivers and Harbors Act, to authorize the proposed dredging and placement of sand on the beach.

9 total sea turtles, with no more than 1 being Kemp's ridleys and the remainder being loggerheads.

On November 15, 2011, we initiated a conference with NASA in accordance with 50 CFR 402.10 on the effects of the SRIPP on Atlantic sturgeon. At that time Atlantic sturgeon were proposed for listing. On February 6, 2012, we published two final rules listing five distinct population segments (DPSs) of Atlantic sturgeon, with an effective date of listing on April 6, 2012. Atlantic sturgeon originating from the New York Bight, Chesapeake Bay, South Atlantic and Carolina DPSs were listed as endangered, while the Gulf of Maine DPS was listed as threatened (77 FR 5880; 77 FR 5914). By February 6, 2012, we had not completed the conference opinion; however, NASA's August 11, 2011, letter and our November 15, 2011, letter documented our mutual understanding that if Atlantic sturgeon were listed, we would issue a Biological Opinion that considers the effects of the SRIPP on all listed species, including Atlantic sturgeon. As such, April 6, 2012, was used to mark the start date of formal consultation. On July 13, 2012, we issued an Opinion to NASA. In this Opinion, we concluded that the SRIPP may adversely affect but is not likely to jeopardize the continued existence of the Northwest Atlantic Ocean Distinct Population Segment (DPS) of loggerhead sea turtle; Kemp's ridley sea turtles; the Gulf of Maine (GOM) DPS of Atlantic sturgeon; New York Bight (NYB) DPS of Atlantic sturgeon; Chesapeake Bay (CB) DPS of Atlantic sturgeon; Carolina DPS of Atlantic sturgeon; or South Atlantic (SA) DPSs of Atlantic sturgeon, and is not likely to adversely affect leatherback or green sea turtles or right, humpback or fin whales. We also conclude that the action will not affect hawksbill turtles as these species are unlikely to occur in the action area. . The Opinion included an Incidental Take Statement (ITS) exempting the incidental taking of no more than 1 sea turtle for approximately every 1.8 million cy of material removed from the borrow areas, which over the life of the project exempted the take of 8 total sea turtles, with no more than 1 being Kemp's ridleys and the remainder being loggerheads. In addition, the ITS exempted the incidental take of no more than 1 Atlantic sturgeon for approximately every 9.4 million cy of material removed from the borrow areas, which over the life of the project exempted the take of 2 subadult Atlantic sturgeon, with the potential that the two sturgeon taken may come from the NYB, CB, GOM, Carolina, or SA DPS.

On June 18, 2012, NASA brought to our attention that the Opinion we recently completed for the SRIPP on July 13, 2012, contained several errors in Section 7.1.2.1 Sea Turtle Entrainment, Table 7, and Section 7.1.2.2 Atlantic Sturgeon Entrainment, Table 9 of the Opinion. The tables provide information on dredging projects in the United States Army Corps of Engineer's North Atlantic District (USACE NAD). The error in the table relates to information provided on a dredge project in Cape May, New Jersey. This dredge project operated from a period ranging from 2004 to 2005, resulting in a total of 2,425,268 cubic yards of material removed at the end of the project in 2005. However, both tables recorded this value twice, once in 2004 and once in 2005 (i.e., as if they were two separate projects instead of one). This error affected the total volume of material removed for all USACE NAD projects being considered in table 7 (i.e., total volume went from 1.8 million cubic yards to 1.6 million cubic yards) and Table 9 (i.e., total volume went from 10.2 million cubic yards to 9.4 million cubic yards). As the number of interactions between dredge equipment and sea turtles and Atlantic sturgeon seems to be best associated with the volume of material removed, the information presented in these tables is used in estimating sea turtle and Atlantic sturgeon incidental takes for the action. This error has the potential to affect the incidental take estimate of sea turtle and Atlantic sturgeon provided in the 2012 Opinion. After correcting for the error, and tabulating the new numbers, the estimated incidental take for sea turtles increased by one turtle (i.e., from 8 to 9 sea turtles), while estimated incidental take for Atlantic sturgeon remained the same.

As this minor error will affect the number of sea turtles estimated to be taken under the action, reconsideration of the July 13, 2012 Opinion's effects and jeopardy analysis needs to be undertaken. On July 18, 2012, we explained this necessity to NASA and via an email dated July 19, 2012, NASA requested reinitiation of consultation, which thereby marked the beginning of formal consultation.

3.0 DESCRIPTION OF THE ACTION

NASA's Wallops Flight Facility (WFF) is located in the northeastern portion of Accomack County, Virginia on the Delmarva Peninsula. NASA has occupied the WFF since the 1940s and is currently used by NASA, the U.S. Coast Guard (USGS), the U.S. Navy, NOAA, and the Mid-Atlantic Regional Spaceport (MARS). Wallops Island is bounded by Chincoteague Inlet to the north and Assawoman Inlet to the south. Chincoteague Inlet is dredged annually to a depth of 12 feet. The predominant direction of longshore sediment movement is from north to south. This longshore movement of sediment has caused sand pits to grow. The consequence of the sand traps is that Wallops Island and the barrier islands to the south have been deprived of sediment and their shorelines have eroded.

From 1857 to 1994, the southern part of Wallops Island has retreated approximately 400 meters (1300 feet), with an average rate of retreat of 12 feet per year. This encroachment of the ocean has threatened the existence of launch pads, infrastructure, and test and training facilities belonging to NASA, the Navy, and to MARS. In the 1960s and 1970s, NASA installed wooden groins to attempt to prevent shoreline retreat and keep sand on the beach. By the mid-1980s, the groins were almost completely gone as a result of the lack of replenishing sand. In 1992, a stone seawall, approximately 15,900 feet long, was constructed along the center of the island; however, the seawall has failed to provide adequate protection against the loss of sand as the current seawall is porous and has allowed sediment to flow out of the area without allowing replenishment. The integrity of the seawall is also at risk due to the lack of protective beach sand. Currently, beach only exists seaward of the northern portion of the seawall. There is no beach along approximately 14,000 feet of the seawall. In 2007, NASA installed geotextile tubes along the shoreline south of the existing seawall as an emergency measure to slow down the transport of sand off the beach and help protect onshore assets from wave action. Despite these efforts, the ocean has continued to encroach toward the infrastructure on Wallops Island. These conditions have lead to the SRIPP by NASA. Under the SRIPP, NASA is proposing to construct and extend the existing seawall, as well as rebuild the beach along the Goddard Space Flight Center's Wallops Flight Facility (WFF), thereby moving the zone of wave break away from launch pads, infrastructure, and testing and training facilities. This will require dredging of offshore borrow sites and/or an area on the northern end of Wallops Island over the life of the SRIPP (50 years) in order to obtain sand to renourish and maintain the newly formed beach.

Within the first 3 years of the 50- year life of the SRIPP, seawall construction and initial beach nourishment will be completed.

3.1 Seawall Extension

Prior to beach nourishment, the seawall extension will be constructed on the beach parallel to the shoreline in the approximate location of the existing geotextile tubes. The new seawall will be constructed landward of the shoreline and will extend 4,600 feet south of the existing seawall and will consist of 5-7 ton rocks placed on the beach. The top of the seawall will be approximately 14 feet above the normal high tide level.

3.2 Dredging and Beach Fill

3.2.1 Description of Borrow areas

Initial site work conducted in May 2007 identified 3 potential offshore shoals (Blackfish Bank Shoal, Unnamed Shoal A and Unnamed Shoal B) (Appendix A) located in Federal waters where beach compatible sand could be removed for the purposes of beach nourishment along the shoreline of the Wallops Flight Facility. In addition, an area located on the northern end of Wallops Island has also been identified as a borrow area for renourishment purposes only. Blackfish Bank Shoal was removed from consideration as a borrow area due to adverse impacts on the Assateague Island shoreline and due to the public perception that dredging within this shoal would negatively impact commercial and recreational fishing communities. As result, NASA identified Unnamed Shoal A as the source of sand for initial beach nourishment along the shoreline of Wallops Flight Facility, and Unnamed Shoal B and the beach area located on the northern portion of Wallops Island (North Wallops Island beach borrow site) as potential sites to obtain sand during subsequent cycles of beach renourishment.

The southwest end of Unnamed Shoal A is located approximately 7 miles east of Assateague Island and approximately 11 miles northeast of Wallops Island. The total predicted volume of sand at Unnamed Shoal A is approximately 31 million m^3 (40 million cubic yards (cy)) and covers an area of approximately 1,800 acres. Depths at Unnamed Shoal A range from 25-40 feet. The sediments within Unnamed Shoal A consist of well sorted medium sand with a median composite grain size of 0.24-0.78mm (USACE 2010a). The borrow area has never been dredged.

Unnamed Shoal B is located approximately 10 miles east of Assateague Island. The southwest end of Unnamed Shoal B is located approximately 12 miles east of Assateague Island and approximately 13 miles northeast of Wallops Island. The total predicted volume of Unnamed Shoal B is approximately 57 million m^3 (70 million cy) and covers an area of approximately 3,900 acres. Depths within Unnamed Shoal B range from 29-50 feet. The sediments within Unnamed Shoal B consist of well sorted, medium sand with a median composite grain size of 0.17-0.0.47mm (USACE 2010a). The borrow area has never been dredged.

The North Wallops Island beach borrow site is being considered by NASA as an additional area for obtaining sand for renourishment cycles. The sediments in this area general consist of poorly graded fine to medium sand with trace shell fragments and silt (USACE 2009b). The median grain sizes of all samples were between 0.18-0.27mm. Although not an optimal grain size for use as beach fill material, the northern end of Wallops Island would offer potential renourishment material without the mobilization and operational costs associated with offshore dredging. Based on current vegetation and wildlife habitat constraints, the total potential area for sand removal is approximately 150 acres. This area of Wallops Island has never been excavated.

3.2.2 Offshore Dredging

From the beginning of April 2012 to the end of June 2012, approximately 3,998,750 cy of sand are expected to be removed from Unnamed Shoal A and placed as beach nourishment along the shoreline of the Wallops Flight Facility, which will aid in restoring the underwater area in front of the seawall to its equilibrium condition (USACE 2010a). Recent beach profiles that were collected in 2011 show that erosion has been continuing, with a majority being sub-aqueous in front of the rock seawall. Therefore, the original amount of material proposed to be removed from the borrow site is no longer sufficient to construct a full project or to completely fill the originally proposed footprint. As such, NASA is considering the removal of an additional 375,000 cy of material to ensure complete beach nourishment. Based on this information, NMFS will assess the impacts of removing a maximum of 4,373,750 cy of material during the initial beach fill phase of the SRIPP. Renourishment cycles are expected to occur every 5 years, with 1,007,500 cy of material removed during each cycle from either of two offshore borrow sites (Unnamed Shoal A and Unnamed Shoal B) and/or the north end of Wallop's Island. Approximately 9 renourishment cycles are proposed to occur over the 50 year life of the SRIPP; this, in combination with the initial cycle of beach nourishment, will result in the removal of a total of approximately 13,440,950 cy of material over the 50-year life of the SRIPP.

A trailer suction hopper dredge will be used to dredge the offshore borrow sites throughout the 50 year life of the SRIPP. These dredges are self propelled and hydraulically operated and are equipped with two dragheads and a hopper. High speed centrifugal pumps are employed to excavate the sediment and dispose of it into a storage hopper. The intake end of the suction pipe is fitted with a draghead, the function of which is to strip off a layer of sediment (approximately 0.3 m (1 foot) in depth) from the seabed and entrain those sediments into the suction pipe. Material dislodged from the ocean floor by the suction is suspended in water in the form of a slurry and then passed through the centrifugal pump to the storage hopper. Once the dredge hopper is filled, the dredge will transport the material to a pump-out buoy or station that will be placed at a water depth of approximately 30 feet, which is located approximately 2 miles offshore of the placement area. The pathway from Unnamed Shoal A and B to the pump-out buoy is not a straight line, but instead is a dogleg shape with a turning point so as to avoid Chincoteague Shoal and Blackfish Bank. The distance from the turning point to the pump-out buoy is approximately 8 miles. The one-way distance from Unnamed Shoal A to the proposed pump-out buoy is approximately 14 miles and the corresponding transit distance from Unnamed Shoal B to the proposed pump-out buoy is approximately 19 miles. Booster pumps may be needed to aid the offloading of sand from the pump-out buoy to the shoreline. Three dredges, with a hopper capacity of approximately 4,000 cy, will be operating at the same time, with two dredges operating at the offshore site and one transiting to the pump out-station. This pattern would alternate within a 24-hour period, with dredges spending approximately 3-4 hours on site at the shoal and the remainder of time traveling and unloading sand. Dredges will be operating at speeds of 3 knots while dredging and 10 knots when transiting to and from the borrow areas.

3.2.3 On-Shore Excavation

The north Wallops Island borrow site will be excavated with a pan excavator. The pan excavator will stockpile the sand, which will be loaded onto dump trucks that will transport the fill material up and down the beach. Bulldozers will then be used to spread the fill material once it is placed on the beach. All heavy equipment will access the beach from existing roads and established access points. No new temporary or permanent roads will be constructed to access the beach or to transport the fill material to renourishment areas. No in water work will be required for this portion of the project.

3.2.4 Beach Fill

Initial beach fill placement is expected in 2012. The beach fill will start approximately 1,500 feet north of the Wallops Island-Assawoman Island property boundary and extend north for 3.7 miles. The initial fill will be placed so that there will be a 6-foot-high berm extending a minimum 70 feet seaward of the existing seawall. The remainder of the fill will be placed at a 20:1 slope underwater for an additional distance seaward; the amount of that distance would vary along the length of the beach fill, but will extend for about an additional 137 m (450 ft), so that the total distance of the fill profile from the seawall will be up to approximately 158 m (520 ft). The beach fill profile will also include a 14-foot-high dune at the seawall. The front sloping face of the dune will rest against the seawall. Sand for initial nourishment will be dredged, as noted above, from Unnamed Shoal A and placed on the beach as described above. For renourishment fill volumes, up to one half of the fill volume may be excavated from the north Wallops Island borrow site, with the remainder of the sand obtained from either Unnamed Shoal A or Unnamed Shoal B.

3.3 Implementation Schedule

As noted above, the extension of the seawall will occur before beach nourishment operations begin (i.e., beginning of April 2012). Following the completion of the initial beach fill (i.e., after June 2012) subsequent renourishment activities (assuming all fill is taken from one of the proposed offshore shoals), will occur approximately once every five years. During each renourishment cycle, 1,007,500 cy of material will be removed from either of two offshore borrow sites (Unnamed Shoal A and Unnamed Shoal B) and/or the north end of Wallop's Island. Renourishment cycles should take approximately 50 days to complete. Over the 50 year life of the SRIPP, approximately 9 renourishment cycles are proposed to occur with a total of approximately 13,440,950 cy of material removed during this period.

3.4 Mitigation Measures

Throughout the action, NASA will implement measures to minimize any potential effects of dredging to listed species of sea turtles and whales throughout the project. Mitigation measures specific to sea turtles were incorporated within the BA and PEIS NMFS received on February 18, 2010. After further analysis of the potential effects of dredging on listed species of whales, specifically in regards to dredge noise, NASA and NMFS devised additional mitigation measures to be put in place throughout the action. These additional mitigation measures were received by NMFS on June 28, 2010 and were incorporated into the final 2010 BA and EIS. The following are the mitigation measures NASA will implement as part of the action:

1. NASA will ensure that during April 1-November 30, hopper dredges are outfitted with state-of-the-art sea turtle deflectors on the drag head and operated in a manner that will reduce the risk of interactions with sea turtles that may be present in the action area.

2. A NMFS-approved observer will be present on board the vessel for any dredging occurring from April 1-Novemeber 30 to monitor for sea turtles.

3. NASA will ensure that dredges are equipped and operated in a manner that provides endangered/threatened species observers with a reasonable opportunity for detecting interactions with listed species and that provides for handling, collection, and resuscitation of turtles injured during project activity.

4. NASA will ensure that all measures are taken to protect any turtles that survive entrainment in the dredge.

5. If a NMFS approved observer or the lookout/bridge watch observes a whale within 1 km (0.62 miles) of the dredge, all pumps will be turned off (i.e., dredging will stop) until the whale leaves the area (i.e., is farther than 1 km (0.62 miles) from the dredge) to avoid acoustic harassment of listed species of whales.

6. All dredge operators will monitor the right whale (*Eubalaena glacialis*) sighting reports (i.e., sighting advisory system (SAS), dynamic management areas (DMA's), seasonal management areas (SMA's)) to remain informed on the whereabouts of right whales within the vicinity of the action area.

7. All dredge operators will conform to the regulations prohibiting the approach of right whales closer than 500 yards (50 CFR 224.103 (c)). Any vessel finding itself within the 500 yard buffer zone around a right whale must depart the area immediately at a safe, slow speed, unless one of the exceptions applies (see 50 CFR 224.103 (c)).

8. For dredging operations at night, the work area will be lit well enough to ensure that the observer/lookout can perform his/her work safely and effectively and that the measures mentioned above can be performed to the extent practicable.

9. NMFS will be contacted before dredging commences and again upon completion of the dredging activity.

10. All whale sightings will be reported to NMFS' Protected Resources Division Section 7 Coordinator.

3.5 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 Wallops Island offshore borrow sites, the waters between and immediately adjacent to these areas where project vessels will travel and dredged material will be transported (see Appendix A for an illustration of the action area) as well as an area extending 4000 feet in all directions from the area to be dredged to account for the sediment plume generated during dredging activities. The action area also includes the northern portion of Wallops Island and the portion of Wallops Island shoreline and nearshore waters that will be affected by the extended seawall and beach fill (i.e., 3.7 miles of shoreline) (see Appendix A for an illustration of the action area). As dredging operations will also produce underwater noise levels that range between 120-160 dB re 1µPa the action area will also include the area around the dredge where effects of increased underwater noise levels will be experienced. Based on the analysis of dredge noise and transmission loss calculations, effects of dredge noise will be experienced within 794 meters from the dredge during loading and pumping.

4.0 LISTED SPECIES IN THE ACTION AREA

NMFS has determined that the action being considered in this biological opinion may affect the following endangered or threatened species under NMFS' 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[2]
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

[2] 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.

Chesapeake Bay DPS	Endangered
South Atlantic DPS	Endangered
Carolina DPS	Endangered

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

4.1 Status of Sea Turtles

Sea turtles continue to be affected by many factors occurring on the nesting beaches and in the water. Poaching, habitat loss, and nesting predation by introduced species affect hatchlings and nesting females while on land. Fishery interactions, vessel interactions, and channel dredging operations, for example, affect sea turtles in the neritic zone (defined as the marine environment extending from mean low water down to 200 m (660 foot) depths, generally corresponding to the continental shelf (Lalli and Parsons 1997; Encyclopedia Britannica 2011)). Fishery interactions also affect sea turtles when these species and the fisheries co-occur in the oceanic zone (defined as the open ocean environment where bottom depths are greater than 200 m (Lalli and Parsons 1997). As a result, sea turtles still face many of the original threats that were the cause of their listing under the ESA.

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 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), 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 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.

As noted above, 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 sub lethal 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 five-year status review of the species, which discussed a variety of threats to loggerheads including climate change, NMFS and FWS determined that loggerhead sea turtles should not be delisted or reclassified as endangered. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPSs should be identified for the loggerhead (NMFS and USFWS 2007a). Genetic differences exist between loggerhead sea turtles that nest and forage in the different ocean basins (Bowen 2003; Bowen and Karl 2007). Differences in the maternally inherited mitochondrial DNA also exist between loggerhead nesting groups that occur within the same ocean basin (TEWG 2000; Pearce 2001; Bowen 2003; Bowen *et al.* 2005; Shamblin 2007; TEWG 2009; NMFS and USFWS 2008). Site fidelity of females to one or more nesting beaches in an area is believed to account for these genetic differences (TEWG 2000; Bowen 2003).

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

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

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

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

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

Presence of Loggerhead Sea Turtles in the Action Area

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

Distribution and Life History

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

(NMFS and USFWS 2008), which is a second revision to the original recovery plan that was approved in 1984 and subsequently revised in 1991.

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

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

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

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

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

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

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

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

Population Dynamics and Status

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

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

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

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

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

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

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

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

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

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

Maier *et al.* (2004) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the Southeast Coast of the U.S. (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 four years of the study (Ehrhart *et* *al.* 2007). However, there was no discernible trend in loggerhead abundance during the 24-year time period of the study (1982-2006) (Ehrhart *et al.* 2007). At St. Lucie Power Plant, data collected from 1977-2004 show an increasing trend of loggerheads at the power plant intake structures (FPL and Quantum Resources 2005).

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

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

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

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

Threats

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

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

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

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

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

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

In addition to improvements in TED designs and TED enforcement, interactions between loggerheads and the shrimp fishery have also been declining because of reductions in fishing effort unrelated to fisheries management actions. The 2002 Opinion take estimates were based in part on fishery effort levels. In recent years, low shrimp prices, rising fuel costs, competition with imported products, and the impacts of 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 were projected in the 2002 Opinion. In 2008, the NMFS Southeast Fisheries Science Center (SEFSC) estimated annual number of interactions between loggerheads and shrimp trawls in the Gulf of Mexico shrimp fishery to be 23,336, with 647 (2.8%) of those interactions resulting in mortality (Memo from Dr. B. Ponwith, Southeast Fisheries Science Center to Dr. R. Crabtree, Southeast Region, December 2008). However, the most recent section 7 consultation on the shrimp fishery, completed in May 2012, was unable to estimate the total annual level of take for loggerheads at present. 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 2012).

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

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

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

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

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

As highly migratory, wide-ranging organisms that are biologically tied to temperature regimes, loggerhead sea turtles are vulnerable to effects of climate change in aspects of their physiology and behavior (Van Houtan 2011; 2009 Loggerhead Status Review Report). Analysis on potential effects of climate change on loggerhead sea turtles in the action area is included below in section 6.0.

Summary of Status for Loggerhead Sea Turtles

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

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

NMFS convened a new Loggerhead Turtle Expert Working Group (TEWG) to review all available information on Atlantic loggerheads in order to evaluate the status of this species in the Atlantic. A final report from the Loggerhead TEWG was published in July 2009. In this report, the TEWG indicated that it could not determine whether the decreasing annual numbers of nests among the Northwest Atlantic loggerhead subpopulations were due to stochastic processes resulting in fewer nests, a decreasing average reproductive output of adult females, decreasing numbers of adult females, or a combination of these factors. Many factors are responsible for past or present loggerhead mortality that could impact current nest numbers; however, no single mortality factor stands out as a likely primary factor. It is likely that several factors compound to create the current decline, including incidental capture (in fisheries, power plant intakes, and dredging operations), lower adult female survival rates, increases in the proportion of first-time nesters, continued directed harvest, and increases in mortality due to disease. Regardless, the TEWG stated that “it is clear that the current levels of hatchling output will result in depressed recruitment to subsequent life stages over the coming decades” (TEWG 2009). However, the report does not provide information on the rate or amount of expected decrease in recruitment but goes on to state that the ability to assess the current status of loggerhead subpopulations is limited due to a lack of fundamental life history information and specific census and mortality data.

While several documents reported the decline in nesting numbers in the NWA DPS (NMFS and USFWS 2008, TEWG 2009), when nest counts through 2010 are analyzed, the nesting trends from 1989-2010 are not significantly different than zero (i.e., stable) for all recovery units within the NWA DPS for which there are enough data to analyze (76 FR 58868, September 22, 2011).

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

Based on this and the current best available information, we believe that the NWA DPS of loggerheads 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 50 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.

4.1.2 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 2007b). 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 suggest 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).

As highly migratory, wide-ranging organisms that are biologically tied to temperature regimes, Kemp's ridley 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 Kemp's ridley sea turtles in the action area is included below in section 6.0.

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 50 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.

4.1.3.1 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).

4.1.3.2 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).

4.1.3.3 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).

4.1.3.4 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 2007d) 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 (2007) report provide summaries of natural as well as anthropogenic threats to leatherback sea turtles. Of the Atlantic sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear, trap/pot gear in particular. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), their diving and foraging behavior, their distributional overlap with the gear, their possible attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, and perhaps to the lightsticks used to attract target species in longline fisheries. Leatherbacks entangled in fishing gear generally have a reduced ability to feed, dive, surface to breathe, or perform any other behavior essential to survival (Balazs 1985). In addition to drowning from forced submergence, they may be more susceptible to boat strikes if forced to remain at the surface, and entangling lines can constrict blood flow resulting in tissue necrosis. The long-term impacts of entanglement on leatherback health remain unclear. Innis *et al*. (2010) conducted a health evaluation of leatherback sea turtles during direct capture (n=12) and disentanglement (n=7). They found no significant difference in many of the measured health parameters between entangled and directly captured turtles. However, blood parameters, including but not limited to sodium, chloride, and blood urea nitrogen, for entangled turtles showed several key differences that were most likely due to reduced foraging and associated seawater ingestion, as well as a general stress response.

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

Leatherbacks have been documented interacting with longline, trap/pot, trawl, and gillnet fishing gear. For instance, between 1992-1999 an estimated 6,363 leatherback sea turtles were documented as caught by the U.S. Atlantic tuna and swordfish longline fisheries (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), whelk/conch (15), black sea bass (10), crab (2), and research pot gear (1).[3] 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 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,

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

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 1885 and 2007) reported plastic within the turtles' stomach contents, and in some cases (8.7% of those cases in which plastic was reported), blockage of the gut was found in a manner that may have caused the mortality (Mrosovsky *et al*. 2009). An increase in reports of plastic ingestion was evident in leatherback necropsies conducted after the late 1960s (Mrosovsky *et al.* 2009). Along the coast of Peru, intestinal contents of 19 of 140 (13%) leatherback carcasses were found to contain plastic bags and film (Fritts 1982). The presence of plastic debris in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items (*e.g.*, jellyfish) and plastic debris (Mrosovsky 1981). Balazs (1985) speculated that plastic objects may resemble food items by their shape, color, size, or even movements as they drift about, and induce a feeding response in leatherbacks.

Although leatherbacks are probably already beginning to be affected by impacts associated with anthropogenic climate change in several ways, no significant climate change-related impacts to leatherback turtle populations have been observed to date (PIRO Longline fisheries BO 2012). However, over the long term, climate change related impacts will likely influence biological trajectories in the future on a century scale (Parmesan and Yohe 2003). Analysis on potential effects of climate change on leatherback sea turtles in the action area is included below in section 6.0.

4.1.3.5 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 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 50 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.

4.1.4.1 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).

4.1.4.2 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).

4.1.4.3 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.

4.1.4.4 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 2007c). 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 6.0.

4.1.4.5 Summary of Status of Green Sea Turtles

A review of 32 Index Sites distributed globally revealed a 48-67% decline in the number of mature females nesting annually over the last three generations (Seminoff 2004).[4] An evaluation

[4] The 32 Index Sites include all of the major known nesting areas as well as many of the lesser nesting areas for

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 2007c). 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 50 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

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which quantitative data are available. Generation times ranged from 35.5 years to 49.5 years for the assessment depending on the Index Beach site.

representative of the population's trend.

4.2 North Atlantic Right Whales

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

Right whales have been listed as endangered under the Endangered Species Act (ESA) since 1973. In December 2006, NMFS completed a comprehensive review of the status of right whales in the North Atlantic and North Pacific Oceans, which at the time were listed as a single species, *Eubalaena glacialis*, or "northern right whale." 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 the 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 right whales in the Northern Hemisphere 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). Right whales in the Southern Hemisphere (*E. australis*) remained listed as endangered as well.

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 1991b). 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 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.* 2010). Like other right whale species, they follow an annual pattern of migration between low latitude winter calving grounds and high latitude summer foraging grounds (Perry *et al.* 1999; Kenney 2002).

The distribution of right whales seems linked to the distribution of their principal zooplankton prey, calanoid copepods (Winn *et al.* 1986; NMFS 2005; Baumgartner and Mate 2005; Waring *et al.* 2010). 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.* 2010). Right whales also frequent Stellwagen Bank and Jeffrey's 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 highlight the high interannual variability in right whale use of some habitats. Calving is known to occur in the winter months in coastal waters off 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. In the North Atlantic it appears that not all reproductively active females return to the calving grounds each year (Kraus *et al.*, 1986; Payne *et al.* 1986). Patrician *et al.* (2009) analyzed photographs of a right whale calf sighted in the Great South Channel in June of 2007 and determined the calf appeared too young to have been born in the known southern calving area. In addition, the location of some portion of the population during the winter months remains unknown (NMFS 2005). However, recent aerial surveys conducted under the North Atlantic Right Whale Sighting Survey (NARWSS) program have indicated that some individuals may reside in the northern Gulf of Maine during the winter. In 2008, 2009, and 2010, right whales were sighted on Jeffrey's and Cashes Ledge, Stellwagen Bank, and Jordan Basin during December to February (Khan *et al*. 2009, 2010, 2011).

While right whales are known to congregate in the aforementioned areas, much is still not understood about their seasonal distribution and movements within and between these areas are extensive (Waring *et al.* 2010). In the winter, only a portion of the known right whale population is seen on the calving grounds. The winter distribution of the remaining right whales remains uncertain (NMFS 2005, Waring *et al.* 2010). 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.* 2010). On multiple days in December 2008, congregations of more than forty 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 of 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) represents one of only two sightings this century of a right whale in Norwegian waters, and the first since 1926. Together, these long-range matches indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not presently well described. Similarly, records from the Gulf of Mexico (Moore and Clark 1963; Schmidly *et al.* 1972) represent either geographic anomalies or a more extensive historic range beyond the sole known calving and wintering ground in the waters of the southeastern United States. The frequency with which right whales occur in offshore waters in the southeastern U.S. remains unclear (Waring *et al.*, 2010).

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 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 299 right whales was estimated in 1998 (Kraus *et al*. 2001), and a review of the photo-ID recapture database on June 24, 2009, indicated that 361 individually recognized whales were known to be alive during 2005 (Waring *et al.* 2010). Because this 2009 review was a nearly complete census, it is assumed this estimate represents a minimum population size. The minimum number alive population index for the years 1990-2005 suggests a positive trend in numbers. These data reveal a significant increase in the number of catalogued whales alive during this period, but with significant variation due to apparent losses exceeding gains during 1998-1999. Mean growth rate for the period was 2.1% (Waring *et al*. 2010).

A total of 297 right whale calves have been born from 1993-2009 (Waring *et al*. 2010). The mean calf production for the 15-year period from 1993-2009 is estimated to be 17.2/year (Waring *et al.* 2010). Calving numbers have been sporadic, with large differences among years, including a second largest calving season in 2000/2001 with 31 right whale births (Waring *et al.* 2010). The three calving years (97/98; 98/99; 99/00) prior to this record year provided low recruitment levels with only 11 calves born. The last nine calving seasons (2000-2009) have been remarkably better with 31, 21, 19, 17, 28, 19, 23, 23, and 39 births, respectively (Waring *et al.* 2010). 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 1983-2005, 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 6 calves each, and 4 cows had at least seven calves. Two of these cows were at an age which 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 included additional 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 Glass *et al.* 2009, below). Of the 15 serious injuries and mortalities between 2003-2007, at least 9 were adult females, three of which were carrying near-term fetuses and 4 of which were just starting to bear calves (Waring *et al.* 2009). Since the average lifetime calf production is 5.25 calves (Fujiwara and Caswell 2001), the deaths of these 9 females represent a loss of reproductive potential of as many as 47 animals. However, it is important to note that not all right whale mothers are equal with regards to calf production. Right whale #1158 had only one calf over a 25-year period (Kraus *et al.* 2007). In contrast, one of the largest right whales on record was a female nicknamed "Stumpy," who was killed in February 2004 of an apparent ship strike (NMFS 2006). She was first sighted in 1975 and known to be a prolific breeder, successfully rearing calves in 1980, 1987, 1990, 1993, and 1996 (Moore *et al.* 2007). At the time of her death, she was estimated to be 30 years of age and carrying her sixth calf; the near-term fetus also died (NMFS 2006).

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 an 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 relative to the 1980s with female survival, in particular, apparently 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 focused on females (Clapham *et al.* 2002). Increased mortalities in 2004 and 2005 were cause for serious concern (Kraus *et. al* 2005). Calculations indicate that this increased mortality rate would reduce population growth by approximately 10% per year (Kraus *et. al* 2005). Despite the preceding, examination of the minimum number alive population index calculated from the individual sightings database, as it existed on 24 June 2009, for the years 1990-2005 suggest a positive trend in numbers (Waring *et al.* 2010). These data reveal a significant increase in the number of catalogued whales alive during this period, but with significant variation due to apparent losses exceeding gains during 1998-1999 (Waring *et al.* 2010). 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, in review). The PVA evaluated several scenarios on how the populations would fare without entanglement mortalities compared to the status quo. Only 2 of 1000 projections (with the status quo simulation) ended with a smaller total population size than they started and zero projections resulted in extinction. As described above, the mean growth rate estimated in the latest stock assessment report, for the period 1990-2005, was 2.1% (Waring *et al.* 2010).

Reproductive Fitness

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 over five years between 1998-2003, and then decreased to just over 3 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 causing an effect on right whales (Kraus *et al.* 2007), there is currently no evidence available to determine their actual effect, if any. The dramatic reduction in the North Atlantic right whale population believed to have occurred due to commercial whaling may have resulted in a loss of genetic diversity which could affect the ability of the current population to successfully reproduce (*i.e.*, decreased conceptions, increased abortions, and increased neonate mortality). One hypothesis is that the low level of genetic variability in this species produces a high rate of mate incompatibility and unsuccessful pregnancies (Frasier *et al.* 2007). Analyses are currently under way to assess this relationship further as well as the influence of genetic characteristics on the potential for species recovery (Frasier *et al.* 2007). Studies by Schaeff *et al.* (1997) and Malik *et al.* (2000) indicate that western North Atlantic right whales are less genetically diverse than southern right whales. However, several apparently healthy populations of cetaceans, such as sperm whales and pilot whales, have even lower genetic diversity than observed for western North Atlantic right whales (IWC 2001a). 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 concentrations were lower than those found in marine mammals proven to be affected by PCBs and DDT (Weisbrod *et al.* 2000). Another suite of contaminants (i.e. antifouling agents and flame retardants) that have been proven to disrupt reproductive patterns and have been found in other marine animals, have raised 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, however 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 disease effects on right whales. Impacts of biotoxins on marine mammals are also poorly understood, yet data is showing that marine algal toxins may play significant roles in mass mortalities of large whales (Rolland *et al.* 2007). Although there are no published data concerning the effects of biotoxins on right whales, researchers are now certain that right whales are being exposed to measurable quantities of paralytic shellfish poisoning (PSP) toxins and domoic acid via trophic transfer through the presence of these biotoxins in prey upon which they feed (Durbin *et al.* 2002, Rolland *et al.* 2007).

Data indicating whether right whales are food-limited 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.* (*in press*)). Miller *et al*. (*in press*) 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 abundances, right whales had significantly thinner blubber than during years of greater prey abundances. The results suggest that blubber thickness is indicative of right whale energy balance and that the marked fluctuations in the North Atlantic right whale reproduction have a nutritional component (Miller *et al*. (*in press*)).

Threats

There is general agreement that right whale recovery is negatively affected by anthropogenic mortality. From 2004-2008, right whales had the highest proportion of entanglement and ship strike events relative to the number of reports for a species (Glass *et al.* 2010). Given the small population size and low annual reproductive rate of right whales, human sources of mortality may have a greater effect to relative population growth rate than for other large whale species (Waring *et al.* 2010). For the period 2004-2008, the annual human-caused mortality and serious injury rate for the North Atlantic right whale averaged 2.8 per year (2.2 in U.S. waters; 0.6 in Canadian waters) (Glass *et al*. 2010). Twenty-one confirmed right whale mortalities were reported along the U.S. east coast and adjacent Canadian Maritimes from 2004-2008 (Glass *et al*. 2010). 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.* 2010).

Considerable effort has been made to examine right whale carcasses for the cause of death (Moore *et al*. 2004). Because they live in an ocean environment, examining right whale carcasses is often very difficult. Some carcasses are discovered floating at sea and cannot be retrieved. Others are in such an advanced stage of decomposition when discovered that a complete examination is not possible. Wave action and post-mortem predation by sharks can also damage carcasses, and preclude a thorough examination of all body parts. It should also be noted that mortality and serious injury event judgments are based upon the best available data and additional information may result in revisions (Glass *et al.* 2010). Of the 21 total, confirmed right whale mortalities (2004-2008) described in Glass *et al.* (2010), 3 were confirmed to be entanglement mortalities (1 adult female, 1 female calf, 1 male calf) and 8 were confirmed to be ship strike mortalities (5 adult females, 1 female of unknown age, 1 male calf, and 1 yearling male). Serious injury involving right whales was documented for 1 entanglement event (adult male) and 2 ship strike events (1 adult female and 1 yearling male).

Although disentanglement is either unsuccessful or not possible for the majority of cases, during the period of 2004-2008, there were at least 4 documented cases of entanglements for which the intervention of disentanglement teams averted a likely serious injury (Waring *et al.* 2010). Even when entanglement or vessel collision does not cause direct mortality, it may weaken or otherwise affect individuals so that further injury or death is likely (Waring *et. al* 2010). Some right whales that have been entangled were subsequently 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 previous 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.* 2010).

Entanglement records from 1990-2008 maintained by NMFS include 47 confirmed right whale entanglement events (Waring *et al.* 2010). 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.* 2010). Data presented in Knowlton *et al.* 2008 indicate the annual rate of entanglement interaction remains at high levels. Four hundred and ninety-three individual, catalogued right whales were reviewed and 625 separate entanglement interactions were documented between 1980 and 2004. Approximately 358 out of 493 animals (72.6% of the population) were entangled at least once; 185 animals bore scars from a single entanglement; however one animal showed scars from 6 different entanglement events. The number of male and female right whales bearing entanglement scars was nearly equivalent (142/202 females, 71.8%; 182/224 males, 81.3%), indicating that 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 percent of the North Atlantic right whale population exhibit signs of injury from vessel strikes. Reports received from 2004-2008 indicate that right whales had the greatest number of ship strike mortalities (n=8) and serious injuries (n=2) compared to other large whales in the Northwest Atlantic (Glass *et al.* 2010). In 2006 alone, four reported mortalities and one serious injury resulted from right whale ship strikes (Glass *et al.* 2010).

As highly migratory, wide-ranging organisms, effects of climate change on cetaceans are possible. Analysis on potential effects of climate change on North Atlantic right whales in the action area is included below in section 6.0. Analysis on potential effects of climate change on North Atlantic right whales in the action area is included below in section 6.0.

Summary of Right Whale Status

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

Previous models estimated that the right whale population in the Atlantic numbered 300 (+/-

10%) (Best *et al*. 2001). However, a review of the photo-ID recapture database on July 24, 2009 indicated that 361 individually recognized right whales were known to be alive in 2005 (Waring *et al.* 2010). The 2000/2001 - 2008/2009 calving seasons have had relatively high calf production (31, 21, 19, 17, 28,19, 23, 23, and 39 calves, respectively) and have included additional first time mothers (*e.g.*, eight new mothers in 2000/2001) (Waring *et al*. 2009, 2010). There are some indications that climate-driven ocean changes impacting the plankton ecology of the Gulf of Maine, may, in some manner, be affecting right whale fitness and reproduction.

However, there is also general agreement that right whale recovery is negatively affected by human sources 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.* 2010).

Over the five-year period 2004-2008, right whales had the highest proportion of entanglements and ship strikes relative to the number of reports for a species: of 64 reports involving right whales, 24 were confirmed entanglements and 17 were confirmed ship strikes. There were 21 verified right whale mortalities, three due to entanglements, and eight due to ship strikes (Glass *et al.* 2010). 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.

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 it existed on 24 June 2009, for the years 1990-2005 suggest a positive trend in numbers of right whales (Waring *et al.* 2010). In addition, calving intervals appear to have declined to 3 years in recent years (Kraus *et al.* 2007), and calf production has been relatively high over the past several seasons.

Based on the information currently available, for the purposes of this Opinion, NMFS believes that the western North Atlantic right whale subpopulation is increasing; as protective measures for large whales are currently in place and continue to be implemented, we expect this trend to continue over the next 50 years.

4.3 Humpback Whales

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

4.3.1 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 respective summer/fall feeding areas to winter/spring calving and mating areas within the North Pacific Basin (Angliss and Outlaw 2007, Carretta *et al.* 2011). Within the Pacific Ocean, NMFS recognizes three management units within the U.S. EEZ for the purposes of managing this species under the MMPA. These are: the California-Oregon-Washington stock (feeding areas off the US 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 which doubles previous population predictions (Calambokidis *et al.* 2008). There are indications that the California-Oregon-Washington stock was growing in the 1980’s and early 1990’s 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 are 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.* 2008a). 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.* 2008a). 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 subpopulations wintering grounds, and the possibility exists that the entire population does not migrate to the wintering grounds (Reilly *et al.* 2008a).

Like other whales, southern hemisphere humpback whales were heavily exploited for commercial whaling. Although they were given protection by the IWC in 1963, Soviet whaling data made available in the 1990's revealed that 48,477 southern hemisphere humpback whales were taken from 1947-1980, 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).

4.3.2 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.* 2010). 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 Jeffrey's Ledge (CeTAP 1982) and peak in May and August. Small numbers of individuals may be present in this area year-round, including the waters of Stellwagen Bank. They feed on a number of species of small schooling fishes, particularly sand lance and Atlantic herring, targeting fish schools and filtering large amounts of water for their associated prey. It is hypothesized humpback whales may also feed on euphausiids (krill) as well as 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 does occur (Waring *et al.* 2010). 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 were most frequent during September through April in North Carolina and Virginia waters, and were composed primarily of juvenile humpback whales of no more than 11 meters in length (Wiley *et al.* 1995).

Abundance Estimates and Trends

Photographic mark-recapture analyses from the Years of the North Atlantic Humpback (YONAH) project gave an ocean-basin-wide estimate of 11,570 animals during 1992/1993 and an additional genotype-based analysis yielded a similar but less precise estimate of 10,400 whales (95% c.i. = 8,000 - 13,600) (Waring *et al.* 2010). 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.* 2010). The best, recent estimate for the Gulf of Maine stock is 847 whales, derived from a 2006 line-transect aerial sighting survey (Waring *et al.* 2010).

Population modeling, using data obtained from photographic mark-recapture studies, estimates the growth rate of the Gulf of Maine stock to be at 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.* 2010). 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 US Mid-Atlantic waters (Waring *et al.* 2010). Regardless, calf survival appears to have increased since 1996, presumably accompanied by an increase in population growth (Waring *et al.* 2010). Stevick *et al.* (2003) calculated an average population growth rate of 3.1% in the North Atlantic population overall for the period 1979-1993.

Threats

As is the case with other large whales, like North Atlantic right whales, the major known sources of anthropogenic mortality and injury of humpback whales occur from fishing gear entanglements and ship strikes. For the period 2004 through 2008, the minimum annual rate of human-caused mortality and serious injury to the Gulf of Maine humpback whale stock averaged 4.6 animals per year (U.S. waters, 4.4; Canadian waters, 0.2) (Waring *et al.* 2010). Between 2004 and 2008 humpback whales were involved in 81 confirmed entanglement events and 14 confirmed ship strike events (Glass *et al.* 2010). Over the five-year period, humpback whales were the most commonly observed entangled whale species; entanglements accounted for 5 mortalities and 11 serious injuries (Glass *et al.* 2010). Of the 14 confirmed ship strikes, 8 of the events were fatal (Glass *et al*. 2010). 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 (Glass *et al.* 2009, Waring *et al.* 2010).

Based on photographs taken between 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 stock humpback whales between 2000 and 2002 suggest a minimum of 49 interactions with gear took place. Based on composite scar patterns, it was believed that male humpback whales were more vulnerable to entanglement than females. Males may be subject to other sources of injury that could affect scar pattern interpretation. Images were obtained from a humpback whale breeding ground; 24% exhibited 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 stock 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 due to trophic effects 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 from 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. It has been suggested that the occurrence of a red tide event is related to an increase in freshwater runoff from coastal development, leading some observers to suggest that such events may become more common among marine mammals as coastal development continues (Clapham *et al.* 1999). There have been three additional known cases of a mass mortality involving large whale species along the East coast between 1998 and 2008. In the 2006 mass mortality event, 21 dead humpback whales were found between July 10 and December 31, 2006, triggering NMFS to declare an unusual mortality event (UME) for humpback whales in the Northeast United States. The UME was officially closed on December 31, 2007 after a review of 2007 humpback whale strandings and mortality showed that the elevated numbers were no longer being observed. The cause of the 2006 UME has not been determined to date, although investigations are ongoing.

Changes in humpback whale distribution in the Gulf of Maine have been found to be associated with changes in herring, mackerel, and sand lance abundance associated with local fishing pressures (Stevick *et al.* 2006, Waring *et al.* 2010). Shifts in relative finfish species abundance correspond to changes in observed humpback whale movements (Stevick *et al.* 2006). However, there is no evidence that humpback whales were adversely affected by these trophic changes.

As highly migratory, wide-ranging organisms, effects of climate change on cetaceans are possible. Analysis on potential effects of climate change on humpback whales in the action area is included below in section 6.0.

4.3.3 Summary of Humpback Whale Status

The best available population estimate for humpback whales in the North Atlantic Ocean is 11,570 animals, and the best, recent estimate for the Gulf of Maine stock is 847 whales (Waring *et al.* 2010). 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 United States 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.* 2010). 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.

Therefore, given the best available information, for the purposes of this biological opinion, NMFS believes that globally, most humpback whale populations are increasing; as protective measures for large whales are currently in place and continue to be implemented, we expect this trend to continue over the next 50 years.

4.4 Fin Whale

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

4.4.1 Pacific Ocean

Within US 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 the US 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 it was estimated from surveys that covered only a portion of the range of the species (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 is estimated to have been 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.

4.4.2 North Atlantic

NMFS has designated one population of fin whales in US waters of the North Atlantic (Waring *et al.* 2010). This species is commonly found from Cape Hatteras northward. A number of 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 between 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.* 2010).

During 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.* 2010). 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 50m isobath past Cape Cod, over Stellwagen Bank, and past Cape Ann to Jeffrey's 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 US 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) as well as squid and planktonic crustaceans (Wynne and Schwartz 1999).

Population Trends and Status

Various estimates have been provided to describe the current status of fin whales in western North Atlantic waters. One method used the catch history and trends in Catch Per Unit Effort 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 US continental shelf waters. The 2010 Stock Assessment Report (SAR) gives a best estimate of abundance for fin whales in the western North Atlantic of 3,985 (CV = 0.24). However, this estimate must be considered extremely conservative in view of the incomplete coverage of the known habitat of the stock and the uncertainties regarding population structure and whale movements between surveyed and unsurveyed areas (Waring *et al*. 2010). The minimum population estimate for the western North Atlantic fin whale is 3,269 (Waring *et al.* 2010). However, there are insufficient data at this time to determine population trends for the fin whale (Waring *et al.* 2010).

Other estimates of the abundance of fin 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.

Threats

The major known sources of anthropogenic mortality and injury of fin whales include entanglement in commercial fishing gear and ship strikes. The minimum annual rate of confirmed human-caused serious injury and mortality to North Atlantic fin whales from 2004-2008 was 3.2 (Glass *et al*. 2010). During this five year period, there were 14 confirmed entanglements (3 fatal; 3 serious injuries) and 13 ship strikes (10 fatal) (Glass *et al.* 2010). Fin whales are believed to be the cetacean most commonly struck by large vessels (Laist *et al.* 2001). In addition, hunting of fin whales continued well into the 20th century. Fin whales were given total protection in the North Atlantic in 1987 with the exception of an aboriginal subsistence whaling hunt for Greenland (Gambell 1993, Caulfield 1993). However, Iceland has increased its whaling activities in recent years and reported a catch of 136 whales in the 1988/89 and 1989/90 seasons (Perry *et al.* 1999), 7 in 2006/07, and 273 in 2009/2010. Fin whales may also be adversely affected by habitat degradation, habitat exclusion, acoustic trauma, harassment, or reduction in prey resources due to trophic effects resulting from a variety of activities.

As highly migratory, wide-ranging organisms, effects of climate change on cetaceans are possible. Analysis on potential effects of climate change on fin whales in the action area is included below in section 6.0.

4.4.3 Summary of Fin Whale Status

Information on the abundance and population structure of fin whales worldwide is limited. NMFS recognizes three fin whale stocks in the Pacific for the purposes of managing this species under the MMPA. Reliable estimates of current abundance for the entire Northeast Pacific fin whale stock are not available (Angliss *et al.* 2001). Stock structure for fin whales in the southern hemisphere is unknown and there are no current estimates of abundance for southern hemisphere fin whales. As noted above, the best population estimate for the western North Atlantic fin whale is 3,985 and the minimum population estimate is 3,269. The 2010 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 restarted 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. Without sufficient data to determine current fin whale population trends, we are unable to predict the potential trend of fin whales over the next 50 years as well.

4.5 Atlantic Sturgeon

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

The Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) is a subspecies of sturgeon distributed along the eastern coast of North America from Hamilton Inlet, Labrador, Canada to Cape Canaveral, Florida, USA (Scott and Scott 1988; ASSRT 2007; T. Savoy, CT DEP, pers. comm.). NMFS has delineated U.S. populations of Atlantic sturgeon into five DPSs (77 FR 5880 and 77 FR 5914).[5] These are: the Gulf of Maine (GOM), New York Bight (NYB), Chesapeake Bay (CB), Carolina, and South Atlantic (SA) DPSs. The results of genetic studies suggest that natal origin influences the distribution of Atlantic sturgeon in the marine environment (Wirgin and King, 2011). However, genetic data as well as tracking and tagging data demonstrate sturgeon from each DPS and Canada occur throughout the full range of the subspecies. Therefore, sturgeon originating from any of the 5 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 NYB, CB, Carolina, and SA DPSs as endangered, and the GOM DPS as threatened (77 FR 5880 and 77 FR 5914). The effective date of the listings was April 6, 2012. The DPSs do not include Atlantic sturgeon that are spawned in Canadian rivers. Therefore, Canadian spawned fish are not included in the listings.

As described below, individuals originating from all of the 5 listed DPSs may occur in the action area. Information general to all Atlantic sturgeon as well as information specific to each of the relevant DPSs, is provided below.

4.5.1 Atlantic sturgeon life history

Atlantic sturgeon are long lived (approximately 60 years), late maturing, estuarine dependent, anadromous fish (Bigelow and Schroeder 1953; Vladykov and Greeley 1963; Mangin 1964; Pikitch *et al.* 2005; Dadswell 2006; ASSRT 2007).[6] They are a relatively large fish, even amongst sturgeon species (Pikitch *et al.*, 2005). Atlantic sturgeons are bottom feeders that suck food into a ventrally-located protruding mouth (Bigelow and Schroeder 1953). Four barbels in front of the mouth assist the sturgeon in locating prey (Bigelow and Schroeder 1953). Diets of adult and migrant subadult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as sand lance (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007; Savoy 2007). Juvenile Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007).

Rate of maturation is affected by water temperature and gender. In general: (1) Atlantic sturgeon that originate from southern systems grow faster and mature sooner than Atlantic sturgeon that

[5] To be considered for listing under the ESA, a group of organisms must constitute a "species." A "species" is defined in section three 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."

[6] 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 FAQ's, available at http://www.nefsc.noaa.gov/faq/fishfaq1a.html, modified June 16, 2011).

originate from more northern systems; (2) males grow faster than females; (3) fully mature females attain a larger size (i.e. length) than fully mature males; and (4) the length of Atlantic sturgeon caught since the mid-late 20th century have typically been less than 3 meters (m) (Smith *et al.* 1982; Smith *et al.* 1984; Smith 1985; Scott and Scott 1988; Young *et al.* 1998; Collins *et al.* 2000; Caron *et al.* 2002; Dadswell 2006; ASSRT 2007; Kahnle *et al.* 2007; DFO, 2011). The largest recorded Atlantic sturgeon was a female captured in 1924 that measured approximately 4.26 m (Vladykov and Greeley 1963). Dadswell (2006) reported seeing seven fish of comparable size in the St. John River estuary from 1973 to 1995. Observations of large sized sturgeon are particularly important given that egg production is correlated with age and body size (Smith *et al.* 1982; Van Eenennaam *et al.* 1996; Van Eenennaam and Doroshov 1998; Dadswell 2006). However, while females are prolific with egg production ranging from 400,000 to 4 million eggs per spawning year, females spawn at intervals of 2-5 years (Vladykov and Greeley 1963; Smith *et al.* 1982; Van Eenennaam *et al.* 1996; Van Eenennaam and Doroshov 1998; Stevenson and Secor 1999; Dadswell 2006). Given spawning periodicity and a female's relatively late age to maturity, the age at which 50 percent of the maximum lifetime egg production is achieved is estimated to be 29 years (Boreman 1997). Males exhibit spawning periodicity of 1-5 years (Smith 1985; Collins *et al.* 2000; Caron *et al.* 2002). While long-lived, Atlantic sturgeon are exposed to a multitude of threats prior to achieving maturation and have a limited number of spawning opportunities once mature.

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

The spawning areas in most U.S. rivers have not been well defined. However, the habitat characteristics of spawning areas have been identified based on historical accounts of where fisheries occurred, tracking and tagging studies of spawning sturgeon, and physiological needs of early life stages. Spawning is believed to occur in flowing water between the salt front of estuaries and the fall line of large rivers, when and where optimal flows are 46-76 cm/s and depths are 3-27 m (Borodin 1925; Dees 1961; Leland,1968; Scott and Crossman, 1973; Crance 1987; Shirey *et al.* 1999; Bain *et al.* 2000; Collins *et al.*, 2000; Caron *et al.* 2002; Hatin *et al.* 2002; ASMFC, 2009). Sturgeon eggs are deposited on hard bottom substrate such as cobble, coarse sand, and bedrock (Dees 1961; Scott and Crossman 1973; Gilbert 1989; Smith and Clugston 1997; Bain *et al.* 2000; Collins *et al.* 2000; Caron *et al.* 2002; Hatin *et al.* 2002; Mohler 2003; ASMFC 2009), and become adhesive shortly after fertilization (Murawski and Pacheco 1977; Van den Avyle 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 4 weeks old, with total lengths (TL) less than 30 mm; Van Eenennaam *et al.* 1996) are assumed to undertake a demersal existence and inhabit the same riverine or estuarine areas where they were spawned (Smith *et al.* 1980; Bain *et al.* 2000; Kynard and Horgan 2002; ASMFC 2009). Studies suggest that age-0 (i.e., young-of-year), age-1, and age-2 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 higher salinity waters as well as low salinity waters (Collins *et al.* 2000). Atlantic sturgeon remain in the natal estuary for months to years before emigrating to open ocean as subadults (Holland and Yelverton 1973; Dovel and Berggen 1983; Waldman *et al.* 1996; Dadswell 2006; ASSRT 2007).

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

4.5.2 Determination of DPS Composition in the Action Area

As explained above, the range of all 5 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. We have determined that Atlantic sturgeon in the action area likely originate from the five DPSs at the following frequencies: NYB 49%; South Atlantic 20%; Chesapeake Bay 14%; Gulf of Maine 11%; and Carolina 4.0%. 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).

Distribution and Abundance

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

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

It is possible, however, to estimate the total number of adults in some other rivers based on the number of mature adults in the Hudson River. We have calculated an estimate of total mature adults and a proportion of subadults for four of the five DPSs. The technique used to obtain these estimates is explained fully in Damon-Randall 2012(b) and is summarized briefly below. We used this method because for these four DPSs, there are: (1) no total population estimates available; (2) with the exception of the Hudson River, no estimates of the number of mature adults; and, (3) no information from directed population surveys which could be used to generate an estimate of the number of spawning adults, total adult population or total DPS population.

Kahnle *et al.* (2007) estimated the number of total mature adults per year in the Hudson River using data from surveys in the 1980s to mid-1990s and based on mean harvest by sex divided by sex specific exploitation rate. While this data is over 20 years old, it is currently the best available data on the abundance of Hudson River origin Atlantic sturgeon. The sex ratio of spawners is estimated to be approximately 70% males and 30% females. As noted above, Kahnle *et al.* (2007) estimated a mean annual number of mature adults at 596 males and 267 females.

We were able to use this estimate of the adult population in the Hudson River and the rate at which Atlantic sturgeon from the Hudson River are intercepted in certain Northeast commercial fisheries to estimate the number of adults in other spawning rivers. As noted above, the method used is summarized below and explained fully in Damon-Randall (2012(b)).[7]

Given the geographic scope of commercial fisheries as well as the extensive marine migrations of Atlantic sturgeon, fish originating from nearly all spawning rivers are believed to be intercepted by commercial fisheries. An estimate of the number of Atlantic sturgeon captured in certain fisheries authorized by NMFS under Federal FMPs in the Northeast is available (NEFSC 2011). This report indicates that based on observed interactions with Atlantic sturgeon in sink gillnet and otter trawl fisheries from 2006-2010, on average 3,118 Atlantic sturgeon are captured in these fisheries each year. Information in the NEFOP database, indicates that 25% of captured Atlantic sturgeon are adults (determined as length greater than 150 cm) and 75% are subadults (determined as length less than 150cm). By applying the mixed stock genetic analysis of individuals sampled by the NEFOP and At Sea Monitoring Program (see Damon-Randall *et al.* 2012a) to the bycatch estimate, we can determine an estimate of the number of Hudson River Atlantic sturgeon that are intercepted by these fisheries on an annual basis.[8]

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[7] Bycatch information was obtained from a report prepared by NMFS' Northeast Fisheries Science Center (NEFSC 2012).

[8] Based on the best available information, we expect 46% of Atlantic sturgeon captured in Northeast commercial fisheries originate from the New York Bight DPS and 91% of those individuals originate from the Hudson River (see Damon-Randall *et al.* 2012a and Wirgin and King 2011).

Given the number of observed Hudson River origin Atlantic sturgeon adults taken as bycatch, we can calculate what percentage of Hudson River origin Atlantic sturgeon mature adults these represent. This provides an interception rate. We assume that fish originating in any river in any DPS are equally likely to be intercepted by the observed commercial fisheries; therefore, we can use this interception rate to estimate the number of Atlantic sturgeon in the other rivers of origin. This type of back calculation allows us to use the information we have for the Hudson River and fill in significant data gaps present for the other rivers. Using this method, for the purposes of this consultation, we have estimated the total adult populations for three DPSs (Gulf of Maine, Chesapeake Bay, and South Atlantic) as follows. It is important to note that this method likely underestimates the total number of adults in the SA DPS because genetic analysis of individuals observed through the NEFOP program indicate that only individuals from the Savannah and Ogeechee are being captured in Northeast fisheries considered in the NEFSC bycatch report. Spawning is known to occur in other rivers in the SA DPS, including the Altamaha (estimate of 343 adult spawners per year).

Given the proportion of adults to subadults in the observer database (ratio of 1:3), we can also estimate a number of subadults originating from each DPS. However, this cannot be considered an estimate of the total number of subadults because it would only consider those subadults that are of a size vulnerable to captured in commercial sink gillnet and otter trawl gear in the marine environment and are present in the marine environment.

Currently, there are an estimated 343 spawning adults in the Altamaha and there are estimated to be less than 300 spawning adults (total of both sexes) in each of the other major river systems occupied by the South Atlantic DPS. Spawning is thought to occur in six rivers in the SA DPS. Adding these estimates together results in a total adult population estimated of less than 1,843 mature adults. Our fishery dependent estimate is 390. This is likely an underestimate of the total number of adults in the SA DPS because genetic analysis of individuals observed through the NEFOP program indicate that only individuals from the Savannah and Ogeechee are being captured in Northeast fisheries considered in the NEFSC bycatch report. Because of this, it is difficult to compare these two estimates. It may be reasonable to consider the estimate of 390 adults to be an estimate of the number of adults in the Savannah and Ogeechee rivers only. This would be consistent with the assumption that there are fewer than 300 adults in each of these two rivers.

We are not able to use this method to calculate an adult population estimate for the Carolina DPS. Based on the results of the genetic mixed stock analysis, fish originating from the Carolina DPS appear rarely in the Northeast Fisheries Observer Program (NEFOP) observer dataset (e.g., 4% of the 173 fish observed). While we are unable to calculate a population estimate using the above methodology, we do have an estimate of 1500 adult spawners/year (5 spawning rivers x 300 spawning adults per river) described in the Atlantic sturgeon status review report. For the South Atlantic DPS, using this method, the estimated number of fish in the South Atlantic DPS would be 1800 spawning adults (6 spawning rivers x 300 spawning adults per river). Therefore, the Carolina DPS has approximately 17% less fish than the South Atlantic DPS. Based on the methodology described above, the estimated number of mean annual mature adults for the South Atlantic DPS is 390 fish. Using the proportion of Carolina DPS fish to

South Atlantic DPS fish, we estimate that the mean number of annual mature adults in the Carolina DPS is 324 (17% less than 390).

Table 1: **Summary of Calculated Population Estimates from NER Fisheries Dependent Data**

DPS	Estimated Adult Population	Estimated Subadults of Size vulnerable to capture in commercial fisheries
GOM	215	645
NYB (Hudson River and Delaware River)	951	2,853
CB	273	819
SA*	390	1,170
Carolina*	324	972

*see note re. South Atlantic and Carolina population sizes in paragraphs above.

Threats faced by Atlantic sturgeon throughout their range

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

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

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

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

Fisheries bycatch in U.S. waters is one of the primary threats faced by all 5 DPSs. At this time, we have an estimate of the number of Atlantic sturgeon captured and killed in sink gillnet and otter trawl fisheries authorized by Federal FMPs (NMFS NEFSC 2011) in the Northeast Region, as well as estimates for the shrimp and Highly Migratory Species fisheries in the Southeast Region (NMFS 2012; A. Herndon, pers. comm.). We also do not have an estimate of the number of Atlantic sturgeon captured or killed in state fisheries. At this time, we are not able to quantify the effects of other significant threats (e.g., vessel strikes, poor water quality, water availability, dams, and dredging) in terms of habitat impacts or loss of individuals. While we have some information on the number of mortalities that have occurred in the past in association with certain activities (e.g., mortalities in the Delaware and James Rivers that are thought to be due to vessel strikes), we are not able to use those numbers to extrapolate effects throughout one or more DPS. This is because of (1) the small number of data points and, (2) lack of information on the percent of incidences that the observed mortalities represent.

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

Global climate change may affect all DPSs of Atlantic. Further analysis on potential effects of climate change on Atlantic sturgeon in the action area is included in section 6.0 below.

Information specific to each DPS is presented in the sections below.

4.5.2.1 Gulf of Maine (GOM) DPS

The GOM 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 and Androscoggin Rivers, and it is possible that it still occurs in the Penobscot River as well. Spawning in the Androscoggin River may also be occurring. Maine Department of Marine Resources reported the capture of a larval Atlantic sturgeon during the 2011 spawning season below the Brunswick Dam; this suggests that spawning may be occurring in this area. 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 GOM DPS as well as likely throughout the entire range (ASSRT 2007; Fernandes *et al.* 2010).

Several threats play a role in shaping the current status of GOM DPS Atlantic sturgeon. Historical records provide evidence of commercial fisheries for Atlantic sturgeon in the Kennebec and Androscoggin Rivers dating back to the 17th century (Squiers *et al.* 1979). In 1849, 160 tons of sturgeon was caught in the Kennebec River by local fishermen (Squiers *et al.* 1979). Following the 1880's, the sturgeon fishery was almost non-existent due to a collapse of the sturgeon stocks. All directed Atlantic sturgeon fishing as well as retention of Atlantic sturgeon by catch has been prohibited since 1998. Nevertheless, mortalities associated with bycatch in fisheries occurring in state and federal waters still occurs. In the marine range, GOM 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 GOM DPS have navigation channels that are maintained by dredging. Dredging outside of Federal channels and in-water construction occurs throughout the GOM DPS. While some dredging projects operate with observers present to document fish mortalities, many do not. To date we have not received any reports of Atlantic sturgeon killed during dredging projects in the Gulf of Maine region; however, as noted above, not all projects are monitored for interactions with fish. At this time, we do not have any information to quantify the number of Atlantic sturgeon killed or disturbed during dredging or in-water construction projects are also not able to quantify any effects to habitat.

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

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

There are no empirical abundance estimates for the GOM DPS. The Atlantic sturgeon SRT (2007) presumed that the GOM 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. As explained above, we have estimated that there is an annual mean of 166 mature adult Atlantic sturgeon in the GOM DPS.

Summary of the Gulf of Maine DPS

Spawning for the GOM 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 GOM DPS. Atlantic sturgeon continue to be present in the Kennebec River; in addition, they are captured in directed research projects in the Penobscot River, and are observed in rivers where they were unknown to occur or had not been observed to occur for many years (e.g., the Saco, Presumpscot, and Charles rivers). These observations suggest that abundance of the GOM DPS of Atlantic sturgeon is sufficient such that recolonization to rivers historically suitable for spawning may be occurring. However, despite some positive signs, there is not enough information to establish a trend for this DPS.

Some of the impacts from the threats that contributed to the decline of the GOM DPS have been removed (e.g., directed fishing), or reduced as a result of improvements in water quality and removal of dams (e.g., the Edwards Dam on the Kennebec River in 1999). 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 GOM 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 GOM 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 GOM DPS is at risk of becoming endangered in the foreseeable future throughout all of its range (i.e., is a threatened species) based on the following: (1) significant declines in population sizes and the protracted period during which sturgeon populations have been depressed; (2) the limited amount of current spawning; and, (3) the impacts and threats that have and will continue to affect recovery.

4.5.2.2 New York Bight (NYB) DPS

The NYB 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 1800's 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 1970's (Kahnle *et al.*, 1998). A decline appeared to occur in the mid to late 1970's followed by a secondary drop in the late 1980's (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 1980's (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 and while 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. 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 1800's indicate that this was historically a large population with an estimated 180,000 adult females prior to 1890 (Secor and Waldman, 1999; Secor, 2002). Sampling in 2009 to target young-of- the year (YOY) Atlantic sturgeon in the Delaware River (i.e., natal sturgeon) resulted in the capture of 34 YOY, ranging in size from 178 to 349 mm TL (Fisher 2009) and the collection of 32 YOY Atlantic sturgeon in a separate study (Brundage and O'Herron in Calvo *et al.* 2010). Genetics information collected from 33 of the 2009 year class YOY indicates that at least 3 females successfully contributed to the 2009 year class (Fisher 2011). Therefore, while the capture of YOY in 2009 provides evidence that successful spawning is still occurring in the Delaware River, the relatively low numbers suggest the existing riverine population is limited in size.

Several threats play a role in shaping the current status and trends observed in the Delaware River and Estuary. In-river threats include habitat disturbance from dredging, and impacts from historical pollution and impaired water quality. A dredged navigation channel extends from Trenton seaward through the tidal river (Brundage and O'Herron 2009), and the river receives significant shipping traffic. Vessel strikes have been identified as a threat in the Delaware River;

however, at this time we do not have information to quantify this threat or its impact to the population or the New York Bight DPS. Similar to the Hudson River, there is currently not enough information to determine a trend for the Delaware River population.

Summary of the New York Bight DPS

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

In the marine range, NYB DPS Atlantic sturgeon are incidentally captured in federal and state managed fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (Stein *et al.* 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 NYB DPS. Individual-based assignment and mixed stock analysis of samples collected from sturgeon captured in Canadian fisheries in the Bay of Fundy indicated that approximately 1-2% were from the NYB DPS. 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.

NYB 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 NYB DPS.

Studies have shown that to rebuild, Atlantic sturgeon can only sustain low levels of anthropogenic mortality (Boreman 1997; ASMFC, 2007; Kahnle *et al.* 2007; Brown and Murphy 2010). There are no empirical abundance estimates of the number of Atlantic sturgeon in the NYB DPS. As explained above, we have estimated that there are an annual mean total of 950 mature adult Atlantic sturgeon in the NYB DPS. NMFS has determined that the NYB 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.

4.5.2.3 Chesapeake Bay (CB) DPS

The CB 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). 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).

Several threats play a role in shaping the current status of CB DPS Atlantic sturgeon. Historical records provide evidence of the large-scale commercial exploitation of Atlantic sturgeon from the James River and Chesapeake Bay in the 19th century (Hildebrand and Schroeder 1928; Vladykov and Greeley 1963; ASMFC 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 CB 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 CB DPS.

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

Summary of the Chesapeake Bay DPS

Spawning for the CB DPS is known to occur in only the James River. Spawning may be occurring in other rivers, such as the York, but has not been confirmed. There are anecdotal reports of increased sightings and captures of Atlantic sturgeon in the James River. However, this information has not been comprehensive enough to develop a population estimate for the James River or to provide sufficient evidence to confirm increased abundance. Some of the impact from the threats that facilitated the decline of the CB DPS have been removed (e.g., directed fishing) or reduced as a result of improvements in water quality since passage of the Clean Water Act (CWA). As explained above, we have estimated that there is an annual mean of 329 mature adult Atlantic sturgeon in the CB DPS. 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 CB 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 CB 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.

4.5.2.4 The South Atlantic (SA) DPS

Distribution and Abundance

The SA 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. The riverine range of the South Atlantic DPS and the adjacent portion of the marine range are shown in Figure 3. 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 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 2). 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 SA DPS for specific life functions, such as spawning, nursery habitat, and foraging. However, fish from the SA DPS likely use other river systems than those listed here for their specific life functions.

Table 2. Major rivers, tributaries, and sounds within the range of the South Atlantic DPS and currently available data on the presence of an Atlantic sturgeon spawning population in each system.

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

The riverine spawning habitat of the SA 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 SA DPS. Currently, the Atlantic sturgeon spawning population in at least two river systems within the SA 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 abundances of the remaining river populations within the DPS, each estimated to have fewer than 300 spawning adults, is estimated 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 SA DPS. Dredging is a present threat to the SA DPS and is contributing to their status by modifying the quality and availability of Atlantic sturgeon habitat. Maintenance dredging is currently modifying Atlantic sturgeon nursery habitat in the Savannah River and modeling indicates that the proposed deepening of the navigation channel will result in reduced DO and upriver movement of the salt wedge, curtailing spawning habitat. Dredging is also modifying nursery and foraging habitat in the St. Johns Rivers. Reductions in water quality from terrestrial activities have modified habitat utilized by the SA DPS. Low DO is modifying sturgeon habitat in the Savannah due to dredging, and non-point source inputs are causing low DO in the Ogeechee River and in the St. Marys River, which completely eliminates juvenile nursery habitat in summer. Low DO has also been observed in the St. Johns River in the summer. Sturgeon are more highly sensitive to low DO and the negative (metabolic, growth, and feeding) effects caused by low DO increase when water temperatures are concurrently high, 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. Known large water withdrawals of over 240 million gallons per day (mgd) of water may be removed from the Savannah River for power generation and municipal uses. However, permits for users withdrawing less than 100,000 gallons per day (gpd) are not required to get permits, so actual water withdrawals from the Savannah and other rivers within the range of the SA 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 SA 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 SA 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 SA DPS. Atlantic sturgeon are more 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 percent of their population to bycatch mortality without suffering population declines. Mortality rates of Atlantic sturgeon taken as bycatch in various types of fishing gear range between 0 and 51 percent, 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. 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 SA 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.

Viability of the South Atlantic DPS

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 SA 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 SA DPS have remained relatively constant at greatly reduced levels (approximately 6 percent of historical population sizes in the Altamaha River, and 1 percent of historical population sizes in the remainder of the DPS) for 100 years. Small numbers of individuals resulting from drastic reductions in populations, such as occurred with Atlantic sturgeon due to the commercial fishery, can remove the buffer against natural demographic and environmental variability provided by large populations (Berry, 1971; Shaffer, 1981; Soulé, 1980). Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon, and they continue to face a variety of other threats that contribute to their risk of extinction. Their late age at maturity provides more opportunities for individual Atlantic sturgeon to be removed from the population before reproducing. While a long life-span also allows multiple opportunities to contribute to future generations, it also results increases the timeframe over which exposure to the multitude of threats facing the SA DPS can occur.

The viability of the SA DPS depends on having multiple self-sustaining riverine spawning populations and maintaining suitable habitat to support the various life functions (i.e., 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 SA DPS of Atlantic Sturgeon

The SA DPS is estimated to number fewer than 6 percent of its historical population size, with all river populations except the Altamaha estimated to be less than 1 percent of historical abundance. There are an estimated 343 spawning adults per year in the Altamaha and less than 300 spawning adults per year (total of both sexes) in each of the other major river systems occupied by the DPS in which spawning still occurs, whose freshwater range occurs in the watersheds (including all rivers and tributaries) of the ACE Basin southward along the South Carolina, Georgia, and Florida coastal areas to the St. Johns River, Florida. Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon. Their late age at maturity provides more opportunities for individuals to be removed from the population before reproducing. While a long life-span also allows multiple opportunities to contribute to future generations, this is hampered within the South Atlantic DPS by habitat alteration, bycatch, and from the inadequacy of existing regulatory mechanisms to address and reduce habitat alterations and bycatch.

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

4.5.2.5 Carolina DPS

Distribution and Abundance

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 3). 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.

Table 3. Major rivers, tributaries, and sounds within the range of the Carolina DPS and currently available data on the presence of an Atlantic sturgeon spawning population in each system.

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

Ashley River, SC	Unknown	

The 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. Prior reductions from the commercial fishery and ongoing threats have drastically reduced the numbers of Atlantic sturgeon within the Carolina DPS. Currently, the Atlantic sturgeon spawning population in at least one river system within the Carolina DPS has been extirpated, with a potential extirpation in an additional system. The abundances of the remaining river populations within the DPS, each estimated to have fewer than 300 spawning adults, is estimated to be less than 3 percent of what they were historically (ASSRT 2007).

Threats

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

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

Overutilization of Atlantic sturgeon from directed fishing caused initial severe declines in Atlantic sturgeon populations in the Southeast, from which they have never rebounded. Further, continued overutilization of Atlantic sturgeon as bycatch in commercial fisheries is an ongoing impact to the Carolina DPS. Atlantic sturgeon are more 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 percent of their population to bycatch mortality without suffering population declines. Mortality rates of Atlantic sturgeon taken as bycatch in various types of fishing gear range between 0 and 51 percent, 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. 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 Carolina DPS, even with existing controls on some pollution sources. Current regulatory regimes are not necessarily effective in controlling water allocation issues (e.g., no restrictions on interbasin water transfers in South Carolina, the lack of ability to regulate non-point source pollution, etc.)

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

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

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

Summary of the Status of the Carolina DPS of Atlantic Sturgeon

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

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

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: 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 vessel operations and gear associated with federally-permitted fisheries on threatened and endangered species in the action area. Each of those consultations sought to develop ways of reducing the probability of adverse impacts of the action on listed species. Formal consultations completed in the action area are summarized below.

Federal Vessel Operations
Potential adverse effects on listed species 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.

Federal Fishery Operations
NMFS authorizes the operation of several fisheries in the action area under the authority of the Magnuson-Stevens Fishery Conservation Act and through Fishery Management Plans and their implementing regulations. Commercial and recreational fisheries in the action area employ gear that is known to harass, injure, and/or kill sea turtles and Atlantic sturgeon. In the Northeast Region (Maine through Virginia), formal ESA section 7 consultations have been conducted on

the American lobster, Atlantic bluefish, Atlantic mackerel/squid/ butterfish, Atlantic sea scallop, monkfish, northeast multispecies, skate, red crab, spiny dogfish, summer flounder/scup/black sea bass, and tilefish fisheries. Of those consultations, only portions of the Atlantic bluefish, Atlantic mackerel/squid/ butterfish, skate, monkfish, northeast multispecies, spiny dogfish, summer flounder/scup/black sea bass, and tilefish fisheries occur within the action area. These consultations have considered effects to loggerhead, green, Kemp's ridley and leatherback sea turtles. We have completed Biological Opinions on the operations of these fisheries. In each of these Opinions, we concluded that the ongoing action was likely to adversely affect but was not likely to jeopardize the continued existence of any sea turtle species. Each of these Opinions included an incidental take statement exempting a certain amount of lethal and/or non-lethal take resulting from interactions with the fishery. These ITSs are summarized in the table below. Further, in each Opinion, we concluded that the potential for interactions (i.e., vessel strikes) between sea turtles and fishing vessels was extremely low and similarly that any effects to sea turtle prey and/or habitat would be insignificant and discountable. We have also determined that the Atlantic herring and surf clam/ocean quahog fisheries do not adversely affect any species of listed sea turtles.

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

FMP	**Date of Most Recent Opinion**	**Loggerhead**	**Kemp's ridley**	**Green**	**Leatherback**
Atlantic bluefish	October 29, 2010	82 (34 lethal)	4	5	4
Monkfish	October 29, 2010	173 (70 lethal)	4	5	4
Multispecies	October 29, 2010	46 in trawls (21 lethal)	4	5	4
Skate	October 29, 2010	39 (17 lethal)	4	5	4
Spiny dogfish	October 29, 2010	2	4	5	4
Mackerel/squid/butterfish	October 29, 2010	62 (25 lethal)	2	2	2
Summer flounder/scup/black sea bass	October 29, 2010	205 (85 lethal)	4	5	6

Pelagic longline under the HMS FMP (per the RPA)	June 1, 2004	1,905 (339 lethal) every 3 years	*105 (18 lethal) every 3 years	*105 (18 lethal) every 3 years	1764 (252 lethal) every 3 years
Tilefish	March 13, 2001	6 (3 lethal)			1

**combination of 105 (18 lethal) Kemp's ridley, green, hawksbill, or Olive ridley*
***combination of 16 turtles total every 3 years with 2 lethal (Kemp's ridley, green, hawksbill, leatherback)*
**** this consultation has been reinitiated and a new Opinion is expected in 2012*

We are in the process of reinitiating consultations that consider fisheries actions that may affect Atlantic sturgeon. Sturgeon originating from the four DPSs considered in this consultation are known to be captured and killed in fisheries operated in the action area. At the time of this writing, no Opinions considering effects of federally authorized fisheries on any DPS of Atlantic sturgeon have been completed. As noted in the Status of the Species section above, the NEFSC prepared a bycatch estimate for Atlantic sturgeon captured in sink gillnet and otter trawl fisheries operated from Maine through Virginia. This estimate indicates that, based on data from 2006-2010, annually, an average of 3,118 Atlantic sturgeon are captured in these fisheries with 1,569 in sink gillnet and 1,548 in otter trawls. The mortality rate in sink gillnets is estimated at approximately 20% and the mortality rate in otter trawls is estimated at 5%. Based on this estimate, a total of 391 Atlantic sturgeon are estimated to be killed annually in these fisheries that are prosecuted in the action area. We are currently in the process of determining the effects of this annual loss to each of the DPSs. Any of these fisheries that operate with sink gillnets or otter trawls are likely to interact with Atlantic sturgeon and be an additional source of mortality in the action area.

5.2 Non-Federally Regulated Actions

Private and Commercial Vessel Operations

Private and commercial vessels, including fishing vessels, operating in the action area of this consultation also have the potential to interact with listed species. 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 certain 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 Virginia; however, very little is known about the level of interactions with listed species in fisheries that operate strictly in state waters, although 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. However, depending on the fishery in question, many state permit holders also hold federal licenses; therefore, section 7 consultations on federal actions in those fisheries address some state-water activity. Impacts of state fisheries on endangered whales are addressed as appropriate through the MMPA take reduction planning process. 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

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 turtles are not very easily affected by changes in water quality or 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 Atlantic sturgeon, 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 also include atmospheric loading of pollutants, stormwater runoff from coastal development, groundwater discharges, industrial development, and debris and materials from launch activities occurring at WFF (i.e., spent rockets, payloads, and rocket-boosted projectiles, as well as non-hazardous expended material such as steel, aluminum, rubber, vinyl, glass, and plastics). 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 primarily as a concern 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 2007). 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 2007). Masking is a major concern about 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. 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. Larger oil spills may result from accidents, although these events would be rare and involve small areas. No direct adverse effects on listed sea turtles resulting from fishing vessel fuel spills have been documented.

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; and, 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 to or mortality of large whales due to incidental entanglement in U.S. commercial trap/pot and gillnet fishing gear. The ALWTRP focuses on the critically endangered North Atlantic right whale, but is also intended to reduce entanglement of endangered humpback and fin whales. The plan is required by the 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 (26°46.5'N lat.). The requirements are year-round in the Northeast, and seasonal in the Mid and South Atlantic.

The plan has been developed in collaboration with the Atlantic Large Whale Take Reduction Team (ALWTRT), which consists of fishing industry representatives, environmentalists, state and federal officials, and other interested parties. The ALWTRP is an evolving plan that changes as NMFS and the ALWTRT learn more about why whales become entangled and how fishing practices might be modified to reduce the risk of entanglement. Regulatory actions are directed at reducing serious entanglement injuries and 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. These components will be discussed in more detail below. The first ALWTRP went into effect in 1997.

5.4.1.1.1 ALWTRP Regulatory Measures to Reduce the Threat of Entanglement on Whales

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

5.4.1.1.2 Non-regulatory components of the ALWTRP

Gear Research and Development

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

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

Large Whale Disentanglement Program

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

Sighting Advisory System (SAS)

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

Educational Outreach

Education and outreach activities are considered one of the primary tools to reduce the threats to all protected species from human activities, including fishing activities. Outreach efforts for fishermen under the ALWTRP are fostering a more cooperative relationship between all parties interested in the conservation of threatened and endangered species. NMFS has also been active in public outreach to educate fishermen regarding sea turtle handling and resuscitation techniques. NMFS has conducted workshops with longline fishermen to discuss bycatch issues including protected species, and to educate them regarding handling and release guidelines. NMFS intends to continue these outreach efforts in an attempt to increase the survival of protected species through education on proper release techniques.

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) continuation of ongoing measures to reduce ship strikes of right whales (*e.g*., SAS, ongoing research into the factors that contribute to ship strikes, and research to identify new technologies that can help mariners and whales avoid each other).

5.4.1.3 Regulatory Measures to Reduce Vessel Strikes to Large Whales

Restricting vessel approach to right whales

In one (1) recovery action aimed at reducing vessel-related impacts, including disturbance, NMFS published a proposed rule in August 1996 restricting vessel approach to right whales (61

FR 41116, August 7, 1996) to a distance of 500 yards. The Recovery Plan for the North Atlantic right whale identified anthropogenic disturbance as one (1) of many factors which had some potential to impede right whale recovery (NMFS 2005). Following public comment, NMFS published an interim final rule in February 1997 codifying the regulations. With certain exceptions, the rule prohibits both boats and aircraft from approaching any right whale closer than 500 yds. Exceptions for closer approach are provided for the following situations, when: (a) compliance would create an imminent and serious threat to a person, vessel, or aircraft; (b) a vessel is restricted in its ability to maneuver around the 500-yard perimeter of a whale; (c) a vessel is investigating or involved in the rescue of an entangled or injured right whale; or (d) the vessel is participating in a permitted activity, such as a research project. If a vessel operator finds that he or she has unknowingly approached closer than 500 yds, the rule requires that a course be steered away from the whale at slow, safe speed. In addition, all aircraft, except those involved in whale watching activities, are exempted from these approach regulations. This rule is expected to reduce the potential for vessel collisions and other adverse vessel-related effects in the environmental baseline.

Mandatory Ship Reporting System (MSR)

In April 1998, the USCG submitted, on behalf of the US, a proposal to the International Maritime Organization (IMO) requesting approval of a mandatory ship reporting system (MSR) in two areas off the east coast of the 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 in the area and information on precautionary measures to take while in the vicinity of right whales.

Vessel Speed Restrictions

A key component of NOAA's right whale ship strike reduction program is the implementation of speed restrictions for vessels transiting the US Atlantic in areas and seasons where right whales predictably occur in high concentrations. The Northeast Implementation Team (NEIT)-funded "Recommended Measures to Reduce Ship Strikes of North Atlantic Right Whales" found that seasonal speed and routing measures could be an effective means of reducing the risk of ship strike along the US east coast. Based on these recommendations, NMFS published an Advance Notice of Proposed Rulemaking (ANPR) in June 2004 (69 FR 30857; June 1, 2004), and subsequently published a proposed rule on June 26, 2006 (71 FR 36299; June 26, 2006). NMFS published regulations on October 10, 2008 to implement a 10-knot speed restriction for all vessels 65 ft (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 these zones is voluntary.

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

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 US Coast Pilots, and mariners have been notified through USCG Notices to Mariners.

Through a joint effort between NOAA and the USCG, the US 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. Overlaying sightings of right whales and all baleen whales on the existing TSS revealed that the existing TSS directly overlaps with areas of high whale densities, while an area slightly to the north showed a considerable decrease in sightings. Separate analyses by the SBNMS and the NEFSC both indicated that the proposed TSS would overlap with 58% fewer right whale sightings and 81% fewer sightings of all large whales, thus considerably reducing the risk of collisions between ships and whales. The proposal was submitted to the IMO in April 2006, and was adopted by the Maritime Safety Committee in December 2006. The shift took effect on July 1, 2007. In 2009 this TSS was modified by narrowing the width of the north-south portion by one (1) mile to 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 ABTA 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 (1) 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. The SAS has also served as the only form of active entanglement monitoring in the Cape Cod Bay and Great South Channel feeding areas. Some of these sighting efforts have resulted in successful disentanglement of right whales. SAS flights have also contributed sightings of dead floating animals that can occasionally be retrieved to increase our knowledge of the biology of the species and effects of human impacts.

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

5.4.1.4 Marine Mammal Health and Stranding Response Program (MMHSRP)

NMFS was designated the lead agency to coordinate the MMHSRP which was formalized by the 1992 Amendments to the MMPA. The program consists of the following components:

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

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

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

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

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

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 (1) 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 is engaged in a number of education and outreach activities aimed specifically at increasing mariner awareness of the threat of ship strike to right whales. 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)

There is an extensive network of STSSN participants along the Atlantic and Gulf of Mexico coasts which not only collects data on dead sea turtles, but also rescues and rehabilitates live stranded turtles, reducing mortality of injured or sick animals. Data collected by the STSSN are used to monitor stranding levels and 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 provide an understanding of sea turtle movements, longevity, and reproductive patterns, all of which contribute to our ability to reach recovery goals for the species.

5.4.2.3 Sea Turtle Disentanglement Network (STD)

NMFS Northeast Region established the Northeast Sea Turtle Disentanglement Network (STDN) in 2002. This program was established in response to the high number of leatherback sea turtles found entangled in pot gear along the U.S. Northeast Atlantic coast. The 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 in order to disentangle and release live animals, thereby reducing serious injury and mortality. In addition, the STDN collects data on these 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. These restrictions were revised in 2006 (71 FR 24776, April 26, 2006). Currently, gillnets with stretched mesh size 7-inches (17.8 cm) or larger are prohibited in the Exclusive Economic Zone (as defined in 50 CFR 600.10) during the following times and in the following areas: (1) north of the NC/SC border to Oregon Inlet at all times, (2) north of Oregon Inlet to Currituck Beach Light, NC from March 16 through January 14, (3) north of Currituck Beach Light, NC to Wachapreague Inlet, VA from April 1 through January 14, and (4) north of Wachapreague Inlet, VA to Chincoteague, VA from April 16 through January 14.
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^{o}46.3' N. lat., north of 35^{o} 00' N. lat., and east of 76^{o} 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. This area is bounded on the north by a line extending along 37° 05'N latitude (Cape Charles, VA) and on the south by a line extending out from the North Carolina-South Carolina border. Vessels north of Oregon Inlet, NC are exempt from the TED requirement from January 15 through March 15 each year (50 CFR 223.206). The TED requirements for the summer flounder trawl fishery do not require the use of the larger escape opening. NMFS is considering increasing the size of the TED escape opening currently required in the summer flounder fishery and implementing sea turtle conservation requirements in other trawl fisheries and in other areas (72 FR 7382, February 15, 2007; 74 FR 21630, May 8, 2009).

Sea Turtle Conservation Requirements in the HMS Fishery

NMFS completed the most recent biological opinion on the FMP for the Atlantic HMS fisheries for swordfish, tuna, and shark on June 1, 2004, and concluded that the Atlantic HMS fisheries, particularly the pelagic longline fisheries, were likely to jeopardize the continued existence of leatherback sea turtles. A RPA was provided to avoid jeopardy to leatherback sea turtles as a result of operation of the HMS fisheries. Although the Opinion did not conclude jeopardy for loggerhead sea turtles, the RPA is also expected to benefit this species by reducing mortalities resulting from interactions with the gear. A number of requirements have been put in place as a result of the Opinion and subsequent research. These include measures related to the fishing gear, bait, disentanglement gear and training.

In 2008, NMFS completed a section 7 consultation on the continued authorization of HMS Atlantic shark fisheries. The commercial fishery uses bottom longline and gillnet gear. The recreational sector of the fishery uses only hook-and-line gear. To protect declining shark stocks the proposed action seeks to greatly reduce the fishing effort in the commercial component of the fishery. These reductions are likely to greatly reduce the interactions between the commercial component of the fishery and sea turtles. The biological opinion concluded that green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles may be adversely affected by operation of the fishery. However, the proposed action was not expected to jeopardize the continued existence of any of these species and an ITS was provided.

Sea Turtle Handling and Resuscitation Requirements

NMFS published as a final rule in the *Federal Register* (66 FR 67495, December 31, 2001) specifying handling and resuscitation requirements for sea turtles that are incidentally caught during scientific research or fishing activities. Persons participating in fishing activities or scientific research are required to handle and resuscitate (as necessary) sea turtles as prescribed in the regulations (50 CFR 223.206). These measures help to prevent mortality of turtles caught in fishing or scientific research gear.

Exception for injured, dead, or stranded specimens

Any agent or employee of NMFS, the USFWS, the U.S. Coast Guard, or any other Federal land or water management agency, or any agent or employee of a state agency responsible for fish and wildlife, when acting in the course of his or her official duties, is allowed to take threatened or endangered sea turtles encountered in the marine environment if such taking is necessary to aid a sick, injured, or entangled endangered sea turtle, or dispose of or salvage a dead endangered or threatened sea turtle (50 CFR 223.206(b); 50 CFR 222.310). This take exemption extends to NMFS' Sea Turtle Stranding and Salvage Network.

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 (i.e., offshore and coastal waters of Virginia/ Atlantic Ocean) and how listed sea turtles, whales, and sturgeon may be affected by those predicted environmental changes over the life of the action (i.e., between now and 2019). Climate change is relevant to the Status of the Species, Environmental Baseline, and Cumulative Effects sections of this Opinion; rather than include partial discussion in several sections of this Opinion, we are synthesizing this information into one discussion. Effects of the action that are relevant to climate change are included in the Effects of the Action section below (section 7.0 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 2007). 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 2007); these trends have been most apparent over the past few decades.

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 JEB Fort Story, 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. In the future, increasing temperatures, sea level rise, changes in ocean productivity, and increased frequency of storm events are expected as a result of climate change and are all potential threats for loggerheads. Increasing temperatures are expected to result in rising sea levels (Titus and Narayanan 1995 in Conant *et al*. 2009), which could result in increased erosion rates along nesting beaches. Sea level rise could result in the inundation of nesting sites and decrease available nesting habitat (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.

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

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, 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

Although leatherbacks are probably already beginning to be affected by impacts associated with anthropogenic climate change in several ways, no significant climate change-related impacts to leatherback turtle populations have been observed to date (PIRO Long Line Fisheries BO 2012). However, 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 prey, jellyfish, 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), which may or may not impact leatherbacks as there is no evidence that any leatherback populations are currently food-limited. Even though there may be a benefit to leatherbacks due to climate change influence on productivity we do not know what impact other climate-related changes may have such as increasing sand temperatures, sea level rise, and increased storm events.

As discussed for loggerheads, increasing temperatures are expected to result in rising sea levels (Titus and Narayanan 1995 in Conant *et al*. 2009), which could result in increased erosion rates along nesting beaches. Sea level rise could result in the inundation of nesting sites and decrease available nesting habitat (Fish *et al*. 2005). This effect would potentially be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents.

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 as warmer sand temperatures at nesting beaches are likely to result in the production of more female embryos. At least one nesting site, Ascension Island, has had an increase in mean sand temperature in recent years (Hays *et al*. 2003 in NMFS and USFWS 2007c). Climate change may also impact nesting beaches through sea level rise which may reduce the availability of nesting habitat and increase the risk of nest inundation. Loss of appropriate nesting habitat may also be accelerated by a combination of other environmental and oceanographic changes, such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion. Oceanic changes related to rising water temperatures could result in changes in the abundance and distribution of the primary food sources of green sea turtles, which in turn could result in changes in behavior and distribution of this species. Seagrass habitats may suffer from decreased productivity and/or increased stress due to sea level rise, as well as salinity and temperature changes (Short and Neckles 1999; Duarte 2002).

6.2.5 Right, Humback, and Fin Whales

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

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

6.2.6 Atlantic Sturgeon

Atlantic sturgeon 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. As such, climate change at normal rates (thousands of years) is not thought to have historically been a problem for sturgeon species. However, at the given rate of global climate change, future affects to Atlantic sturgeon are possible. Rising sea level may result in the salt wedge moving upstream in affected rivers. Atlantic sturgeon spawning occurs in fresh water reaches of rivers because early life stages have little to no tolerance for salinity. Similarly, juvenile Atlantic sturgeon have limited tolerance to salinity and remain in waters with little to no salinity. If the salt wedge moves further upstream, Atlantic sturgeon spawning and rearing habitat could be restricted. In river systems with dams or natural falls that are impassable by sturgeon, the extent that spawning or rearing may be shifted upstream to compensate for the shift in the movement of the saltwedge would be limited. While there is an indication that an increase in sea level rise would result in a shift in the location of the salt wedge, at this time there are no predictions on the timing or extent of any shifts that may occur; thus, it is not possible to predict any future loss in spawning or rearing habitat. However, in all river systems, spawning occurs miles upstream of the saltwedge. It is unlikely that shifts in the location of the saltwedge would eliminate freshwater spawning or rearing habitat. If habitat was severely restricted, productivity or survivability may decrease.

The increased rainfall predicted by some models in some areas may increase runoff and scour spawning areas and flooding events could cause temporary water quality issues. Rising temperatures predicted for all of the U.S. could exacerbate existing water quality problems with DO and temperature. While this occurs primarily in rivers in the southeast U.S. and the Chesapeake Bay, it may start to occur more commonly in the northern rivers. Atlantic sturgeon are tolerant to 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 extremely limited. Scientists from George Mason University and Center for Ocean-Land-Atmosphere Studies in Maryland found that from 2000 to 2099 the average warming for Virginia and the adjoining areas would be 3.1°C (5.6°F) and that precipitation would increase by 11% (Bryant 2008). NOAA tide gauge data reported by the State indicates that the sea level within Virginia portion of the Chesapeake Bay has risen, on average, at a rate of approximately 4.5 mm/yr since recordings began in 1927. Similarly, Zervas (2004) observed sea level rise rates of 4.4 mm/yr in the Mid-Atlantic region between northern New Jersey and Northeastern North Carolina. In addition, offshore waters of Virginia range between 7^{o}C to 28^{o}C (http://www.surf-forecast.com/breaks/Virginia-Beach/seatemp, last visited 5/30/2012), with an expected rise in sea surface temperature over the next 100 years of up to 3^{o}C (Nicholls *et al.* 2007).

6.4 Effects of Climate Change in the Action Area to Listed Species of Sea Turtles, Whales, and 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 whales, sea turtles, and Atlantic sturgeon; however, we have considered the available information to consider likely impacts to these species in the action area. The action will be under taken for a period 50 years; thus, we consider here, likely effects of climate change during the period from now until 2062.

6.4.1 Whales

As described above, the impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of seawater due to melting ice and increased rainfall, sea level rise, the loss of polar habitats, and potential shifts in the distribution and abundance of prey species. These impacts, in turn, are likely to affect the distribution of species of whales. Based on the location of the action area (i.e., Atlantic Ocean, coastal waters of Virginia), the most likely effect to whales in the action area 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 life of the action (through 2062) it is unlikely that this possible shift in range will be observed due the extremely small increase in water temperature predicted to occur during the lifetime of the project (i.e., approximately 1.5^{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 over the life of the action.

6.4.2 Atlantic Sturgeon

Although climate change has the potential to impact Atlantic sturgeon in various ways (see section 6.2.6), due to the location of the action area (i.e., coastal, offshore waters), the most likely effect to Atlantic sturgeon in the action area from climate change would be if warming temperatures led to changes in their range and migratory patterns. Warming temperatures predicted to occur over the next 100 years 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, over the life of the action (i.e., through 2062), this increase in sea surface temperature would be minimal (i.e., approximately 1.5^{o}C) and thus, it is unlikely that this expanded range will be observed over the next 50 years. If any shift does occur, it is likely to be minimal and thus, it seems unlikely that this small increase in temperature will cause a significant effect to Atlantic sturgeon or a significant modification to the number of sturgeon likely to be present in the action area over the life of the action.

Although the action area is not a spawning ground for Atlantic sturgeon, sturgeon are likely to migrating through the action area to reach their natal rivers to spawn. Elevated temperatures could modify cues for spawning migration, resulting in an earlier spawning season, and thus, altering the time of year sturgeon may or may not be present within the action area. This may cause an increase or decrease in the number of sturgeon present in the action area. However, because spawning is not triggered solely by water temperature, but also by day length (which would not be affected by climate change) and river flow (which could be affected by climate change), it is not possible to predict how any change in water temperature alone will affect the seasonal movements of sturgeon through the action area.

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

6.4.3 Sea Turtles

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

Over the time period considered in this Opinion, sea surface temperatures are expected to rise less than 1.5°C. Warming temperatures would likely result in a shift in the seasonal distribution of sea turtles in the action area, such that sea turtles may begin northward migrations from their southern overwintering grounds earlier in the spring and thus would be present in the action area earlier in the year. Likewise, if water temperatures were warmer in the fall, sea turtles could remain in the action area later in the year. Sea turtles are known to enter Virginia waters when sea surface temperatures are at or above 11°C, and current ranges of sea surface temperatures in Virginia waters range from 7°C to 28°C. As increases in sea surface temperatures are expected to be small over the next 50 years (i.e., approximately 1.5°C), it is unlikely that a shift in sea turtle distribution will be seen over the timeframe of the action. 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 sea turtles or a significant modification to the number of sea turtles likely to be present in the action area over the next 50 years.

It has also been speculated that the nesting range of some sea turtle species may shift northward with increasing temperature. Nesting in the mid-Atlantic generally is extremely rare and no nesting has been documented along the Wallops Island shoreline. As noted above, predicted increases in water temperatures between now and 2062 are not expected to be large enough to cause a significant shift in the distribution of sea turtles. As such, it is unlikely that there will be a significant shift in nesting trends in Virginia to suggest that an increase in nesting will occur along the shorelines of the Virginia Coast, let alone the action area. As there have been no documented reports of sea turtle nests in the action area, the shoreline of Wallops Island does not appear to be a nesting site for sea turtles. As such, it is unlikely that any climatic changes that may occur over the next 50 years will alter the habitat in any way that will cause sea turtles to begin nesting along the shoreline of Wallops Island. However, should that shift occur, the action will not affect the environment in any way that would prevent sea turtles from using the action area as a nesting ground, even in the face of sea level rise. As the SRIPP serves to replenish the beach along the Wallops Island shoreline, and will in fact extend the beach approximately 520 feet from the seawall to be constructed, the action will not contribute to the loss of any potential beach habitat, but instead, will serve to create beach habitat that could potentially be used by sea turtles to nest in the future. As the beaches along the Wallops Island shoreline will be maintained over 50 years, with continual renourishment cycles being undertaken approximately every 5 years, future renourishment activities along the Wallops Island shoreline will ensure that this beach habitat, which could be used by nesting sea turtles in the future, is maintained. Therefore, if, over the next 50 years any sea turtles begin to shift to more northern areas to nest, available nesting habitat would be present on the beaches of Wallops Island due to the creation and maintenance of this beach habitat. Additionally, the seawall will not contribute to the loss of any potential beach habitat, but instead, will serve as a reference point of where material is to be placed from shoreline to sea. However, as noted above, sea level rise has the potential to remove possible beach nesting habitat. Based on NOAA tide gauge data, sea level is expected to rise approximately 4.5 mm/yr in the action area; over a 50-year period, this equates to an approximately 0.74-foot increase in sea level along the shoreline of the action area. The small increase in sea level along the shorelines of the action area will not remove a significant area of the beach and thus, potential nesting areas would remain present over 50 years, even with the presence of the seawall.

Changes in water temperature may also alter the forage base and thus, foraging behavior of sea turtles. Changes in the foraging behavior of sea turtles in the action area and thus, could lead to either an increase or decrease in the number of sea turtles in the action area, depending on whether there was an increase or decrease in the forage base and/or a seasonal shift in water temperature. For example, if there was a decrease in sea grasses in the action area resulting from increased water temperatures or other climate change related factors, it is reasonable to expect that there may be a decrease in the number of foraging green sea turtles in the action area. Likewise, if the prey base for loggerhead, Kemp's ridley or leatherback sea turtles was affected, there may be changes in the abundance and distribution of these species in the action area. However, as noted above, because we do not know the adaptive capacity of these individuals or how much of a change in temperature would be necessary to cause a shift in distribution, it is not possible to predict changes to the foraging behavior of sea turtles over the next seven years. If sea turtle distribution shifted along with prey distribution, it is likely that there would be minimal, if any, impact on the availability of food. Similarly, if sea turtles shifted to areas where different forage was available and sea turtles were able to obtain sufficient nutrition from that new source of forage, any effect would be minimal. The greatest potential for effect to forage resources would be if sea turtles shifted to an area or time where insufficient forage was available; however, the likelihood of this happening seems low because sea turtles feed on a wide variety of species and in a wide variety of habitats.

7.0 EFFECTS OF THE ACTION

This section of an Opinion assesses the direct and indirect effects of the 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). This Opinion examines the likely effects (direct and indirect) of the action on Atlantic sturgeon, whales and sea turtles in the action area and their habitat within the context of the species current status, the environmental baseline and cumulative effects. As explained in the Description of the Action, the action under consideration in this Opinion includes the extension and construction of a seawall during Year One of the SRIPP; the initial dredging cycle needed to renourish the 3.7 mile stretch of shoreline/beach along the Goddard Space Flight Center's WFF, which will be conducted within the second and third year of the SRIPP; the subsequent nine renourishment cycles required to maintain beach nourishment, which are expected to occur every 5 years; the transport of material to and from the borrow areas throughout the 50-year life of the SRIPP; and, the placement of sand along the WFF shoreline and its impact on nearshore waters and benthos.

7.1 Effects of Seawall Construction and Extension

The construction and extension of the seawall will occur on the beach parallel to the shoreline in the approximate location of the geotextile tubes. The new seawall will be constructed landward of the existing shoreline and will be comprised of 5-7 tons of rock that will placed on the beach, with the top of the seawall approximately 14-feet above the normal high tide water level. As this portion of the project will occur on land where listed species under NMFS jurisdiction will not be present, no direct or indirect effects are expected to be incurred on Atlantic sturgeon, sea turtles or whales during this phase of the SRIPP.

7.2 Effects of Dredging Operations

As explained in the Description of the Action section above, over the 50-year life of the SRIPP, hopper dredges will be used for both initial and renourishment cycles of dredging. Below, the effects of hopper dredging 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 and foraging behavior due to dredging; (3) suspended sediment associated with dredging operations; (4) underwater noise generated during dredging operations; and (5) the potential for interactions between project vessels and individual whales, Atlantic sturgeon, or sea turtles.

As noted above, sea turtles are likely to occur in the action area from April-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-May; humpbacks from September-April; and fin whales from October-January; however, individual transient whales could be present in the action area outside of these time frame as this area 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.

7.2.1 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). These areas are believed to be where Atlantic sturgeon overwinter and/or forage (Laney *et. al* 2007; Erickson *et al*. 2011; Dunton *et al*. 2010). Areas between these sites serve as migration corridors to and from these areas, as well as to spawning grounds found within natal rivers.

The borrow sites where dredging will occur are at least 13 or more miles away from the nearest identified aggregation area (i.e., nearshore waters between Chesapeake Bay and Delaware Bay; southern Virginia and North Caolina); based on the location of known aggregation areas and information available on habitat at the borrow areas and distribution of Atlantic sturgeon, it is unlikely that the borrow sites are used for overwintering and/or foraging. While opportunistic foraging may occur at these sites, it is more likely that the borrow areas are used by migrating invidividuals as they move from foraging, overwintering, and spawning grounds.

Sea Turtles

Sea turtles occur in the action area from April through November each year with the largest numbers present from June through October of any year (Stetzar 2002). One of the main factors influencing sea turtle presence in northern waters is seasonal temperature patterns (Ruben and Morreale 1999). Temperature is correlated with the time of year, with the warmer waters in the late spring, summer, and early fall being the most suitable for cold-blooded sea turtles. Sea turtles are most likely to occur in the action area between April and November when water temperatures are above 11°C. Sea turtles have been documented in the action area by the CETAP aerial and boat surveys as well as by surveys conducted by NMFS Northeast Science Center and observers on commercial fishing vessels. Additionally, satellite tracked sea turtles have been documented in the action area (seaturtle.org tracking database). The majority of sea turtle observations have been of loggerhead sea turtles, although Kemp's ridley, green and leatherback sea turtles have also been documented in the area.

In addition to temperature, water depth also affects sea turtle distribution. Water depths in and around the borrow sites range from approximately 25-50 feet. Satellite tracking studies of sea turtles found that foraging turtles mainly occurred in areas where the water depth was between approximately 16 and 49 feet (Morreale and Standora 1990; Ruben and Morreale 1999). This depth was interpreted not to be as much an upper physiological depth limit for turtles, as a natural limiting depth where light and food are most suitable for foraging turtles (Morreale and Standora 1990). The areas to be dredged and the depths preferred by sea turtles do overlap, suggesting that if suitable foraging items were present, loggerheads and Kemp's ridleys may be foraging in the offshore shoals where dredging will occur. Green sea turtles feed almost exclusively on sea grasses, as there are no SAV beds in any of the borrow areas where dredging will occur, green sea turtles are not likely to use the areas to be dredged for foraging.[9]

Alteration of Foraging Habitat

Dredging can affect Atlantic sturgeon and sea turtles by reducing prey species through the alteration of the existing biotic assemblages. As noted above, the borrow areas are not believed to be an area where Atlantic sturgeon concentrate to forage. However, opportunistic foraging may occur at these sites as surveys of the borrow areas indicate the presence of potential Atlantic sturgeon foraging items (i.e., moon shell, whelks, polycheates, and amphipods). 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 offshore borrow sites are not known to be an area where sea turtles concentrate to forage; however, based on surveys conducted at the borrow sites, potential sea turtle foraging items appear to be present, including jellyfish, comb jellies, crabs (portly spider (*Libinia emarginata*) and Atlantic rock crabs (*Cancer irroratus*)), moon shell, and whelks. Of the listed species found in the action area, loggerhead and Kemp's ridley sea turtles are the

[9] According to the 2008 SAV online mapper prepared by the Virginia Institute of Marine Science (VIMS), the nearest mapped SAV bed to the SRIPP project area is in New Virginia Cove, approximately 11 km (7 miles) from the northern most point of the proposed beach fill on Wallops Island shoreline.

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 at the borrow areas, green sea turtles will not use the borrow sites 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 occur on the benthos of the borrow areas and depths within the borrow areas are suitable for use by these species of sea turtles, some loggerhead and Kemp's ridley sea turtle foraging likely occurs at these sites and therefore, may be affected by dredging activities within this portion of the action area.

While some offshore areas may be more desirable to certain turtles or sturgeon due to prey availability, there is no information to indicate that the borrow areas proposed for dredging have more abundant turtle or sturgeon prey or better foraging habitat than other surrounding areas. The assumption can be made that sea turtles and sturgeon are not likely to be more attracted to the borrow areas 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 in 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. 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. As such, recolonization of the borrow areas should be complete within 1 year after the initial dredge cycle (approximately the summer of 2013). It also should be noted that only a small percentage of the available sand at each borrow area (e.g., if Unnamed Shoal A is used for the initial dredge cycle and all renourishment cycles, SRIPP will remove approximately 33% of the total volume of available sand on Unnamed Shoal A (40 million cy) through 2050) is proposed to be removed and suitable foraging items should continue to be available at each borrow area at all times.

When the shoals targeted for dredging are considered within the context of the entire complex of shoals off the Delmarva coast, it can be concluded that they are not necessarily unique habitats. A recent study by Dibajnia and Nairn (*in press*) identified 181 shoals between Delaware and Chesapeake Bays that were between the 10 m (33 ft) and 40 m (130 ft) depth contours and greater than 2 kilometers (1.2 miles) in length, all of which fit the general characteristics of Unnamed Shoals A and B. Assuming that these shoals are rectangular in shape, their surface area is estimated to be in excess of 238,765 ha (590,000 acres). It should be noted, however, that this is only a first-order approximation; the referenced study only focuses on shoals deemed to be economically viable for dredging and excludes shoreface attached shoals, shorter shoals, and those in deeper waters. Accordingly, available shoal habitat is larger. However, even under this conservative evaluation, the SRIPP will affect only 0.2 percent of the shoals within Dibajnia and Nairn's study area. Additionally, in total, there is nearly 2,560,000 acres of seafloor offshore of Maryland and Virginia. Cumulatively, the reasonably foreseeable, future dredging projects offshore will affect less than 0.05% of the nearshore seafloor in the region (NASA 2010). NMFS anticipates that while the dredging activities may temporarily disrupt normal feeding behaviors for sea turtles and Atlantic sturgeon by causing them to move to alternate areas, the action is not likely to remove critical amounts of prey resources from the action area 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 sea turtles, Atlantic sturgeon, or whales from using the action area as a migratory pathway to other near-by areas that may be more suitable for foraging.

7.2.2 Entrainment

7.2.2.1 Sea Turtles

Because of 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. Therefore, this section of the Opinion will only consider the effects of entrainment on loggerhead, Kemp's ridley and green sea turtles.

The National Research Council's Committee on Sea Turtle Conservation (1990) estimated that dredging mortalities, along with boat strikes, were second only to fishery interactions as a source of probable mortality of sea turtles. Experience has shown that injuries sustained by sea turtles entrained in hopper dredge dragheads are usually fatal. Mortality in hopper dredging operations most often occurs when turtles are entrained in 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 as the dredge is being placed or removed from the bottom, creating suction in the draghead, or when the dredge is operating on an uneven or rocky substrate causing the draghead to rise off the bottom, it is likely that only those species feeding or resting on or near the bottom would be vulnerable to entrainment. 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. However, it is possible to operate the dredge in a manner that minimizes potential for such incidents as noted in the Monitoring Specifications for Hopper Dredges (Appendix B).

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 467 sea turtles have been entrained in hopper dredges since 1980 and in the Gulf Region over 186 sea turtles have been killed since 1995. Records of sea turtle entrainment in the USACE NAD begin in 1994. Since this time, at least 72 sea turtles deaths (see Table 4) related to hopper dredge activities have been recorded in waters north of the North Carolina/Virginia border (USACE Sea Turtle Database).[10]

Table 4. Sea Turtle Takes in USACE NAD Dredging Operations

Project Location	**Year of Operation**	**Cubic Yardage Removed**	**Observed Takes**
York Spit, VA	2011	NA	1 Loggerhead
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

[10] 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.

Cape Henry	1994	552,671	4 loggerheads 1 unknown
York Spit Channel	1994	61,299	4 loggerheads
Delaware Bay	1994	NA	1 Loggerhead
Cape May NJ	1993	NA	1 Loggerhead
Off Ocean City MD	1992	1,592,262	3 Loggerheads
			TOTAL = 73 Turtles

Official records of sea turtle mortality in dredging activities in the USACE NAD begin in the early 1990s. Before this time, 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 Chesapeake Bay. 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. To date, no hopper dredge operations (other than the Currituck) have occurred in the New England District in areas or at times when sea turtles are likely to be present.

Most of the available information on the effects of hopper dredging on sea turtles in the USACE NAD has come from operations in Virginia waters, particularly in the entrance channels to the Chesapeake Bay. Since 1994, 63 sea turtles mortalities have been observed on hopper dredges operating in Virginia waters. In Thimble Shoals Channel, maintenance dredging took several turtles during the warmer months of 1996 (1 loggerhead) and 2000 (2 loggerheads, 1 unknown). A total of 6 turtles (5 loggerhead, 1 unknown) were taken in association with dredging in Thimble Shoal Channel during 2001, and one turtle was taken in May 2002 (1 loggerhead). Nine sea turtle takes were reported during dredging conducted in September and October 2003 (7 loggerhead, 1 Kemp's ridley, 1 unknown) and one sea turtle take (1 loggerhead) was reported in the summer of 2006. Most recently, Thimble Shoals Channel was dredged in the spring of 2009, with 3 loggerheads killed during this operation.

Incidental takes have occurred in the Cape Henry and York Spit Channels as well. In May and June 1994, parts of at least five sea turtles were observed (at least 4 loggerheads and 1 unknown) during dredging at Cape Henry. In September and October 2001, 3 turtle takes were observed (1 Kemp's ridley and 2 loggerheads). Eight turtle takes were observed during dredging at Cape Henry in April, May, June and October 2002 (1 green, 1 Kemp's and 6 loggerhead). Three loggerheads were killed during the dredging of the Cape Henry Channel in the summer of 2006. At York Spit, four loggerheads were taken in dredging operations occurring during one week in June 1994. Nine turtles were taken in dredging operations at York Spit in 2002 (8 loggerheads, 1 Kemp's ridley). York Spit was last dredged in the summer of 2007, with the take of 1 Kemp's ridley reported. In 1998, dredging in the York River Entrance Channel took 5 loggerheads. No turtles had been observed in dredging operations in Rappahannock Shoal Channels or the Sandbridge Shoals borrow area.

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.

Sea turtles have been found resting in deeper waters, which could increase the likelihood of interactions from 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. While sea turtle brumation has not been documented in mid-Atlantic or New England waters, it is possible that this phenomenon occurs in these waters.

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 crushed strandings and dredging activities, and if those strandings need to be factored into an incidental take level. More research also needs to be conducted to determine if sea turtles are in fact undergoing brumation in mid-Atlantic or New England waters. 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 in the examples of sea turtle takes above. 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.

Few interactions with listed sea turtles have been recorded during dredging at offshore borrow areas. This is likely due to the transitory nature of most sea turtles occurring in offshore borrow areas as well as the widely distributed nature of sea turtles in offshore waters. This lack of information is also largely due to the infrequency of dredging in offshore borrow areas in the USACE NAD, which makes it even more difficult to predict the likely number of interactions between this action and listed sea turtles. However, as sea turtles have been documented in the action area and suitable habitat and forage items are present, it is likely that sea turtles will be present in the action area when dredging takes place. As sea turtles are likely to be less concentrated in the action area than they are while foraging in Virginia waters such as the entrance channels to the Chesapeake Bay, the level of interactions during this project are likely to be fewer than those recorded during dredging in the Chesapeake Bay area (i.e., the Thimble Shoals and Cape Henry projects noted above).

In the USACE Sea Turtle Database, records for 38 projects occurring during “sea turtle season” (i.e., April 1 – November 30) are available that report the cubic yardage removed during a project (see Table 5). As noted above, the most complete information is available for the Norfolk district. Records for 22 projects occurring in the April – November time frame that report cubic yards removed are available for channels in the Chesapeake Bay (see Table 6). NMFS has made calculations from that data which indicate that, in the Chesapeake Bay, an average of 1 sea turtle is killed for approximately every 387,000 cy removed. This calculation has been based on a number of assumptions including the following: that sea turtles are evenly distributed throughout all channels and borrow areas for which takes have occurred, that all dredges will take an identical number of sea turtles, and that sea turtles are equally likely to be encountered throughout the April to November time frame.

Table 5. Dredging projects in USACE NAD with recorded cubic yardage

Project Location	Year of Operation	Cubic Yards Removed	Observed Takes
York Spit/Thimble Shoals, VA	2011	1,630,713	0
Cape Henry, VA	2011	2,472,000	0
York Spit Channel, VA	2009	372,533	0
Dewey and Bethany Beach, DE	2009	397,956	0
York Spit, VA	2007	608,000	1 Kemp’s Ridley
Atlantic Ocean Channel, VA	2006	1,118,749	0
Thimble Shoal Channel	2006	300,000	1 loggerhead

Dewey Beach/Cape Henlopen (DE Bay)	2005	1,134,329	0
Delaware Bay	2005	50,000	2 Loggerheads
Cape May Point, NJ	2005	2,425,268	0
Thimble Shoal Channel, VA	2004	139,200	0
VA Beach Hurricane Protection Project	2004	844,968	0
Thimble Shoal Channel, VA	2003	1,828,312	7 Loggerheads 1 Kemp’s ridley 1 unknown
York River Entrance Channel, VA	2003	343,092	0
Off Ocean City MD	2002	744,827	0
Cape Henry, VA	2002	1,407,814	6 Loggerheads 1 Kemp’s ridley 1 green
York Spit Channel, VA	2002	911,406	8 Loggerheads 1 Kemp’s ridley
Chincoteague Inlet, VA	2002	84,479	0
Cape Henry, VA	2001	1,641,140	2 loggerheads 1 Kemp’s ridley
Thimble Shoal Channel, VA	2000	831,761	2 loggerheads 1 unknown
Cape Henry , VA	2000	759,986	0
York River Entrance Channel, VA	1998	672,536	6 loggerheads
Off Ocean City MD	1998	1,289,817	0
York Spit Channel, VA	1998	296,140	0
Cape Henry, VA	1998	740,674	0
Atlantic Coast of NJ	1997	1,000,000	1 Loggerhead
Thimble Shoal Channel, VA	1996	529,301	1 loggerhead
Delaware Bay	1995	218,151	1 Loggerhead
Cape Henry Channel, VA	1995	485,885	0
Bethany Beach (DE Bay)	1994	184,451	0
York Spit Channel, VA	1994	61,299	4 loggerheads
Cape Henry , VA	1994	552,671	4 loggerheads 1 unknown
Dewey Beach (DE Bay)	1994	624,869	0
Off Ocean City MD	1994	1,245,125	0
Off Ocean City MD	1992	1,592,262	3 Loggerheads
Off Ocean City MD	1991	1,622,776	0
Off Ocean City MD	1990	2,198,987	0
	TOTAL	***33,361,477 cy***	***57 Turtles***

Table 6. Projects in USACE NAD with recorded cubic yardage – Chesapeake Bay Only

Project Location	Year of Operation	Cubic Yards Removed	Observed Takes
York Spit/Thimble Shoals	2011	1,630,713	0
Cape Henry	2011	2,472,000	0
York Spit Channel	2009	372,533	0
York Spit	2007	608,000	1 Kemp's Ridley
Atlantic Ocean Channel	2006	1,118,749	0
Thimble Shoal Channel	2006	300,000	1 loggerhead
Thimble Shoal Channel	2004	139,200	0
Thimble Shoal Channel	2003	1,828,312	7 Loggerheads 1 Kemp's ridley 1 unknown
York River Entrance Channel	2003	343,092	0
Cape Henry	2002	1,407,814	6 Loggerheads 1 Kemp's ridley 1 green
York Spit Channel	2002	911,406	8 Loggerheads 1 Kemp's ridley
Cape Henry	2001	1,641,140	2 loggerheads 1 Kemp's ridley
Cape Henry	2001	1,641,140	0
Thimble Shoal Channel	2000	831,761	2 loggerheads 1 unknown
Cape Henry	2000	759,986	0
York River Entrance Channel	1998	672,536	6 loggerheads
York Spit Channel	1998	296,140	0
Cape Henry	1998	740, 674	0
Thimble Shoal Channel	1996	529,301	1 loggerhead
Cape Henry Channel	1995	485,885	0
York Spit Channel	1994	61,299	4 loggerheads
Cape Henry	1994	552,671	4 loggerheads 1 unknown
	TOTAL	***19,344,352 cy***	***50 turtles***

As noted above, sea turtles are likely to be less concentrated in the action area for this consultation than they are in the Chesapeake Bay area. Based on this information, NMFS believes that hopper dredges operating in the offshore borrow areas are less likely to interact with sea turtles than hopper dredges operating in the Chesapeake Bay area. Based on habitat characteristics and geographic area, the level of interactions during this project may be more comparable to the level of interactions recorded for dredging projects in Delaware Bay or offshore of New York and New Jersey (i.e., Cape May, Sea Girt, lower Delaware Bay).

Records for 17 projects occurring during “sea turtle season” (i.e., April 1 – November 30) in the Baltimore, Philadelphia and New York District (all offshore) are available that report the cubic yardage removed during a project; however an important caveat is that observer coverage at these projects has ranged from 0 to 50% (see Table 7).

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 7 as "adjusted entrainment number."

Table 7. Projects in USACE NAD with recorded cubic yardage (with Chesapeake Bay projects removed)

Project Location	**Year of Operation**	**Cubic Yards Removed**	**Observed Entrainment**	**Adjusted Entrainment Number**
Dewey and Bethany Beach (DE)	2009	397,956	0	0
Dewey Beach/Cape Henlopen (DE Bay)	2005	1,134,329	0	0
Delaware Bay	2005	50,000	2 Loggerhead	2 Loggerhead
Cape May Point, NJ	2005	2,425,268	0	0
VA Beach Hurricane Protection Program	2004	844,968	0	0
Off Ocean City MD	2002	744,827	0	0
Chincoteague Inlet	2002	84,479	0	0
Offshore New Jersey	1997	1,000,000	1 Loggerhead	1 Loggerhead
Off Ocean City MD	1998	1,289,817	0	0
Delaware Bay	1995	218,151	1 Loggerhead	1 Loggerhead
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
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	***15,658,265***	***7 Loggerheads***	***10 Loggerheads***

As information available (number of days dredged, cubic yards removed) on projects outside of the Chesapeake Bay is incomplete and observer coverage has been relatively low, it is difficult to estimate the number of sea turtles likely to be taken in these areas. It is reasonable, based on the available information, to calculate the number of sea turtles entrained during projects where cubic yardage is available, not just for projects where entrainment has occurred (which would overestimate the likelihood of interactions). Using this method, and based on the adjusted entrainment number in Table 7, an estimate of 1 sea turtle per 1.6 million cubic yards is calculated. As noted above, it is likely that including the Chesapeake Bay data would overestimate the number of interactions in offshore borrow areas likely due to the concentration of sea turtles in the Chesapeake Bay and differences in habitat between the Chesapeake Bay entrance channels and the offshore channels or borrow areas considered above. Based on this approach, we estimate that dredging in offshore borrow areas outside of the Chesapeake Bay is likely to result in the entrainment of 1 sea turtle for every 1.6 million cubic yards of material removed by a hopper dredge. This calculation is based on a number of assumptions including the following: that sea turtles are evenly distributed throughout all borrow areas, that all 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 73 sea turtles, 63 have been loggerhead, 5 have been Kemp's ridleys, 1 green and 4 unknown. Overall, of those identified to species, approximately 90% of the sea turtles taken in dredges operating in the USACE North Atlantic Division have been loggerheads. 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, we expect that 1 sea turtle is likely to be injured or killed for approximately every 1.6 million cy of material removed from the proposed borrow area and that at least 90% will be loggerheads. We anticipate that no more than 3 sea turtles are likely to be entrained in the initial dredge cycle when 4,373,450 cy of material is removed. Maintenance dredging operations are expected to remove up to 1,007,500 cy of sand every 5 years. Over the 50 year life of the SRIPP nine maintenance cycles will occur removing approximately 9,067,500 cubic yards of material from the shoals resulting in the entrainment of 6 or fewer sea turtles

Due to the nature of the injuries expected to result from entrainment, all of the turtles are expected to die.

NMFS expects that nearly all of the sea turtles will be loggerheads and that the entrainment of a Kemp’s ridley during a particular 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 90% of the sea turtles taken in dredges operating in the USACE North Atlantic Division have been loggerheads. Based on that ratio, NMFS anticipates that over the life of the project, for every 10 sea turtle interactions only 1 of them is likely to be with a Kemp’s ridley. Over the life of the project (i.e., through 2062), NMFS anticipates that up to 9 sea turtles could be killed, with no more than 1 being a Kemp’s ridley and the remainder being loggerheads.

As explained in the Status of the Species section, it is likely that the sea turtles entrained in hopper dredges operating in the waters off Virginia originate from several of the recovery units, primarily from the PFRU, NRU, and GCRU, with smaller amounts possible from the DTRU and NGMRU. Based on the best available information on sea turtles in the action area, NMFS anticipates that a loggerhead entrained at the Wallops Island borrow site is likely to be either a benthic immature or sexually mature turtle. There is no information to suggest that either sex is disproportionately taken in hopper dredges. Therefore, either a male or female loggerhead may be entrained in the dredge.

7.2.2.2 Atlantic Sturgeon

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, adults are unlikely to be vulnerable to entrainment. USACE reports that from 1990-2011, 30 interactions with sturgeon occurred during dredge operations. Of these, 17 were reported as Atlantic sturgeon, with 15 of these entrained in hopper dredges. Of the 7 Atlantic sturgeon for which size is available, all were juveniles. Information on these interactions is presented in Table 8. Most of these interactions occurred within rivers and harbors; however, to date, few records exist for interactions between hopper dredges and Atlantic sturgeon along coastal/offshore borrow sites (Table 9).

Table 8. USACE Atlantic Sturgeon Entrainment Records from Hopper Dredge Operations 1990-2011

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	2
Brunswick Harbor (GA)	SAD/SAS	Feb-12	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
York Spit (VA)	NAD/NAN	Apr-11	700,000	2
		Total	**17,553,816**	**15**

* 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 9: Atlantic Sturgeon Entrainment: Coastal/Offshore Projects in USACE NAD Since 1998 with Recorded Cubic Yardage

*a: 14 Atlantic sturgeon removed during pre-dredge trawl/relocation trawling (September and November, 2003).

*b: 1 Atlantic sturgeon removed during pre-dredge trawl/relocation trawling on 10/26/02.

*c: 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
York Spit Channel, VA	2011	1,630,713	2	2
Cape Henry, VA	2011	2,472,000	0	0
York Spit Channel, VA	2009	372,533	0	0
Dewey and Bethany Beach, DE	2009	397,956	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 Channel	2006	300,000	0	0
Dewey Beach/Cape Henlopen	2005	1,134,329	0	0
Cape May Point, NJ	2005	2,425,268	0	0
Thimble Shoal Channel, VA	2004	139,200	0	0
VA Beach Hurricane Protection Project	2004	844,968	0	0
Thimble Shoal Channel, VA (*a)	2003	1,828,312	0	0
Off Ocean City MD	2002	744,827	0	0
Cape Henry, VA (*b)	2002	1,407,814	0	0
York Spit Channel, VA (*c)	2002	911,406	0	0
Cape Henry, VA	2001	1,641,140	0	0
Thimble Shoal Channel, VA	2000	831,761	0	0
Cape Henry , VA	2000	759,986	0	0
Off Ocean City MD	1998	1,289,817	0	0
York Spit Channel, VA	1998	296,140	0	0
Cape Henry, VA	1998	740,674	0	0
Atlantic Coast of NJ	1997	1,000,000	0	0
Thimble Shoal Channel, VA	1996	529,301	0	0
Cape Henry Channel,	1995	485,885	0	0

VA				
Bethany Beach,DE	1994	184,451	0	0
York Spit Channel, VA	1994	61,299	0	0
Cape Henry , VA	1994	552,671	0	0
Dewey Beach, DE	1994	624,869	0	0
Off Ocean City MD	1994	1,245,125	0	0
Off Ocean City MD	1992	1,592,262	0	0
	TOTAL	28,194,956	3	3

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

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 1992. 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 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. Dredging in the offshore environment, such as where this project will occur, is very rare in the winter months.

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 dredging will occur in an open ocean environment, 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 borrow sites as a migration corridor and are not aggregated in this area, the density of Atlantic sturgeon in this area is likely to be very low. 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 borrow areas, 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 dredge operating under the SRIPP 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 3 Atlantic sturgeon have been observed entrained in a hopper dredge (see Table 9). The low level of interactions may be, in part, due to the use of pre-trawl/dredge relocation trawling (see Table 9; just because 0 Atlantic sturgeon were entrained in some locations, Atlantic sturgeon were still documented prior to dredging operations) or the infrequency of dredging offshore borrow/coastal areas in the USACE NAD. It is also possible that interactions with Atlantic sturgeon have occurred and not been 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.

Based on what we know about Atlantic sturgeon behavior in coastal/offshore areas such as the borrow areas, it is reasonable to consider that the risk of entrainment at these borrow areas is similar to that at other non-riverine/harbor areas. Some of the areas considered in this analysis (see Table 9) are closer to shore than the borrow areas and may be more heavily used than the borrow 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 in the borrow areas.

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 non-riverine/harbor projects 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. Using this method, and using the dataset presented in Table 9, we have calculated an entrainment rate of 1 Atlantic sturgeon is likely to be injured or killed for approximately every 9.4 million cy of material removed from the proposed borrow area. This calculation is based on a number of assumptions including the following: that Atlantic sturgeon are evenly distributed throughout the action area, that all 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 no more than 1 Atlantic sturgeon is likely to be entrained in the initial dredge cycle when 4,373,450 cy of material is removed over a period of approximately 3 months (i.e., during spring or early summer 2012).

Maintenance dredging operations may occur at any time of year and are expected to remove up to 1,007,500 cy of sand every 5 years. Over the 50 year life of the SRIPP nine maintenance cycles will occur removing approximately 9,067,500 cy of material from the shoals, mostly in Shoal A, resulting in the entrainment of no more than 1 Atlantic sturgeon. The USACE has indicated that over the life of the project, a total of approximately 13,440,950 cy of material will be removed from the borrow area. Due to the nature of the injuries expected to result from entrainment, all of the sturgeon are expected to die. As such, over the life of the project (i.e., through 2062), NMFS anticipates that up to 2 Atlantic sturgeon could be killed. 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 both of these sturgeon will be subadults.

7.2.3 Interactions with the Sediment Plume

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).

No information is available on the effects of total suspended solids (TSS) on juvenile and adult sea turtles or whales; however, studies of the effects of turbid waters on fish suggest that concentrations of suspended solids can reach thousands of milligrams per liter before an acute toxic reaction is expected (Burton 1993). TSS is most likely to affect sea turtles or whales if a plume causes a barrier to normal behaviors or if sediment settles on the bottom affecting sea turtle prey. As Atlantic sturgeon, sea turtles and whales are highly mobile they are likely to be able to avoid any sediment plume and any effect on Atlantic sturgeon, sea turtle or whale movements is likely to be insignificant. Additionally, the TSS levels expected 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)).

While the increase in suspended sediments may cause Atlantic sturgeon, sea turtles or whales to alter their normal movements, any change in behavior is likely to be insignificant as it will only involve movements to alter their course out of the sediment plume. Based on this information, any increase in suspended sediment is not likely to affect the movement of Atlantic sturgeon, sea turtles or whales between foraging areas or while migrating or otherwise negatively affect listed species in the action area. Based on this information, it is likely that the effect of the suspension of sediment resulting from dredging operations will be insignificant.

7.2.4 Collisions with Dredges

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 action 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 operations. It is more likely that contact injuries during actual dredging would involve the propeller of the vessel and 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, the dredge in transit would be moving at faster speeds (i.e., 10 knots) than during dredging operations (i.e., 3 knots), particularly when empty and returning to the borrow area. The speed of the dredge while empty is not expected to exceed 10 knots.

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 April through November; Atlantic sturgeon primarily during fall, winter, and spring months (approximately October-March); right whales primarily from November-May; humpbacks from September-April; and fin whales from October-January; however, individual transient right whales could be present in the action area outside of these time frame as this area has been used by whales migrating between calving/mating grounds and foraging grounds.

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 STSSN stranding data from 2001-2008, at least 520 sea turtles (loggerhead, green, Kemp's ridley and leatherbacks) that stranded on beaches within the NMFS Northeast Region (Maine through Virginia) showed evidence of propeller wounds and were, therefore, probable vessel strikes. In the vast majority of cases, it is unknown whether these injuries occurred pre- or post-mortem; however, in 18 cases there was evidence that the turtle was alive at the time of the strike.

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 3 knots while dredging or transiting to the pump site with a full load and is expected to operate at a maximum speed of 10 knots while empty. As such, the 10 knot or less speed of the dredge vessel is likely to reduce the chances of collision with a sea turtle. 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 SRIPP, the greatest risk of vessel collision will occur during transit between shore and the offshore Wallops Island borrow sites to be dredged. Sea turtles present in these shallow nearshore waters are most likely to be foraging along the bottom, thereby reducing the likelihood of interaction with a vessel as they will be found primarily on the bottom and away from the surface of the water column near the hull of the vessel. 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 to a discountable level the potential for interaction with vessels.

Effects of Vessel Collisions on Atlantic Sturgeon

Although there have been no documented reports of dredge vessels colliding with Atlantic sturgeon, vessel strikes have been identified as a threat to Atlantic sturgeon and this species is known to be vulnerable to interactions with vessels. While the exact number of Atlantic sturgeon killed as a result of being struck by boat hulls or propellers is unknown, it is an area of concern in the Delaware and James Rivers. Brown and Murphy (2010) examined twenty-eight dead Atlantic sturgeon observed in the Delaware River from 2005-2008. Fifty-percent of the mortalities resulted from apparent vessel strikes and 71% of these (10 of 14) had injuries consistent with being struck by a large vessel (Brown and Murphy 2010). Eight of the fourteen vessel struck sturgeon were adult-sized fish (Brown and Murphy 2010). Given the time of year in which the fish were observed (predominantly May through July; Brown and Murphy 2010), it is likely that many of the adults were migrating through the river to the spawning grounds.

Similarly, five sturgeon were reported to have been struck by commercial vessels within the James River, VA in 2005, and one strike per five years is reported for the Cape Fear River. Locations that support large ports and have relatively narrow waterways seem to be more prone to ship strikes (e.g., Delaware and James Rivers) (ASSRT 2007).

The factors relevant to determining the risk to Atlantic sturgeon from vessel strikes are currently unknown, but they may be related to size and speed of the vessels, navigational clearance (i.e., depth of water and draft of the vessel) in the area where the vessel is operating, and the behavior of Atlantic sturgeon in the area (e.g., foraging, migrating, etc.). It is important to note that vessel strikes have only been identified as a significant concern in the Delaware and James rivers and current thinking suggests that there may be unique geographic features in these areas (e.g., potentially narrow migration corridors combined with shallow/narrow river channels) that increase the risk of interactions between vessels and Atlantic sturgeon. These geographic features are not present in the waters of the action area and thus, vessel strike is not considered to be a significant threat in the open waters of the ocean. Additionally, in contrast to the Delaware and James rivers where several vessel-struck individuals are identified each year, very few Atlantic sturgeon with injuries consistent with vessel strike have been observed in the ocean environment. Although the likelihood of a vessel collision with Atlantic sturgeon in the ocean environment is expected to be low, we cannot discount the possibility of such an interaction and as such, will discuss below the risk of such an interaction.

As described above, although Atlantic sturgeon may be found foraging in the action area, Atlantic sturgeon are likely to be primarily using the action area as a migration corridor to and from spawning, overwintering, and/or foraging sites along the U.S. eastern coastline. Based on available information, it is believed that when migrating, Atlantic sturgeon are found primarily at mid-water depths (Cameron 2010) and while foraging, within the bottom meter of the water column. As depths within the portion of the action area that dredges will be operating (i.e., borrow sites to pump-out buoy) will be between 25 to more than 40 feet, there should be sufficient clearance between the underkeel of the dredge and the bottom that Atlantic sturgeon should be able to continue essential behaviors (e.g., migration, foraging) without an interaction with a dredge to occur. However, Atlantic sturgeon are not restricted to these depths, and on occasion, have been known to occur in the upper water column. Similar to sea turtles, it may be assumed that Atlantic 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 3 knots while dredging or transiting to the pump site with a full load and is expected to operate at a maximum speed of 10 knots while empty. As such, the 10 knot or less speed of the dredge vessel is likely to reduce the chances of collision with an Atlantic sturgeon. In addition, as noted above, locations that support large ports and have relatively narrow waterways seem to be more prone to ship strikes. Neither of these characteristics applies to the action area, which is located in waters offshore of Virginia, and as such, further reduces the likelihood of an interaction/strike of a dredge vessel with an Atlantic sturgeon. Based on this and the best available information, the potential interaction of a dredge/vessel and an Atlantic sturgeon is likely to be discountable.

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, under the SRIPP, the speed of the dredge is not expected to exceed 3 knots while dredging or while transiting to the pump out site with a full load, and it is expected to operate at a maximum speed of 10 knots while empty. In addition, all vessels operators and observers 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.

Synthesis of the Effects of Vessel Collisions on Listed Species

Although the threat of vessel collision exists anywhere listed species and vessel activity overlap, ship strike is more likely to occur in areas where high vessel traffic coincides with high species density. In addition, ship strikes are more likely to occur and more likely to result in serious injury or mortality when vessels are traveling at speeds greater than ten knots. As noted above, with dredge vessels moving at speeds of 10 knots or less, dredge vessels in the action area are not likely to pose a vessel strike risk to listed species of whales, Atlantic sturgeon, and sea turtles. In addition, the onboard observer will be able to watch for whales and sea turtles while the vessel is in transit and provide information to both dredges operating in the action area about the location of sea turtles and whales nearby, thereby allowing vessels to reduce their speeds further and/or alter their course accordingly. Additionally, based on the draft of the vessel and the depths of the action area, sufficient clearance is expected to allow for the movement of Atlantic sturgeon beneath the vessels without the risk of an interaction with the vessel. Based on the best available information on sea turtle, Atlantic sturgeon, and whale interactions with vessels, and the fact that vessel strike avoidance measures will be in place, NMFS concludes that the likelihood of dredge related vessel traffic resulting in the collision with a whale, Atlantic sturgeon, or sea turtle is discountable.

7.2.5 Dredge Noise and Effects of Exposure to Increased Underwater Noise Levels

The level of a sound in water can be expressed in several different ways, but always in terms of dB relative to 1 micro-Pascal (μPa). Decibels, a log scale, is used to "compress" very large differences of sound level (e.g., from a whisper to cracking of thunder) into more manageable numbers. Each 10 dB increase is a ten-fold increase in sound pressure. Accordingly, a 10 dB increase is a 10x increase in sound pressure, and a 20 dB increase is a 100x increase in sound pressure.

Several measures of sound will be considered here:

- Peak sound pressure level (SPL) is the maximum sound pressure level in a signal measured in dB re 1 μPa (micropascal).

- Sound exposure level (SEL) is the integration over time of the squared instantaneous sound pressure normalized to a 1-sec period. This measure is an indication of the total acoustic energy received by an organism from a particular source. Measured in dB re 1μPa2-s.

- Root mean square (RMS) pressure level is the square root of the time average of the squared pressures. Measured in dB re 1 μPa.

Sound levels are analyzed in several different ways. The most common approach is "root mean square" (rms); however, one may measure "Peak" sound level, which is the highest level of sound within a signal. Peak is most often used to give an indication of the maximum level of a sound, but it does not give a good picture of the overall sound pressure in a signal. SEL is the integration over time of the square of the acoustic pressure in the signal and is thus an indication of the total acoustic energy received by an organism from a particular source.

7.2.5.1 Summary of available information on hearing ability of listed species

Right, Humpback, and Fin Whale Hearing

In order for right, humpback, and fin whales to be adversely affected by dredge noise, they must be able to perceive the noises produced by the activities. If a species cannot hear a sound, or hears it poorly, then the sound is unlikely to have a significant effect (Ketten 1998). Baleen whale hearing has not been studied directly, and there are no specific data on sensitivity, frequency or intensity discrimination, or localization (Richardson *et al*. 1995) for these whales. Thus, predictions about probable impact on baleen whales are based on assumptions about their hearing rather than actual studies of their hearing (Richardson *et al*. 1995; Ketten 1998). Ketten (1998) summarized that the vocalizations of most animals are tightly linked to their peak hearing sensitivity. Hence, it is generally assumed that baleen whales hear in the same range as their typical vocalizations, even though there are no direct data from hearing tests on any baleen whale. Most baleen whale sounds are concentrated at frequencies less than 1 kHz (Richardson *et al*. 1995), although humpback whales can produce songs up to 8 kHz (Payne and Payne 1985). Based on indirect evidence, at least some baleen whales are quite sensitive to frequencies below 1 kHz but can hear sounds up to a considerably higher but unknown frequency. Most of the man made sounds that elicited reactions by baleen whales were at frequencies below 1 kHz (Richardson *et al*. 1995). Some or all baleen whales may hear infrasounds, sounds at frequencies well below those detectable by humans. Functional models indicate that the functional hearing of baleen whales extends to 20 Hz, with an upper range of 30 Hz. Even if the range of sensitive hearing does not extend below 20-50 Hz, whales may hear strong infrasounds at considerably lower frequencies. Based on work with other marine mammals, if hearing sensitivity is good at 50 Hz, strong infrasounds at 5 Hz might be detected (Richardson *et al*. 1995). Fin whales are predicted to hear at frequencies as low as 10-15 Hz. The right whale uses tonal signals in the frequency range from roughly 20 to 1000 Hz, with broadband source levels ranging from 137 to 162 dB (RMS) re 1 μPa at 1 m (Parks & Tyack 2005). One of the more common sounds made by right whales is the “up call,” a frequency-modulated upsweep in the 50–200 Hz range (Mellinger 2004). The following table summarizes the range of sounds produced by right, humpback, and fin whales (from Au *et al*. 2000):

Table 10. Summary of known right, humpback, and fin whale vocalizations

Species	**Signal type**	**Frequency Limits (Hz)**	**Dominant Frequencies (Hz)**	**Source Level (dB re 1μPa RMS)**	**References**
Northern right	Moans	< 400	--	--	Watkins and Schevill (1972)
	Tonal	20-1000	100-2500	137-162	Parks and Tyack (2005)
	Gunshots		50-2000	174-192	Parks et al. (2005)
Humpback	Grunts	25-1900	25-1900	--	Thompson, Cummings, and Ha (1986)
	Pulses	25-89	25-80	176	Thompson, Cummings, and Ha (1986)
	Songs	30-8000	120-4000	144-174	Payne and Payne (1985)
Fin	FM moans	14-118	20	160-186	Watkins (1981), Edds (1988), Cummings and Thompson (1994)
	Tonal	34-150	34-150		Edds (1988)
	Songs	17-25	17-25	186	Watkins (1981)

Most species also have the ability to hear beyond their region of best sensitivity. This broader range of hearing probably is related to their need to detect other important environmental phenomena, such as the locations of predators or prey. Considerable variation exists among marine mammals in hearing sensitivity and absolute hearing range (Richardson *et al*. 1995; Ketten 1998); however, from what is known of right, humpback, and fin whale hearing, hearing ranges of these species are likely to have peak sensitivities in low frequency ranges.

Sea Turtle Hearing

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

Atlantic Sturgeon Hearing

There is no data both in terms of hearing sensitivity and structure of the auditory system for Atlantic sturgeon; however, there are a few studies or published data available on hearing in other sturgeon species, such as the closely related lake sturgeon (Lovell *et al*. 2005; Meyer *et al*. 2010). Initial studies by Meyer and Popper (2002) measuring responses of the ear using physiological methods suggest that a species of *Acipenser* may be able to detect sounds from below 100 Hz to possibly higher than 1,000 Hz. Lovell *et al*. (2005) suggests that lake sturgeon can hear sounds from below 100 Hz to about 500 Hz, whereas Meyer *et al*. (2010) reported evidence to suggest that the same species may hear up to 800 Hz. Since both studies examined responses of the ear and did not examine whether fish would behaviorally respond to sounds detected by the ear, it is hard to determine thresholds for hearing (that is, the lowest sound levels that an animal can hear at a particular frequency).

In addition, due to the lack of an acoustic coupling between the swim bladder and inner ear (characteristic of hearing specialist), sturgeon are considered hearing “generalists,” meaning that they are unlikely to detect sound at frequencies above 1 to 1.5 kilohertz (kHz), and compared to “hearing specialists,” they have a higher sound detection threshold (i.e., require higher intensity before detection) for the same frequencies of sound (Popper and Schilt 2008). Additionally, as hearing generalists, sturgeon rely primarily on particle motion to detect sounds (Lovell *et al*. 2005), which does not propagate as far from the sound source as does pressure. Based on this and the best available information, hearing thresholds for Atlantic sturgeon are expected to range from 100 Hz to 1000 Hz (Meyer and Popper 2002; Popper 2005; Lovell *et al*. 2005).

7.2.5.2 Criteria for Assessing Potential for Physiological and Behavior Effects

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

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

1. Behavioral reactions – Range from brief startle responses, to changes or interruptions in feeding, diving, or respiratory patterns, to cessation of vocalizations, to temporary or permanent displacement from habitat.

2. Masking – Reduction in ability to detect communication or other relevant sound signals due to elevated levels of background noise.

3. Temporary threshold shift (TTS) – Temporary, fully recoverable reduction in hearing sensitivity caused by exposure to sound. TTS may occur within specified frequency range or across all frequency ranges.

4. Permanent threshold shift (PTS) – Permanent, irreversible reduction in hearing sensitivity due to damage or injury to ear structures caused by prolonged exposure to sound or temporary exposure to very intense sound. PTS may occur within a specified frequency range or across all frequency ranges.

5. Non-auditory physiological effects – Effects of sound exposure on tissues in non-auditory systems either through direct exposure or as a consequence of changes in behavior (e.g., resonance of respiratory cavities or growth of gas bubbles in body fluids).

Under the SRIPP, dredging will produce sound that may affect listed species of sea turtles, whales and Atlantic sturgeon. The criteria described below will be used to assess the physiological and behavior effects of dredge noise on listed species of whales, sea turtles, and Atlantic sturgeon.

Whales

NMFS is in the process of developing a comprehensive acoustic policy that will provide guidance on assessing the impacts of anthropogenically produced sound on marine mammals. In the interim, NMFS' current thresholds for determining impacts to marine mammals typically center around root-mean-square (RMS) received levels of 180 dB re 1µPa for potential injury, 160 dB re 1µPa for behavioral disturbance/harassment from an impulsive noise source (e.g., seismic survey), and 120 dB re 1µPa for behavioral disturbance/harassment from a continuous noise source (e.g., dredging). These thresholds are based on a limited number of experimental studies on captive odontocetes and pinnipeds, a limited number of controlled field studies on wild marine mammals, observations of marine mammal behavior in the wild, and inferences from studies of hearing in terrestrial mammals. In addition, marine mammal responses to sound can be highly variable, depending on the individual hearing sensitivity of the animal, the behavioral or motivational state at the time of exposure, past exposure to the noise which may have caused habituation or sensitization, demographic factors, habitat characteristics, environmental factors that affect sound transmission, and non-acoustic characteristics of the sound source, such as whether it is stationary or moving (NRC 2003). Nonetheless, the threshold levels referred to above are considered conservative and are based on the best available scientific information and will be used as guidance in the analysis of effects on listed species of whales for this Opinion.

Sea Turtles

Currently there are no established thresholds for injury or behavioral disturbance/harassment for sea turtles. As noted above, the hearing capabilities of sea turtles are poorly known and there is little available information on the effects of noise on sea turtles; however, McCauley *et al.* (2000) noted that decibel levels of 166 dB re 1 µPa RMS were required before any behavioral reaction (e.g., increased swimming speed) was observed, and decibel levels above 175 dB re 1 µPa RMS elicited avoidance behavior of sea turtles. Based on this and the best available information, NMFS believes any underwater noise levels at or above 166 dB re 1µPa RMS has the potential to adversely affect sea turtles (e.g., injury, temporary threshold shifts, behavior alteration) and thus, will be used as guidance in the analysis of effects on listed species of sea turtles for this BO.

Atlantic Sturgeon

No information on the effects of dredge noise on fish is currently available; however, information on the effects of noise exposure from other underwater activities, such as pile driving, are available and as such, serve as the best available information on underwater noise levels and potential effects to Atlantic sturgeon.

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

- Peak SPL: 206 decibels relative to 1 micro-Pascal (dB re 1 μPa).
- cSEL:187 decibels relative to 1 micro-Pascal-squared second (dB re 1μPa^2-s) for fishes above 2 grams (0.07 ounces). [11]
- cSEL: 183 dB re 1μPa^2-s for fishes below 2 grams (0.07 ounces).

NMFS has relied on these criteria in determining the potential for physiological effects in ESA Section 7 consultations conducted on the U.S. West Coast. At this time, they represent the best available information on the thresholds at which physiological effects to sturgeon are likely to occur. It is important to note that physiological effects may range from minor injuries from which individuals are anticipated to completely recover with no impact to fitness to significant injuries that will lead to death. The severity of injury is related to the distance from the noise source and the duration of exposure (i.e., the closer to the source and the greater the duration of the exposure, the higher likelihood of significant injury). As such, for the purposes of this Opinion, we consider exposure to underwater noise levels of 206 dB re 1 µPa Peak and 187 dB 1 µPa^2·s cSEL a conservative estimate of the level of dredge noise that has the potential to incur physiological effects upon Atlantic sturgeon. Please note, use of the 183 dB 1 µPa^2·s cSEL threshold, is not appropriate for this consultation because all Atlantic sturgeon in the action area will be larger than 2 grams. As explained here, physiological effects could range from minor injuries that a fish is expected to completely recover from with no impairment to survival to major injuries that increase the potential for mortality, or result in death.

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

In order to be detected, a sound must be above the “background” level. Additionally, results from some studies suggest that sound may need to be biologically relevant to an individual to elicit a behavioral response. For example, in an experiment on responses of American shad to sounds produced by their predators (dolphins), it was found that if the predator sound is detectable, but not very loud, the shad will not respond (Plachta and Popper 2003). But, if the sound level is raised an additional 8 or 10 dB, the fish will turn and move away from the sound source. Finally, if the sound is made even louder, as if a predator were nearby, the American shad go into a frenzied series of motions that probably helps them avoid being caught. It was speculated by the

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

researchers that the lowest sound levels were those recognized by the American shad as being from very distant predators, and thus, not worth a response. At somewhat higher levels, the shad recognized that the predator was closer and then started to swim away. Finally, the loudest sound was thought to indicate a very near-by predator, eliciting maximum response to avoid predation. Similarly, results from Doksaeter *et al.* (2009) suggest that fish will only respond to sounds that are of biological relevance to them. This study showed no responses by free-swimming herring (*Clupea* spp.) when exposed to sonars produced by naval vessels; but, sounds at the same received level produced by major predators of the herring (killer whales) elicited strong flight responses. Sound levels at the fishes from the sonar in this experiment were from 197 dB to 209 dB re 1 μPa RMS at 1,000 to 2,000Hz.

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

As hearing generalists, sturgeon rely primarily on particle motion to detect sounds (Lovell *et al*. 2005), which does not propagate as far from the sound source as does pressure. However, a clear threshold for particle motion was not provided in this study. In addition, flanking of the sounds through the substrate may result in higher levels of particle motion at greater distances than would be expected from the non-flanking sounds. Unfortunately, data on particle motion from pile driving, and even dredging, is not available at this time, and we are forced to rely on sound pressure level criteria. Although we agree that more research is needed, other studies have been conducted that support use of this level as an indication for when behavioral effects could be expected (e.g., Mueller-Blenke *et al.* 2010; Purser and Radford 2011; Wysocki *et al.* 2007). Given the available information from studies on other fish species, we consider 150 dB re 1 μPa RMS to be a reasonable estimate of the noise level at which exposure may result in behavioral modifications and as such, we will use 150 dB re 1 μPa RMS as a guideline for assessing when behavioral responses to dredge noise may be expected. The effect of any anticipated response on individuals will be considered in the effects analysis below.

7.2.5.3 Noise Associated with Dredging

Noise generated by dredges are considered continuous and low in frequency (i.e., no rapid rise times; frequency bandwidth between 50 and 1000 Hertz (Hz)) (Richardson *et al*. 1995; Defra 2003; MALSF 2009; 74FR 46090, September 8, 2009) and as such, are within the audible range of listed species of whales and sea turtles and Atlantic sturgeon likely to occur in the action area (e.g., auditory bandwidth for right, humpback, and fin whales are 7 Hz-22kHz (Southall *et al*. 2007); hearing thresholds for sea turtles are 100-1000 Hz (Ketten and Bartol 2005); approximately 100-500 Hz for sturgeon (Meyer and Popper 2002; Popper 2005; Lovell *et al*. 2005)). Low frequency noise tends to carry long distances in water, but due to spreading loss, is attenuated as the distance from the source increases. Under the SRIPP, underwater noise will be generated through the use of a hopper dredge. The primary noise produced from a hopper dredge is associated with the suction pipes and pumps used to remove the fill from the seabed; however, these noise levels fluctuate with the operational status of the dredge, with the highest levels occurring during loading operations (i.e., during the removal of the substrate) (Greene 1985a, 1987). Greene (1987) measured hopper dredge noise during the removal of gravel in the Beaufort Sea and reported received levels of 142 dB re 1μPa at 0.93 kilometers (km) (0.58 miles) for loading operations at a depth of 20 meters, 127 dB re 1μPa at 2.4 km (1.5 miles) while underway, and 117 dB re 1μPa at 13.3 km (8.3 miles) while pumping at a depth of 13 meters. Based on this information, NASA calculated a worst case estimate of underwater noise levels to the 120 dB re 1μPa RMS threshold (i.e., the threshold for continuous noise sources); however, based on the review of the paper by Greene (1987) and a document by the USACE (Clarke *et al*. 2003), which dealt with the removal of sand substrate via a hopper dredge, NMFS has determined that the most appropriate document to use in the analysis of dredge noise, for the purposes of this action, is the information presented by Clarke *et al.* (2003), as it deals with the removal of similar substrate and the recorded levels of underwater noise are in accordance with thresholds established by NMFS (i.e., RMS values) for marine mammals. Additionally, in the analysis of dredge noise and propagation undertaken by NMFS, a transmission loss of 15 log R was used over 10 log R as the latter is more appropriate to use for dredging operations occurring in extremely shallow waters (e.g., less than 25 feet). Based on this information, NMFS has calculated that within 794 meters from the dredge, noise levels could reach 120 dB re 1 μPa RMS, with source levels of approximately 164 dB re 1 μPa RMS (approximately 154 dB re 1 μPa^2-s cSEL; 179 dB re 1μPa Peak) being produced approximately 1 meter from the dredge. It should be noted that to date, equations that take into account other factors affecting perceived underwater noise levels and the propagation of noise (e.g., water depth, frequency, absorptive bottom substrate, ambient noise levels, level of activity in the area, etc.) have not been developed and as such, the estimated distances by NASA and NMFS are most likely overestimates of where increased underwater noise levels will be experienced. Based on the best available information, listed species of whales and sea turtles and Atlantic sturgeon may be exposed to increased underwater noise levels within the action area; however, the audibility and behavioral response of listed species of whales and sea turtles and Atlantic sturgeon is dependent on many factors, such as the physical environment (e.g., depth), existing ambient noise, acoustic characteristics of the sound (e.g., frequency), hearing ability of the animal, as well as behavioral context of the animal (e.g., feeding, migrating, resting) (Southall *et al*. 2007).

7.2.5.4 Effects of Exposure to Dredge Noise

Exposure to Injurious Levels of Sound

As described above, NMFS considers 180 dB re 1 µPa RMS to be the onset of potential for injury for cetaceans; 166 dB re 1 µPa RMS for sea turtles; and 206 dB re 1 µPa Peak and 187 dB re 1 µPa2-s cSEL for the onset of potential injury/mortality to Atlantic sturgeon. However, based on the scientific literature, injury likely occurs at some level well above this level. Therefore, these levels are considered conservative. Regardless, hopper dredging under the SRIPP will not generate source levels in excess of 180 dB re 1μPa RMS (approximately 195 dB re 1μPa Peak) and thus is not likely to cause injury to whales, sea turtles or Atlantic sturgeon. The predominant noise source associated with hopper dredging is caused by the noise generated by suction pipes and pumps. Although source levels of some dredging operations have been reported to reach source levels of 180 dB re 1μPa RMS (195 dB re 1μPa Peak) within 10 meters or less of the dredge, it is extremely unlikely that whales, Atlantic sturgeon, or sea turtles would be exposed to such injurious sound levels as the dredges are moving at very slow speeds (i.e., 10 knots or less), minimizing the likelihood that a sea turtle, Atlantic sturgeon, or whale would be unable to move away from an approaching vessel before the received level reaches a potentially injurious threshold. Based on this information, and the fact that the source levels of dredge noise under the SRIPP will not exceed 164 dB re 1 μPa RMS (154 dB re 1 μPa2-s cSEL; 179 dB re 1μPa Peak), sea turtles, Atlantic sturgeon, and whales are not likely to be exposed to levels of dredge related noise that will result injury.

Exposure to Behaviorally Disturbing Levels of Sound

Sea Turtles

There is very little information about sea turtle behavioral reactions to levels of sound below the thresholds suspected to cause injury or TTS. However, as noted above, McCauley (2000) noted that dB levels of 166 dB re 1μPa RMS were required before any behavioral reaction was observed. As underwater noise levels produced by dredging operations throughout the 50 year life of the SRIPP will not exceed 166 dB re 1μPa RMS (i.e., maximum underwater noise levels will be 164 dB re 1μPa RMS within 1 meter of the dredge) under water noise levels are not likely to reach levels that will disturb sea turtles.

Atlantic Sturgeon

As noted above, 150 dB re 1μPa RMS is believed to be a reasonable estimate of the noise level at which exposure may result in behavioral modifications. As dredging operations will produce underwater noise levels above 150 dB re 1μPa RMS within 10 meters of the dredge, and as the hearing threshold of Atlantic sturgeon overlaps with that of dredges, it is likely that if present in the action area, Atlantic sturgeon will be able to detect the presence of the dredge, resulting in possible behavioral modification. However, based on a recent study done in the James River, Atlantic sturgeon continued normal behavior within the river, regardless of the presence of a dredge and showed no signs of impeded movement, up or downriver, due to the presence of the dredge and in fact, actively moved past the dredge (Cameron 2010). Additionally, an avoidance response (e.g., due to dredge noise) was never observed by Atlantic sturgeon as indicated by Atlantic sturgeon remaining in close proximity to the dredge following tagging release. (i.e., the fish remained in proximity to the dredge for 3.5 to 21.5 hours following release). Based on this information, it is unlikely that the elevated levels of underwater noise will cause significant behavioral changes of Atlantic sturgeon that may be present in the offshore borrow site and thus, if any minor movements away from the area being dredged do occur, it is extremely unlikely that these movements will cause substantial changes to essential Atlantic sturgeon behaviors (e.g., reproduction, foraging, resting, and migration). Additionally, as noted above, the extent of underwater noise is not likely to present a barrier to Atlantic sturgeon movements and as such, if individuals are present within the vicinity of the action area, they are likely to continue normal behaviors (e.g., feeding, resting, and migrating) in other portions of the action area and/or in other locations within Virginia coastal waters. Based on this and the best available information, NMFS concludes that dredge noise is not likely to cause significant behavior modification to Atlantic sturgeon.

Whales

As described above, dredging noise is not expected to cause injury to whales; however, there is potential for whales to be exposed to behaviorally disturbing levels of sound produced by these activities. Potentially disturbing levels of construction-related noise (120-160 dB re 1μPa RMS) are expected to propagate over distances ranging from 1.0 to 794 meters from the source. As dredging operations are proposed to occur year round and humpbacks are likely to occur in the action area from September-April; right whales from November-May; and Fin whales from October-January; and, individual transient whales could be present in the action area outside of these time frame as this area is used by whales migrating between calving/mating grounds and foraging grounds, there is a potential for listed species to be exposed to increased underwater noise levels at anytime throughout the year. Based on this information, the remainder of the acoustics portion of the analysis will focus on the effects of dredge noise on listed species of whales.

Characterizing the effects of noise on whales involves assessing the species' sensitivity to the particular frequency range of the sound; the intensity, duration, and frequency of the exposure; the potential physiological effects caused by the animals response to the increase in underwater noise; and, the potential behavioral responses that could lead to impairment of feeding, breeding, nursing, breathing, sheltering, migration, or other biologically important functions. To date, few studies have been done that analyze and assess the effects of dredge noise and operations on marine mammals. Much of any analysis involving the effects of anthropogenic sounds on listed species relates to how an animal may change behavior upon exposure to vessel noise and operations (e.g., drillships and seismic vessels) and as such, will be used as the best available information in referencing potential effects of dredge noise on listed species of whales.

The most commonly observed marine mammal behavioral responses to vessel noise and activities include increased swim speed (Watkins 1981), horizontal and vertical (diving) avoidance (Baker *et al*. 1983; Richardson *et al*. 1985), changes in respiration or dive rate (Baker *et al*. 1982; Bauer and Herman 1985; Richardson *et al*. 1985; Baker and Herman 1989; Jahoda *et al.* 2003), and interruptions or changes in feeding or social behaviors (Richardson *et al*. 1985; Baker *et al.* 1982; Jahoda *et al.* 2003). However, Watkins (1981) noted that the passage of a tanker within 800 m did not disrupt feeding humpback whales and Brewer *et al*. (1993) and Hall *et al*. (1994) reported numerous sightings of marine mammals, including bowhead whales, in the vicinity of offshore drilling operations in the Beaufort Sea, with one whale sighted 400 m of the drilling vessel. Additionally, based on the review of a number of papers describing the response of marine mammals to non-pulsed sound, Southall *et al*. (2007) reported that in general, behavioral responses of marine mammals did not occur until sounds were higher than 120 dB and that many animals had no observable response at all when exposed to anthropogenic sound at levels of 120 dB re 1μPa RMS or even higher.

Although the above studies demonstrate that a high degree of variability exists in the intensity of responses of marine mammals to vessel noise and activities, it is still unclear whether these responses are due solely to the increase in underwater noise levels, the physical presence of a nearby vessel, or a combination of both. Often, specific acoustic features of the sound and contextual variables (i.e., proximity, durations, or recurrence of the sound or the current behavior that the marine mammal is engaged in or its prior experience), as well as entirely separate factors such as the physical presence of a nearby vessel, may be more relevant to the animal's response than the received level alone (75 FR Register 20482, April 19, 2010). For instance, Baker *et al*. (1982) found that abrupt changes in engine speed and aggressive maneuvers such as circling the whale or crossing directly behind or in front of the whale or its projected path elicited much stronger responses than unobtrusive maneuvering (tracking in parallel to the whale and changing vessel speed only when necessary to maintain a safe distance from the whale). Reactions were even less intense during a simple straight line passby, which most closely represents the type of vessel transit that will take place as a result of the construction activities (i.e., not targeted toward viewing whales).

Richardson *et al*. (1985) observed strong reactions in bowhead whales to approaching boats and subtler reactions to drillship playbacks, but also found that bowhead whales often occurred in areas where low frequency underwater noise from drillships, dredges, or seismic vessels was readily detectable, suggesting that bowheads may react to transient or recently begun industrial activities, but may tolerate noise from operations that continue with little change for extended periods of time (hours or days).

Watkins (1986) compiled and summarized whale responses to human activities in Cape Cod Bay over 25 years, and found that the types of reactions had shifted over the course of time, generally from predominantly negative responses to an increasing number of uninterested or positive responses, although trends varied by species and only emerged over relatively long spans of time (i.e., individual variability from one experience to the next remains high). Watkins also noted that whales generally appeared to habituate rapidly to stimuli that were relatively non-disturbing.

One playback experiment on right whales recorded behavioral reactions on summer foraging grounds to different stimuli, including an alert signal, vessel noise, other whale social sounds, and a silent control (Nowacek *et al*. 2004). No significant response was observed in any case except the alert signal broadcast ranging from 500 to 4,500 Hz. In response to the alert signal, which had measured received levels between 130 and 150 dB, whales abandoned current foraging dives, began a high power ascent, remained at or near the surface for the duration of the exposure, and spent more time at subsurface depths (1 to 10 m) (Nowacek *et al*. 2004). The only whale that did not respond to this signal was the sixth and final whale tested, which had potentially already been exposed to the sound five times. The lack of response to a vessel noise stimulus from a container ship and from passing vessels indicated that whales are unlikely to respond to the sounds of approaching vessels even when they can hear them (Nowacek *et al*. 2004). This non-avoidance behavior could be an indication that right whales have become habituated to the vessel noise in the ocean and therefore do not feel the need to respond to the noise or may not perceive it as a threat. In another study, scientists played a recording of a tanker using an underwater sound source and observed no response from a tagged whale 600 meters away (Johnson and Tyack 2003). These studies may suggest that if right whales are startled or disturbed by novel construction sounds, they may temporarily abandon feeding activities, but may habituate to those sounds over time, particularly if the sounds are not associated with any aversive conditions.

The evidence presented above indicates that animals do respond and modify behavioral patterns in the presence of vessel noise and activity, although adequate data does not yet exist to quantitatively assess or predict the significance of minor alterations in behavior to the health and viability of marine mammal and sea turtle populations. Based on this information it is reasonable to assume that the potential exists that dredge noise and operations under the SRIPP may similarly cause behavioral changes to listed species of whales in the action area. However, in previous studies the areas of research were known to be sites where whales concentrated and as such had a higher probability of being exposed to elevated underwater noise levels that resulted in behavioral alterations. The action area is not known as an area where listed species of whales congregate for the purposes of foraging, resting, or reproduction. Instead, the action area is primarily used for migration to and from foraging and calving grounds throughout the year. As such, the behavioral responses observed in previous studies due to vessel noise and operations are extremely unlikely to occur under the SRIPP as it is extremely unlikely that whales will be found in high concentrations in the action area, resulting in an extremely low probability that a whale will be within 794 meters of the dredge at any one time and therefore, exposed to levels of underwater noise levels that could adversely affect and/or cause behavioral changes to the animal in a manner that disrupts essential behaviors (e.g., feeding, resting, migrating, reproducing). In addition, in the unlikely event that a whale approaches the area where the dredge is in operation, the mitigation measures NASA has established as part of the action (e.g., NMFS approved sea turtle/marine mammal observer on board all dredge vessels from April-November and a designated lookout/bridge watch on board all dredge vessels from December 1- March 31; shut down of dredge pumps when a whale is observed within 1 km of the dredge; 500 yard restriction on vessel approach to right whales; compliance with SAS operations), will ensure that whales will not be exposed to underwater noise levels greater than or equal to 120 dB re 1μPa RMS. Based on the best available information, NMFS concludes that the effects of dredge noise on listed species of whales will be insignificant and discountable.

In addition, it should be noted that when assessing the potential effects of anthropogenic noise on marine mammals, it is important to consider that there are “zones of audibility” and “zones of responsiveness” that will affect marine mammal responses to anthropogenic noise. The most extensive zone is the zone of audibility, the area within which the mammal might hear noise (Richardson *et al*. 1995). The zone of responsiveness is the region within which the animal reacts behaviorally (i.e., stop feeding) or physiologically (i.e., increase in respiratory rates) (Richardson *et al.* 1995). Marine mammals usually do not respond overtly to audible, but weak man made sounds and therefore, the zone of responsiveness is usually much smaller than the zone of audibility (Richardson *et al.* 1995). It is believed that marine mammals will not remain in areas where received levels of continuous underwater noise are 140 + dB at frequencies to which the animals are most sensitive (Richardson *et al.* 1995). As such, although underwater noise levels of 120 dB re 1μPa RMS may be audible to listed species of whales within 794 meters of the dredge, the behavioral response to elevated noise levels most likely will occur within 40 meters or less from the dredge where underwater noise levels will be greater than or equal to 140 dB re 1μPa RMS. As noted above, it is extremely unlikely for whales to be within 1 km of the dredge and therefore, extremely unlikely for a whale to be within 40 meters or less of the dredge where responses to underwater noise levels are believed to occur. In addition, with the mitigation measures in place, listed species of whales will not be exposed to levels greater than or equal 120 dB re 1μPa RMS as all pumps will be turned off upon a whale observed within 1 km of the dredge. As such, based on the best available information, NMFS concludes that the effects of dredge noise on listed species of whales are discountable.

7.2.6 Fuel Oil Spills

Fuel oil spills could occur from the dredge plant or tender vessel. A fuel oil spill would be an unintended, unpredictable event. Marine animals, including whales, Atlantic sturgeon, and sea turtles, are known to be negatively impacted by exposure to oil and other petroleum products. Without an estimate of the amount of fuel oil released it is difficult to predict the likely effects on listed species. No accidental spills of diesel fuel are expected during dredging operations; however, if such an incident does occur, implementation of the USCG-approved safety response plans or procedures outlined in the WFF Integrated Contingency Plan (ICP) to prevent and minimize any impacts associated with a spill will be implemented by all personnel to ensure a rapid response to any spill. As the effects of a possible spill are likely to be localized and temporary, sea turtles, Atlantic sturgeon, and whales are not likely to be exposed to oil and any effects would be discountable. Additionally, should a response be required by the United States Environmental Protection Agency or the USCG, there would be an opportunity for NMFS to conduct a consultation with the lead Federal agency on the oil spill response.

7.3 Effects of Sand Placement/Beach Renourishment

As noted in the Description of the Action, 3.7 miles of the Wallops Island shoreline will receive beach fill and renourishment over the 50 year life of the SRIPP. The initial fill will be placed so that there will be a 6-foot high berm extending a minimum of 70-feet seaward of the existing seawall. The remainder of the fill will be placed at a 20:1 slope underwater for an additional distance seaward; the amount of that distance would vary along the length of the beach fill, but will extend for about an additional 137 m (450 ft), so that the total distance of the fill profile from the seawall will be up to approximately 158 m (520 ft). The primary effects under consideration are: (1) reduction in Atlantic sturgeon and sea turtle prey and alteration of foraging behavior; and (2) suspended sediment associated with beach fill operations.

7.3.1 Interactions with the Sediment Plume

The placement of sand along the 3.7 mile area along the Wallops Island shoreline will cause an increase in localized turbidity associated with the beach nourishment operations in the nearshore environment and from the anchoring of the dredge and pump-out stations. 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 offshore borrow 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 (TSS) 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 TSS concentrations related to nourishment activities were 64 mg/L and 34 mg/L, which were only slightly higher than background maximum bottom TSS 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 TSS concentrations associated with the active beach nourishment site were limited to within 400 m (1,310 ft) 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 TSS levels are expected to be limited to a narrow area of the swash zone up to 500 m (1,640 ft) downcurrent 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 Wallops Island 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.

As noted above, 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 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 sea turtle movements is likely to be insignificant. Additionally, the TSS levels expected 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 movement or 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 fill operations on sea turtles and Atlantic sturgeon will be insignificant.

7.3.2 Alteration of Foraging Habitat

Sea Turtles

Of the listed species found in the action area, loggerhead and Kemp's ridley sea turtles are the most likely to utilize the nearshore area for feeding, foraging mainly on benthic species, namely crabs and mollusks (Morreale and Standora 1992, Bjorndal 1997). As no seagrass beds exist along the nearshore area of Wallops Island, green sea turtles will not use the nearshore area as foraging areas and as such, sand placement and beach nourishment are not likely to disrupt normal feeding behaviors of green sea turtles. Additionally, leatherback sea turtles are primarily pelagic, feeding on jellyfish and may come into shallow water if there is an abundance of jellyfish nearshore. However, as the nearshore area along Wallops Island is not known to be an area where jellyfish concentrate, leatherback sea turtles are unlikely to be found foraging in the nearshore area where disposal activities will occur. As such, beach nourishment activities are not likely to disrupt leatherback foraging behavior. However, as suitable loggerhead and Kemps ridley foraging items occur on the benthos of the nearshore area and depths within this portion of the action area are suitable for use by sea turtles, some loggerhead and Kemp ridley sea turtle foraging likely occurs at these sites.

Atlantic Sturgeon

As described above, Atlantic sturgeon concentrate in several distinct areas along the eastern coastline of the United States, with the nearshore waters between the Chesapeake Bay and the Delaware Bay being one of these identified areas (i.e., Stein *et al.* 2004a; Laney *et. al* 2007; Erickson *et al*. 2011; Dunton *et al*. 2010; NEFOP and ASM data 2006-2010; NEMAP data 2007-2011; NMFS inshore Trawl data 1972-2011); The portion of the action area where beach nourishment operations will take place is located within the range of this concentration area. Based on this and the best available information, the portion of the action area where beach nourishment operations will take place is likely to be used by foraging, overwintering, and/or migrating sturgeon throughout year, with the spring months likely to be months of highest sturgeon use (survey data from NEFOP and ASM 2006-2010; NEMAP 2007-2011; NMFS inshore Trawl 1972-2011). As such, the placement of fill along the Wallops Island shoreline could affect available Atlantic sturgeon food sources and thus, the foraging ability of Atlantic sturgeon.

Sea Turtle and Atlantic Sturgeon Foraging Effects

Beach nourishment can affect Atlantic sturgeon and sea turtles by reducing prey species through the alteration of the existing biotic assemblages. The placement of dredged sand along the Wallops shoreline will bury existing subtidal benthic organisms (i.e., crabs, clams, mussels) along the 14,000 feet of seawall as well as the area extending seaward, approximately 520-feet from the seawall. In total, approximately 1.2 acres of hard bottom, intertidal habitat will be permanently buried. In addition, approximately 225 acres of the sub-tidal benthic community along the existing seawall will be buried during initial fill placement.

While some nearshore areas may be more desirable to certain turtles or Atlantic sturgeon due to prey availability, there is no information to indicate that the nearshore areas proposed for beach nourishment have more abundant sturgeon and 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 nearshore waters along the Wallops Island shoreline than to other foraging areas and should be able to find sufficient prey in alternate areas. Depending on the species, recolonization of a newly renourished beach are 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 the Wallops shoreline is similar in grain size as the indigenous beach sand, it is expected that recolonization of the nearshore benthos will occur within 2-6 months after initial beach fill or renourishment 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).

NMFS anticipates that while the beach nourishment activities may temporarily disrupt normal feeding behaviors for sturgeon and sea turtles by causing them to move to alternate areas, the beach nourishment 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. In addition, the placement of sand seaward of the existing seawall, where previously no beach area existed, will have beneficial effects on benthic organisms by restoring and creating new beach habitat and therefore, providing additional sources of prey along the Wallops Island shoreline that previously were not present. As such, based on the best available information, the placement of sand is not likely to remove critical amounts of prey resources from the action area and any disruption to normal foraging is likely to be insignificant.

7.3.3 Fuel Oil Spills

Throughout the project, construction vehicles will be present on the existing roads and also during the use of heavy machinery on the beach or at the north end of Wallops Island throughout different phases of the SRIPP. The nearshore marine environment may be affected if a spill or leak from construction vehicles or heavy machinery occurs. Construction-related impacts are expected to be temporary and will not likely be adverse because any accidental release of contaminants or liquid fuels will be addressed in accordance with the existing WFF ICP emergency response and clean-up measures. Additionally, implementation of Best Management Practices (BMPs) for equipment and vehicle fueling and maintenance and spill prevention and control measure will reduce the potential impacts on surface water during construction. As the effects of a possible spill are likely to be localized and temporary, sturgeon, sea turtles and whales are not likely to be exposed to oil and any effects would be discountable. Additionally, should a response be required by the United States Environmental Protection Agency or the USCG, there would be an opportunity for NMFS to conduct a consultation with the lead Federal agency on the oil spill response.

7.4 Climate change related effects of SRIPP

In sections 6.0, we considered effects of global climate change, generally, on listed species of whales, sea turtles, and Atlantic sturgeon. Given the likely rate of climate change, it is unlikely that there will be any noticeable effects to sea turtles, whales, or Atlantic sturgeon in the action area over the 50-year life or the SRIPP. As explained in section 6.0, based on currently available information and predicted habitat changes, these effects are most likely to be changes in distribution/seasonal migrations of sea turtles, whales, and Atlantic sturgeon throughout the coastal waters of Virginia. Additionally, the SRIPP will not affect the ability of these species to adapt to climate change or affect their movement or distribution along the coastline of Virginia or within waters of the Atlantic Ocean.

8.0 CUMULATIVE EFFECTS

Cumulative effects as defined in 50 CFR 402.02 to include the effects of future State or private activities, not involving Federal activities, that are reasonably certain to occur within the action area. Future Federal actions are not considered in the definition of “cumulative effects.” Ongoing Federal actions are considered in the “Environmental Baseline” section above.

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 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.

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

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

9.0 INTEGRATION AND SYNTHESIS OF EFFECTS

NMFS has estimated that the over the life of the SRIPP, up to 9 sea turtles will be entrained in hopper dredging operations, with no more than 1 likely to be a Kemp's ridley, and the remainder being loggerheads. Additionally, NMFS has estimated that over the life of the SRIPP, up to 2 subadult Atlantic sturgeons will be entrained. As explained in the "Effects of the Action" section, effects of habitat alteration, dredge noise, suspended sediment, vessel interactions, and fuel spills on sea turtles, whales, or Atlantic sturgeon as a result of dredging and beach nourishment operations 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 offshore shoals, 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, NMFS considers whether the effects of the action reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of the listed species in the wild by reducing the reproduction, numbers, or distribution of the species. The purpose of this analysis is to determine whether the action would jeopardize the continued existence of the 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 each of the listed species that may be affected by the SRIPP, NMFS summarizes the status of the species and considers whether the action will result in reductions in reproduction, numbers or distribution of that species and then considers whether any reductions in reproduction, numbers or distribution resulting from the SRIPP would reduce appreciably the likelihood of both the survival and recovery of that species, as those terms are defined for purposes of the federal Endangered Species Act.

9.1 Kemp's Ridley Sea Turtles

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 2007b). 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 turtles 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. Based on the number of nests laid in 2006 and the remigration interval for Kemp's ridley sea turtles, there were an estimated 7,000-8,000 adult female Kemp's ridleys in 2006 (NMFS and USFWS 2007b), which represents an increase in the nesting trend for Kemp's ridleys.

The most recent review of the Kemp's ridley as a species suggests that it 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). More female Kemp's ridley sea turtles are maturing and subsequently nesting, and/or are surviving to an older age and producing more nests across their lifetime, resulting in a positive population trend globally.

Despite the threats faced by individual Kemp's ridley sea turtles inside and outside of the action area, the SRIPP 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 action. While NMFS is not able to predict with precision how climate change will continue to impact Kemp's ridley sea turtles in the action area or how the species will adapt to climate-change related environmental impacts, we have considered the effects of the action 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.

As noted above, NMFS has estimated that over the 50 year life of the SRIPP, up to 9 sea turtles will be entrained and killed in hopper dredge operations, with no more than 1 likely to be a Kemp's ridley, and the remainder being loggerheads. The mortality of 1 Kemp's ridley represents a very small percentage of the Kemp's ridleys worldwide. Even taking into account just nesting females, the death of 1 Kemp's ridley represents less than 0.01% of the population. While the death of 1 Kemp's ridley will reduce the number of Kemp's ridleys compared to the number that would have been present absent the SRIPP, 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 1 Kemp's ridley 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 individual that would be killed as a result of the action, 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 SRIPP 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 now or over 50 years (i.e., see section 6.0).

The SRIPP is not likely to reduce distribution because the action 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 SRIPP, 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 1 Kemp's ridley sea turtle over the 50 year life of the SRIPP 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 1 Kemp's ridley represents an extremely small percentage of the species as a whole; (3) the death of 1 Kemp's ridley will not change the status or trends of the species as a whole; (4) the loss of this Kemp's ridley is not likely to have an effect on the levels of genetic heterogeneity in the population; (5) the loss of this Kemp's ridley is likely to have such a small effect on reproductive output that the loss of this individual will not change the status or trends of the species; (5) the action 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, (6) the action 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 certain instances an action that does not appreciably reduce the likelihood of a species survival (persistence) may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the SRIPP will not appreciably reduce the likelihood that Kemp's ridleys will survive in the wild. Here, NMFS considers the potential for the action to reduce the likelihood of recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., "endangered"), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., "threatened") because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

The SRIPP is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of Kemp's ridley sea turtles in any geographic area and thus, it will not affect the overall distribution of Kemp's ridley sea turtles. The SRIPP will not utilize Kemp's ridley sea turtles for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The SRIPP is likely to result in the mortality of 1 Kemp's ridley; however, as explained above, the loss of this individual and what would have been their progeny is not expected to affect the persistence of Kemp's ridleys. As the reduction in numbers and future reproduction is very small, the loss of this individual will not change the status or trend of

Kemp's ridleys, which is stable to increasing. The effects of the action will not hasten the extinction timeline or otherwise increase the danger of extinction since the action will cause the mortality of only a very small percentage of the species as a whole and these mortalities are not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the SRIPP will not appreciably reduce the likelihood that Kemp's ridleys can be brought to the point at which they are no longer listed as endangered or threatened. Based on the analysis presented herein, the SRIPP, resulting in the entrainment and mortality of 1 individual Kemp's ridley, is not likely to appreciably reduce the survival and recovery of this species.

9.2 Northwest Atlantic Ocean DPS of Loggerhead Sea Turtles

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, 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.

Based on the information provided in this Opinion, NMFS anticipates the entrainment and mortality of no more than 9 loggerhead sea turtles over the life of the SRIPP (i.e., through 2062). The lethal removal of up to 9 loggerhead sea turtle from the action area 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 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 sea turtles entrained in hopper dredges operating in the waters off Virginia originate from several of the recovery units. Limited information is available on the genetic makeup of sea turtles in the mid-Atlantic. 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 likely to occur in the action area from the DTRU or the NGMRU, it is extremely unlikely that any of the up to 9 loggerheads that are likely to be entrained during dredging operations are likely to have originated from either of these recovery units. The majority, at least 80% of the loggerheads entrained, are likely to have originated from the PFRU, with the remainder from the NRU and GCRU. As such, 7 of the sea turtles are expected to be from the PFRU and 2 from the NRU or the GCRU.

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, Yucatan, 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 Yucatan 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 7 loggerheads over a 50 year time 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 7 individuals would represent approximately 0.04% of the population. Similarly, the loss of two loggerheads over a 50 year period from the NRU or GCRU represents an extremely small percentage from either recovery unit. Even if the total NRU population was limited to 1,272 loggerheads, the loss of two individuals would represent approximately 0.16% of the NRU population, while the loss of two loggerheads over a 50 year time period from the GCRU, which is expected to support at least 1,000 nesting females, represents less than 0.2 % of the population. The loss of such a small percentage of individuals from any of these recovery units represents an even smaller percentage of the species as a whole. As such, it is unlikely that the death of these individuals 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. Additionally, this action is not likely to reduce the distribution of loggerheads as the action will not impede loggerheads from accessing suitable foraging grounds or disrupt other migratory behaviors.

In general, while 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 loggerhead sea turtles because: the species is widely distributed geographically, it is not known to have low levels of genetic diversity, and there are several thousand individuals in the population.

Based on the information provided above, the death of up to 9 loggerhead sea turtles over a 50 year time period as a result of the SRIPP 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 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 death of up to 9 loggerheads represents an extremely small percentage of the species as a whole; (2) the loss of these loggerheads will not change the status or trends of any nesting aggregation, recovery unit or the species as a whole; (3) the loss of up to 9 loggerheads is not likely to have an effect on the levels of genetic heterogeneity in the population; (3) the loss of up to 9 loggerheads is likely to have an undetectable effect on reproductive output of any nesting aggregation or the species as a whole; and, (4) the action will have no effect on the distribution of loggerheads in the action area or throughout its range; and, (6) the action will have no effect on the ability of loggerheads to shelter and only an insignificant effect on individual foraging loggerheads.

In certain 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, NMFS has determined that the action will not appreciably reduce the likelihood that loggerheads will survive in the wild. Here, NMFS considers the potential for the action to reduce the likelihood of recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate.

Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., "endangered"), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., "threatened") because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

The action will not appreciably reduce the likelihood of survival of the loggerhead sea turtle species. Also, it is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of loggerheads in any geographic area and since it will not affect the overall distribution of loggerheads other than to cause minor temporary adjustments in movements in the action area. The action will not utilize loggerheads for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect any of these species of sea turtles, or affect their continued existence. As explained above, the action is likely to result in the mortality of up to 9 loggerheads over the life of the SRIPP (i.e., through 2062); however, as explained above, the loss of these individuals over this time period is not expected to affect the persistence of loggerhead sea turtles. In summary, the effects of the action will not hasten the extinction timeline or otherwise increase the danger of extinction; further, the action will not prevent the species from growing in a way that leads to recovery and the action will not change the rate at which recovery can occur. This is the case because while the action 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, these effects will be undetectable over the long-term and the action is 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 action 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 action 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 action. While NMFS is not able to predict with precision how climate change will continue to impact loggerhead sea turtles in the action area or how the species will adapt to climate-change related environmental impacts, we have considered the effects of the action 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 action, resulting in the mortality of up to 9 loggerheads, is not likely to appreciably reduce the survival and recovery of the NWA DPS of loggerhead sea turtles.

9.3 Atlantic Sturgeon

As explained above, the action is likely to result in the mortality of 2 or fewer Atlantic sturgeon. As noted above, we expect that both of these sturgeon will be subadults and that both Atlantic sturgeon may come from any of the five DPSs.

Gulf of Maine DPS

Individuals originating from the GOM DPS are likely to occur in the action area. The GOM DPS has been listed as threatened. While Atlantic sturgeon occur in several rivers in the GOM DPS, recent spawning has only been documented in the Kennebec River and possibly the Androscoggin River. No total population estimates are available. We have estimated, based on fishery-dependent data, that there are approximately 645 subadults in the GOM DPS. GOM DPSAtlantic 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.

NMFS has estimated that the SRIPP will result in the possible mortality of up to 2 subadult Atlantic sturgeon, of which, one may be a GOM DPS Atlantic sturgeon. The following analysis applies to the worst case scenario of all the sturgeon coming from the GOM DPS. In addition, as described above, the total population size of the GOM DPS is unknown at this time; however, in the absence of an estimate of the overall GOM DPS population, NMFS has provided a subadult population estimate for the GOM DPS (see above). This represents the best available information on subadult population numbers for the GOM DPS and will therefore, allow us to consider the loss of these individuals against the life stage for which we have an estimated population size.

The mortality of up to 2 subadult Atlantic sturgeon from the GOM DPS subadult population represents a very small percentage of the subadult population (i.e., less than 0.3% of the population). While the death of up to 2 subadult Atlantic sturgeon will reduce the number of GOM DPS Atlantic sturgeon compared to the number that would have been present absent the action, it is not likely that this reduction in numbers will change the status of this species as this loss represents a very small percentage of the subadult (less than 0.3%) population). In addition, as described above, subadults, based on their smaller size (i.e., 40-150 cm), are more likely to be entrained than full sized adults (>150 cm). As such, the reproductive potential of the GOM DPS is not expected to be significantly affected in any way other than through a reduction in numbers of individuals. A reduction in the number of GOM DPS Atlantic sturgeon 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 a small reduction in the number of eggs laid or larvae produced in future years and similarly, a very 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 action, any effect to future year classes is anticipated to be very small and would not change the status of this species. Additionally, as the action will occur outside of the rivers where GOM DPS fish are expected to spawn (e.g., the Kennebec River in Maine), the action will not affect their spawning habitat in any way and will not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds.

The action is 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 temporal and geographic scale of the SRIPP.

Based on the information provided above, the death of up to 2 subadult GOM DPS Atlantic sturgeon over the life of the SRIPP 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 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, including reproduction, sustenance, and shelter. This is the case because: (1) the death of up to 2 subadult GOM DPS Atlantic sturgeon represents an extremely small percentage of the species as a whole; (2) the death of up to 2 subadult GOM DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these subadult GOM DPS Atlantic sturgeon are 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 are 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 action 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 action will have no effect on the ability of GOM DPS Atlantic sturgeon to shelter and only an insignificant effect on individual foraging GOM DPS Atlantic sturgeon.

In certain instances an action that does not appreciably reduce the likelihood of a species survival (persistence) may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the action will not appreciably reduce the likelihood that GOM DPS Atlantic sturgeon will survive in the wild. Here, NMFS considers the potential for the action to reduce the likelihood of recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

The action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of GOM DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of GOM DPS Atlantic sturgeon. The action will not utilize GOM DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The action is likely to result in the mortality of up to 2 subadult GOM DPS Atlantic sturgeon; however, as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the GOM DPS. As the reduction in numbers and future reproduction is very small, the loss of these individuals will not change the status of GOM DPS Atlantic sturgeon. The effects of the action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will cause the mortality of only a very small percentage of the species as a whole and these mortalities are not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the action will not appreciably reduce the likelihood that GOM DPS can be brought to the point at which they are no longer listed as endangered or threatened.

Despite the threats faced by individual GOM DPS Atlantic sturgeon inside and outside of the action area, the action will not increase the vulnerability of individual sturgeon to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the action. While we are not able to predict with precision how climate change will continue to impact Atlantic sturgeon in the action area or how the species will adapt to climate-change related environmental impacts, we have considered the effects of the action 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 action, resulting in the entrainment and mortality of up to 2 subadult GOM DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

New York Bight DPS

Individuals originating from the NYB DPS are likely to occur in the action area. The NYB DPS has been listed as endangered. While Atlantic sturgeon occur in several rivers in the NYB DPS, recent spawning has only been documented in the Delaware and Hudson Rivers. The vast majority of spawning occurs in the Hudson River, with Delaware River origin Atlantic sturgeon making up less than 20% of the NYB DPS adult population. We have estimated, based on fishery-dependent data, that there are approximately 2,853 subadults in the New York Bight DPS. NYB DPS 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, for the Hudson or Delaware River spawning populations, or for the DPS as a whole.

NMFS has estimated that the SRIPP will result in the possible mortality of up to 2 subadult Atlantic sturgeon, of which, one may be a NYB DPS Atlantic sturgeon. The following analysis applies to the worst case scenario of all the sturgeon coming from the NYB DPS. In addition, as described above, the total population size of the NYB DPS is unknown at this time; however, in the absence of an estimate of the overall NYB DPS population, NMFS has provided a subadult population estimate for the NYB DPS (see above). This represents the best available information on subadult population numbers for the NYB DPS and will therefore, allow us to consider the loss of these individuals against the life stage for which we have an estimated population size.

The mortality of up to 2 subadult Atlantic sturgeon from the NYB DPS subadult population represents a very small percentage of the subadult population (i.e., less than 0.07% of the population). While the death of up to 2 subadult Atlantic sturgeon will reduce the number of NYB DPS Atlantic sturgeon compared to the number that would have been present absent the action, it is not likely that this reduction in numbers will change the status of this species as this loss represents a very small percentage of the subadult (less than 0.07%) population). In addition, as described above, subadultss, based on their smaller size (i.e., 40-150 cm), are more likely to be entrained than full sized adults (>150 cm). As such, the reproductive potential of the NYB DPS is not expected to be significantly affected in any way other than through a reduction in numbers of individuals. A reduction in the number of NYB DPS Atlantic sturgeon 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 a small reduction in the number of eggs laid or larvae produced in future years and similarly, a very 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 action, any effect to future year classes is anticipated to be very small and would not change the status of this species. Additionally, as the action will occur outside of the rivers where NYB DPS fish are expected to spawn (e.g., the Hudson and Delaware Rivers), the action will not affect their spawning habitat in any way and will not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds.

The action is not likely to reduce distribution because the action 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 temporal and geographic scale of the SRIPP.

Based on the information provided above, the death of up to 2 subadult NYB DPS Atlantic sturgeon over the life of the SRIPP 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 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, including reproduction, sustenance, and shelter. This is the case because: (1) the death of up to 2 subadult NYB DPS Atlantic sturgeon represents an extremely small percentage of the species as a whole; (2) the death of up to 2 subadult NYB DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these subadult NYB DPS Atlantic sturgeon are not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of these subadult NYB DPS Atlantic sturgeon are 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 action 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, (6) the action 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 certain instances an action that does not appreciably reduce the likelihood of a species survival (persistence) may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the action will not appreciably reduce the likelihood that NYB DPS Atlantic sturgeon will survive in the wild. Here, NMFS considers the potential for the action to reduce the likelihood of recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

The action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of NYB DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of NYB DPS Atlantic sturgeon. The action will not utilize NYB DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The action is likely to result in the mortality of up to 2 subadult NYB DPS Atlantic sturgeon; however, as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the NYB DPS. As the reduction in numbers and future reproduction is very small, the loss of these individuals will not change the status of NYB DPS Atlantic sturgeon. The effects of the action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will cause the mortality of only a very small percentage of the species as a whole and these mortalities are not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the action will not appreciably reduce the likelihood that NYB DPS can be brought to the point at which they are no longer listed as endangered or threatened.

Despite the threats faced by individual NYB DPS Atlantic sturgeon inside and outside of the action area, the action will not increase the vulnerability of individual sturgeon to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the action. While we are not able to predict with precision how climate change will continue to impact Atlantic sturgeon in the action area or how the species will adapt to climate-change related environmental impacts, we have considered the effects of the action 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 action, resulting in the entrainment and mortality of up to 2 subadult NYB DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

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. While Atlantic sturgeon occur in several rivers in the CB DPS, recent spawning has only been documented in the James River. Using fishery-dependent data, we have estimated that there are 819 subadults in the CB DPS. Chesapeake Bay DPS 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, for the James River spawning population or for the DPS as a whole.

NMFS has estimated that the SRIPP will result in the possible mortality of up to 2 subadult Atlantic sturgeon, of which, one may be a CB DPS Atlantic sturgeon. The following analysis applies to the worst case scenario of all the sturgeon coming from the CB DPS. In addition, as described above, the total population size of the CB DPS is unknown at this time; however, in the absence of an estimate of the overall CB DPS population, NMFS has provided a subadult population estimate for the CB DPS (see above). This represents the best available information on subadult population numbers for the CB DPS and will therefore, allow us to consider the loss of these individuals against the life stage for which we have an estimated population size.

The mortality of up to 2 subadult Atlantic sturgeon from the CB DPS subadult population represents a very small percentage of the subadult population (i.e., less than 0.24% of the population). While the death of up to 2 subadult Atlantic sturgeon will reduce the number of CB DPS Atlantic sturgeon compared to the number that would have been present absent the action, it is not likely that this reduction in numbers will change the status of this species as this loss represents a very small percentage of the subadult (less than 0.24%) population). In addition, as described above, subadults, based on their smaller size (i.e., 40-150 cm), are more likely to be entrained than full sized adults (>150 cm). As such, the reproductive potential of the CB DPS is not expected to be significantly affected in any way other than through a reduction in numbers of individuals. A reduction in the number of CB DPS Atlantic sturgeon 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 a small reduction in the number of eggs laid or larvae produced in future years and similarly, a very 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 action, any effect to future year classes is anticipated to be very small and would not change the status of this species. Additionally, as the action will occur outside of the rivers where CB DPS fish are expected to spawn (i.e., James River), the action will not affect their spawning habitat in any way and will not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds.

The action is 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 temporal and geographic scale of the SRIPP.

Based on the information provided above, the death of up to 2 subadult CB DPS Atlantic sturgeon over the life of the SRIPP 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 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, including reproduction, sustenance, and shelter. This is the case because: (1) the death of up to 2 subadult CB DPS Atlantic sturgeon represents an extremely small percentage of the species as a whole; (2) the death of up to 2 subadult CB DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these subadult CB DPS Atlantic sturgeon are 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 are 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 action 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 action will have no effect on the ability of CB DPS Atlantic sturgeon to shelter and only an insignificant effect on individual foraging CB DPS Atlantic sturgeon.

In certain instances an action that does not appreciably reduce the likelihood of a species survival (persistence) may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the action will not appreciably reduce the likelihood that CB DPS Atlantic sturgeon will survive in the wild. Here, NMFS considers the potential for the action to reduce the likelihood of recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

The action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of CB DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of CB DPS Atlantic sturgeon. The action will not utilize CB DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The action is likely to result in the mortality of up to 2 subadult CB DPS Atlantic sturgeon; however, as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the CB DPS. As the reduction in numbers and future reproduction is very small, the loss of these individuals will not change the status of CB DPS Atlantic sturgeon. The effects of the action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will cause the mortality of only a very small percentage of the species as a whole and these mortalities are not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the action will not appreciably reduce the likelihood that CB DPS can be brought to the point at which they are no longer listed as endangered or threatened.

Despite the threats faced by individual CB DPS Atlantic sturgeon inside and outside of the action area, the action will not increase the vulnerability of individual sturgeon to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the action. While we are not able to predict with precision how climate change will continue to impact Atlantic sturgeon in the action area or how the species will adapt to climate-change related environmental impacts, we have considered the effects of the action 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 action, resulting in the entrainment and mortality of up to 2 subadult CB DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

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. Spawning occurs in multiple rivers (e.g., Altamaha River) in the SA DPS but spawning populations have been extirpated in some river in the SA DPS. There is no published population estimate for the DPS or total estimate for any river within the DPS. We have estimated, based on fishery-dependent data, that there are approximately 1,170 subadults in the SA DPS. SA DPS 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, for any spawning population or for the DPS as a whole.

NMFS has estimated that the SRIPP will result in the possible mortality of up to 2 subadult Atlantic sturgeon, of which, one may be a SA DPS Atlantic sturgeon. The following analysis applies to the worst case scenario of all the sturgeon coming from the SA DPS. In addition, as described above, the total population size of the SA DPS is unknown at this time; however, in the absence of an estimate of the overall SA DPS population, NMFS has provided a subadult population estimate for the SA DPS (see above). This represents the best available information on subadult population numbers for the SA DPS and will therefore, allow us to consider the loss of these individuals against the life stage for which we have an estimated population size.

The mortality of up to 2 subadult Atlantic sturgeon from the SA DPS subadult population represents a very small percentage of the subadult population (i.e., less than 0.17% of the population). While the death of up to 2 subadult Atlantic sturgeon will reduce the number of SA DPS Atlantic sturgeon compared to the number that would have been present absent the action, it is not likely that this reduction in numbers will change the status of this species as this loss represents a very small percentage of the subadult (less than 0.17%) population). In addition, as described above, subadults, based on their smaller size (i.e., 40-150 cm), are more likely to be entrained than full sized adults (>150 cm). As such, the reproductive potential of the SA DPS is not expected to be significantly affected in any way other than through a reduction in numbers of individuals. A reduction in the number of SA DPS Atlantic sturgeon 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 a small reduction in the number of eggs laid or larvae produced in future years and similarly, a very 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 action, any effect to future year classes is anticipated to be very small and would not change the status of this species. Additionally, as the action will occur outside of the rivers where SA DPS fish are expected to spawn (i.e., Altamaha River), the action will not affect their spawning habitat in any way and will not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds.

The action is not likely to reduce distribution because the action 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 temporal and geographic scale of the SRIPP.

Based on the information provided above, the death of up to 2 subadult SA DPS Atlantic sturgeon over the life of the SRIPP 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 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, including reproduction, sustenance, and shelter. This is the case because: (1) the death of up to 2 subadult SA DPS Atlantic sturgeon represents an extremely small percentage of the species as a whole; (2) the death of up to 2 subadult SA DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these subadult SA DPS Atlantic sturgeon are 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 are 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 action 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 action will have no effect on the ability of SA DPS Atlantic sturgeon to shelter and only an insignificant effect on individual foraging SA DPS Atlantic sturgeon.

In certain instances an action that does not appreciably reduce the likelihood of a species survival (persistence) may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the action will not appreciably reduce the likelihood that SA DPS Atlantic sturgeon will survive in the wild. Here, NMFS considers the potential for the action to reduce the likelihood of recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

The action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of SA DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of SA DPS Atlantic sturgeon. The action will not utilize SA DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The action is likely to result in the mortality of up to 2 subadult SA DPS Atlantic sturgeon; however, as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the SA DPS. As the reduction in numbers and future reproduction is very small, the loss of these individuals will not change the status of SA DPS Atlantic sturgeon. The effects of the action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will cause the mortality of only a very small percentage of the species as a whole and these mortalities are not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the action will not appreciably reduce the likelihood that SA DPS can be brought to the point at which they are no longer listed as endangered or threatened.

Despite the threats faced by individual SA DPS Atlantic sturgeon inside and outside of the action area, the action will not increase the vulnerability of individual sturgeon to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the action. While we are not able to predict with precision how climate change will continue to impact Atlantic sturgeon in the action area or how the species will adapt to climate-change related environmental impacts, we have considered the effects of the action 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 action, resulting in the entrainment and mortality of up to 2 subadult SA DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

Carolina DPS

Individuals originating from the Carolina DPS are likely to occur in the action area. The Carolina DPS has been listed as endangered. Spawning occurs in multiple rivers in the Carolina DPS but spawning populations have been extirpated in some river in the Carolina DPS. There is no published population estimate for the DPS or total estimate for any river within the DPS. We have estimated, based on fishery-dependent data, that there are approximately 972 subadults in the Carolina DPS. Carolina DPS 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, for any spawning population or for the DPS as a whole.

NMFS has estimated that the SRIPP will result in the possible mortality of up to 2 subadult Atlantic sturgeon, of which, one may be a Carolina DPS Atlantic sturgeon. The following analysis applies to the worst case scenario of all the sturgeon coming from the Carolina DPS. In addition, as described above, the total population size of the Carolina DPS is unknown at this time; however, in the absence of an estimate of the overall Carolina DPS population, NMFS has provided a subadult population estimate for the Carolina DPS (see above). This represents the best available information on subadult population numbers for the Carolina DPS and will therefore, allow us to consider the loss of these individuals against the life stage for which we have an estimated population size.

The mortality of up to 2 subadult Atlantic sturgeon from the Carolina DPS subadult population represents a very small percentage of the subadult population (i.e., less than 0.21% of the population). While the death of up to 2 subadult Atlantic sturgeon will reduce the number of Carolina DPS Atlantic sturgeon compared to the number that would have been present absent the action, it is not likely that this reduction in numbers will change the status of this species as this loss represents a very small percentage of the subadult (less than 0.21%) population). In addition, as described above, subadults, based on their smaller size (i.e., 40-150 cm), are more likely to be entrained than full sized adults (>150 cm). As such, the reproductive potential of the SA DPS is not expected to be significantly affected in any way other than through a reduction in numbers of individuals. A reduction in the number of Carolina DPS Atlantic sturgeon would have the effect of reducing the amount of potential reproduction as any dead Carolina DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in a small reduction in the number of eggs laid or larvae produced in future years and similarly, a very 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 action, any effect to future year classes is anticipated to be very small and would not change the status of this species. Additionally, as the action will occur outside of the rivers where Carolina DPS fish are expected to spawn, the action will not affect their spawning habitat in any way and will not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds.

The action is not likely to reduce distribution because the action will not impede Carolina 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 temporal and geographic scale of the SRIPP.

Based on the information provided above, the death of up to 2 subadult Carolina DPS Atlantic sturgeon over the life of the SRIPP 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 Carolina 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, including reproduction, sustenance, and shelter. This is the case because: (1) the death of up to 2 subadult Carolina DPS Atlantic sturgeon represents an extremely small percentage of the species as a whole; (2) the death of up to 2 subadult Carolina DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these subadult Carolina DPS Atlantic sturgeon are 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 are 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 action will have only a minor and temporary effect on the distribution of Carolina DPS Atlantic sturgeon in the action area and no effect on the distribution of the species throughout its range; and, (6) the action will have no effect on the ability of SA DPS Atlantic sturgeon to shelter and only an insignificant effect on individual foraging SA DPS Atlantic sturgeon.

In certain instances an action that does not appreciably reduce the likelihood of a species survival (persistence) may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the action will not appreciably reduce the likelihood that Carolina DPS Atlantic sturgeon will survive in the wild. Here, NMFS considers the potential for the action to reduce the likelihood of recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate. Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., “endangered”), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., “threatened”) because of any of the following five listing factors: (1) The present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, (5) other natural or manmade factors affecting its continued existence.

The action is not expected to modify, curtail or destroy the range of the species since it will result in an extremely small reduction in the number of Carolina DPS Atlantic sturgeon in any geographic area and thus, it will not affect the overall distribution of Carolina DPS Atlantic sturgeon. The action will not utilize Carolina DPS Atlantic sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species or affect its continued existence. The action is likely to result in the mortality of up to 2 subadult Carolina DPS Atlantic sturgeon; however, as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the Carolina DPS. As the reduction in numbers and future reproduction is very small, the loss of these individuals will not change the status of Carolina DPS Atlantic sturgeon. The effects of the action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will cause the mortality of only a very small percentage of the species as a whole and these mortalities are not expected to result in the reduction of overall reproductive fitness for the species as a whole. The effects of the action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the action will not appreciably reduce the likelihood that Carolina DPS can be brought to the point at which they are no longer listed as endangered or threatened.

Despite the threats faced by individual Carolina DPS Atlantic sturgeon inside and outside of the action area, the action will not increase the vulnerability of individual sturgeon to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the action. While we are not able to predict with precision how climate change will continue to impact Atlantic sturgeon in the action area or how the species will adapt to climate-change related environmental impacts, we have considered the effects of the action 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 action, resulting in the entrainment and mortality of up to 2 subadult Carolina DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

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 action may adversely affect but is not likely to jeopardize the continued existence of the loggerhead and Kemp's ridley sea turtle or Atlantic sturgeon, 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 and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without special exemption. 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. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

The measures described below are non-discretionary, and must be undertaken so that they become binding conditions for the exemption in section 7(o)(2) to apply. Failure to implement the terms and conditions through enforceable measures may result in a lapse of the protective coverage of section 7(o)(2).

Amount or Extent of Take

The dredging project has the potential to directly affect loggerhead and Kemp's ridley sea turtles or Atlantic sturgeon by entraining these species in the dredge. These interactions are likely to cause injury and/or mortality to the affected sea turtles and sturgeon. Based on the distribution of sea turtles and Atlantic sturgeon in the action area and information available on historic interactions between sea turtles or Atlantic sturgeon and dredging operations, NMFS believes that it is reasonable to expect that no more than 1 sea turtle is likely to be injured or killed for approximately every 1.6 million cy of material removed from the borrow areas. NMFS has estimated that at least 90% of these turtles will be loggerheads. As such, over the course of the project life, NMFS expects that a total of 9 sea turtles will be killed, with no more than 1 being a Kemp's ridley and the remainder being loggerheads. In regards to Atlantic sturgeon, NMFS believes it is reasonable to expect that no more than 1 Atlantic sturgeon is likely to be injured or killed for approximately every 9.4 million cy of material removed from the borrow areas. As such, over the course of the project life, NMFS expects that a total of 2 subadult Atlantic sturgeon will be killed, with the potential that the two sturgeon taken may come from the NYB, CB, GOM, Carolina, or SA DPS. Due to the nature of the injuries expected by entrainment, any entrained Atlantic sturgeon or sea turtle is expected to die.

NMFS also expects that the maintenance dredging may collect an additional unquantifiable number of parts from previously dead sea turtles or Atlantic sturgeon. 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 previously dead sea turtle or Atlantic sturgeon parts (as well as live turtles or Atlantic sturgeon) that may be on the bottom through the high level of suction.

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 the 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 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 have 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 an unquantifiable yet 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. 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, NMFS has determined that the use of draghead deflectors and certain operating guidelines (as outlined below) are necessary and appropriate to minimize the take of sea turtles and likely Atlantic sturgeon during the dredging of the two borrow areas.

In order to effectively monitor the effects of this action, it is necessary to examine the sea turtles or Atlantic sturgeon entrained in the dredge. Monitoring provides information on the characteristics of the sturgeon or turtles encountered and may provide data which will help develop more effective measures to avoid future interactions with listed species. For example, measurement data may reveal that draghead deflectors or trawl gear is most effective for a particular size class of turtle. In addition, data from genetic sampling of dead sturgeon or sea turtles can definitively identify the species of turtle or Atlantic sturgeon DPS. Reasonable and prudent measures and implementing terms and conditions requiring this monitoring are outlined below.

Reasonable and Prudent 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. It should be noted that this Opinion results from the reinitiation of consultation that lead to the July 22, 2010 Opinion. The action agencies have incorporated the reasonable and prudent measures from the 2010 Opinion as well as all associated specifications and requirements for monitoring hopper dredge operations (Appendix B); sea turtle handling and resuscitation (Appendix C); protocols for collecting genetic samples from sea turtles (Appendix D) and Atlantic sturgeon (Appendix E) for genetic analysis; endangered species observer forms (Appendix F); and incident report forms for sea turtle and Atlantic sturgeon takes (Appendix G and Appendix H) as part of this consultation's action's mitigation measures (see pages 7-8).

1. NMFS must be contacted within 3 days prior to commencement of dredging and again within 3 days following completion of the dredging activity. Upon contacting NMFS, NASA shall report to NMFS whether:
    a. 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 which may be present in the action area;
    b. NMFS-approved observer is present on board the vessel for any dredging occurring in the April 1 – November 30 time frame;
    c. NMFS-approved observer is present on board the vessel for any dredging occurring from December 1-March 31 for Atlantic sturgeon.
    d. All dredges are equipped and operated in a manner that provides endangered/threatened species observers with a reasonable opportunity for detecting interactions with listed species and that provides for handling, collection, and resuscitation of turtles injured during project activity. Full cooperation with the endangered/threatened species observer program is essential for compliance with the ITS; and,
    e. Measures are taken to protect any turtles or sturgeon that survive entrainment in the dredge.
2. All Atlantic sturgeon captured must have a fin clip taken for genetic analysis following established NMFS protocols. This sample must be transferred to NMFS.
3. All Atlantic sturgeon that are captured during the project must be scanned for the presence of Passive Integrated Transponder (PIT) tags. Tag numbers must be recorded and reported to NMFS.

4. Any dead sturgeon must be transferred to NMFS or an appropriately permitted research facility NMFS will identify so that a necropsy can be undertaken to attempt to determine the cause of death. Sturgeon should be held in cold storage.

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

6. All sturgeon and turtle captures, injuries or mortalities associated with 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, the NASA, USACE, and BOEM 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. Because NASA is the lead agency for this consultation, the terms and conditions are directed to it, except where noted.

1. To implement RPM #1(a-d), the NASA must contact NMFS ((978)281-9328 or mail: Protected Resources Division, 55 Great Republic Drive, Gloucester, MA 01930)). This correspondence will serve both to alert NMFS of the commencement and cessation of dredging activities, to give NMFS an opportunity to provide NASA with any updated contact information or reporting forms, and to provide NMFS with information of any incidences with listed species.

2. To implement RPM #1(a), hopper dredges must be equipped with the rigid deflector draghead as designed by the ACOE 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.

3. To implement RPM #1(b-c), observer coverage on hopper dredges operating in the borrow areas must be sufficient for 100% monitoring of hopper dredging operations. This monitoring coverage must involve the placement of a NMFS-approved observer on board the dredge for every day that dredging is occurring. While onboard, observers shall provide the required inspection coverage on a rotating basis so that combined monitoring periods represent 100% of total dredging through the project period. The NASA must ensure that USACE dredge operators and/or any dredge contractor adhere to the attached "Monitoring Specifications for Hopper Dredges" with trained NMFS-approved observers, in accordance with the attached "Observer Protocol" and "Observer Criteria"

(Appendix B). No observers can be deployed to the dredge site until USACE has written confirmation from NMFS that they have met the qualifications to be a “NMFS-approved observer” as outlined in Appendix B. If substitute observers are required during dredging operations, USACE must ensure that NMFS approval is obtain before those observers are deployed on dredges.

4. To implement RPM #1(b-c), NASA shall require of the dredge operator that, when the observer is off watch, the cage shall not be opened unless it is clogged. The NASA shall also require that if it is necessary to clean the cage when the observer is off watch, 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 shall be the only one allowed to clean off the overflow screen.

5. To implement RPM #1(c), NASA must ensure that any initial dredge cycles that occur during the months of December through March must have 100% observer coverage for Atlantic sturgeon. After this time period, NASA and NMFS will reconvene and assess whether 100% observer coverage year round is appropriate or whether modifications to observer coverage are necessary.

6. To implement RPM #1(d), if sea turtles are present during dredging or material transport, vessels transiting the area must post a bridge watch/observer, avoid intentional approaches closer than 100 yards when in transit, and reduce speeds to below 4 knots if the bridge watch/observer identifies a listed species in the immediate vicinity of the dredge.

7. To implement RPM #1(d), the NASA must ensure that all contracted personnel involved in operating hopper dredges receive thorough training on measures of dredge operation that will minimize takes of sea turtles. Training shall include measures discussed in Appendix B.

8. To implement RPM #1(e), the procedures for handling live sea turtles must be followed in the unlikely event that a sea turtle survives entrainment in the dredge (Appendix C).

9. To implement RPM#1(e), the NASA, in coordination with the USACE, and contractors as appropriate, must remove, via a net, any sturgeon observed in the hopper/basket of the dredge and if alive, inspected for injuries, placed on board the vessel with a flow through live well, and returned to the ocean away from the project site.

10. To implement RPM #2, the NASA must ensure that fin clips are taken (according to the procedure outlined in Appendix E) of any sturgeon captured during the project and that the fin clips are sent to NMFS for genetic analysis. Fin clips must be taken prior to preservation of other fish parts or whole bodies.

11. To implement RPM #3, all collected sturgeon must be inspected for a PIT tag with an appropriate PIT tag reader. Any tag numbers must be recorded and reported to NMFS.

12. To implement RPM #4, 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 H (sturgeon salvage form) must be completed and submitted to NMFS.

13. To implement RPM #4, if a decomposed Atlantic sturgeon or Atlantic sturgeon body part is entrained during any dredging operations, the NASA must ensure that an incident report is completed and the specimen is photographed. Any sturgeon 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 the NASA anticipates that they will not be counted towards the ITS) must be frozen. The NASA must submit an incident report for the decomposed sturgeon part, as well as photographs, to NMFS within 24 hours of the take (see Appendix B and H) and request concurrence that this take should not be attributed to the Incidental Take Statement. NMFS shall have the final say in determining if the take should count towards the Incidental Take Statement.

14. To implement RPM #5, 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 G must be completed and submitted to NMFS.

15. To implement RPM #5, if a decomposed turtle or turtle part is 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 NASA 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 NASA must ensure that the observer submits the incident report for the decomposed turtle part, as well as photographs, to NMFS within 24 hours of the take (see Appendix B and G) and request concurrence that this take should not be attributed to the Incidental Take Statement. NMFS shall have the final say in determining if the take should count towards the Incidental Take Statement.

16. To implement RPM #6, the NASA 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 NASA should contact Danielle Palmer: by email (danielle.palmer@noaa.gov) or phone (978)-282-8468 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.

17. To implement RPM #6, the NASA must photograph and measure 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 dredge, hopper or scow)

and the corresponding form (Appendix G and/or H) must be completed and submitted to NMFS **within 24 hours** by fax ((978)-281-9394) or e-mail (incidental.take@noaa.gov).

18. To implement RPM #6, any time a take occurs the NASA must immediately contact NMFS to review the situation. At that time, the NASA 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 NASA and the USACE should discuss with NMFS whether any new management measures could be implemented to prevent the total incidental take level from being exceeded and will work with NMFS to determine 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 USACE to report any take in a reasonable amount of time, as well as implement measures to monitor for entrainment during dredging. The NASA 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 by the NASA.

RPM #1 and #6 and Term and Condition #1, #16-18, 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 any incidences of interactions of 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 NASA 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 NASA and NMFS staff.

RPM #1(a) 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.

RPM #1(b)(c) and Terms and Conditions (#3-4) are necessary and appropriate to ensure the proper handling and documentation of any interactions with listed species as well as requiring that these interactions are reported to NMFS in a timely manner with all of the necessary information. This is essential for monitoring the level of incidental take associated with the action. The inclusion of these RPMs and Terms and Conditions is only a minor change as the NASA included observer coverage in the original project description and the increase in coverage (i.e., the addition of the months from December through March) will represent only a small increase in the cost of the project 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 observers.

RPM #1(d) and Terms and Conditions (#6-7), are necessary and appropriate as they will require that dredge operators use best management practices, including slowing down to 4 knots should listed species be observed, that will minimize the likelihood of take. This represents only a minor change as following these procedures should not increase the cost of the dredging operation or result in any delays of reduction of efficiency of the dredging project.

RPM #1(e) and Terms and Conditions (# 8-9), are necessary and appropriate to ensure that any sea turtles or Atlantic sturgeon that survive entrainment in a hopper dredge 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#2-3 and Term and Condition #10-11 are necessary and appropriate to maximize the potential for detection of any affected sturgeon. The taking of fin clips allows NMFS to run genetic analysis to determine the DPS of origin for Atlantic sturgeon. This allows us to determine if the actual level of take has been exceeded. Sampling of fin tissue is used for genetic sampling. This procedure does not harm sturgeon and is common practice in fisheries science. Tissue sampling does not appear to impair the sturgeon's ability to swim and is not thought to have any long-term adverse impact. Checking and tagging fish with PIT tags allows NASA to determine the identity of detected fish and determine if the same fish is detected more than once. PIT tagging is not known to have any adverse impact to fish. NMFS has received no reports of injury or mortality to any sturgeon sampled or tagged in this way. 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 #4 and Term and Condition #12-13, are necessary and appropriate to determine the cause of death of any dead sturgeon observed during the bridge replacement project. 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 #5 and Terms and Condition #14-15, are necessary and appropriate as future analysis may be needed on the dead sea turtle. 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 NASAs 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 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 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 an action on listed species or critical habitat, to help implement recovery plans, or to develop information.

1. When endangered species observers are required on hopper dredges, 100% overflow screening is recommended. While monitoring 100% of the inflow screening is required as a term and condition of this project's Incidental Take Statement, observing 100% of the overflow screening would ensure that any takes of sea turtles or sturgeon are detected and reported.

2. To facilitate future management decisions on listed species occurring in the action area, NASA 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. NASA should support ongoing and/or future research to determine the abundance and distribution of sea turtles and Atlantic sturgeon in offshore Virginia waters.

4. NASA should work with the USACE to investigate, support, and/or develop additional technological solutions to further reduce the potential for sea turtle or Atlantic sturgeon takes in hopper dredges. 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 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. NMFS believes that development and use of effective pre-deflectors could reduce the need for sea turtle relocation trawling.

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. 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 so 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 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. Copies of NMFS genetic sampling protocols for live and dead turtles or Atlantic sturgeon are attached as Appendix D and Appendix E.

7. NASA and 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.

8. When whales are present in the action area, vessels transiting the area should post a bridge watch, avoid intentional approaches closer than 100 yards (or 500 yards in the case of right whales) when in transit, and reduce speeds to below 4 knots.

13.0 REINITIATION OF CONSULTATION

This concludes formal consultation on NASA’s Wallops Island Shoreline Restoration and Infrastructure Protection Program. 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, NASA must immediately request reinitiation of formal consultation.

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APPENDIX A
Map of Action Area

APPENDIX B

MONITORING SPECIFICATIONS FOR HOPPER DREDGES

I. EQUIPMENT SPECIFICATIONS

A. Baskets or screening

Baskets or screening must be installed over the hopper inflows with openings no smaller than 4 inches by 4 inches to provide 100% coverage of all dredged material and shall remain in place during all dredging operations of any calendar year. Baskets/screening will allow for better monitoring by observers of the dredged material intake for sea turtles and their remains. The baskets or screening must be safely accessible to the observer and designed for efficient cleaning.

B. 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; and
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.

C. Floodlights

Floodlights must be installed to allow the NMFS-approved observer to safely observe and monitor the baskets or screens.

D. Intervals between dredging

Sufficient time must be allotted between each dredging cycle for the NMFS-approved observer to inspect and thoroughly clean the baskets and screens for sea turtles and/or turtle parts and document the findings. Between each dredging cycle, the NMFS-approved observer should also examine and clean the dragheads and document the findings.

II. OBSERVER PROTOCOL

A. Basic Requirement

A NMFS-approved observer with demonstrated ability to identify sea turtlespecies 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 and/or parts being entrained or present in the vicinity of dredge operations.

B. Duty Cycle

NMFS-approved observers are to be onboard for every week of the dredging project until project completion. While onboard, observers shall provide the required inspection coverage on a rotating basis so that combined monitoring periods represent 100% of total dredging through the project period.

C. Inspection of Dredge Spoils

During the required inspection coverage, the trained NMFS-approved observer shall inspect the galvanized screens and baskets at the completion of each loading cycle for evidence of sea turtles or Atlantic sturgeon. The Endangered Species Observation Form shall be completed for each loading cycle, whether listed species are present or not (Appendix F). If any whole (alive or dead) sea turtles or Atlantic sturgeon, or turtle or sturgeon parts are taken incidental to the project(s), the NMFS Section 7 Coordinator (978-281-9328) must be contacted within 24 hours of the take. An incident report for sea turtle and/or Atlantic sturgeon take (Appendix G and Appendix H) shall also be completed by the observer and sent to Danielle Palmer via FAX (978) 281-9394 within 24 hours of the take. Incident reports shall be completed for every take regardless of the state of decomposition. NMFS will determine if the take should be attributed to the incidental take level, after the incident report is received. Every incidental take (alive or dead, decomposed or fresh) should be photographed, and photographs shall be sent to NMFS either electronically (danielle.palmer@noaa.gov) or through the mail. Weekly reports, including all completed load sheets, photographs, and relevant incident reports, as well as a final report, shall be submitted to NMFS NER, Protected Resources Division, 55 Great Republic Drive, Gloucester, MA 01930-2298.

D. Information to be Collected

For each sighting of any endangered or threatened marine species, record the following information on the Endangered Species Observation Form (Appendix F):

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

E. 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 or Mark Murray-Brown (978) 281-9306 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 G or Appendix H). 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 NASAs 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 C). No live turtles should be released back into the water without first being checked by a qualified veterinarian or a rehabilitation facility.

III. OBSERVER REQUIREMENTS

Submission of resumes of endangered species observer candidates to NMFS for final approval ensures that the observers placed onboard the dredges are qualified to document takes of endangered and threatened species, to confirm that incidental take levels are not exceeded, and to provide expert advice on ways to avoid impacting endangered and threatened species. NMFS does not offer certificates of approval for observers, but approves observers on a case-by-case basis.

A. Qualifications

Observers must be able to:

1) differentiate between leatherback (*Dermochelys coriacea*), loggerhead *Caretta caretta*), Kemp's ridley *(Lepidochelys kempii*), green (*Chelonia mydas*), and hawksbill *(Eretmochelys imbricata*) turtles and their parts, and shortnose (*Acipenser brevirostrum*) and Atlantic (*Acipenser oxyrinchus oxyrinchus*) sturgeon and their parts;
2) handle live sea turtles and sturgeon and resuscitate and release them according accepted procedures;
3) correctly measure the total length and width of live and whole dead sea turtle and sturgeon species;
4) observe and advise on the appropriate screening of the dredge’s overflow, skimmer funnels, and dragheads; and
5) identify marine mammal species and behaviors.

B. Training

Ideally, the applicant will have educational background in marine biology, general experience aboard dredges, and hands-on field experience with the species of concern. For observer candidates who do not have sufficient experience or educational background to gain immediate approval as endangered species observers, the below observer training is necessary to be considered admissible by NMFS. We can assist the USACE by identifying groups or individuals capable of providing acceptable observer training. Therefore, at a minimum, observer training must include:

1) instruction on how to identify sea turtles and sturgeon and their parts;
2) instruction on appropriate screening on hopper dredges for the monitoring of sea turtles and sturgeon (whole or parts);

3) demonstration of the proper handling of live sea turtles and sturgeon incidentally captured during project operations. Observers may be required to resuscitate sea turtles according to accepted procedures prior to release;

4) instruction on standardized measurement methods for sea turtle and sturgeon lengths and widths; and

5) instruction on how to identify marine mammals; and

6) instruction on dredging operations and procedures, including safety precautions onboard a vessel.

APPENDIX C

Sea Turtle Handling and Resuscitation

It is unlikely that sea turtles will survive entrainment 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 G).

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 foreflippers.
  Keep clear of slashing foreflippers.
- 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.

<u>Live sea turtles within dredge gear</u>
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:

1) Contact the nearest rehabilitation facility to inform them of the incident. If the

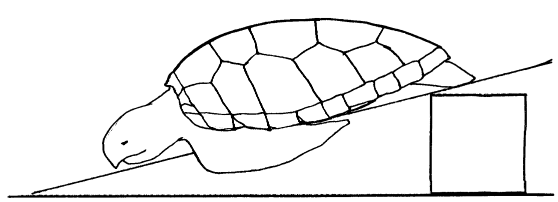
rehabilitation personnel cannot be reached immediately, please contact NMFS stranding hotline at 866-755-6622 or NMFS Sea Turtle Stranding Coordinate at 978-282-8470.

2) 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.
3) 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 (up to 24) should be transported to a suitable facility for necropsy (if the condition of the sea turtle allows).



<u>Dead sea turtles</u>
The procedures for handling dead sea turtles and parts are described in Appendix B-II-E.

<u>Stranding/rehabilitation contacts</u>

- Virginia Marine Science Museum Hotline: (757)-437-6159
- Virginia Aquarium Stranding Program Hotline: (757)-385-7576; General: (757)-385-7575)
- National Aquarium of in Baltimore (**for live animals only**) Hotline: (410)-373-0083)
- NMFS Stranding Hotline at (866)-755-6622

APPENDIX D

Protocol for Collecting Tissue from Sea Turtles for Genetic Analysis

Materials for collecting genetic samples:

- surgical gloves
- alcohol swabs
- betadine swabs
- sterile disposable biopsy punches
- sterile disposable scalpels
- permanent marker to externally label the vials
- scotch tape to protect external labels on the vials
- pencil to write on internal waterproof label
- waterproof label, 1/4" x 4"
- screw-cap vial of saturated NaCl with 20% DMSO*, wrapped in parafilm
- piece of parafilm to wrap the cap of the vial after sample is taken
- vial storage box

* The 20% DMSO buffer within the vials is nontoxic and nonflammable. Handling the buffer without gloves may result in exposure to DMSO. This substance soaks into skin very rapidly and is commonly used to alleviate muscle aches. DMSO will produce a garlic/oyster taste in the mouth along with breath odor. The protocol requires that you wear gloves each time you collect a sample and handle the buffer vials. DO **NOT** store the buffer where it will experience extreme heat. The buffer must be stored at room temperature or cooler, such as in a refrigerator.

Please collect two small pieces of muscle tissue from ***all*** live or dead sea turtles. A muscle sample can be obtained no matter what stage of decomposition a carcass is in. Please utilize the equipment in these kits for genetic sampling of *turtles only* and contact Kate Sampson when you need additional supplies.

Sampling protocol for live turtles:

1. Stabilize the turtle on its plastron. When turtles are placed on their carapace they tend to flap their flippers aggressively and injuries can happen. Exercise caution around the head and jaws.

2. The biopsy location is the dorsal surface of the rear flipper, 5-10 cm from the posterior (trailing) edge and close to the body. Put on a pair of surgical gloves and wipe this area with a Betadine swab.

3. Wipe the hard surface (plastic dive slate, biopsy vial cap or other available clean surface) that will be used under the flipper with an alcohol swab and place this surface underneath the Betadine treated flipper.

4. Using a new (sterile and disposable) plastic skin biopsy punch, gently press the biopsy punch

into the flesh, as close to the posterior edge of the rear flipper as possible. Press down with moderate force and rotate the punch one or two complete turns to make a circular cut all the way through the flipper. The biopsy tool has a sharp cutting edge so exercise caution at all times.

5. Repeat the procedure on the other rear flipper (one sample per rear flipper) with the same biopsy punch so that you now have two samples from this animal.
6. Remove the tissue plugs by knocking them directly from the biopsy punch into a single vial containing 20% DMSO saturated with salt. It is important to ensure that the tissue samples do not come into contact with any other surface or materials during this transfer.

7. Wipe the biopsy area with another Betadine swab.

8. Dispose of the used biopsy punch in a sharps container. It is very important to use a new biopsy punch and gloves for each animal to avoid cross contamination.

Sampling protocol for dead turtles:

1. The best place to obtain the muscle sample is on the ventral side where the front flippers insert near the plastron. It is not necessary to cut very deeply to get muscle tissue.

2. Using a new (sterile and disposable) scalpel cut out two pieces of muscle of a size that will fit in the vial.

3. Transfer both samples directly from the scalpel to a single vial of 20% DMSO saturated with salt.

4. Dispose of the used scalpel in a sharps container. It is very important to use a new scalpel and gloves for each animal to avoid cross contamination.

Labeling of sample vials:

1. Use a pencil to write stranding ID, date, species and SCL on a waterproof label and place it in the vial with the samples.

2. Use a permanent marker to label stranding ID, date, species and SCL on the outside of the vial.

3. Apply a piece of clear scotch tape over the label on the outside of the vial to protect it from being erased or smeared.

4. Wrap Parafilm around the cap of the vial by stretching as you wrap.

5. Place the vial in the vial storage box.

6. Complete the Sea Turtle Biopsy Sample Collection Log (Appendix I).

7. Attach a copy of the STSSN form (Appendix J) to the Collection Log - be sure to indicate on the STSSN form that a genetic sample was taken.

At the end of the calendar year submit all genetic samples to:

Kate Sampson
NOAA/NMFS/NER
Protected Resources Division
55 Great Republic Drive
Gloucester, MA 01930
O: (978) 282-8470
C: (978) 479-9729

APPENDIX E

Procedure for obtaining fin clips from sturgeon for genetic analysis

Obtaining Sample

1. Wash hands and use disposable gloves. Ensure that any knife, scalpel or scissors used for sampling has been thoroughly cleaned and wiped with alcohol to minimize the risk of contamination.

2. For any sturgeon, after the specimen has been measured and photographed, take a one-cm square clip from the pelvic fin.

3. Each fin clip should be placed into a vial of 95% non-denatured ethanol and the vial should be labeled with the species name, date, name of project and the fork length and total length of the fish along with a note identifying the fish to the appropriate observer report. All vials should be sealed with a lid and further secured with tape Please use permanent marker and cover any markings with tape to minimize the chance of smearing or erasure.

Storage of Sample

1. If possible, place the vial on ice for the first 24 hours. If ice is not available, please refrigerate the vial. Send as soon as possible as instructed below.

Sending of Sample

1. Vials should be placed into Ziploc or similar resealable plastic bags. Vials should be then wrapped in bubble wrap or newspaper (to prevent breakage) and sent to:

Julie Carter
NOAA/NOS – Marine Forensics
219 Fort Johnson Road
Charleston, SC 29412-9110
Phone: 843-762-8547

a. Prior to sending the sample, contact Russ Bohl at NMFS Northeast Regional Office (978-282-8493) to report that a sample is being sent and to discuss proper shipping procedures.

APPENDIX F

ENDANGERED SPECIES OBSERVER FORM
Borrow Area Dredging
NASA Wallops Island Project

Daily Report

Date: ____________________________________
Geographic Site:____
Location: Lat/Long _______________________ Vessel Name _______________________________

Weather conditions:____
____
Water temperature: Surface ______________ Below midwater (if known) ______________

Condition of screening apparatus: ____
____
____
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, unusual circumstances, etc:)
____
____
____

Observer's Name: ____
Observer's Signature: ____

Species	# of Sightings	# of Animals	Comments

APPENDIX G

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)
____
____
____

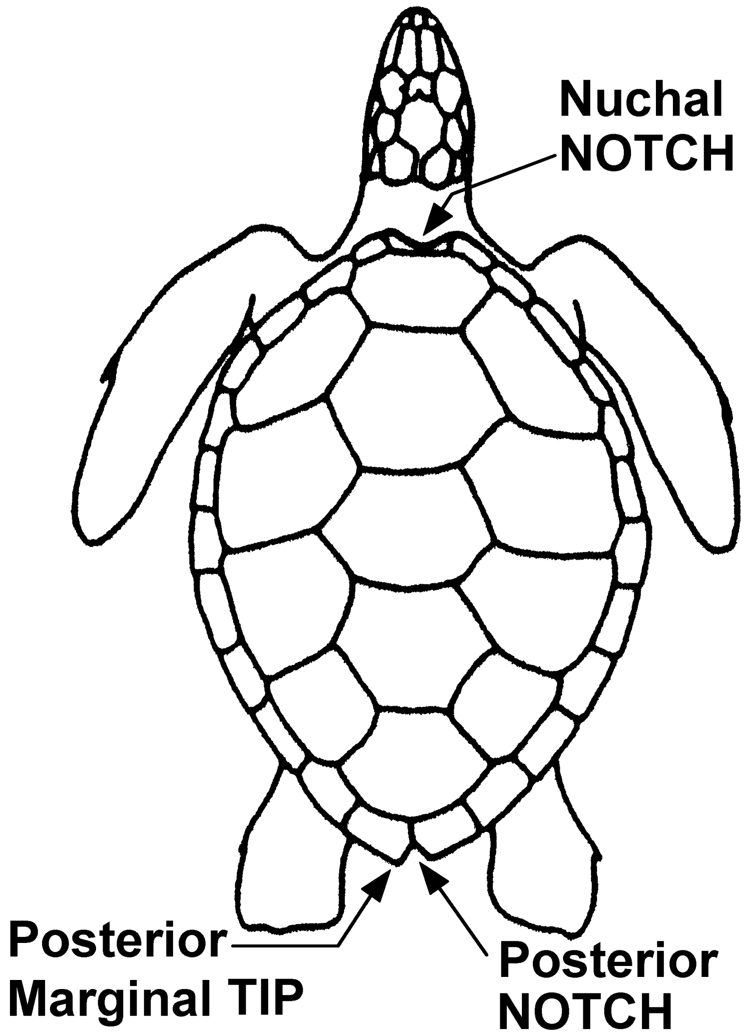
Turtle Decomposed: NO SLIGHTLY MODERATELY SEVERELY

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) ______________________
____
____
Observer's Name ____
Observer’s Signature ____________________________________

APPENDIX G, Continued
Incident Report of Sea Turtle Take

Draw wounds, abnormalities, tag locations on diagram and briefly describe below.



Description of animal:

APPENDIX H

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 SRIPP.

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) _______________

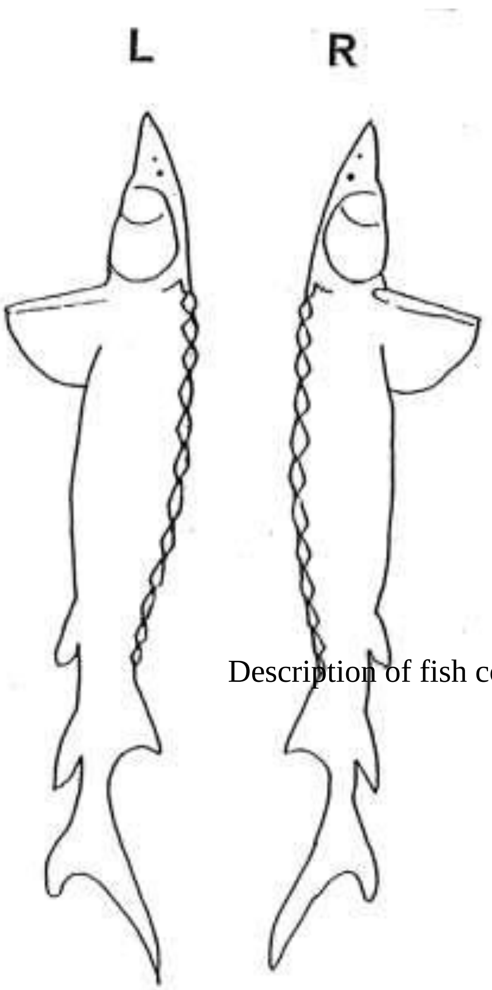
Species Information: *(please designate cm/m or inches.)*
Fork length (or total length) _____________________ Weight _____________________

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)
____
____
Observer's Name ___________________Observer's Signature_____________________

Appendix H, continued

Draw wounds, abnormalities, tag locations on diagram and briefly describe below



Description of fish condition: