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ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
BIOLOGICAL OPINION

Action Agency: U.S. Fish and Wildlife Service

Activity: Funding of State of New Jersey Marine Surveys through the Dingell-Johnson Fish Restoration Grant
F/NER/2012/00151

Date Issued: 5/11/12

Approved by:
D.S. MORRIS
Acting Regional Administrator

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1.0 INTRODUCTION

The U.S. Fish and Wildlife Service (FWS) provides funds to the State of New Jersey (NJ) Department of Environmental Protection, Division of Fish and Wildlife, through the Dingell-Johnson Sport Fish Restoration Grant (grant number F-15-R-52). NJ uses these funds to carry out two studies, an ocean trawling study and a striped bass tagging program. This Opinion is based on information provided by FWS and NJ in March 2012 and other available information. A complete administrative record of this consultation will be kept on file at NMFS Northeast Regional Office.

2.0 CONSULTATION HISTORY

On February 6, 2012, we published two rules listing five Distinct Population Segments (DPS) of Atlantic sturgeon (77 *Federal Register* 5880 and 5914). The effective date of these rules was April 6, 2012. In response to the listing, FWS reviewed State programs that they fund to assess which programs may interact with Atlantic sturgeon. Through this process, the FWS identified programs carried out by the State of NJ that are funded by FWS through the Dingell-Johnson Sport Fish Restoration Program. On March 2, 2012, FWS requested consultation with us to consider effects of this program on Atlantic sturgeon.

3.0 DESCRIPTION OF THE PROPOSED ACTION

FWS provides funds to NJ to carry out five surveys, which are described fully below. FWS provides funds for these studies on a five year cycle. The 5-year period for which these studies are approved runs until May 31, 2014. Thus, this consultation will consider effects of these studies to be carried out from now until May 31, 2014.

3.1 NJ Ocean Trawl Survey

The Ocean Trawl stock assessment program monitors the occurrence, distribution, and relative abundance of fishes inhabiting the nearshore coastal waters of New Jersey and has been ongoing since August 1988 (see Figure 1 for location). The data collected in the Ocean Trawl survey are used in the coastwide stock assessments for summer flounder, winter flounder, striped bass, bluefish, black sea bass, scup, tautog and weakfish. The survey is also used to meet the Atlantic States Marine Fisheries Commission (ASMFC) compliance criteria for the Interstate Fishery Management Plan for winter flounder.

The survey is a random stratified sampling design with a total of five cruises per year. Annually, 186 trawl samples are performed during January (30), April (39), June (39), August (39), October (39). Samples are collected with a three-in-one trawl, so named because all the tapers are three to one. The net is a two-seam trawl with forward netting of 12 cm (4.7 inches) stretch mesh and rear netting of 8 cm (3.0 inches) and is lined with a 6.4 mm (0.25 inch) bar mesh liner. The headrope is 25 m (82 feet) long and the footrope is 30.5 m (100 feet) long. The trawl bridle is 20 fathoms long, the top leg consisting of 0.5 inch wire rope and the bottom leg comprised of 0.75 inch wire rope covered with 2 3/8-inch diameter rubber cookies. A 10-fathom groundwire, also made of 0.75-inch wire rope covered with 2 3/8-inch diameter rubber cookies, extends

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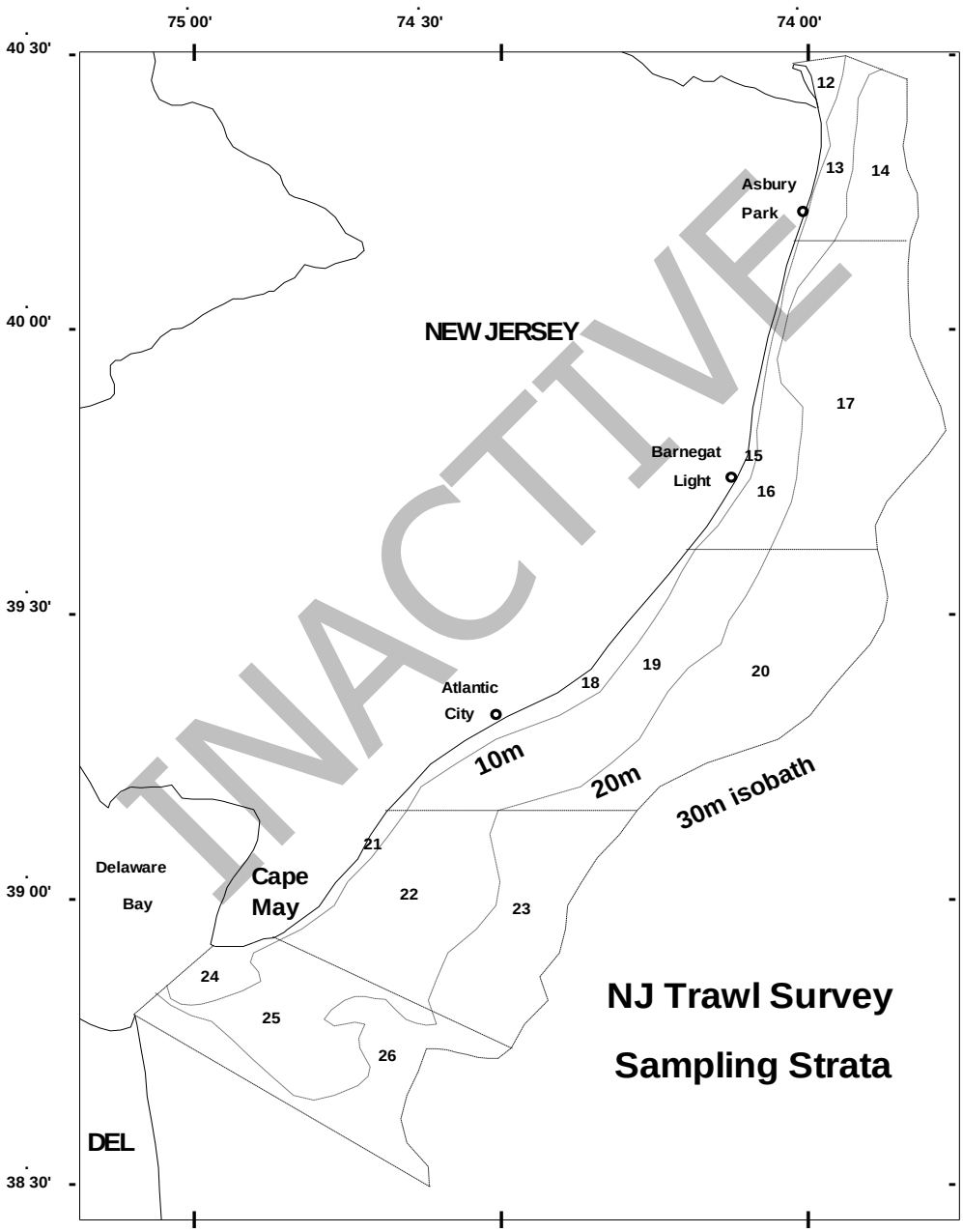
between the bridle and trawl doors. The trawl doors are wood with steel shoes, 8 ft. x 4 ft. 2 in., and weigh approximately 1000 lbs. each.

Figure 1. New Jersey Trawl Survey Sampling Area



Trawl samples are collected by towing the net for 20 minutes (approximately 1 nautical mile), timed from the moment the winch brakes are set to stop the deployment of tow wire to the beginning of haulback. Enough tow wire is released to provide a wire length to depth ratio of at least 3:1, but in shallow (< 10 m) water this ratio is often much greater, in order to provide separation between the vessel and the net. Following haulback, the catch is placed into a 4 x 8-ft. sorting table where fishes and macroinvertebrates are sorted by species into plastic buckets and fish baskets. The depth of tow is contingent on the water depth at the station location. The total weight of each species is measured with metric scales and the length of all individuals comprising each species caught, or a representative sample by weight for large catches, is measured to the nearest cm. Fork or total length, depending on tail shape, is measured for all fishes except stingrays, which have disk width measured instead. For invertebrates, carapace width is measured on crabs, carapace length (in mm) on lobster, and mantle length on squid. Catches containing large numbers of relatively small specimens are often mixed and the mix subsampled by weight. The mix is then sorted and measured and species components later extrapolated, based upon their representation in the subsample, to determine contribution to the total catch.

The survey area consists of New Jersey coastal waters from Ambrose Channel, or the entrance to New York Harbor, south to Cape Henlopen Channel, or the entrance to Delaware Bay, and from about the 3 fathom isobath inshore to approximately the 15 fathom isobath offshore (Figure 1). This area is divided into 15 sampling strata. Latitudinal boundaries are identical to those that define the sampling strata of the National Marine Fisheries Service (NMFS) Northwest Atlantic groundfish survey. Exceptions are those strata at the extreme northern and southern ends of New Jersey. Where NMFS strata extended into New York or Delaware waters, truncated boundaries were drawn which included only waters adjacent to New Jersey, except for the ocean waters off the mouth of Delaware Bay, which were also included.

Longitudinal boundaries consist of the 5, 10, and 15 fathom isobaths. Where these bottom contours were irregular, stratum boundaries were smoothed by eye. As a result, the longitudinal strata boundaries for the New Jersey survey area are similar, but not identical, to the corresponding NMFS boundaries.

Each stratum is divided by grid lines into blocks which represent potential sampling sites; each block is identified by a number assigned sequentially within each stratum. The dimensions of mid-shore (5-10 fathoms) and offshore (10-15 fathoms) blocks are 2.0 minutes longitude by 2.5 minutes latitude; inshore (3-5 fathoms) blocks were 1.0 minutes longitude by 1.0 minutes latitude. Inshore block dimensions were smaller because inshore strata were narrower and of much less area compared to mid- and offshore strata; small block size permits a greater number of potential sampling sites than would be possible with the larger dimensions. This is important for statistical analysis and follows the strategy of NMFS Northeast Fisheries Science Center (NEFSC) for their groundfish survey.

Dimensions of blocks transected by stratum boundaries have less area than described above; blocks reduced in area by more than one-half were generally not assigned a number. Sampling

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sites in 1988-91 were determined by blindly picking disks numbered to correspond to stratum blocks and mixed to assure randomness. In 1992 this method was replaced by using a computer to generate random numbers.

3.2 NJ Striped Bass Tagging Program

In 1989, New Jersey Division of Fish and Wildlife (NJDFW) began collaborating with other agencies by entering the USFWS Cooperative Coastal Striped Bass Tagging Program. Sampling was initiated in areas of lower Delaware Bay near Bidwell's Creek/Reeds Beach, New Jersey where striped bass had been reported as bycatch in the shad gill net fishery (Figure 2). In 1995, this program became a mandatory compliance issue under the ASMFC Atlantic Striped Bass Interstate Fishery Management Plan. Failure to complete this program annually could result in a closure to New Jersey's recreational striped bass fishery.

The program currently utilizes 0.40 mm to 0.47 mm diameter monofilament gill nets, ranging from 5 to 6 inch stretch mesh from early March through early May. Nets are 600 feet in length, 8 to 12 feet in depth and typically set in water depths of 6 to 12 feet. The average soak time in recent years has been 0.5 hours (Table 4). In the mid 1990s, the NJDFW began the switch over from anchored gear to the use of drifting gear resulting in a decrease in average annual soak times since 2000. Usually, only one net is set at a time, and all nets are monitored to diminish potential mortalities to any species.

Although the survey specifically targets striped bass, it has developed into a valuable assessment mechanism for collecting multispecies biological information. All species, especially Atlantic sturgeon and horseshoe crabs, are examined for tags or other markings, while otoliths are collected from bluefish and weakfish. Additional data will be collected from black drum for use in the upcoming ASMFC Fishery Management Plan for that species.

American shad collected during the survey are an essential component in the development of the Delaware River Sustainable Fishing Plan for American Shad. American shad caught in good condition are tagged while a subsample of fish provides scale and otolith samples for age determination and fin clips for genetic analysis.

In 2005, NJDFW began tagging Atlantic sturgeon captured in good condition while targeting striped bass during this program. All sturgeon were processed according to USFWS tagging protocols in the following manner: fork and total length (millimeters) recorded, scanned for pit tags, tagged using dart and pit tags provided by the USFWS, fin clipped and then released alive.

Striped bass in good condition are processed as follows: fork and total lengths (millimeters) recorded, scale samples taken, tagged using internal anchor/external streamer tags provided by the USFWS and then released. A subsample of tagged fish is weighed. In addition, a subsample of fish caught is retained for biological characterization including otolith removal. Basic water quality parameters, net specifications, duration of the sets and other data as outlined by the USFWS are also recorded.

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3.3 Delaware River Juvenile Striped Bass Seine Survey

Since 1980, NJDFW has conducted a juvenile striped bass survey in the Delaware River to provide an annual index of striped bass juvenile abundance. Field sampling utilizes a bagged, 100-foot long by 6-foot deep by ¼-inch mesh beach seine. The seine is set by boat in nearshore waters normally less than six feet in depth and therefore soak times are typically less than ten minutes. All striped bass, as well as other target species, caught are quantified and measured. Basic water quality parameters that include water temperature, salinity and dissolved oxygen are also recorded.

This program was identified in 1989 as an essential tool for the management of the coastwide stocks of striped bass. The ASMFC mandated that the NJDFW continue this program as a compliance criterion. Data collected for American shad was added as a compliance criterion in 1999. As with the striped bass tagging program in Delaware Bay, discontinuation of this program would be costly in regards to New Jersey's recreational fishing industry.

Although the survey specifically targets striped bass, it has always been a valuable tool for collecting multispecies information essential for ASMFC stock assessments and management plans. Annual abundance indices are developed for the following species: American shad (ASMFC compliance), alewife, blueback herring, Atlantic croaker, weakfish, bluefish, spot, American eel, white perch, menhaden and black drum. The survey has provided samples for various species including striped bass, blueback herring, alewife, American shad, hogchoker and bunker for research at various universities across the US and Canada. In addition, this survey is available tool for providing information for use in waterfront development projects and dredging operations.

Although the juvenile survey has been modified throughout the time series the current fixed station format has been followed since 2002. The NJDFW samples 32 stations from mid-June to mid-November for a total of 320 annual hauls (see Figure 3). During the time series, the sampling area has ranged from river mile 44.9 to 129.7. Since 1998, sampling stations are located from Augustine Beach to Newbold Island. Occasionally due to tidal extremes, sediment, or construction, alternate sites are sampled.

The Delaware River recruitment survey area is divided into three distinct habitats:

1) Region I -- brackish, tidal water extending from the springtime saltwater/freshwater interface to the Delaware Memorial Bridges
2) Region II -- brackish to fresh tidal water extending from the Delaware Memorial Bridges to the Schuylkill River at the Philadelphia Naval Yard, and
3) Region III -- tidal freshwater from Philadelphia to the fall line at Trenton

Saltmarsh vegetation predominates along the Region I shoreline while Region II is primarily urban with a shoreline heavily developed for commerce and industry. Region III is sporadically developed by industry with considerable freshwater marsh.

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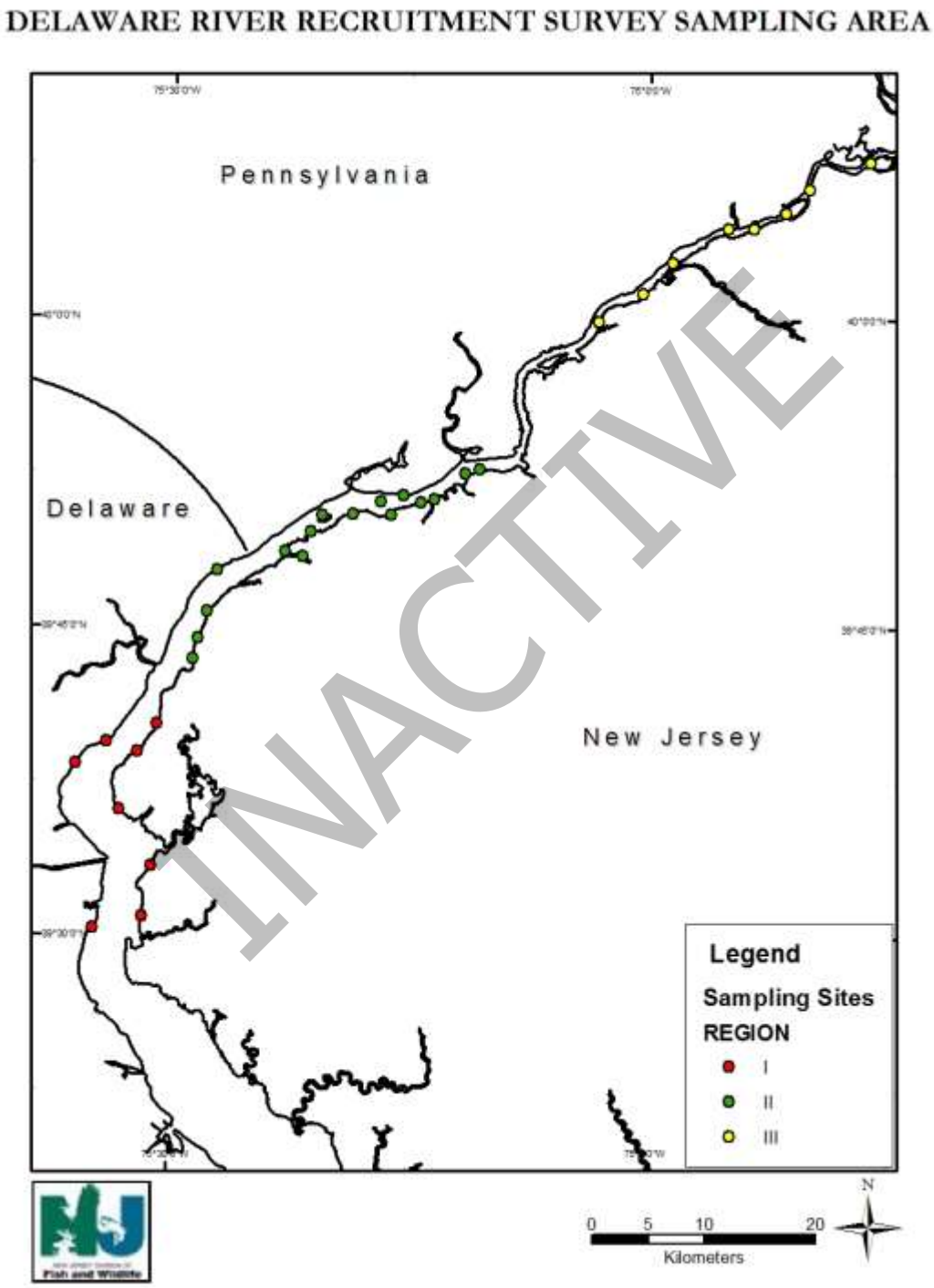
Figure 3. Striped Bass Seine Survey Locations

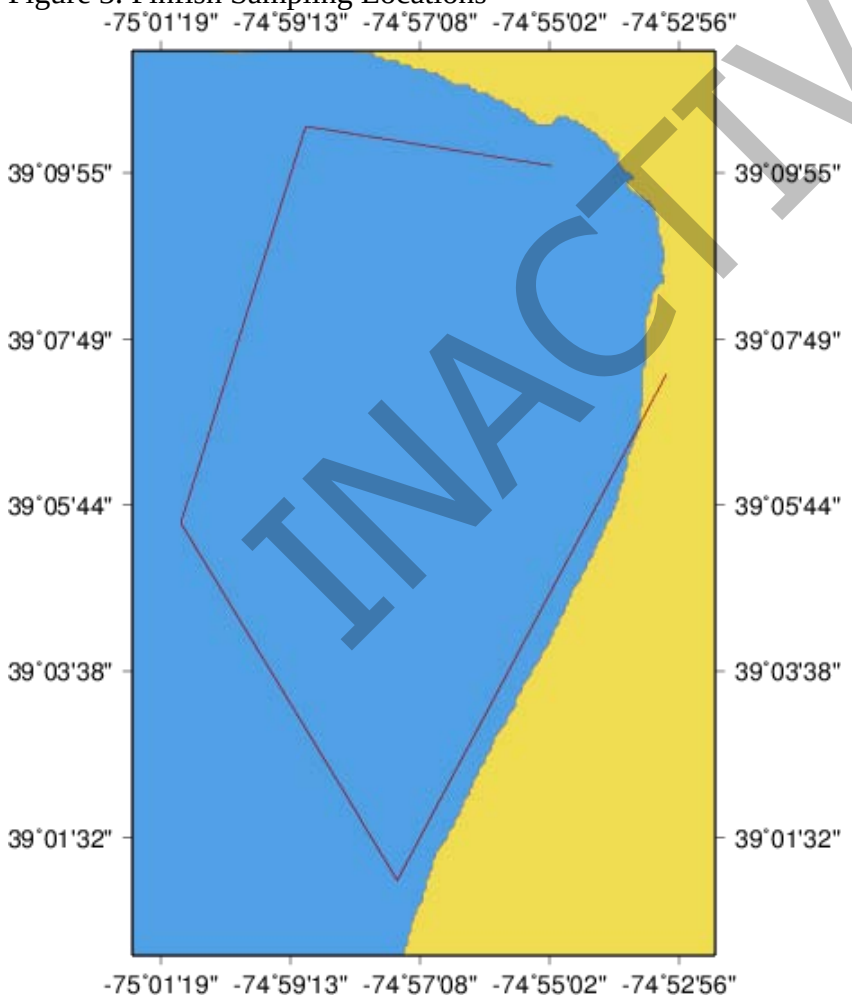


3.4 Relative Abundance of Selected Finfish Species in Delaware Bay

The New Jersey Division of Fish and Wildlife initiated an estuarine finfish sampling program in Delaware Bay in 1991 to identify and develop relative abundance estimates for finfish utilizing this estuary. The estimated year class strength of important finfish and the creation of a time series provide data necessary to assess trends in relative abundance for select species, assess spawning success via juvenile abundance and assess the effects of various management strategies instituted for these species.

Sampling is conducted monthly from April to October at eleven fixed stations on the New Jersey side of Delaware Bay (Figure 3). The area sampled ranges from the Villas in Cape May to the mouth of the Cohansey River. All samples collected during this program are taken on shoals located near shore.

Figure 3. Finfish Sampling Locations



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A 4.9-m (16-foot) otter trawl with a 3.8 cm (1.5 inch) stretch body mesh and 3.2 cm (1.25 inch) stretch mesh in the cod end is used for sampling. The cod end is lined with a 1.3 cm (0.5 inch) knotless stretch mesh net. The headrope is buoyed with several can-shaped molded fish net floats. The bottom of the net's mouth is weighted with a 0.3 cm (0.125 inch) galvanized chain looped along the footrope. The door dimensions are 30.5 cm (12 inches) x 61.0 cm (24 inches) and were constructed of 1.9 cm (0.75 inch) marine plywood with 1.3 cm (0.5 inch) by 5.1 cm (2 inch) steel shoes. The doors are attached to 1.6 cm (0.625 inch) twisted three strand nylon towlines, by a 0.5 cm (0.188 inch) galvanized chain bridle with 1.0 cm (0.375 inch) swivels.

Single ten-minute tows are conducted against the prevailing tide at each station. All stations are sampled once during the second or third week of the month. The engine tow speed is usually set depending on tidal velocity, to maintain a speed-over-ground of approximately 3.9 km/hr (or 2.1 knots). Speed-over-ground, tow distance and depth are monitored using a Garmin 2010 GPS Receiver/Depthfinder. Engine speed is constantly monitored and adjusted during the sampling period to maintain trawl speed. The estimated distance towed (nautical miles) is calculated from the average speed over ground (knots) and multiplying it by the duration (in hours) of each tow (Distance = Speed x Time).

On board the trawl net is manually deployed with 60 feet of towline tied to the stern cleats and retrieved with the towlines being spooled through blocks at the end of a 4.6 m (15 foot) A-frame made of 7.62 cm (3 inch) inside diameter aluminum, marine grade pipe. On retrieval, the A-frame and net are hauled at the transom using a Gearmatic GH5 Hydraulic Winch installed on the mast located aft of the wheelhouse bulkhead. The cod end of the net is manually retrieved and the contents emptied onto a sorting table affixed to the stern of the vessel.

All fish collected are identified to the lowest possible taxonomic level, enumerated and measured to the nearest millimeter. When large numbers (>50) of a single species are taken, fifty individuals were randomly selected and measured. Annual relative abundance (catch per tow = c/t) for all species combined and for each single species are calculated as the total number of individual fish collected over the total number of tows.

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3.5 Action Area

The action area for Section 7 consultations is defined as all of the areas directly or indirectly affected by the Federal action, and not merely the immediate area involved in the action. NMFS anticipates that the only effects on ESA-listed species and their habitat as a result of the survey are the direct effects of interaction between listed species and sampling gear that will be used for the survey, and the effects on other marine organisms (*i.e.*, prey) on or very near the seafloor from the sampling gear. The trawl survey area consists of New Jersey coastal waters from Ambrose Channel, or the entrance to New York Harbor, south to Cape Henlopen Channel, or the entrance to Delaware Bay, and from about the 3 fathom (5.5 m) isobath inshore to approximately the 15 fathom (27 m) isobath offshore (Figure 1). The action area also includes areas in the Delaware River and Delaware Bay where beach seine, gillnet and trawl surveys take place. Therefore, for the purpose of this consultation, the action area for the proposed action is defined by the area in which sampling gear for the project will be operated.

4.0 STATUS OF THE SPECIES

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

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

We have determined that the actions being considered in the Opinion are not likely to adversely affect shortnose sturgeon (*Acipenser brevirostrum*), hawksbill sea turtles (*Eretmochelys imbricata*), North Atlantic right whales (right whales) (*Eubalaena glacialis*), humpback whales (*Megaptera novaeangliae*), fin whales (*Balaenoptera physalus*), sei whales (*Balaenoptera borealis*), blue whales (*Balaenoptera musculus*), and sperm whales (*Physeter macrocephalus*), all of which are listed as endangered species under the ESA. Thus, these species will not be considered further in this Opinion. Below, we present our rationale for these determinations.

4.1.1 Shortnose sturgeon

Shortnose sturgeon are benthic fish that occur in large coastal rivers of eastern North America. They range from as far south as the St. Johns River, Florida (possibly extirpated from this system) to as far north as the Saint John River in New Brunswick, Canada. Shortnose sturgeon occur in 19 rivers along the U.S. Atlantic coast, including the Delaware River and Delaware Bay. Shortnose sturgeon occur within some portions of the action area. NJ has monitored for interactions with shortnose sturgeon since these surveys began. No shortnose sturgeon have been encountered in the ocean trawl survey, striped bass gillnet survey or the finfish survey in Delaware Bay. Based on the duration of these surveys and that there will be no changes in survey protocol in 2012-2014 that would increase the likelihood of an interaction with a shortnose sturgeon, we do not anticipate interactions between shortnose sturgeon and the ocean trawl, striped bass gillnet or finfish survey.

Between 1980 and 2011, 6,278 seine hauls have been conducted with one interaction with a shortnose sturgeon. This interaction occurred with a 115 mm shortnose sturgeon on August 24, 2004 at Oldman's Point. Given the rare nature of this event, the past capture rate (1 fish/6,278 hauls) and the number of hauls to be carried out in the period considered in this Opinion (640) it is extremely unlikely that there will be any interactions between shortnose sturgeon and the beach seine survey. No other effects to shortnose sturgeon are anticipated to result from the proposed action. As such, all effects to shortnose sturgeon will be insignificant and discountable and this species will not be considered further in this Opinion.

4.1.2 Hawksbill sea turtle

The hawksbill sea turtle is listed as endangered. This species is uncommon in the waters of the continental U.S. Hawksbills prefer coral reef habitats, such as those found in the Caribbean and Central America. Mona Island (Puerto Rico) and Buck Island (St. Croix, U.S. Virgin Islands)

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contain especially important foraging and nesting habitat for hawksbills. Within the continental U.S., nesting is restricted to the southeast coast of Florida and the Florida Keys, but nesting is rare in these areas. Hawksbills have been recorded from all the Gulf States and along the east coast of the U.S. as far north as Massachusetts, but sightings north of Florida are rare. Aside from Florida, Texas is the only other U.S. state where hawksbills are sighted with any regularity. Since hawksbill sea turtles are not expected to be present in the areas where the survey will take place, it is highly unlikely that the proposed action will affect this sea turtle species. The lack of any captures of hawksbill sea turtles in the NJ surveys to date supports this determination.

4.1.4 Large Whales

Sperm whales and blue whales are listed as endangered. These species are unlikely to occur in areas where bottom otter trawl gear for the survey will operate. During surveys for the Cetacean and Turtle Assessment Program (CeTAP), sperm whales were observed along the shelf edge, centered around the 1,000 m depth contour but extending seaward out to the 2,000 m depth contour (CeTAP 1982). Although blue whales are occasionally seen in U.S. waters, they are more commonly found in Canadian waters and are rare in continental shelf waters of the eastern U.S. (Waring *et al.* 2000). Given the predominantly offshore distribution of these two cetacean species, both are highly unlikely to occur in the action area or to be affected by the NJ surveys.

North Atlantic right whales, humpback whales, fin whales, and sei whales may occur in the area where the ocean trawl surveys will be conducted; however, given the shallow depths (less than 15 fathoms) at which these surveys operate, occurrence is expected to be rare. None of these species are expected to be affected by the use of bottom otter trawl gear for the survey given the following. While these species may occur in the action area, large cetaceans have the speed and maneuverability to get out of the way of oncoming mobile gear, including trawl gear. The slow speed of the trawl survey (3.1 knots) and the short tow times (20 minutes) further reduce the potential for entanglement or any other interaction. Observations of many fishing trips using mobile gear (*e.g.*, dredge, trawl gear) have shown that entanglement or capture of large whales in these gear types is extremely rare and unlikely. No interactions with any species of whale has occurred during the NJ surveys or any similar surveys (*i.e.*, the NMFS Northeast Fisheries Science Center (NEFSC) spring and fall bottom trawl surveys or the Virginia Institute of Marine Science's (VIMS) Northeast Area Monitoring and Assessment Program (NEAMAP) Near Shore Trawl Program. Because of this, we have determined that it is extremely unlikely that any large whale would interact with the trawl gear during 2012.

We have also determined that in-water work for the survey will not have any adverse effects on cetacean prey. Right and sei whales feed on copepods (Horwood 2002; Kenney 2002). The use of trawl gear for the proposed project will not affect the availability of copepods for foraging right and sei whales. This is because copepods are very small organisms that will pass through the gear rather than being captured in it. Blue whales feed on euphausiids (krill) (Sears 2002) which, likewise, are too small to be captured in the gear. Humpback and fin whales also feed on krill as well as small schooling fish (*e.g.*, sand lance, herring, and mackerel) found within the water column (Aguilar 2002; Clapham 2002). The trawl gear used for the survey will operate on or very near the bottom. Therefore, the fish species caught in such gear would be species that

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live in benthic habitats (on or very near the bottom) such as flounders and other groundfish. Schooling fish such as herring and mackerel that occur within the water column are unlikely to be captured with this gear. Sperm whales feed on larger organisms that inhabit the deeper ocean regions (Whitehead 2002) outside of the action area. Based on this analysis, it is extremely unlikely that the trawl surveys will affect the availability of prey for any whale species.

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

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

Common name	Scientific name	ESA Status
Northwest Atlantic DPS of loggerhead sea turtle	*Caretta caretta*	Threatened
Kemp's ridley sea turtle	*Lepidochelys kempii*	Endangered
Green sea turtle	*Chelonia mydas*	Endangered[1]
Leatherback sea turtle	*Dermochelys coriacea*	Endangered
GOM DPS of Atlantic sturgeon	*Acipenser oxyrinchus oxyrinchus*	Threatened
New York Bight DPS of Atlantic sturgeon	*Acipenser oxyrinchus oxyrinchus*	Endangered
Chesapeake Bay DPS of Atlantic sturgeon	*Acipenser oxyrinchus oxyrinchus*	Endangered
South Atlantic DPS of Atlantic sturgeon	*Acipenser oxyrinchus oxyrinchus*	Endangered

4.2.1 Status of Sea Turtles

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

2010 BP Deepwater Horizon Oil Spill

[1] Green sea turtles in U.S. waters are listed as threatened except for the Florida breeding population, which is listed as endangered. Due to the inability to distinguish between these populations away from the nesting beach, green sea turtles are considered endangered wherever they occur in U.S. waters

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The April 20, 2010, explosion of the Deepwater Horizon oil rig affected sea turtles in the Gulf of Mexico. There is an on-going assessment of the long-term effects of the spill on Gulf of Mexico marine life, including sea turtle populations. Following the spill, juvenile Kemp's ridley, green, and loggerhead sea turtles were found in *Sargassum* algae mats in the convergence zones, where currents meet and oil collected. Sea turtles found in these areas were often coated in oil and/or had ingested oil. Approximately 536 live adult and juvenile sea turtles were recovered from the Gulf and brought into rehabilitation centers; of these, 456 were visibly oiled (these and the following numbers were obtained from http://www.nmfs.noaa.gov/pr/health/oilspill/). To date, 469 of the live recovered sea turtles have been successfully returned to the wild, 25 died during rehabilitation, and 42 are still in care but will hopefully be returned to the wild eventually. During the clean-up period, 613 dead sea turtles were recovered in coastal waters or on beaches in Mississippi, Alabama, Louisiana, and the Florida Panhandle. As of February 2011, 478 of these dead turtles had been examined. Many of the examined sea turtles showed indications that they had died as a result of interactions with trawl gear, most likely used in the shrimp fishery, and not as a result of exposure to or ingestion of oil.

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

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

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4.2.1.1. Northwest Atlantic DPS of loggerhead sea turtle

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

Listing History

Loggerhead sea turtles were listed as threatened throughout their global range on July 28, 1978. Since that time, several status reviews have been conducted to review the status of the species and make recommendations regarding its ESA listing status. Based on a 2007 5-year status review of the species, which discussed a variety of threats to loggerheads including climate change, NMFS and FWS determined that loggerhead sea turtles should not be delisted or reclassified as endangered. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPSs should be identified for the loggerhead (NMFS and USFWS 2007a). Genetic differences exist between loggerhead sea turtles that nest and forage in the different ocean basins (Bowen 2003; Bowen and Karl 2007).

Differences in the maternally inherited mitochondrial DNA also exist between loggerhead nesting groups that occur within the same ocean basin (TEWG 2000; Pearce 2001; Bowen 2003; Bowen *et al.*2005; Shamblin 2007; TEWG 2009; NMFS and USFWS 2008). Site fidelity of females to one or more nesting beaches in an area is believed to account for these genetic differences (TEWG 2000; Bowen 2003).

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

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

On March 16, 2010, NMFS and USFWS published a proposed rule (75 FR 12598) to divide the worldwide population of loggerhead sea turtles into nine DPSs, as described in the 2009 Status Review. Two of the DPSs were proposed to be listed as threatened and seven of the DPSs, including the Northwest Atlantic Ocean DPS, were proposed to be listed as endangered. NMFS and the USFWS accepted comments on the proposed rule through September 13, 2010 (75 FR 30769, June 2, 2010). On March 22, 2011 (76 FR 15932), NMFS and USFWS extended the date by which a final determination 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.

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

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

Presence of Loggerhead Sea Turtles in the Action Area

The effects of this proposed action are only experienced within the Atlantic Ocean. NMFS has considered the available information on the distribution of the 9 DPSs to determine the origin of any loggerhead sea turtles that may occur in the action area. As noted in Conant *et al.*(2009), the range of the four DPSs occurring in the Atlantic Ocean are as follows: NWA DPS – north of the equator, south of 60° N latitude, and west of 40° W longitude; Northeast Atlantic Ocean (NEA) DPS – north of the equator, south of 60° N latitude, east of 40° W longitude, and west of 5° 36' W longitude; South Atlantic DPS – south of the equator, north of 60° S latitude, west of 20° E longitude, and east of 60° W longitude; Mediterranean DPS – the Mediterranean Sea east of 5° 36' W longitude. These boundaries were determined based on oceanographic features, loggerhead sightings, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. While adults are highly structured with no overlap, there may be some degree of overlap by juveniles of the NWA, NEA, and Mediterranean DPSs on oceanic foraging grounds (Laurent *et al.*1993, 1998; Bolten *et al.*1998; LaCasella *et al.*2005; Carreras *et al.*2006, Monzón-Argüello *et al.*2006; Revelles *et al.*2007). Previous literature (Bowen *et al.*2004) has suggested that there is the potential, albeit small, for some juveniles from the Mediterranean DPS to be present in U.S. Atlantic coastal foraging grounds. These conclusions must be interpreted with caution however, as they may be representing a shared common haplotype and lack of representative sampling at Eastern Atlantic rookeries rather than an actual presence of Mediterranean DPS turtles in US Atlantic coastal waters. A re-analysis of the data by the Atlantic loggerhead Turtle Expert Working Group has

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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

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coastal waters to waters offshore of North Carolina, particularly off of Cape Hatteras, and waters further south where the influence of the Gulf Stream provides temperatures favorable to sea turtles (Shoop and Kenney 1992; Epperly *et al.*1995b).

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

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

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

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Table 1. Key Life History Parameters for Loggerheads

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).

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Population Dynamics and Status

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

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

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

The Recovery Team evaluated the status and trends of the Northwest Atlantic loggerhead

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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

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nesting annually. The 2008 recovery plan compiled information on mean number of loggerhead nests and the approximated counts of nesting females per year for four of the five identified recovery units (*i.e.*, nesting groups). They are: (1) for the NRU, a mean of 5,215 loggerhead nests per year (from 1989-2008) with approximately 1,272 females nesting per year; (2) for the PFRU, a mean of 64,513 nests per year (from 1989-2007) with approximately 15,735 females nesting per year; (3) for the DTRU, a mean of 246 nests per year (from 1995-2004, excluding 2002) with approximately 60 females nesting per year; and (4) for the NGMRU, a mean of 906 nests per year (from 1995-2007) with approximately 221 females nesting per year. For the GCRU, the only estimate available for the number of loggerhead nests per year is from Quintana Roo, Yucatán, Mexico, where a range of 903-2,331 nests per year was estimated from 1987-2001 (NMFS and USFWS 2007a). There are no annual nest estimates available for the Yucatán since 2001 or for any other regions in the GCRU, nor are there any estimates of the number of nesting females per year for any nesting assemblage in this recovery unit. Note that the above values for average nesting females per year were based upon 4.1 nests per female per Murphy and Hopkins (1984).

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

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

Maier *et al.*(2004) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the southeast coast of the United States (Winyah Bay, South Carolina to St. Augustine, Florida) during the period 2000-2003. A comparison of loggerhead catch data

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

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

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

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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.

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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.

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

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

Loggerhead sea turtles are also known to interact with non-shrimp trawl, gillnet, longline, dredge, pound net, pot/trap, and hook and line fisheries. The NRC (1990) report stated that other U.S. Atlantic fisheries collectively accounted for 500 to 5,000 loggerhead deaths each year, but recognized that there was considerable uncertainty in the estimate. The reduction of sea turtle captures in fishing operations is identified in recovery plans and 5-year status reviews as a priority for the recovery of all sea turtle species. In the threats analysis of the loggerhead recovery plan, trawl bycatch is identified as the greatest source of mortality. While loggerhead bycatch in U.S. Mid-Atlantic bottom otter trawl gear was previously estimated for the period 1996-2004 (Murray 2006, 2008), a recent bycatch analysis estimated the number of loggerhead sea turtle interactions with U.S. Mid-Atlantic bottom trawl gear from 2005-2008 (Warden 2011a). Northeast Fisheries Observer Program data from 1994-2008 were used to develop a model of interaction rates and those predicted rates were applied to 2005-2008 commercial fishing data to estimate the number of interactions for the trawl fleet. The number of predicted average annual loggerhead interactions for 2005-2008 was 292 (CV=0.13, 95% CI=221-369), with an additional 61 loggerheads (CV=0.17, 95% CI=41-83) interacting with trawls but being

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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

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historical averages since 2001 (Garrison and Stokes 2010). This fishery represents just one of several longline fisheries operating in the Atlantic Ocean. Lewison *et al.*(2004) estimated that 150,000-200,000 loggerheads were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries as well as others).

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

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

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

Climate change has the potential to result in changes at nesting beaches that may affect loggerhead sex ratios. Loggerhead sea turtles exhibit temperature-dependent sex determination. Rapidly increasing global temperatures may result in warmer incubation temperatures and highly female-biased sex ratios (*e.g.*, Glen and Mrosovsky 2004; Hawkes *et al.*2009); however, to the extent that nesting can occur at beaches further north where sand temperatures are not as warm, these effects may be partially offset. The BRT specifically identified climate change as a threat

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to loggerhead sea turtles in the neritic/oceanic zone where climate change may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution. In the threats matrix analysis, climate change was considered for oceanic juveniles and adults and eggs/hatchlings. The report states that for oceanic juveniles and adults, "although the effect of trophic level change from…climate change…is unknown it is believed to be very low." For eggs/hatchlings the report states that total mortality from anthropogenic causes, including sea level rise resulting from climate change, is believed to be low relative to the entire life stage. However, only limited data are available on past trends related to climate effects on loggerhead sea turtles; current scientific methods are not able to reliably predict the future magnitude of climate change, associated impacts, whether and to what extent some impacts will offset others, or the adaptive capacity of this species.

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

While there is a reasonable degree of certainty that certain climate change related effects will be experienced globally (*e.g.*, rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects to sea turtles resulting from climate change are not predictable or quantifiable at this time (Hawkes *et al.*2009). However, given this uncertainty and the likely rate of change associated with climate impacts (*i.e.*, the century scale), it is unlikely that climate related impacts will have a significant effect on the status of loggerhead sea turtles over the temporal scale of the proposed action (*i.e.*, through 2012).

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

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recovery units for loggerheads in the Northwest Atlantic, including the PFRU, which is the largest (in terms of number of nests laid) in the Atlantic Ocean. The nesting trends for the other two recovery units could not be determined due to an absence of long term data.

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

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

4.2.1.2 Status of 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).

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

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

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

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

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

Population Dynamics and Status

The majority of Kemp's ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; NMFS and USFWS 2007c; NMFS *et al.*2011). There is a limited amount of scattered nesting to the north and south of the primary nesting beach (NMFS

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

Threats

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

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

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

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

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

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features that are relevant to Kemp's ridleys, as well as change rain regimes and levels of nearshore runoff.

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

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

As with the other sea turtle species discussed in this section, while there is a reasonable degree of certainty that certain climate change related effects will be experienced globally (*e.g.*, rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects of climate change on this species are not predictable or quantifiable at this time (Hawkes *et al.*2009). However, given the likely rate of change associated with climate impacts (*i.e.*, the century scale), it is unlikely that climate change will have a significant effect on the status of Kemp's ridley sea turtles over the temporal scale of the proposed action (*i.e.*, through 2012).

Summary of Status for Kemp's Ridley Sea Turtles

The majority of Kemp's ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; NMFS and USFWS 2007c; NMFS *et al.*2011). The number of nesting females in the Kemp's ridley population declined dramatically from the late 1940s through the mid-1980s, with an estimated 40,000 nesting females in a single *arribada* in 1947 and fewer than 300 nesting females in the entire 1985 nesting season (TEWG 2000; NMFS *et al.*2011). However, the total annual number of nests at Rancho Nuevo gradually began to increase in the 1990s (NMFS and USFWS 2007c). Based on the number of nests laid in 2006

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and the remigration interval for Kemp's ridley sea turtles (1.8-2 years), there were an estimated 7,000-8,000 adult female Kemp's ridley sea turtles in 2006 (NMFS and USFWS 2007c). The number of adult males in the population is unknown, but sex ratios of hatchlings and immature Kemp's ridleys suggest that the population is female-biased, suggesting that the number of adult males is less than the number of adult females (NMFS and USFWS 2007c). While there is cautious optimism for recovery, events such as the Deepwater Horizon oil release, and stranding events associated increased skimmer trawl use and poor TED compliance in the northern Gulf of Mexico may dampen recent population growth.

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

4.2.1.3 Status of Green Sea Turtles

Green sea turtles are distributed circumglobally, and can be found in the Pacific, Indian, and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 1991, 2007d; Seminoff 2004). In 1978, the Atlantic population of the green sea turtle was listed as threatened under the ESA, except for the breeding populations in Florida and on the Pacific coast of Mexico, which were listed as endangered. As it is difficult to differentiate between breeding populations away from the nesting beaches, all green sea turtles in the water are considered endangered.

Pacific Ocean

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

Historically, green sea turtles were used in many areas of the Pacific for food. They were also commercially exploited, which, coupled with habitat degradation, led to their decline in the Pacific (NMFS and USFWS 1998b). Green sea turtles in the Pacific continue to be affected by

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poaching, habitat loss or degradation, fishing gear interactions, and fibropapillomatosis, which is a viral disease that causes tumors in affected turtles (NMFS and USFWS 1998b; NMFS 2004b).

Indian Ocean

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

Mediterranean Sea

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

Atlantic Ocean

Distribution and Life History

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

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

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

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

Population Dynamics and Status

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

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

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

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

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

Threats

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

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

Finkbeiner *et al.*(2011) compiled cumulative sea turtle bycatch information in U.S. fisheries from 1990 through 2007, before and after implementation of bycatch mitigation measures. Information was obtained from peer reviewed publications and NMFS documents (e.g., Biological Opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, occurred annually (since implementation of bycatch mitigation measures). Kemp's ridleys interacted with fisheries most frequently, with the

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highest level of mean annual mortality (2,700), followed by loggerheads (1,400), greens (300), and leatherbacks (40). The Southeast/Gulf of Mexico shrimp trawl fishery was responsible for the vast majority of U.S. interactions (up to 98%) and mortalities (more than 80%). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations.

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

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

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

Summary of Status of Green Sea Turtles

A review of 32 Index Sites[2] distributed globally revealed a 48-67% decline in the number of mature females nesting annually over the last three generations[3] (Seminoff 2004). An evaluation

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

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

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

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

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

4.2.1.4 Status of Leatherback Sea Turtles

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

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

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

Pacific Ocean

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

The largest, extant leatherback nesting group in the Indo-Pacific lies on the North Vogelkop coast of West Papua, Indonesia, with 3,000-5,000 nests reported annually in the 1990s (Suárez *et al.*2000). However, in 1999, local villagers started reporting dramatic declines in sea turtles near their villages (Suárez 1999). Declines in nesting groups have been reported throughout the western Pacific region where observers report that nesting groups are well below abundance levels that were observed several decades ago (*e.g.*, Suárez 1999).

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

In the eastern Pacific Ocean, major leatherback nesting beaches are located in Mexico and Costa Rica, where nest numbers have been declining. According to reports from the late 1970s and early 1980s, beaches located on the Mexican Pacific coasts of Michoacán, Guerrero, and Oaxaca sustained a large portion, perhaps 50%, of all global nesting by leatherbacks (Sarti *et al.*1996). A dramatic decline has been seen on nesting beaches in Pacific Mexico, where aerial survey data was used to estimate that tens of thousands of leatherback nests were laid on the beaches in the 1980s (Pritchard 1982), but a total of only 120 nests on the four primary index beaches (combined) were counted in the 2003-2004 season (Sarti Martinez *et al.*2007). Since the early 1980s, the Mexican Pacific population of adult female leatherback turtles has declined to slightly more than 200 during 1998-1999 and 1999-2000 (Sarti *et al.*2000). Spotila *et al.*(2000) reported the decline of the leatherback nesting at Playa Grande, Costa Rica, which had been the fourth largest nesting group in the world and the most important nesting beach in the Pacific. Between

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1988 and 1999, the nesting group declined from 1,367 to 117 female leatherback sea turtles. Based on their models, Spotila *et al.*(2000) estimated that the group could fall to less than 50 females by 2003-2004. Another, more recent, analysis of the Costa Rican nesting beaches indicates a decline in nesting during 15 years of monitoring (1989-2004) with approximately 1,504 females nesting in 1988-1989 to an average of 188 females nesting in 2000-2001 and 2003-2004 (NMFS and USFWS 2007b), indicating that the reductions in nesting females were not as extreme as the reductions predicted by Spotila *et al.*(2000).

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

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

Indian Ocean

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

Mediterranean Sea

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

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

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

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

In 1979, the waters adjacent to Sandy Point, St. Croix, U.S. Virgin Islands were designated as critical habitat for the leatherback sea turtle. On February 2, 2010, NMFS received a petition to revise the critical habitat designation for leatherback sea turtles to include waters adjacent to a major nesting beach in Puerto Rico. NMFS published a 90-day finding on the petition on July 16, 2010, which found that the petition did not present substantial scientific information

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indicating that the petitioned revision was warranted. The original petitioners submitted a second petition on November 2, 2010 to revise the critical habitat designation to again include waters adjacent to a major nesting beach in Puerto Rico, including additional information on the usage of the waters. NMFS determined on May 5, 2011, that a revision to critical habitat off Puerto Rico may be warranted, and an analysis is underway. Note that on August 4, 2011, FWS issued a determination that revision to critical habitat along Puerto Rico should be made and will be addressed during the future planned status review.

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

Population Dynamics and Status

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

In the United States, the Florida Statewide Nesting Beach Survey program has documented an increase in leatherback nesting numbers from 98 nests in 1988 to between 800 and 900 nests in the early 2000s (NMFS and USFWS 2007b). Stewart *et al.*(2011) evaluated nest counts from 68 Florida beaches over 30 years (1979-2008) and found that nesting increased at all beaches with trends ranging from 3.1%-16.3% per year, with an overall increase of 10.2% per year. An analysis of Florida's index nesting beach sites from 1989-2006 shows a substantial increase in leatherback nesting in Florida during this time, with an annual growth rate of approximately 1.17

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(TEWG 2007). The TEWG reports an increasing or stable nesting trend for all of the seven populations or groups of populations with the exception of the Western Caribbean and West Africa. The leatherback rookery along the northern coast of South America in French Guiana and Suriname supports the majority of leatherback nesting in the western Atlantic (TEWG 2007), and represents more than half of total nesting by leatherback sea turtles worldwide (Hilterman and Goverse 2004). Nest numbers in Suriname have shown an increase and the long-term trend for the Suriname and French Guiana nesting group seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests for Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). The TEWG (2007) report indicates that using nest numbers from 1967-2005, a positive population growth rate was found over the 39-year period for French Guinea and Suriname, with a 95% probability that the population was growing. Given the magnitude of leatherback nesting in this area compared to other nest sites, negative impacts in leatherback sea turtles in this area could have profound impacts on the entire species.

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

Threats

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

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nitrogen, for entangled turtles showed several key differences that were most likely due to reduced foraging and associated seawater ingestion, as well as a general stress response.

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

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

Leatherbacks are susceptible to entanglement in the lines associated with trap/pot gear used in several fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine (Dwyer *et al.*2002). Additional leatherbacks stranded wrapped in line of unknown origin or with evidence of a past entanglement (Dwyer *et al.*2002). More recently, from 2002 to 2010, NMFS received 137 reports of sea turtles entangled in vertical lines from Maine to Virginia, with 128 events confirmed (verified by photo documentation or response by a trained responder; NMFS 2008a). Of the 128 confirmed events during this period, 117 events involved leatherbacks. NMFS identified the gear type and fishery for 72 of the 117 confirmed

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events, which included lobster (42[4]), whelk/conch (15), black sea bass (10), crab (2), and research pot gear (1). A review of leatherback mortality documented by the STSSN in Massachusetts suggests that vessel strikes and entanglement in fixed gear (primarily lobster pots and whelk pots) are the principal sources of this mortality (Dwyer *et al.*2002).

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

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

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

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

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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.

Global climate change has been identified as a factor that may affect leatherback habitat and biology (NMFS and USFWS 2007b); however, no significant climate change related impacts to leatherback sea turtle populations have been observed to date. Over the long term, climate change related impacts will likely influence biological trajectories in the future on a century scale (Parmesan and Yohe 2003). Changes in marine systems associated with rising water temperatures, changes in ice cover, salinity, oxygen levels and circulation including shifts in ranges and changes in algal, plankton, and fish abundance could affect leatherback prey distribution and abundance. Climate change is expected to expand foraging habitats into higher latitude waters and some concern has been noted that increasing temperatures may increase the female:male sex ratio of hatchlings on some beaches (Morosovsky *et al.*1984 and Hawkes *et al.*2007 in NMFS and USFWS 2007b). However, due to the tendency of leatherbacks to have individual nest placement preferences and deposit some clutches in the cooler tide zone of beaches, the effects of long-term climate on sex ratios may be mitigated (Kamel and Mrosovsky 2004 in NMFS and USFWS 2007b). Additional potential effects of climate change on leatherbacks include range expansion and changes in migration routes as increasing ocean temperatures shift range-limiting isotherms north (Robinson *et al.*2008). Leatherbacks have expanded their range in the Atlantic north by 330 km in the last 17 years as warming has caused

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the northerly migration of the 15°C sea surface temperature (SST) isotherm, the lower limit of thermal tolerance for leatherbacks (McMahon and Hays 2006). Leatherbacks are speculated to be the best able to cope with climate change of all the sea turtle species due to their wide geographic distribution and relatively weak beach fidelity. Leatherback sea turtles may be most affected by any changes in the distribution of their primary jellyfish prey, which may affect leatherback distribution and foraging behavior (NMFS and USFWS 2007b). Jellyfish populations may increase due to ocean warming and other factors (Brodeur *et al.*1999; Attrill *et al.*2007; Richardson *et al.*2009). However, any increase in jellyfish populations may or may not impact leatherbacks as there is no evidence that any leatherback populations are currently food-limited.

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

Summary of Status for Leatherback Sea Turtles

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

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

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

4.2.2 Status of Atlantic sturgeon

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

The Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) is a subspecies of sturgeon distributed along the eastern coast of North America from Hamilton Inlet, Labrador, Canada to Cape Canaveral, Florida, USA (Scott and Scott, 1988; ASSRT, 2007; T. Savoy, CT DEP, pers. comm.). NMFS has delineated U.S. populations of Atlantic sturgeon into five DPSs (77 FR 5880 and 77 FR 5914). These are: the Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs (see Figure 1). 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 New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs as "endangered," and the Gulf of Maine DPS as "threatened" (77 FR 5880 and 77 FR 5914). The effective date of the listings was April 6, 2012. The DPSs do not include Atlantic sturgeon that are spawned in Canadian rivers. Therefore, Canadian spawned fish are not included in the listings.

As described below, individuals originating from 4 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.

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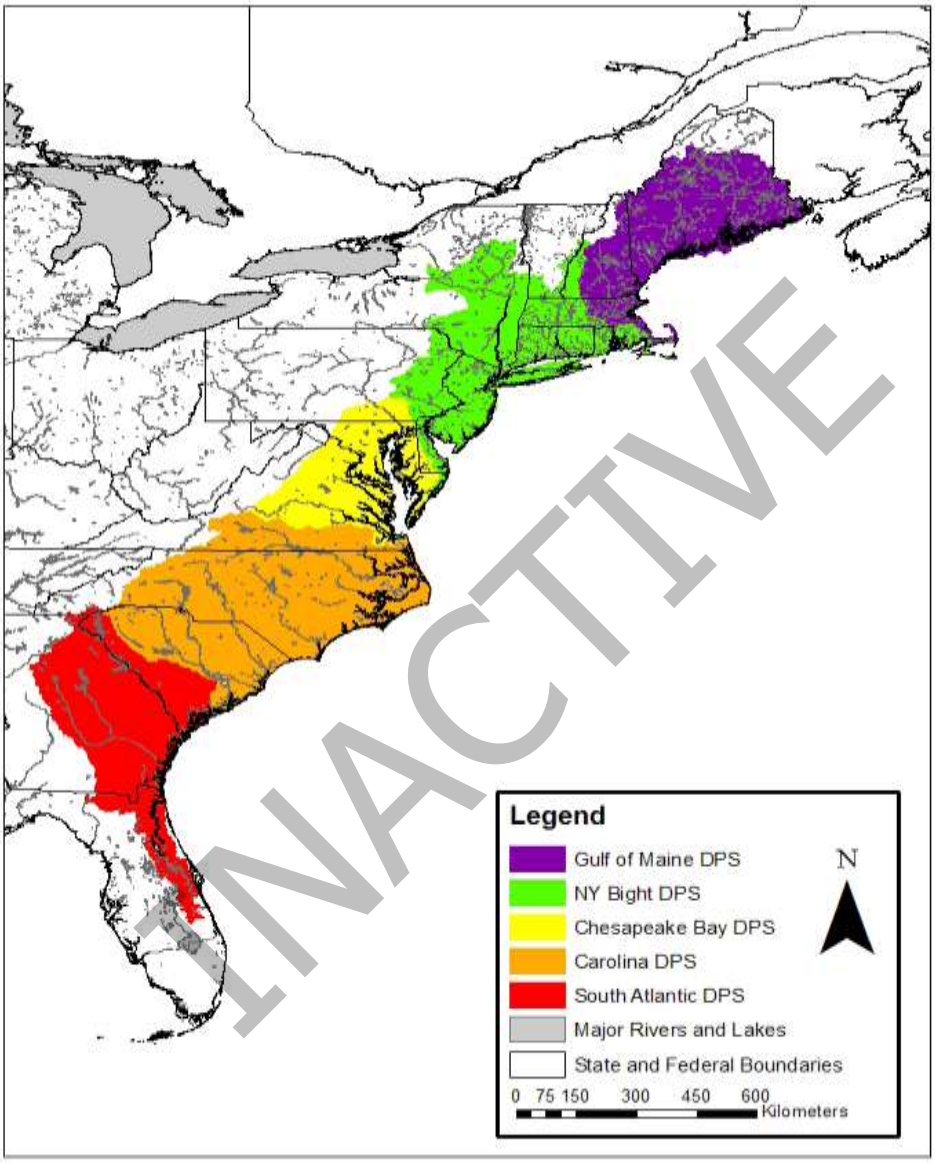
Figure 1. Map Depicting the Boundaries of the five Atlantic sturgeon DPSs



Atlantic sturgeon life history

Atlantic sturgeon are long lived (approximately 60 years), late maturing, estuarine dependent, anadromous[5] fish (Bigelow and Schroeder, 1953; Vladykov and Greeley 1963; Mangin, 1964; Pikitch *et al.*, 2005; Dadswell, 2006; ASSRT, 2007). 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 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

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[5] 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)

spawning grounds throughout the spawning season (Bain, 1997). Females begin spawning migrations when temperatures are closer to 12° C to 13° C (54° to 55° F) (Dovel and Berggren, 1983; Smith, 1985; Collins *et al.*, 2000), make rapid spawning migrations upstream, and quickly depart following spawning (Bain, 1997).

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

Larval Atlantic sturgeon (*i.e.* less than 4 weeks old, with total lengths (TL) less than 30 mm; Van Eenennaam *et al.*1996) are assumed to undertake a demersal existence and inhabit the same riverine or estuarine areas where they were spawned (Smith *et al.*, 1980; Bain *et al.*, 2000; Kynard and Horgan, 2002; ASMFC, 2009). Studies suggest that age-0 (*i.e.*, young-of-year), age-1, and age-2 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

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fishermen in nearshore waters along the Atlantic coast as far south as Cape Hatteras, North Carolina from November through early March. In the spring, a portion of the tagged fish re-entered the Delaware River estuary. However, many fish continued a northerly coastal migration through the Mid-Atlantic as well as into southern New England waters where they were recovered throughout the summer months. Movements as far north as Maine were documented. A southerly coastal migration was apparent from tag returns reported in the fall. The majority of these tag returns were reported from relatively shallow near shore fisheries with few fish reported from waters in excess of 25 m (C. Shirey, Delaware Department of Fish and Wildlife, unpublished data reviewed in ASMFC, 2009). Areas where migratory Atlantic sturgeon commonly aggregate include the Bay of Fundy (e.g., Minas and Cumberland Basins), Massachusetts Bay, Connecticut River estuary, Long Island Sound, New York Bight, Delaware Bay, Chesapeake Bay, and waters off of North Carolina from the Virginia/North Carolina border to Cape Hatteras at depths up to 24 m (Dovel and Berggren, 1983; Dadswell *et al.*, 1984; Johnson *et al.*, 1997; Rochard *et al.*, 1997; Kynard *et al.*, 2000; Eyler *et al.*, 2004; Stein *et al.*, 2004; Wehrell, 2005; Dadswell, 2006; ASSRT, 2007; Laney *et al.*, 2007). These sites may be used as foraging sites and/or thermal refuge.

Determination of DPS Composition in the Action Area
As explained above, the range of all 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 area where the NJ surveys will operate likely originate from four of the five DPSs at the following frequencies: Gulf of Maine 7%; NYB 58%; Chesapeake Bay 18%; and South Atlantic 17%. These percentages are based on genetic sampling of individuals (n=105, captured from April 2009-2010) sampled in targeted surveys for Atlantic sturgeon carried out in Delaware coastal waters. This area is the nearest geographic area to the action area for which mixed stock analysis is available. 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.

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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

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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[6] 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).

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 Northeast Fisheries Observer Program (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[7] 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.

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, 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).

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 do not appear in the NEFOP dataset and based on this, as well as genetics

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

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

information on fish captured in other coastal sampling programs in the Northeast[8] are assumed to not be intercepted in Northeast fisheries. 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 598. 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 598 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.

Table 2: 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	166	498
NYB (Hudson River and Delaware River)	950	2,850
CB	329	987
SA*	598	1,794

*see note re. South Atlantic population size in paragraph 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).

[8] We reviewed genetics information available for 701 individuals sampled in a variety of coastal sampling programs from Maine to Virginia. Only two fish were identified as Carolina DPS origin (collected in central Long Island Sound) and no fish in the NEFOP database (n=89 for genetic samples) were identified as Carolina DPS origin.

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

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

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

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

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

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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%, with the exception of monkfish gear which has a higher mortality rate of approximately 27%. Mortality rates in otter trawl gear are believed to be lower at approximately 5%. Comparing the estimated annual average mortalities to the adult population estimates for each of the 4 DPSs encountered in Northeast fisheries, we estimate that at least 4% of adults from each DPS are being killed as a result of interactions with fisheries authorized by Northeast FMPs each year.

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

4.2.2.1 Status of Gulf of Maine DPS of Atlantic sturgeon

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

Bigelow and Schroeder (1953) surmised that Atlantic sturgeon likely spawned in Gulf of Maine Rivers in May-July. More recent captures of Atlantic sturgeon in spawning condition within the Kennebec River suggest that spawning more likely occurs in June-July (Squiers *et al.*, 1981; ASMFC, 1998; NMFS and USFWS, 1998). Evidence for the timing and location of Atlantic sturgeon spawning in the Kennebec River includes: (1) the capture of five adult male Atlantic

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sturgeon in spawning condition (*i.e.*, expressing milt) in July 1994 below the (former) Edwards Dam; (2) capture of 31 adult Atlantic sturgeon from June 15,1980, through July 26,1980, in a small commercial fishery directed at Atlantic sturgeon from the South Gardiner area (above Merrymeeting Bay) that included at least 4 ripe males and 1 ripe female captured on July 26,1980; and, (3) capture of nine adults during a gillnet survey conducted from 1977-1981, the majority of which were captured in July in the area from Merrymeeting Bay and upriver as far as Gardiner, ME (NMFS and USFWS, 1998; ASMFC 2007). The low salinity values for waters above Merrymeeting Bay are consistent with values found in other rivers where successful Atlantic sturgeon spawning is known to occur.

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

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

Connectivity is disrupted by the presence of dams on several rivers in the Gulf of Maine region, including the Penobscot and Merrimack Rivers. While there are also dams on the Kennebec, Androscoggin and Saco Rivers, these dams are near the site of natural falls and likely represent the maximum upstream extent of sturgeon occurrence even if the dams were not present. Because no Atlantic sturgeon are known to occur upstream of any hydroelectric projects in the Gulf of Maine region, passage over hydroelectric dams or through hydroelectric turbines is not a source of injury or mortality in this area. While not expected to be killed or injured during

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passage at a dam, the extent that Atlantic sturgeon are affected by the existence of dams and their operations in the Gulf of Maine region is currently unknown. The documentation of an Atlantic sturgeon larvae downstream of the Brunswick Dam in the Androscoggin River suggests that Atlantic sturgeon spawning may be occurring in the vicinity of at least that project and therefore, may be affected by project operations. The range of Atlantic sturgeon in the Penobscot River is limited by the presence of the Veazie and Great Works Dams. Together these dams prevent Atlantic sturgeon from accessing approximately 29 km of habitat, including the presumed historical spawning habitat located downstream of Milford Falls, the site of the Milford Dam. While removal of the Veazie and Great Works Dams is anticipated to occur in the near future, the presence of these dams is currently preventing access to significant habitats within the Penobscot River. While Atlantic sturgeon are known to occur in the Penobscot River, it is unknown if spawning is currently occurring or whether the presence of the Veazie and Great Works Dams affects the likelihood of spawning occurring in this river. The Essex Dam on the Merrimack River blocks access to approximately 58% of historically accessible habitat in this river. Atlantic sturgeon occur in the Merrimack River but spawning has not been documented. Like the Penobscot, it is unknown how the Essex Dam affects the likelihood of spawning occurring in this river.

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

There are no empirical abundance estimates for the Gulf of Maine DPS. The Atlantic sturgeon SRT (2007) presumed that the Gulf of Maine DPS was comprised of less than 300 spawning adults per year, based on abundance estimates for the Hudson and Altamaha River riverine populations of Atlantic sturgeon. Surveys of the Kennebec River over two time periods, 1977-1981 and 1998-2000, resulted in the capture of nine adult Atlantic sturgeon (Squiers, 2004). However, since the surveys were primarily directed at capture of shortnose sturgeon, the capture gear used may not have been selective for the larger-sized, adult Atlantic sturgeon; several hundred subadult Atlantic sturgeon were caught in the Kennebec River during these studies. 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 Gulf of Maine DPS is known to occur in two rivers (Kennebec and Androscoggin) and possibly in a third. Spawning may be occurring in other rivers, such as the Sheepscot or Penobscot, but has not been confirmed. There are indications of increasing abundance of Atlantic sturgeon belonging to the Gulf of Maine DPS. Atlantic sturgeon continue to be present in the Kennebec River; in addition, they are captured in directed research projects

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

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

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

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4.2.2.2 Status of New York Bight DPS of Atlantic sturgeon

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

The abundance of the Hudson River Atlantic sturgeon riverine population prior to the onset of expanded exploitation in the 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.

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Summary of the New York Bight DPS

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

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

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

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

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

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

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

4.2.2.3 Status of Chesapeake Bay DPS of Atlantic sturgeon

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

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

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

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

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

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

Summary of the Chesapeake Bay DPS

Spawning for the Chesapeake Bay DPS is known to occur in only the James River. Spawning

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may be occurring in other rivers, such as the York, but has not been confirmed. There are anecdotal reports of increased sightings and captures of Atlantic sturgeon in the James River. However, this information has not been comprehensive enough to develop a population estimate for the James River or to provide sufficient evidence to confirm increased abundance. Some of the impact from the threats that facilitated the decline of the Chesapeake Bay DPS have been removed (e.g., directed fishing) or reduced as a result of improvements in water quality since passage of the Clean Water Act (CWA). As explained above, we have estimated that there is an annual mean of 329 mature adult Atlantic sturgeon in the Chesapeake Bay 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 Chesapeake Bay DPS of Atlantic sturgeon. Studies have shown that Atlantic sturgeon can only sustain low levels of bycatch mortality (Boreman, 1997; ASMFC, 2007; Kahnle *et al.*, 2007). The Chesapeake Bay DPS is currently at risk of extinction given (1) precipitous declines in population sizes and the protracted period in which sturgeon populations have been depressed; (2) the limited amount of current spawning; and, (3) the impacts and threats that have and will continue to affect the potential for population recovery.

4.2.2.4 Status of South Atlantic DPS of Atlantic sturgeon

Distribution and Abundance

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

Rivers known to have current spawning populations within the range of the South Atlantic DPS include the Combahee, Edisto, Savannah, Ogeechee, Altamaha, and Satilla Rivers. We determined spawning was occurring if young-of-the-year (YOY) were observed, or mature adults were present, in freshwater portions of a system (Table 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. 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

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populations. This represents our current knowledge of the river systems utilized by the South Atlantic DPS for specific life functions, such as spawning, nursery habitat, and foraging. However, fish from the South Atlantic DPS likely use other river systems than those listed here for their specific life functions.

Table 3. 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	

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The riverine spawning habitat of the South Atlantic DPS occurs within the South Atlantic Coastal Plain ecoregion (TNC 2002b), which includes fall-line sandhills, rolling longleaf pine uplands, wet pine flatwoods, isolated depression wetlands, small streams, large river systems, and estuaries. Other ecological systems in the ecoregion include maritime forests on barrier islands, pitcher plant seepage bogs and Altamaha grit (sandstone) outcrops. Other ecological systems in the ecoregion include maritime forests on barrier islands, pitcher plant seepage bogs and Altamaha grit (sandstone) outcrops. The primary threats to biological diversity in the South Atlantic Coastal Plain listed by TNC are intensive silvicultural practices, including conversion of natural forests to highly managed pine monocultures and the clear-cutting of bottomland hardwood forests. Changes in water quality and quantity, caused by hydrologic alterations

(impoundments, groundwater withdrawal, and ditching), and point and nonpoint pollution, are threatening the aquatic systems. Development is a growing threat, especially in coastal areas. Agricultural conversion, fire regime alteration, and the introduction of nonnative species are additional threats to the ecoregion's diversity. The South Atlantic DPS' spawning rivers, located in the South Atlantic Coastal Plain, are primarily of two types: brownwater (with headwaters north of the Fall Line, silt-laden) and blackwater (with headwaters in the coastal plain, stained by tannic acids).

Secor (2002) estimates that 8,000 adult females were present in South Carolina prior to 1890. Prior to the collapse of the fishery in the late 1800s, the sturgeon fishery was the third largest fishery in Georgia. Secor (2002) estimated from U.S. Fish Commission landing reports that approximately 11,000 spawning females were likely present in the state prior to 1890. Reductions from the commercial fishery and ongoing threats have drastically reduced the numbers of Atlantic sturgeon within the South Atlantic DPS. Currently, the Atlantic sturgeon spawning population in at least two river systems within the South Atlantic DPS has been extirpated. The Altamaha River population of Atlantic sturgeon, with an estimated 343 adults spawning annually, is believed to be the largest population in the Southeast, yet is estimated to be only 6 percent of its historical population size. The 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 South Atlantic DPS. Dredging is a present threat to the South Atlantic DPS and is contributing to their status by modifying the quality and availability of Atlantic sturgeon habitat. Maintenance dredging is currently modifying Atlantic sturgeon nursery habitat in the Savannah River and modeling indicates that the proposed deepening of the navigation channel will result in reduced DO and upriver movement of the salt wedge, curtailing spawning habitat. Dredging is also modifying nursery and foraging habitat in the St. Johns Rivers. Reductions in water quality from terrestrial activities have modified habitat utilized by the South Atlantic DPS. Low DO is modifying sturgeon habitat in the Savannah due to dredging, and non-point source inputs are causing low DO in the Ogeechee River and in the St. Marys River, which completely eliminates juvenile nursery habitat in summer. Low DO has also been observed in the St. Johns River in the summer. Sturgeon are more sensitive to low DO and the negative (metabolic, growth, and feeding) effects caused by low DO increase when water temperatures are concurrently high, as they are within the range of the South Atlantic DPS. Additional stressors arising from water allocation and climate change threaten to exacerbate water quality problems that are already present throughout the range of the South Atlantic DPS. Large withdrawals of over 240 million

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gallons per day mgd of water occur in the Savannah River for power generation and municipal uses. However, users withdrawing less than 100,000 gallons per day (gpd) are not required to get permits, so actual water withdrawals from the Savannah and other rivers within the range of the South Atlantic DPS are likely much higher. The removal of large amounts of water from the system will alter flows, temperature, and DO. Water shortages and "water wars" are already occurring in the rivers occupied by the South Atlantic DPS and will likely be compounded in the future by population growth and potentially by climate change. Climate change is also predicted to elevate water temperatures and exacerbate nutrient-loading, pollution inputs, and lower DO, all of which are current stressors to the South Atlantic DPS.

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

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

The recovery of Atlantic sturgeon along the Atlantic Coast, especially in areas where habitat is limited and water quality is severely degraded, will require improvements in the following areas:

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(1) elimination of barriers to spawning habitat either through dam removal, breaching, or installation of successful fish passage facilities; (2) operation of water control structures to provide appropriate flows, especially during spawning season; (3) imposition of dredging restrictions including seasonal moratoriums and avoidance of spawning/nursery habitat; and, (4) mitigation of water quality parameters that are restricting sturgeon use of a river (*i.e.*, DO). Additional data regarding sturgeon use of riverine and estuarine environments is needed.

A viable population able to adapt to changing environmental conditions is critical to Atlantic sturgeon, and the low population numbers of every river population in the South Atlantic DPS put them in danger of extinction throughout their range; none of the populations are large or stable enough to provide with any level of certainty for continued existence of Atlantic sturgeon in this part of its range. Although the largest impact that caused the precipitous decline of the species has been curtailed (directed fishing), the population sizes within the South Atlantic DPS have remained relatively constant at greatly reduced levels (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 South Atlantic DPS can occur.

Summary of the Status of the Carolina DPS of Atlantic Sturgeon

The South Atlantic DPS is estimated to number fewer than 6 percent of its historical population size, with all river populations except the Altamaha estimated to be 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 South Atlantic DPS by modifying spawning, nursery, and foraging habitat. Habitat modifications through reductions in water quality are also contributing to the status of the South Atlantic DPS through reductions in DO, particularly

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

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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.

5.1 Federal Actions that have Undergone Section 7 Consultation

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

5.1.1 Authorization of Fisheries through Fishery Management Plans

NMFS authorizes the operation of several fisheries in the action area under the authority of the Magnuson-Stevens Fishery Conservation Act and through Fishery Management Plans and their implementing regulations. Commercial and recreational fisheries in the action area employ gear that is known to harass, injure, and/or kill sea turtles and Atlantic sturgeon. In the Northeast Region (Maine through Virginia), formal ESA section 7 consultations have been conducted on the American lobster, Atlantic bluefish, Atlantic mackerel/squid/ butterfish, Atlantic sea scallop, monkfish, northeast multispecies, red crab, spiny dogfish, summer flounder/scup/black sea bass, and tilefish fisheries. These consultations have considered effects to loggerhead, green, Kemp's ridley and leatherback sea turtles. 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 (ITS) 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.

Table 4. Information on Fisheries Opinions conducted by NMFS NERO and SERO for federally managed fisheries that operate in the action area

FMP	**Date of Most Recent Opinion**	**Loggerhead**	**Kemp's ridley**	**Green**	**Leatherback**
American lobster	October 29, 2010	1	0	0	5
Atlantic bluefish	October 29, 2010	82 (34 lethal)	4	5	4
Monkfish	October 29, 2010	173 (70 lethal)	4	5	4

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Multispecies	October 29, 2010	46 in trawls (21 lethal)	4	5	4
Skate	October 29, 2010	39 (17 lethal)	4	5	4
Spiny dogfish	October 29, 2010	2	4	5	4
Mackerel/squid/butterfish	October 29, 2010	62 (25 lethal)	2	2	2
Summer flounder/scup/black sea bass	October 29, 2010	205 (85 lethal)	4	5	6
Shark fisheries as managed under the Consolidated HMS FMP	May 20, 2008	679 (349 lethal) every 3 years	2 (1 lethal) every 3 years	2 (1 lethal) every 3 years	74 (47 lethal) every 3 years
Atlantic sea scallop	March 15, 2008 (amended Feb 5, 2009)	1,083 (615 lethal)	3	3	2
Coastal migratory pelagic	August 13, 2007	33 every 3 years	4 every 3 years	14 every 3 years	2 every 3 years
Red Crab	February 6, 2002	1	0	0	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*

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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. At this time, there is no bycatch estimate for fisheries that are regulated by NMFS SERO. 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. Also, as noted above, NMFS SERO has reinitiated the consultation for shrimp trawling; consultation on the smooth dogfish fishery is also currently being conducted by SERO in coordination with NMFS HMS.

5.1.2 Hopper Dredging

The construction and maintenance of federal navigation channels and sand mining ("borrow") areas have also been identified as sources of sea turtle mortality. Atlantic sturgeon may also be killed during hopper dredging operations, although this is rare. All hopper dredging projects are authorized or carried out by the U.S. Army Corps of Engineers. In the action area, these projects are under the jurisdiction of the districts within the North Atlantic Division. Hopper dredging projects in this area have resulted in the recorded mortality of approximately 87 loggerheads, 4 greens, 9 Kemp's ridleys and 4 unidentified hard shell turtles since observer records began in 1993. Nearly all of these interactions resulted in the death of the turtle. To date, nearly all of these interactions have occurred in nearshore coastal waters with very few interactions in the open ocean. Similarly, few interactions between hopper dredges and Atlantic sturgeon have been observed, with just 3 records documenting interactions between hopper dredges and Atlantic sturgeon in the action area (2 in Virginia near the Chesapeake Bay entrance, and one in New York Bight). We have completed several ESA Section 7 consultations with the Corps to consider effects of these hopper dredging projects on listed sea turtles. Many of these consultations will be reinitiated to consider effects to Atlantic sturgeon. The table below provides information on Biological Opinions considering dredging projects in the action area and the associated ITS for sea turtles (unless otherwise noted, take estimates are per dredge cycle):

Table 5. Information on Consultations conducted by NMFS for dredging projects that occur in the action area

Project	Date of Opinion	Loggerhead	Kemp's ridley	Green	Leatherback	Notes
Long Island NY to Manasquan NJ Beach Nourishment	12/15/1995	5 turtles total: combination of any species				
Sandy Hook Channel Dredging	6/10/1996	2	1	2	1	2 loggerheads/green inclusive; and 1 Kemp's/leatherback
ACOE Philadelphia District Dredging	11/26/1996	4	1	1	0	Annual Estimate

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Ambrose Channel, NJ Sand Mining	10/11/2002	2	1	1	1	1 leatherback OR Kemp's
Delaware River Deepening	07/17/2009	20 sea turtles total over the life of the project: no more than 2 Kemp's the remainder loggerhead		0	0	

5.1.3 Salem and Hope Creek Nuclear Generating Stations

PSEG Nuclear operates two nuclear power plants pursuant to licenses issued by the US Nuclear Regulatory Commission (NRC). These facilities are the Salem and Hope Creek Generating Stations (Salem and HCGS), which are located on adjacent sites within a 740-acre parcel of property at the southern end of Artificial Island in Lower Alloways Creek Township, Salem County, New Jersey. Consultation pursuant to Section 7 of the ESA between NRC and NMFS on the effects of the operation of these facilities has been ongoing since 1979. A Biological Opinion was issued by NMFS in April 1980 in which NMFS concluded that the ongoing operation of the facilities was not likely to jeopardize the continued existence of shortnose sturgeon. Consultation was reinitiated in 1988 due to the documentation of impingement of sea turtles at the Salem facility. An Opinion was issued on January 2, 1991 in which NMFS concluded that the ongoing operation was not likely to jeopardize shortnose sturgeon, Kemp's ridley, green or loggerhead sea turtles. Consultation was reinitiated in 1992 due to the number of sea turtle impingements at the Salem intake exceeding the number exempted in the 1991 Incidental Take Statement. A new Opinion was issued on August 4, 1992. Consultation was again reinitiated in January 1993 when the number of sea turtle impingements exceeded the 1992 ITS with an Opinion issued on May 14, 1993. In 1998 the NRC requested that NMFS modify the Reasonable and Prudent Measures and Terms and Conditions of the ITS, and, specifically, remove a sea turtle study requirement. NMFS responded to this request in a letter dated January 21, 1999. Accompanying this letter was a revised ITS which served to amend the May 14, 1993 Opinion. The 1999 ITS exempts the annual take (capture at intake with injury or mortality) of 5 shortnose sturgeon, 30 loggerhead sea turtles, 5 green sea turtles, and 5 Kemp's ridleys. With the exception of 1991 and 1992, when 23 and 10 sea turtles were captured at the intakes, the actual level of take has been far lower than the exempted level. Inclusive of 1991 and 1992, for the period between 1979 and 1992, a total of 2 green, 23 Kemp's ridley and 60 loggerheads have been captured at the intakes. Since monitoring of the intakes was initiated in 1978, 18 shortnose sturgeon have been recovered from the Salem intakes. No shortnose sturgeon or sea turtles have been observed at the Hope Creek intakes. No sea turtles have been captured at Salem since 2001. Two Atlantic sturgeon, both previously dead, have been observed at the Salem intakes; none have been observed at the Hope Creek intakes. Consultation was reinitiated in 2011; we

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are currently in the process of preparing an updated Biological Opinion considering effects of ongoing operations on sea turtles and Atlantic sturgeon.

5.2 Non-federally regulated fisheries

Like federally authorized fisheries, Atlantic sturgeon and sea turtles may be vulnerable to capture, injury and mortality in fisheries occurring in state waters. The action area includes portions of New Jersey state waters. Information on the number of Atlantic sturgeon captured or killed in state fisheries is extremely limited and as such, efforts are currently underway to obtain more information on the numbers of Atlantic sturgeon captured and killed in state water fisheries.

Atlantic sturgeon are taken incidentally to the operation of fisheries targeting other anadromous species along the East Coast. New Jersey and Delaware do permit commercial fisheries to operate in designated portions of the Delaware River (Miller 2000, pers. Comm.; Boriek 2000, pers. comm.). American shad, eel, and blue crab are the species targeted by commercial fisherman, however, in the action area the level of commercial fishing is low (Miller 2000, pers. Comm.; Boriek 2000, pers. comm.). Recreational hook and line fisheries, that target largemouth bass, striped bass, white catfish and channel catfish, are permitted throughout the Delaware River and Delaware Bay (Coughman 2000, pers. comm.; Boriek 2000, pers. comm.). Very little is known about the level of listed species take in fisheries that operate strictly in state waters and at this time we are not able to quantify the number of interactions between Atlantic sturgeon or sea turtles and state authorized commercial or recreational fisheries operating in the action area.

Atlantic croaker fishery

An Atlantic croaker fishery using trawl and gillnet gear also occurs within the action area and turtle takes have been observed in the fishery. The average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the Atlantic croaker fishery was estimated to be 70 loggerhead sea turtles (Warden 2011). Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the Atlantic croaker fishery, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear used in the Atlantic croaker fishery, based on VTR data from 2002-2006, was estimated to be 11 per year with a 95% CI of 3-20 (Murray 2009b). A quantitative assessment of the number of Atlantic sturgeon captured in the croaker fishery is not available. Mortality rates of Atlantic sturgeon in commercial trawls has been estimated at 5%. A review of the NEFOP observer database indicates that from 2006-2010, 60 Atlantic sturgeon (out of a total of 726 observed interactions) were captured during observed trips where the trip target was identified as croaker. This represents a minimum number of Atlantic sturgeon captured in the croaker fishery during this time period as it only considers observed trips.

Weakfish fishery

The weakfish fishery occurs in both state and Federal waters but the majority of commercially and recreationally caught weakfish are caught in state waters (ASMFC 2002). The dominant commercial gears include gill nets, pound nets, haul seines, and trawls, with the majority of landings occurring in the fall and winter months (ASMFC 2002). Weakfish landings were

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dominated by the trawl fishery through the mid-1980s after which gill net landings began to account for most weakfish landed (ASMFC 2002). North Carolina has accounted for the majority of the annual landings since 1972 while Virginia ranks second, followed by New Jersey (ASMFC 2002). As described in section 3.1.1, sea turtle bycatch in the weakfish fishery has occurred (Warden 2011; Murray 2009a, 2009b). The average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the weakfish fishery was estimated to be 1 loggerhead sea turtle (Warden 2011). Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the weakfish fishery, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear used in the weakfish fishery, based on VTR data from 2002-2006, was estimated to be one (1) per year with a 95% CI of 0-1 (Murray 2009b). A quantitative assessment of the number of Atlantic sturgeon captured in the weakfish fishery is not available. Mortality rates of Atlantic sturgeon in commercial trawls has been estimated at 5%. A review of the NEFOP observer database indicates that from 2006-2010, 36 Atlantic sturgeon (out of a total of 726 observed interactions) were captured during observed trips where the trip target was identified as weakfish. This represents a minimum number of Atlantic sturgeon captured in the weakfish fishery during this time period as it only considers observed trips.

Whelk fishery

A whelk fishery using pot/trap gear is known to occur in several parts of the action area, including waters off of Maine, Connecticut, Massachusetts, Delaware, Maryland, and Virginia. Landings data for Delaware suggests that the greatest effort in the whelk fishery for waters off of that state occurs in the months of July and October; times when sea turtles are present. Whelk pots, which unlike lobster traps are not fully enclosed, have been suggested as a potential source of entrapment for loggerhead sea turtles that may be enticed to enter the trap to get the bait or whelks caught in the trap (Mansfield *et al.*2001). Leatherback and loggerhead sea turtles as well as right, humpback, and fin whales are known to become entangled in lines associated with trap/pot gear used in several fisheries including lobster, whelk, and crab species (NMFS SEFSC 2001; Dwyer *et al.*2002: NMFS 2007a). Whelk pots are not known to interact with Atlantic sturgeon

Crab fisheries

Various crab fisheries, such as horseshoe crab and blue crab, also occur in Federal and state waters. Atlantic sturgeon are not known to be captured in crab pot gear. The crab fisheries may have detrimental impacts on sea turtles beyond entanglement in the fishing gear itself. Loggerheads are known to prey on crab species, including horseshoe and blue crabs. In a study of the diet of loggerhead sea turtles in Virginia waters from 1983-2002, Seney and Musick (2007) found a shift in the diet of loggerheads in the area from horseshoe and blue crabs to fish, particularly menhaden and Atlantic croaker. The authors suggested that a decline in the crab species have resulted in the shift and loggerheads are likely foraging on fish captured in fishing nets or on discarded fishery bycatch (Seney and Musick 2007). The physiological impacts of this shift are uncertain although it was suggested as a possible explanation for the declines in loggerhead abundance noted by Mansfield (2006). Other studies have detected seasonal declines in loggerhead abundance coincident with seasonal declines of horseshoe and blue crabs in the

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same area (Maier *et al.*2005). While there is no evidence of a decline in horseshoe crab abundance in the Southeast during the period 1995-2003, declines were evident in some parts of the Mid-Atlantic (ASMFC 2004; Eyler *et al.*2007). Given the variety of loggerheads prey items (Dodd 1988; Burke *et al.*1993; Bjorndal 1997; Morreale and Standora 1998) and the differences in regional abundance of horseshoe crabs and other prey items (ASMFC 2004; Eyler *et al.*2007), a direct correlation between loggerhead sea turtle abundance and horseshoe crab and blue crab availability cannot be made at this time. Nevertheless, the decline in loggerhead abundance in Virginia waters (Mansfield 2006), and possibly Long Island waters (Morreale *et al.*2005), commensurate with noted declines in the abundance of horseshoe crab and other crab species raises concerns that crab fisheries may be impacting the forage base for loggerheads in some areas of their range.

American lobster trap fishery

An American lobster trap fishery also occurs in state waters of New England and the Mid-Atlantic and is managed under the ASMFC's ISFMP. Like the Federal waters component of the fishery, the state waters fishery has also been identified as a source of gear causing injuries to and mortality of loggerhead and leatherback sea turtles as a result of entanglement in vertical buoy lines of the pot/trap gear. Between 2002 and 2008, the lobster trap fishery in state waters was verified as the fishery involved in at least 27 leatherback entanglements in the Northeast Region. All entanglements involved the vertical line of the gear. These verified/confirmed entanglements occurred in Maine, Massachusetts, and Rhode Island state waters from June through October (Northeast Region STDN database). Atlantic sturgeon are not known to interact with lobster trap gear.

Incidental captures of loggerheads in fish traps have also been reported from several Atlantic coast states (Shoop and Ruckdeschel 1989; W. Teas, pers. comm.). Long haul seines and channel nets are also known to incidentally capture loggerheads and other sea turtles in sounds and other inshore waters along the U.S. Atlantic coast, although no lethal takes have been reported (NMFS SEFSC 2001). No information on interactions between Atlantic sturgeon and fish traps, long haul seines or channel nets is currently available; however, depending on where this gear is set and the mesh size, the potential exists for Atlantic sturgeon to be entangled or captured in this gear.

Observations of state recreational fisheries have shown that loggerhead, leatherback, and green sea turtles are known to bite baited hooks, and loggerheads frequently ingest the hooks. Hooked sea turtles have been reported by the public fishing from boats, piers, beaches, banks, and jetties, and from commercial fishermen fishing for snapper, grouper, and sharks with both single rigs and bottom longlines (NMFS SEFSC 2001). A summary of known impacts of hook-and-line captures on loggerhead sea turtles can be found in the TEWG (1998, 2000, 2009) reports. Atlantic sturgeon have been observed captured in hook and line gear; the number of interactions that occur is unknown. While most Atlantic sturgeon are likely to be released alive, we currently have no information on post-release survival.

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5.3 Vessel Activity and Military Operations

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

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

Similarly, operations of vessels by other Federal agencies within the action area (NOAA, EPA, and ACOE) may adversely affect sea turtles. However, vessel activities of those agencies are often limited in scope, as they operate a limited number of vessels or are engaged in research/ operational activities that are unlikely to contribute a large amount of risk. From 2009 on, NOAA research vessels conducting fisheries surveys for the NEFSC are estimated to take no more than nine sea turtles per year (eight alive, one dead). This includes up to seven loggerheads as well as an additional loggerhead, leatherback, Kemp's ridley, or green sea turtle per year during bottom trawl surveys and one loggerhead, leatherback, Kemp's ridley, or green sea turtle per year during scallop dredge surveys (NMFS 2007c).

5.4 Other Activities

5.4.1 Maritime Industry

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

5.4.2 Pollution

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

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

5.4.3 Coastal development

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

5.4.4 Global climate change and ocean acidification

The global mean temperature has risen 0.76ºC over the last 150 years, and the linear trend over the last 50 years is nearly twice that for the last 100 years (IPCC 2007a) and precipitation has increased nationally by 5%-10%, mostly due to an increase in heavy downpours (NAST 2000). There is a high confidence, based on substantial new evidence, that observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover,

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salinity, oxygen levels, and circulation. Ocean acidification resulting from massive amounts of carbon dioxide and pollutants released into the air can have major adverse impacts on the calcium balance in the oceans. Changes to the marine ecosystem due to climate change include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007b). These trends are most apparent over the past few decades.

Climate model projections exhibit a wide range of plausible scenarios for both temperature and precipitation over the next century. Both of the principal climate models used by the National Assessment Synthesis Team (NAST) project warming in the southeast by the 2090s, but at different rates (NAST 2000): the Canadian model scenario shows the southeast U.S. experiencing a high degree of warming, which translates into lower soil moisture as higher temperatures increase evaporation; the Hadley model scenario projects less warming and a significant increase in precipitation (about 20%). The scenarios examined, which assume no major interventions to reduce continued growth of world greenhouse gases (GHG), indicate that temperatures in the U.S. will rise by about 3°-5°C (5°-9°F) on average in the next 100 years which is more than the projected global increase (NAST 2000). A warming of about 0.2^{o}C 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 3 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 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 is 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

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and marine resources on smaller geographic scales, such as the action area, 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 United States. Additional information on potential effects of climate change specific to the action area is discussed below. Warming is very likely to continue in the U.S. during 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 they 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 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 will experience greater changes in discharge and water stress than unimpacted, free-flowing rivers (Palmer *et al.*2008).
While debated, researchers anticipate: 1) the frequency and intensity of droughts and floods will change across the nation; 2) a warming of about 0.2^{o}C 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

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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, and between 1985 and 1995 more than 32,000 acres of coastal salt marsh was lost in the southeastern U.S. due to a combination of human development activities, sea level rise, natural subsidence and erosion.

Effects on sea turtles and Atlantic sturgeon globally
Sea turtle species and 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 a problem for sea turtle or sturgeon species. As explained in the "Status of the Species" sections 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, increased risk of nest inundation, and changes in the abundance and distribution of forage species which could result in changes in the foraging behavior and distribution of sea turtle species. Atlantic sturgeon could be affected by changes in river ecology resulting from increases in precipitation and changes in water temperature which may affect recruitment and distribution in these rivers. Changes in oceanic conditions could also affect the marine distribution of Atlantic sturgeon or their marine and estuarine prey resources. However, as noted in the "Status of the Species" section above, with the exception of green sea turtles, information on current effects of global climate change on sea turtles and Atlantic sturgeon is not available and while it is speculated that future climate change may affect these species, it is not possible to quantify the extent to which effects may occur. However, given the short duration of the proposed action (*i.e.*, to be completed by the end of 2012) it is not likely that there will be any new effects of climate change in the action area that may affect any of these species in a manner that was not already considered in the Status of the Species sections above.

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5.5 Reducing Threats to ESA-listed Sea Turtles

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

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

In response to the observed capture of sea turtles in scallop dredge gear, including serious injuries and sea turtle mortality as a result of capture, NMFS proposed a modification to scallop

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

5.5.2 Sea Turtle Handling and Resuscitation Techniques

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

5.5.3 Sea Turtle Entanglements and Rehabilitation

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

5.5.4 Education and Outreach Activities

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

5.5.5 Sea Turtle Stranding and Salvage Network (STSSN)

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

5.6 Reducing Threats to Atlantic sturgeon

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

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6.0 EFFECTS OF THE ACTION

As discussed in the *Description of the Proposed Action*, the proposed Federal action is the FWS funding of fish surveys carried out by the State of New Jersey through May 31, 2014. Sea turtles and Atlantic sturgeon could be affected by the proposed action in a number of ways. This includes: (1) capture in sampling gear; (2) interactions with the research vessels; (3) effects to prey; and (4) effects to habitat. The analysis will be organized along these topics.

6.1 Summary of information on distribution of sea turtles and Atlantic sturgeon in the action area

As described in sections 3.1 – 3.4, the occurrence of loggerhead, Kemp's ridley, green, and leatherback sea turtles in New England, Mid-Atlantic, and south Atlantic waters is primarily temperature dependent (Thompson 1984; Keinath *et al.*1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998; Mitchell *et al.*2003; Braun-McNeill and Epperly 2004; James *et al.*2005a; Morreale and Standora 2005). In general, sea turtles move up the U.S. Atlantic coast from southern wintering areas as water temperatures warm in the spring (Keinath *et al.*1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998; Mitchell *et al.*2003; Braun-McNeill and Epperly 2004; James *et al.*2005a; Morreale and Standora

2005). The trend is reversed in the fall as water temperatures cool. By December, sea turtles have passed Cape Hatteras, returning to more southern waters for the winter (Keinath *et al.*1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998; Mitchell *et al.*2003; Braun-McNeill and Epperly 2004; James *et al.*2005a; Morreale and Standora 2005). Recreational anglers have reported sightings of sea turtles in waters defined as inshore waters (bays, inlets, rivers, or sounds; Braun-McNeill and Epperly 2004) as far north as New York as early as March-April, but in relatively low numbers (Braun-McNeill and Epperly 2004). Greater numbers of loggerheads, Kemp's ridleys, and greens are found in inshore, nearshore, and offshore waters of North Carolina and Virginia from May through November and in inshore, nearshore, and offshore waters of New York from June through October (Keinath *et al.*1987; Morreale and Standora 1993; Braun-McNeill and Epperly 2004). The hard-shelled sea turtles (loggerheads, Kemp's ridleys, and greens) appear to be temperature limited to water no further north than Cape Cod. Leatherback sea turtles have a similar seasonal distribution but have a more extensive range in the Gulf of Maine compared to the hard-shelled species (Shoop and Kenney 1992; Mitchell *et al.*2003; STSSN database).

Extensive survey effort of the continental shelf from Cape Hatteras to Nova Scotia, Canada in the 1980s (CeTAP 1982) revealed that loggerheads were observed at the surface in waters from the beach to waters with bottom depths of up to 4,481 m. However, they were generally found in waters where bottom depths ranged from 22-49 m deep (the median value was 36.6 m; Shoop and Kenney 1992). Leatherbacks were sighted at the surface in waters with bottom depths ranging from 1-4,151 m deep (Shoop and Kenney 1992). However, 84.4% of leatherback sightings occurred in waters where the bottom depth was less than 180 m (Shoop and Kenney 1992), whereas 84.5% of loggerhead sightings occurred in waters where the bottom depth was less than 80 m (Shoop and Kenney 1992). Neither species was commonly found in waters over Georges Bank, regardless of season (Shoop and Kenney 1992). The CeTAP study did not include Kemp's ridley and green sea turtle sightings, given the difficulty of sighting these smaller sea turtle species (CeTAP 1982).

The Southeast Turtle Survey (SeTS), an aerial survey research program initiated by the NMFS Southeast Fisheries Science Center (SEFSC) in 1982 through 1984, was conducted from Cape Hatteras to Key West over coastal waters from the coastline to the approximate mean western boundary of the Gulf Stream (Thompson 1984). Seasonal surveys that corresponded to spring (April-May) and summer (July-August) were completed in all three years. Fall (October-November) surveys were completed in 1982 and 1983 and a single winter survey was completed in January/February 1983 (Thompson and Huang 1993). The study area was designed as a southern extension of the CeTAP aerial surveys. These surveys showed that sea turtles in the south Atlantic region are distributed randomly from the coast out to the Gulf Stream except in the winter. During the winter, sea turtles appear to aggregate within the western Gulf Stream boundary waters which can be 5°-6°C warmer than coastal waters (Thompson 1988).

Given the seasonal occurrence patterns of sea turtles off the Mid-Atlantic coast, sea turtles are likely to occur in the action area from May – November. All sea turtle interactions in past NJ

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studies have occurred in June, August and October, which is consistent with our understanding of the seasonal distribution of sea turtles in the action area.

Subadult and adult Atlantic sturgeon may be present in the action area year-round. In the marine environment, Atlantic sturgeon are most often captured in depths less than 50 meters. Some information suggests that captures in otter trawl gear is most likely to occur in waters with depths less than 30 m (ASMFC 2007). Given the past capture of Atlantic sturgeon in nearly all of the ocean trawl surveys carried out since 1988, which have occurred in January, April, June, August, October and December, it is reasonable to anticipate that Atlantic sturgeon will be present in the action area year round. A similar seasonal distribution is anticipated in Delaware Bay where the other surveys will take place. As described above, we expect that Atlantic sturgeon in the action are will originate from the NYB (58%), SA (17%), CB (18%), and GOM (7%) DPSs.

6.2 Capture in trawl gear

6.2.1 Capture in trawl gear – sea turtles

The potential for capture of sea turtles in bottom otter trawl gear is well established (see for example, Lutcavage *et al.*1997, Henwood and Stuntz 1987, NRC 1990). Here, we establish the expected number of sea turtles that will be captured in the NJ ocean trawl surveys. As noted above, these surveys take place in January, April, June, August and October. This consultation considers activities to be carried out through the end of May 2014. Thus, we will consider effects of the following: 2012 – June, August and October; 2013 – January, April, June, August and October; 2014 – January and April. In sum, this represents two complete years (2 surveys in each of the 5 months) of ocean trawl survey effort.

6.2.1.1 Estimated Number of captures in 2012-2014 trawl surveys

The table below provides information on all sea turtles captured in past NJ trawl surveys conducted since the program began in 1989 (n=10, 9 loggerheads, 1 leatherback).

Table 6. Captures of Sea Turtles in NJ Ocean Trawl Survey 1989-2011

Year	Month	Species	Weight (kg)	Alive	Injured
1991	August	*loggerhead*	80	YES	NO
1993	June	*loggerhead*	19.87	YES	NO
1993	June	*loggerhead*	28.32	YES	NO
2002	October	*loggerhead*	30	YES	NO
2005	June	*loggerhead*	NA	YES	NO
2005	August	*leatherback*	227.27	YES	NO
2005	August	*loggerhead*	160	YES	NO
2007	October	*loggerhead*	19.78	YES	NO
2009	August	*loggerhead*	117.22	YES	NO

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2011	June	*loggerhead*	41.93	YES	NO

This study has been ongoing for 23 years; the mathematical average capture of sea turtles per year is 0.43 turtles/year. The number of sea turtles captured is highly variable from year to year, with most years having zero captures; however in 2005, three turtles were captured. The capture of sea turtles has become more frequent since 2002, as compared to the 1980s or 1990s. Applying the annual average over the complete time series to the 10 surveys considered here, we would anticipate the capture of no more than 1 sea turtle between now and May 31, 2014. However, the capture rate is higher in more recent years. The capture of 3 sea turtles in 2005 suggests that future interaction rates could be as high as 3 sea turtles per year. The maximum number of sea turtles captured in any particular survey has been two (August 2005), with no more than 1 sea turtle captured in all other surveys. No sea turtles have been captured in the January or April surveys which is consistent with our knowledge of the seasonal distribution of sea turtles in the action area. Based on this, we expect that an average of one sea turtle will be captured during each survey in June, August, and October. Each of these surveys will be carried out twice during the period considered here. Therefore, we anticipate that a total of six sea turtles will be captured during the NJ ocean trawl survey between now and May 31, 2014.

With the exception of one capture of a leatherback in 2005, all other captures of sea turtles in the ocean trawl survey have been loggerheads. However, we know that Kemp's ridley and green sea turtles occur in the action area and are vulnerable to capture in trawl gear. Because these species have been captured in trawl gear operating in nearby areas in similar surveys (*i.e.*, the NEAMAP surveys carried out by VIMS), we anticipate that future NJ surveys could capture Kemp's ridley or green sea turtles. Based on past interactions, we anticipate that the majority of the six captured sea turtles will be loggerheads, with no more than 1 leatherback, Kemp's ridley or green sea turtle.

6.2.1.2 Potential for Mortality Resulting from Capture in Trawls – Sea Turtles

Sea turtles forcibly submerged in any type of restrictive gear can eventually suffer fatal consequences from prolonged anoxia and/or seawater infiltration of the lung (Lutcavage *et al.*1997). A study examining the relationship between tow time and sea turtle mortality in the shrimp trawl fishery showed that mortality was strongly dependent on trawling duration, with the proportion of dead or comatose sea turtles rising from 0% for the first 50 minutes of capture to 70% after 90 minutes of capture (Henwood and Stuntz 1987). However, metabolic changes that can impair a sea turtle's ability to function can occur within minutes of a forced submergence. While most voluntary dives appear to be aerobic, showing little if any increases in blood lactate and only minor changes in acid-base status, the story is quite different in forcibly submerged sea turtles, where oxygen stores are rapidly consumed, anaerobic glycolysis is activated, and acid-base balance is disturbed, sometimes to lethal levels (Lutcavage and Lutz 1997). Forced submergence of Kemp's ridley sea turtles in shrimp trawls resulted in an acid-base imbalance after just a few minutes (times that were within the normal dive times for the species) (Stabenau *et al.*1991). Conversely, recovery times for acid-base levels to return to normal may be

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prolonged. Henwood and Stuntz (1987) found that it took as long as 20 hours for the acid-base levels of loggerhead sea turtles to return to normal after capture in shrimp trawls for less than 30 minutes. This effect is expected to be worse for sea turtles that are recaptured before metabolic levels have returned to normal.

Following the recommendations of the NRC to reexamine the association between tow times and sea turtle deaths, the data set used by Henwood and Stuntz (1987) was updated and re-analyzed (Epperly *et al.*2002; Sasso and Epperly 2006). Seasonal differences in the likelihood of mortality for sea turtles caught in trawl gear were apparent. For example, the observed mortality exceeded 1% after 10 minutes of towing in the winter (defined in Sasso and Epperly (2006) as the months of December-February), while the observed mortality did not exceed 1% until after 50 minutes in the summer (defined as March-November; Sasso and Epperly 2006). In general, tows of short duration (<10 minutes) in either season have little effect on the likelihood of mortality for sea turtles caught in the trawl gear and would likely achieve a negligible mortality rate (defined by the NRC as <1%). Intermediate tow times (10-200 minutes in summer and 10-150 minutes in winter) result in a rapid escalation of mortality, and eventually reach a plateau of high mortality, but will not equal 100%, as a sea turtle caught within the last hour of a long tow will likely survive (Epperly *et al.*2002; Sasso and Epperly 2006). However, in both seasons, a rapid escalation in the mortality rate did not occur until after 50 minutes (Sasso and Epperly 2006) as had been found by Henwood and Stuntz (1987). Although the data used in the reanalysis were specific to bottom otter trawl gear in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries, the authors considered the findings to be applicable to the impacts of forced submergence in general (Sasso and Epperly 2006).

Sea turtle behaviors may influence the likelihood of them being captured in bottom trawl gear. Video footage recorded by the NMFS, Southeast Fisheries Science Center (SEFSC), Pascagoula Laboratory indicated that sea turtles will keep swimming in front of an advancing shrimp trawl, rather than deviating to the side, until they become fatigued and are caught by the trawl or the trawl is hauled up (NMFS 2002a). Sea turtles have also been observed to dive to the bottom and hunker down when alarmed by loud noise or gear (Memo to the File, L. Lankshear, December 4, 2007), which could place them in the path of bottom gear such as a bottom otter trawl. With respect to oceanographic features, a review of the data associated with the 11 sea turtles captured by the scallop dredge fishery in 2001 concluded that the sea turtles appeared to have been near the shelf/slope front (D. Mountain, pers. comm.).

Tows for the NJ marine trawl surveys will be 20 minutes in duration. Based on the analysis by Sasso and Epperly (2006) and Epperly *et al.*(2002) as well as information on captured sea turtles from past NJ trawl surveys, the NEAMAP and NEFSC trawl surveys, as well as the NEFSC FSB observer program, a 20-minute tow time for the bottom otter trawl gear to be used in the survey will likely eliminate the risk of death from forced submergence for sea turtles caught in the bottom otter trawl survey gear.

During spring and fall bottom otter trawl surveys conducted by the NEFSC from 1963-2009, a total of 71 loggerhead sea turtles were observed captured. Only one of the 71 loggerheads

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suffered injuries (cracks to the carapace) causing death (Wendy Teas, SEFSC, pers. comm. to Linda Despres, NEFSC, 2007). All others were alive and returned to the water unharmed. The one leatherback sea turtle captured in the NEFSC trawl survey was released alive and uninjured. NEFSC trawl survey tows are approximately 30 minutes in duration. All sea turtles captured in the NEAMAP surveys as well as the NJ trawl surveys have also been released alive and uninjured. Based on past results and the short duration of the tows, we do not anticipate that any of the six sea turtles (5 loggerheads and 1 Kemp's ridley, green or leatherback) captured during the New Jersey surveys will be injured or killed.

6.2.2 Capture in trawl gear – Atlantic sturgeon

The capture of Atlantic sturgeon in otter trawls used for commercial fisheries is well documented (see for example, Stein *et al.*2004 and ASMFC 2007). Atlantic sturgeon are also captured incidentally in trawls used for scientific studies. No information on bycatch rates that could be applied to the NJ study to predict future catch is available from the literature. However, NJ has recorded all sturgeon interactions since 1988. This information allows us to predict future interactions. To date, a total of 322 Atlantic sturgeon captures have been recorded, with an average encounter rate of 3.4% (*i.e.*, the percent of trawl samples that captured an Atlantic sturgeon) with a maximum of 35 captures in a given year (range of 0-35, mean of 13.4 captures per year).

Table 7. Captures of Atlantic Sturgeon in NJ Ocean Trawl 1988-2011

Year	**No. Sturgeon Caught**	**Total Samples**	**Samples with Sturgeon**
1988	2	68	2
1989	33	193	13
1990	15	171	10
1991	25	189	7
1992	27	191	8
1993	10	187	3
1994	0	186	0
1995	6	188	3
1996	3	189	3
1997	12	187	3
1998	1	188	1
1999	11	186	6
2000	1	186	1
2001	4	186	3
2002	5	188	5
2003	16	188	10
2004	23	187	8
2005	18	186	10
2006	35	186	10
2007	24	187	12

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2008	26	186	12
2009	12	186	7
2010	10	186	9
2011	3	186	3
Total	**322**	**4,361**	**149**

The number of Atlantic sturgeon captured each year is variable; thus, the maximum number of Atlantic sturgeon captured in a given survey is a reasonable indicator of the likely number of captures during the 2012-2014 surveys. Because these surveys will follow identical protocols to the past and operate in the same areas, it is reasonable to anticipate similar catch levels in these years. Surveys take place five times per year. Between zero and 18 Atlantic sturgeon were captured during each of the 118 surveys carried out since August 1988. The mathematical average is about 3 Atlantic sturgeon per survey or about 0.07 sturgeon per sample. We have considered whether this average has changed over time. From 1989-1999, 143 Atlantic sturgeon were caught in 2,055 samples, with an average of 0.07 Atlantic sturgeon per sample. From 2000-2011, 179 Atlantic sturgeon were captured in 2,306 samples, with an average of 0.08 Atlantic sturgeon per sample. While the number of captures is highly variable from year to year, the average per sample is consistent over time. Therefore, it is reasonable to consider that surveys carried out from June 2012 – May 2014, will result in the same average capture of Atlantic sturgeon. During this period, there will be ten surveys with approximately 372 samples. Using the average capture rate per sample, we anticipate the capture of 26 Atlantic sturgeon during this time. Using the average capture rate per survey, we would anticipate the capture of 30 Atlantic sturgeon. However, because more than this number of individuals has been captured in some years, the number of captures is variable over time, and we are only considering two years of sampling, we have considered that it is possible that the maximum number of Atlantic sturgeon caught in any year (35) could be captured during the two years of surveys considered here. Based on this, we anticipate that no more than 70 Atlantic sturgeon will be captured from June 2012 – May 2014.

Based on the mixed stock analysis, we expect that 58% of the captured Atlantic sturgeon will originate from the NYB DPS (40 individuals), 18% from the CB DPS (13 individuals), 17% from the SA DPS (12 individuals), and 7% from the GOM DPS (5 individuals).

The short duration of the tow and careful handling of any sturgeon once on deck is likely to result in a low potential for mortality. None of the 322 Atlantic sturgeon captured in past NJ ocean trawl surveys have had any evidence of injury and there have been no recorded mortalities. The NEFSC surveys have recorded the capture of 110 Atlantic sturgeon since 1972; the NEAMAP survey has captured 102 Atlantic sturgeon since 2007. To date, there have been no recorded injuries or mortalities. In the Hudson River, a trawl survey that incidentally captures shortnose and Atlantic sturgeon has been ongoing since the late 1970s. To date, no injuries or mortalities of any sturgeon have been recorded. Based on this information, we expect that all Atlantic sturgeon captured in the 2012-2014 NJ ocean trawl surveys will be alive and will be released uninjured.

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6.3 Effects of Striped Bass Gillnet Survey

The gillnet survey for striped bass has been ongoing since 1989. Gillnets are set in water depths of 6 to 12-feet in areas of lower Delaware Bay near Bidwell's Creek and Reeds Beach, New Jersey. Nets are 5-6" stretch mesh. The survey takes place from early March through early May. Since the mid-1990s, the survey has operated with drift gill nets rather than anchored gear; average soak time is about 30 minutes. Only one 600-foot net is set at a time and all nets are monitored/tended throughout the study.

Since 1989, 397 trips have occurred with 2,863 hours of sampling. No interactions with any sea turtles have occurred. Sea turtles are vulnerable to capture in gillnets; however, the location of the deployment makes interactions with these species unlikely. Because no captures of these species have occurred in this study in the past and there are no changes to the study proposed that would increase the potential for interactions, we do not anticipate any future interactions with any species of sea turtles in the striped bass gillnet survey.

To date, 55 Atlantic sturgeon have been captured in the striped bass survey. All Atlantic sturgeon have been captured since 1997. With the exception of 2005 when 33 individuals were captured, the number of captures has been less than 6 per year. The average catch per unit effort of Atlantic sturgeon since 1989 is 0.17 sturgeon/set; since 1996 the average CPUE is 0.24 sturgeon/set (2,354 in this time period); considering only the last 10 years (2001-2011), the CPUE is 0.034 sturgeon/set (1,454 sets in this time period). Because the CPUE has changed over time, with could be related to either changes in the way the gear is deployed (anchored nets vs. drift nets) and/or changes in Atlantic sturgeon abundance or distribution, it is reasonable to apply the CPUE from the most recent time period (*i.e.*, the last 10 years) to predict future interactions with Atlantic sturgeon. The average number of sets per year since 2001 is 145; considering three years of sampling with this number of sets per year and the CPUE of 0.034 sturgeon per set, we calculate that 15 Atlantic sturgeon will be captured during the striped bass gillnet survey from 2012-2014, with an average of 5 captures per year. However, we must also consider that past observations suggest that there is the potential for some years to have higher levels of interactions (e.g., there were 10 captures in 2010 and 33 in 2005). The maximum number of interactions in any consecutive 3 year period is 39 (2005-2007). Because the number of captures is variable over time, and we are only considering three years of sampling, we have considered that it is possible that the maximum number of Atlantic sturgeon caught in any three year period (39) could be captured during the three years of surveys considered here. Based on this, we anticipate that no more than 39 Atlantic sturgeon will be captured from 2012 – May 2014.

Based on the mixed stock analysis, we expect that 58% of the captured Atlantic sturgeon will originate from the NYB DPS (22 individuals), 18% from the CB DPS (7 individuals)17% from the SA DPS (7 individuals), and 7% from the GOM DPS (3 individuals).

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The short duration of the net sets, constant monitoring/tending of the gear and careful handling of any sturgeon once the net is hauled is likely to result in a low potential for mortality. None of the 55 Atlantic sturgeon captured in past gillnet surveys have had any evidence of injury and there have been no recorded mortalities. Information available from the NEFOP database suggests that mortality of Atlantic sturgeon in commercially fished sink gillnets is, on average, approximately 20%; however, mortality of sturgeon in gillnets set for fisheries research is much lower, on average around 1%. The duration of gillnet deployment is likely a primary factor in mortality rates. Based on the short duration of net sets (average of 30 minutes) and the constant observation/tending of the net, and past monitoring which indicates that no mortalities have occurred, we expect that the likelihood of an Atlantic sturgeon captured in the 2012-2014 striped bass gillnet surveys suffering serious injury or mortality is very low (around 1% based on other research using gillnets to capture sturgeon); therefore, we expect that no more than one of the 39 Atlantic sturgeon will die. All other captured Atlantic sturgeon will be alive and will be released uninjured.

6.4 Effects of Juvenile Striped Bass Seine Survey

Between 1980 and 2011, a total of 6,278 seine hauls have taken place. No Atlantic sturgeon or sea turtles have been captured. We have reviewed information on interactions between these species and other seine surveys conducted in New England and the Mid-Atlantic and have found no records of any interactions between Atlantic sturgeon or sea turtles and this type of sampling gear. As such, we do not anticipate any future interactions with Atlantic sturgeon or any species of sea turtles in the juvenile striped bass seine survey.

6.5 Effects of Delaware Bay Finfish Survey

Since 1991, a total of 1,552 bottom trawl samples have taken place. These are short 10 minute tows. As evidenced above, Atlantic sturgeon and sea turtles are vulnerable to capture in bottom trawls. However, because no captures of these species have occurred in this study in the past and there are no changes to the study proposed that would increase the potential for interactions, we do not anticipate any future interactions with Atlantic sturgeon or any species of sea turtles in the Delaware Bay finfish survey.

6.6 Interactions with the research vessel

Sea turtles are known to be injured and/or killed as a result of being struck by vessels on the water and as a result of capture in or physical contact with fishing gear. With respect to the NJ surveys, the effects to sea turtles as a result of vessel activities are discountable. The single vessel that will operate on the water as a result of the proposed action is unlikely to strike sea turtles in the action area given that: (a) the vessel will operate/travel at a slow speed such that a sea turtle would have the speed and maneuverability to avoid contact with the vessel and (b) sea turtles spend part of their time at depths out of range of a vessel collision.

As noted in the 2007 Status Review and the proposed rule, in certain geographic areas vessel strikes have been identified as a threat to Atlantic sturgeon. 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

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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.

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. The risk of vessel strikes between Atlantic sturgeon and research vessels operating in the open ocean is likely to be low given that the research vessels are likely to be operating at slow speeds and there are no restrictions forcing Atlantic sturgeon into close proximity with the vessel as may be present in some rivers.

Given the large volume of vessel traffic in the action area and the wide variability in traffic in any given day, the increase in traffic (one vessel, traveling at relatively slow speeds) associated with the NJ surveys is extremely small. Given the small and localized increase in vessel traffic that would result from the NJ surveys, it is unlikely that there would be any detectable increase in the risk of vessel strike. As such, effects to Atlantic sturgeon from the increase in vessel traffic are likely to be discountable.

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6.7 Effects to Prey

Sea turtles could be negatively affected by the loss of prey as a result of mobile fishing gear that removes or incidentally kills such prey during commercial fishing or marine survey activities The use of trawls and gillnets for the surveys carried out by NJ will not reduce the availability of prey for loggerhead, Kemp's ridley, green, or leatherback sea turtles. The sampling gear is expected to catch a variety of organisms including fish and crab species. None of these are typical prey species of leatherback sea turtles or of neritic juvenile or adult green sea turtles (Rebel 1974; Mortimer 1982; Bjorndal 1985, 1997; USFWS and NMFS 1992). Those organisms that are caught in the surveys will be sampled according to the survey protocol. Species that meet the sampling criteria will be sampled for scientific purposes and not returned to the water, while the other species will be returned to the water alive, dead, or injured to the extent that they will subsequently die. All of the species that will be retained for further study are fish. Crabs, on the other hand, which are the preferred prey of loggerhead and Kemp's ridley sea turtles, will not be retained for further study, and thus would still be available as prey for loggerheads and Kemp's ridleys when returned to the water, as both of these species of sea turtles are known to eat a variety of live prey as well as scavenge dead organisms (Lutcavage and Musick 1985;

Keinath *et al.*1987; Dodd 1988; Burke *et al.*1993, 1994; Morreale and Standora 2005). Thus, the NJ ocean trawl survey is not expected to affect the availability of prey for loggerhead and Kemp's ridley sea turtles in the action area given that: (a) the sea turtle food items that are returned to the water could still be preyed upon by loggerheads and Kemp's ridleys, (b) the number of trawl tows for the study are limited in scope and duration, and (c) the priority species that will be retained for scientific analysis are all fish species, which are not the preferred prey for loggerhead and Kemp's ridley sea turtles (Keinath *et al.*1987; Lutcavage and Musick 1985; Burke *et al.*1993, 1994; Morreale and Standora 2005).

Atlantic sturgeon feed primarily on small benthic invertebrates and occasionally on small fish such as sand lance. Because of the small size or benthic nature of these prey species, it is unlikely that the surveys will capture any Atlantic sturgeon prey items. Thus, the NJ surveys will not affect the availability of prey for Atlantic sturgeon. Any effects to prey will be limited to minor disturbances to the bottom from the sampling gear. Because of this, we have determined that any effects to Atlantic sturgeon prey or foraging Atlantic sturgeon will be insignificant and discountable.

6.8 Effects to Habitat

The area to be surveyed is principally sand substrate (NEFMC 2007). A panel of experts has previously concluded that the effects of even light weight otter trawl gear would include: (1) the scraping or plowing of the doors on the bottom, sometimes creating furrows along their path, (2) sediment suspension resulting from the turbulence caused by the doors and the ground gear on the bottom, (3) the removal or damage to benthic or demersal species, and (4) the removal or damage to structure forming biota. The panel also concluded that the greatest impacts from otter trawls occur in high and low energy gravel habitats and in hard clay outcroppings, and that sand habitats were the least likely to be impacted (NREFHSC 2002). The areas to be surveyed for the NJ surveys include very few habitats that are purely gravel or hard clay—so few that the area encompassed by these habitats is insignificant compared to the area encompassed by sand and silt type habitats, which are more resilient to bottom trawling. For sea turtles and sturgeon, the effects on habitat due to bottom otter trawl gear would be felt as an effect on their benthic prey species. Similarly, effects to habitat of gillnet gear are expected to be minor. As stated above, the effects on sea turtle and Atlantic sturgeon prey items are expected to be insignificant.

7.0 CUMULATIVE EFFECTS

Cumulative effects as defined in 50 CFR 402.02 to include the effects of future State, tribal, local or private actions that are reasonably certain to occur within the action area considered in the biological opinion. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA. Ongoing Federal actions are considered in the "Environmental Baseline" section above.

Sources of human-induced mortality, injury, and/or harassment of Atlantic sturgeon and/or sea turtles in the action area that are reasonably certain to occur in the future include incidental mortalities in state-regulated fishing activities, vessel collisions, ingestion of plastic debris,

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pollution, global climate change, coastal development, and catastrophic events. While the combination of these activities may affect Atlantic sturgeon and sea turtles, preventing or slowing a species' recovery, the magnitude of these effects is currently unknown.

State Water Fisheries - Future recreational and commercial fishing activities in state waters may capture, injure or kill Atlantic sturgeon and sea turtles. However, it is not clear to what extent these future activities would affect listed species differently than the current state fishery activities described in the Environmental Baseline section. Atlantic sturgeon are captured and killed in fishing gear operating in the action area; however, at this time we are not able to quantify the number of interactions that occur. However, this Opinion assumes effects in the future would be similar to those in the past and are, therefore, reflected in the anticipated trends described in the status of the species and environmental baseline sections.

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, including 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 interactions with state water fisheries as well as the number of sea turtles injured or killed as a result of these interactions. 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. However, this Opinion assumes effects in the future would be similar to those in the past and are, therefore, reflected in the anticipated trends described in the status of the species and environmental baseline sections.

Vessel Interactions – NMFS' STSSN data indicate that vessel interactions are responsible for a large number of sea turtles strandings within the action area each year. Such collisions are reasonably certain to continue into the future. Collisions with boats can stun or easily kill sea turtles, and many stranded turtles have obvious propeller or collision marks (Dwyer *et al.*2003). However, it is not always clear whether the collision occurred pre- or post-mortem. NMFS believes that sea turtles takes by vessel interactions will continue in the future. An estimate of the number of sea turtles that will likely be killed by vessels is not available from data at this time. Similarly, we are unable at this time to assess the risk that vessel operations in the action area pose to Atlantic sturgeon. While vessel strikes have been documented in several rivers, the extent that interactions occur in the marine environment is currently unknown. However, this Opinion assumes effects in the future would be similar to those in the past and are, therefore,

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reflected in the anticipated trends described in the status of the species and environmental baseline sections.

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 sea turtles and Atlantic sturgeon. 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.

In the future, *global climate change* is expected to continue and may impact sea turtles and shortnose sturgeon and their habitat in the action area. However, as noted in the "Status of the Species" and "Environmental Baseline" sections above, given the likely rate of change associated with climate impacts (*i.e.*, the century scale), it is unlikely that climate related impacts will have a significant effect on the status of any species of sea turtles or Atlantic sturgeon over the temporal scale of the proposed action (*i.e.*, through 2014) or that in this time period, the abundance, distribution, or behavior of these species in the action area will change as a result of climate change related impacts.

8.0 INTEGRATION AND SYNTHESIS OF EFFECTS

NMFS has estimated that the surveys to be carried out by NJ from now until May 31, 2014, will result in the capture of 5 NWA DPS loggerhead sea turtles and no more than 1 leatherback or Kemp's ridley or green sea turtle. We also anticipate that the surveys will result in the capture of 109 Atlantic sturgeon consisting of 62 NYB DPS, 19 SA DPS, 20 CB DPS, and 8 GOM DPS Atlantic sturgeon. No injuries or mortality is anticipated during the trawl surveys and all affected sturgeon and sea turtles are expected to recover from capture without any reduction in fitness or impact on survival. We expect no more than one Atlantic sturgeon to be killed as a result of capture in the striped bass gillnet survey; given the distribution of Atlantic sturgeon in the action area and the location of the gillnet survey in the Delaware River and Bay, we expect that the mortality will be of a NYB DPS Atlantic sturgeon. As explained in the "Effects of the Action" section, all other effects to sea turtles and Atlantic sturgeon, including to their prey, will be insignificant or discountable.

In the discussion below, NMFS considers whether the effects of the proposed action reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of the listed species in the wild by reducing the reproduction, numbers, or distribution of loggerhead, Kemp's ridley, green or leatherback sea turtles and each of the four DPSs of Atlantic sturgeon. The purpose of this analysis is to determine whether the proposed action, in the context established by the status of the species, environmental baseline, and cumulative effects, would jeopardize the continued existence of any listed species.

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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 proposed action, NMFS summarizes the status of the species and considers whether the proposed action will result in reductions in reproduction, numbers or distribution of that species and then considers whether any reductions in reproduction, numbers or distribution resulting from the proposed action would reduce appreciably the likelihood of both the survival and recovery of that species, as those terms are defined for purposes of the federal Endangered Species Act.

8.1 Northwest Atlantic DPS of loggerhead sea turtles

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

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

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In this Opinion, NMFS has considered the potential impacts of the proposed action on the NWA DPS of loggerhead sea turtles. We have estimated that a total of 5 loggerheads are likely to be captured in the NJ ocean trawl surveys to be carried out from now until May 31, 2014. All five captured turtles are expected to be safely removed from the trawl gear and returned to the ocean without any injury or mortality. All other effects to loggerhead sea turtles, including effects to prey, are expected to be insignificant and discountable.

As there will be no injury or mortality to any individual loggerhead sea turtle and no effects to the prey base that would cause sea turtles to leave the action area to forage elsewhere, the NJ ocean trawl survey is not likely to reduce the numbers of loggerhead sea turtles in the action area, the numbers of loggerheads in any subpopulation or the species as a whole. Similarly, as the proposed action will not affect the fitness of any individual, no effects to reproduction are anticipated. The action is also not likely to affect the distribution of loggerhead sea turtles in the action area or affect the distribution of sea turtles throughout their range. Because effects are limited to capture, with no injury or mortality, there are not anticipated to be any population level impacts. Despite the threats faced by individual loggerhead sea turtles inside and outside of the action area, the proposed 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 proposed 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, no additional effects related to climate change to loggerhead sea turtles in the action area are anticipated over the life of the proposed action (*i.e.*, through 2012). NMFS has considered the effects of the proposed action in light of cumulative effects explained above, including climate change, and has concluded that even in light of the ongoing impacts of these activities and conditions, the conclusions reached above do not change.

Based on the information provided above, the capture of five or fewer NWA DPS loggerhead sea turtles in the NJ ocean trawl surveys will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of NWA DPS sea turtles; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of NWA DPS loggerhead sea turtles in the action area (related to the temporary capture and handling of captured individuals) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the proposed action will not appreciably reduce the likelihood that the NWA DPS 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

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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 proposed action is not expected to modify, curtail or destroy the range of the species since it will not result in a reduction in the number of NWA DPS loggerhead sea turtles and since it will not affect the overall distribution of the species other than to cause minor temporary adjustments in movements in the action area. The proposed action will not utilize NWA DPS loggerhead sea turtles for recreational, scientific or commercial purposes or affect the adequacy of existing regulatory mechanisms to protect this species. The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the NWA DPS. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the NWA DPS can be brought to the point at which they are no longer listed as threatened. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of this species.

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8.2 Leatherback sea turtles

Leatherback sea turtles are listed as "endangered" under the ESA. Leatherbacks are widely distributed throughout the oceans of the world, and are found in waters of the Atlantic, Pacific, and Indian Oceans, the Caribbean Sea, Mediterranean Sea, and the Gulf of Mexico (Ernst and Barbour 1972). Leatherback nesting occurs on beaches of the Atlantic, Pacific, and Indian Oceans as well as in the Caribbean (NMFS and USFWS 2007b). Leatherbacks face a multitude of threats that can cause death prior to and after reaching maturity. Some activities resulting in leatherback mortality have been addressed. There are some population estimates for leatherback sea turtles although there appears to be considerable uncertainty in the numbers. The most recent population size estimate for the North Atlantic alone is 34,000-94,000 adult leatherbacks (TEWG 2007; NMFS and USFWS 2007b).

Leatherback nesting in the eastern Atlantic (*i.e.*, off Africa) and in the Caribbean appears to be stable, but there is conflicting information for some sites and it is certain that some nesting groups (*e.g.*, St. John and St. Thomas, U.S. Virgin Islands) have been extirpated (NMFS and USFWS 1995). Data collected for some nesting beaches in the western Atlantic, including leatherback nesting beaches in the U.S., clearly indicate increasing numbers of nests (NMFS SEFSC 2001; NMFS and USFWS 2007b). However, declines in nesting have been noted for

beaches in the western Caribbean (NMFS and USFWS 2007b). The largest leatherback rookery in the western Atlantic remains along the northern coast of South America in French Guiana and Suriname. More than half the present world leatherback population is estimated to nest on the beaches in and close to the Marowijne River Estuary in Suriname and French Guiana (Hilterman and Goverse 2004). 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). Studies by Girondot *et al.*(2007) also suggest that the trend for the Suriname - French Guiana nesting population over the last 36 years is stable or slightly increasing.

Increased nesting by leatherbacks in the Atlantic is not expected to affect leatherback abundance in the Pacific where the abundance of leatherback sea turtles on nesting beaches has declined dramatically over the past 10 to 20 years (NMFS and USFWS 2007b). Although genetic analyses suggest little difference between Atlantic and Pacific leatherbacks (Bowen and Karl 2007), it is generally recognized that there is little to no genetic exchange between these turtles.

In this Opinion, NMFS has considered the potential impacts of the proposed action on leatherback sea turtles. We anticipate that one leatherback will be captured in the NJ ocean trawl surveys to be carried out from now through May 31, 2014. All captured turtles are expected to be safely removed from the trawl gear and returned to the ocean without any injury or mortality. All other effects to leatherback sea turtles, including effects to prey, are expected to be insignificant and discountable.

As there will be no injury or mortality to any individual leatherback sea turtle and no effects to the prey base that would cause sea turtles to leave the action area to forage elsewhere, the NJ ocean trawl survey is not likely to reduce the numbers of leatherback sea turtles in the action area, the numbers of leatherbacks in any subpopulation or the species as a whole. Similarly, as the proposed action will not affect the fitness of any individual, no effects to reproduction are anticipated. The action is also not likely to affect the distribution of leatherback sea turtles in the action area or affect the distribution of leatherback sea turtles throughout their range. Because effects are limited to capture, with no injury or mortality, there are not anticipated to be any population level impacts. Despite the threats faced by individual leatherback sea turtles inside and outside of the action area, the proposed 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 proposed action. While NMFS is not able to predict with precision how climate change will continue to impact leatherback sea turtles in the action area or how the species will adapt to climate-change related environmental impacts, no additional effects related to climate change to leatherback sea turtles in the action area are anticipated over the life of the proposed action (*i.e.*, through 2014). NMFS has considered the effects of the proposed action in light of cumulative effects explained above, including climate change, and has concluded that even in light of the ongoing impacts of these activities and conditions, the conclusions reached above do not change.

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Based on the information provided above, the capture of 1 leatherback sea turtle in the NJ ocean trawl surveys will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of leatherback sea turtles; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of leatherback sea turtles in the action area (related to the temporary capture and handling of captured individuals) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the proposed action will not appreciably reduce the likelihood that the leatherback sea turtle species 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 proposed action is not expected to modify, curtail or destroy the range of the species since it will not result in a reduction in the number of leatherback sea turtles and since it will not affect the overall distribution of the species other than to cause minor temporary adjustments in movements in the action area. The proposed action will not utilize leatherback sea turtles for recreational, scientific or commercial purposes or affect the adequacy of existing regulatory mechanisms to protect this species. The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of leatherback sea turtles. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that leatherback sea turtles can be brought to the point at which they are no longer listed as endangered. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of this species.

8.3 Kemp's ridley sea turtles

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

Nest count data provides the best available information on the number of adult females nesting each year. As is the case with the other sea 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. Estimates of the adult female nesting population reached a low of approximately 250-300 in 1985 (USFWS and NMFS 1992; TEWG 2000). From 1985 to 1999, the number of nests observed at Rancho Nuevo and nearby beaches increased at a mean rate of 11.3% per year (TEWG 2000). Current estimates suggest an adult female population of 7,000-8,000 Kemp's ridleys (NMFS and USFWS 2007c).

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

In this Opinion, NMFS has considered the potential impacts of the proposed action on Kemp's ridley sea turtles. As there will be no injury or mortality to any individual Kemp's ridley sea turtle and no effects to the prey base that would cause sea turtles to leave the action area to forage elsewhere, the capture of no more than 1 Kemp's ridley sea turtle during the NJ ocean trawl surveys carried out from now through May 31, 2014, is not likely to reduce the numbers of Kemp's ridley sea turtles in the action area, the numbers of Kemp's ridleys in any subpopulation or the species as a whole. Similarly, as the proposed action will not affect the fitness of any individual, no effects to reproduction are anticipated. The action is also not likely to affect the distribution of Kemp's ridley sea turtles in the action area or affect the distribution of sea turtles throughout their range. Because effects are limited to capture, with no injury or mortality, there are not anticipated to be any population level impacts. Despite the threats faced by individual Kemp's ridley sea turtles inside and outside of the action area, the proposed 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 proposed action. While

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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, no additional effects related to climate change to leatherback sea turtles in the action area are anticipated over the life of the proposed action (*i.e.*, through 2014). NMFS has considered the effects of the proposed action in light of cumulative effects explained above, including climate change, and has concluded that even in light of the ongoing impacts of these activities and conditions, the conclusions reached above do not change.

Based on the information provided above, the capture of two Kemp's ridley sea turtles in the NJ ocean trawl surveys will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of Kemp's ridley sea turtles; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of Kemp's ridley sea turtles in the action area (related to the temporary capture and handling of captured individuals) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the proposed action will not appreciably reduce the likelihood that the Kemp's ridley sea turtle species 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 proposed action is not expected to modify, curtail or destroy the range of the species since it will not result in a reduction in the number of Kemp's ridley sea turtles and since it will not affect the overall distribution of the species other than to cause minor temporary adjustments in movements in the action area. The proposed action will not utilize Kemp's ridley sea turtles for recreational, scientific or commercial purposes or affect the adequacy of existing regulatory mechanisms to protect this species. The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of Kemp's ridley sea turtles. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive

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fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that Kemp's ridley sea turtles can be brought to the point at which they are no longer listed as endangered. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of this species.

8.4 Green sea turtles

Green sea turtles are listed as both threatened and endangered under the ESA. Breeding colony populations in Florida and on the Pacific coast of Mexico are considered endangered while all others are considered threatened. Due to the inability to distinguish between these populations away from the nesting beach, for this Opinion, green sea turtles are considered endangered wherever they occur in U.S. waters. 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; Seminoff 2004; NMFS and USFWS 2007d). As is also the case with the other sea turtle species, green sea turtles face numerous threats on land and in the water that affect the survival of all age classes.

A review of 32 Index Sites distributed globally revealed a 48% to 67% decline in the number of mature females nesting annually over the last three generations (Seminoff 2004). For example, in the eastern Pacific, the main nesting sites for the green sea turtle are located in Michoacan, Mexico, and in the Galapagos Islands, Ecuador, where the number of nesting females exceeds 1,000 females per year at each site (NMFS and USFWS 2007d). Historically, however, greater than 20,000 females per year are believed to have nested in Michoacan alone (Cliffton *et al.*1982; NMFS and USFWS 2007d). However, the decline is not consistent across all green sea turtle nesting areas. Increases in the number of nests counted and, presumably, the numbers of mature females laying nests were recorded for several areas (Seminoff 2004; NMFS and USFWS 2007d). Of the 32 index sites reviewed by Seminoff (2004), the trend in nesting was described as: increasing for 10 sites, decreasing for 19 sites, and stable (no change) for 3 sites. Of the 46 green sea turtle nesting sites reviewed for the 5-year status review, the trend in nesting was described as increasing for 12 sites, decreasing for 4 sites, stable for 10 sites, and unknown for 20 sites (NMFS and USFWS 2007d). The greatest abundance of green sea turtle nesting in the western Atlantic occurs on beaches in Tortuguero, Costa Rica (NMFS and USFWS 2007d). Nesting in the area has increased considerably since the 1970s and nest count data from 1999-2003 suggest nesting by 17,402-37,290 females per year (NMFS and USFWS 2007d). One of the largest nesting sites for green sea turtles worldwide is still believed to be on the beaches of Oman in the Indian Ocean (Hirth 1997; Ferreira *et al.*2003; NMFS and USFWS 2007d). However, nesting data for this area has not been published since the 1980s and updated nest numbers are needed (NMFS and USFWS 2007d).

The results of genetic analyses show that green sea turtles in the Atlantic do not contribute to green sea turtle nesting elsewhere in the species' range (Bowen and Karl 2007). Therefore, increased nesting by green sea turtles in the Atlantic is not expected to affect green sea turtle abundance in other ocean basins in which the species occurs. However, the ESA-listing of green

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sea turtles as a species across ocean basins means that the effects of a proposed action must, ultimately, be considered at the species level for section 7 consultations. NMFS recognizes that the nest count data available for green sea turtles in the Atlantic clearly indicates increased nesting at many sites. However, NMFS also recognizes that the nest count data, including data for green sea turtles in the Atlantic, only provides information on the number of females currently nesting, and is not necessarily a reflection of the number of mature females available to nest or the number of immature females that will reach maturity and nest in the future. Given the late age to maturity for green sea turtles (20 to 50 years) (Balazs 1982; Frazer and Ehrhart 1985; Seminoff 2004), caution is urged regarding the trend for any of the nesting groups since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d).

In this Opinion, NMFS has considered the potential impacts of the proposed action on green sea turtles. As there will be no injury or mortality to the 1 green sea turtle we expect to be captured and no effects to the prey base that would cause sea turtles to leave the action area to forage elsewhere, the NJ ocean trawl surveys to be carried out from now until May 31, 2014, are not likely to reduce the numbers of green sea turtles in the action area, the numbers of greens in any subpopulation or the species as a whole. Similarly, as the proposed action will not affect the fitness of any individual, no effects to reproduction are anticipated. The action is also not likely to affect the distribution of green sea turtles in the action area or affect the distribution of sea turtles throughout their range. Because effects are limited to capture, with no injury or mortality, there are not anticipated to be any population level impacts. Despite the threats faced by individual green sea turtles inside and outside of the action area, the proposed 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 proposed action. While NMFS is not able to predict with precision how climate change will continue to impact green sea turtles in the action area or how the species will adapt to climate-change related environmental impacts, no additional effects related to climate change to green sea turtles in the action area are anticipated over the life of the proposed action (*i.e.*, through 2014). NMFS has considered the effects of the proposed action in light of cumulative effects explained above, including climate change, and has concluded that even in light of the ongoing impacts of these activities and conditions, the conclusions reached above do not change.

Based on the information provided above, the capture of one green sea turtle in the NJ ocean trawl surveys will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of green sea turtles; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the species; (3) and, the action will have only a minor and temporary effect on the distribution of green sea turtles in the action area (related to the temporary capture and handling of captured individuals) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the proposed action will not appreciably reduce

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the likelihood that the green sea turtle species 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 proposed action is not expected to modify, curtail or destroy the range of the species since it will not result in a reduction in the number of green sea turtles and since it will not affect the overall distribution of the species. The proposed action will not utilize green sea turtles for recreational, scientific or commercial purposes or affect the adequacy of existing regulatory mechanisms to protect this species. The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the species. There will not be a change in the status or trend of the species. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of green sea turtles. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that green sea turtles can be brought to the point at which they are no longer listed as endangered. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of this species.

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8.5 Atlantic sturgeon

8.5.1 Determination of DPS Composition

As explained above, the proposed action is likely to result in the capture of 109 or fewer Atlantic sturgeon. These captures are likely to occur in the ocean trawl surveys (70 individuals) and the striped bass gillnet surveys (39 individuals). We have considered the best available information to determine from which DPSs these individuals are likely to have originated. Using mixed stock analysis explained above, we have determined that Atlantic sturgeon in the action area likely originate from four DPSs at the following frequencies: NYB 58%, CB 18%, SA 17%, and GOM 7%. As such, of the 109 or fewer Atlantic sturgeon that will be captured during the course of these surveys between now and May 31, 2014, we expect that 62 of these Atlantic sturgeon would be New York Bight DPS origin, 19 will originate from the South Atlantic DPS, 20 will originate from the Chesapeake Bay DPS and 8 will originate from the Gulf of Maine DPS. We do not anticipate any mortality in the trawl survey. We expect that the mortality rate for Atlantic sturgeon caught in the striped bass gillnet survey will be very low (1%) and anticipate that no

more than 1 Atlantic sturgeon will be killed as a result of capture in the gillnet. Based on the location of the gillnet survey and the mixed stock analysis, we expect that the one mortality will be from the NYB DPS. Below, we consider these effects to each of the four DPSs.

8.5.2 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. The capture of a larvae in the Androscoggin River suggests that spawning may also be occurring in this river. No total population estimates are available. We have estimated, based on fishery-dependent data, that there are approximately 166 mature adults in the GOM DPS. Approximately 1/3 of adults are likely to spawn each year. GOM origin Atlantic sturgeon are affected by numerous sources of human induced mortality and habitat disturbance throughout the riverine and marine portions of their range. While there are some indications that the status of the GOM DPS may be improving, there is currently not enough information to establish a trend for any life stage or for the DPS as a whole.

NMFS has estimated that the proposed NJ marine surveys will result in the capture of 109 or fewer Atlantic sturgeon between now and May 31, 2014, of which 8 are expected to be GOM DPS Atlantic sturgeon. These captures are likely to occur in the ocean trawl surveys and the striped bass gillnet surveys. We do not anticipate any interactions in the striped bass seine survey or the Delaware Bay finfish trawl survey. The following analysis applies to anticipated effects on 8 individual from the GOM DPS, but given the nature of the effects (*i.e.*, non-lethal), it applies equally well to the worst case, which is the unlikely scenario of all 109 being from the GOM DPS. No injury or mortality is anticipated. The survival of any GOM DPS Atlantic sturgeon will not be affected by these surveys. As such, there will be no reduction in the numbers of GOM DPS Atlantic sturgeon and no change in the status of this species or its trend.

Reproductive potential of the GOM DPS is not expected to be affected in any way. As all sturgeon are anticipated to fully recover from capture and the short duration of any capture and handling (*i.e.*, less than 45 minutes total, 20-30 tow or gillnet set plus up to 10-15 minutes of handling time) will not cause a delay or disruption of any essential behavior including spawning, there will be no reduction in individual fitness or any future reduction in numbers of individuals. Additionally, as the proposed action will occur outside of the rivers where GOM DPS fish are expected to spawn (*i.e.*, the Kennebec River in Maine), the proposed 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 proposed 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 in the action area or elsewhere. Any effects to distribution will be minor and temporary and limited to the temporary capture and handling of individuals.

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Based on the information provided above, the capture of 8 or fewer GOM DPS Atlantic sturgeon surveys will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of GOM DPS Atlantic sturgeon; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the GOM DPS of Atlantic sturgeon; (3) and, the action will have only a minor and temporary effect on the distribution of GOM DPS Atlantic sturgeon in the action area (related to the temporary capture and handling of captured individuals) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the proposed action will not appreciably reduce the likelihood that the GOM DPS 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 proposed action is not expected to modify, curtail or destroy the range of the species since it will not result in a reduction in the number of GOM DPS Atlantic sturgeon and since it will not affect the overall distribution of GOM DPS Atlantic sturgeon. The proposed action will not utilize GOM DPS Atlantic sturgeon for recreational, scientific or commercial purposes or affect the adequacy of existing regulatory mechanisms to protect this species. The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, it is not expected to affect the persistence of the GOM DPS of Atlantic sturgeon. There will not be a change in the status or trend of the GOM DPS of Atlantic sturgeon. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the GOM DPS of Atlantic sturgeon. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the GOM DPS of Atlantic sturgeon can be brought to the point at which they are no longer listed as threatened. Based on the analysis presented herein, the proposed action, is not likely to appreciably reduce the survival and recovery of this species.

8.5.3 New York Bight DPS

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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. Kahnle *et al.*(2007) estimated that there is a mean annual total mature adult population of 863 Hudson River Atlantic sturgeon. Using fishery-dependent data we have estimated that there are 87 Delaware River origin adults; combined, we estimate a total adult population of 950 in the New York Bight DPS. As explained in the Status of the Species, we estimate that there are at least 2,850 subadults belonging to this 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 proposed NJ marine surveys will result in the capture of 109 or fewer Atlantic sturgeon between now and May 31, 2014, of which 62 are expected to be NYB DPS Atlantic sturgeon. These captures are likely to occur in the ocean trawl surveys and the striped bass gillnet surveys. We do not anticipate any interactions in the striped bass seine survey or the Delaware Bay finfish trawl survey. The following analysis applies to anticipated effects on 62 individuals, but given the nature of the effects, it applies equally well to the worst case, the unlikely scenario of all 109 being from the NYB DPS. The majority of individuals are likely to be Hudson River origin, but some may be Delaware River origin. We anticipate the mortality of one NYB DPS individual in the gillnet survey; given the mesh size used in the survey, we expect that the mortality will be a subadult. No injury or mortality is anticipated for any other captured sturgeon.

The mortality of one subadult Atlantic sturgeon from the NYB DPS subadult population represents a very small percentage of the subadult population (*i.e.*, less than 0.04% of the population). While the death of one subadult Atlantic sturgeon will reduce the number of NYB DPS Atlantic sturgeon compared to the number that would have been present absent the proposed action, it is not likely that this reduction in numbers will change the status of this species as this loss represents a very small percentage of the subadult (less than 0.035%) population and an even smaller percentage of the overall population of the DPS (juveniles, subadults and adults combined).

The reproductive potential of the NYB DPS will not be affected in any way other than through a reduction in numbers of individuals. The loss of one subadult would have the effect of reducing the amount of potential reproduction as any dead NYB DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in an extremely small reduction in the number of eggs laid or larvae produced in future years and similarly, an extremely small effect on the strength of subsequent year classes. Even considering the potential future spawners that would be produced by the individual that would be killed as a result of the proposed action, any effect to future year classes is anticipated to be extremely small and would not change the status of this species. Reproductive potential of other captured individuals is not expected to be affected in any way. As no injury is anticipated and

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there will be no impacts on fitness of these captured individuals, and any captured fish will be released within 45 minutes (20-30 tow or gillnet set plus up to 10-15 minutes of handling time), there will not be any delay or disruption of any essential behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. Additionally, any delay in migration to the spawning grounds will be limited to less than 45 minutes and is not anticipated to impact the success of reproduction.

The proposed action will also not affect the spawning grounds within either the Delaware or Hudson rivers where NYB DPS fish spawn. The action will also not create any barrier to pre-spawning sturgeon accessing the overwintering sites or the spawning grounds.

The proposed 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 in the Hudson River or elsewhere. Any effects to distribution will be minor and temporary and limited to the temporary capture in the trawl or gillnet.

Based on the information provided above, the death of one suabult NYB DPS Atlantic sturgeon 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 one subadult NYB DPS Atlantic sturgeon represents an extremely small percentage of the species as a whole; (2) the death of one subadult NYB DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of this subadult NYB DPS Atlantic sturgeon is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of this subadult NYB DPS Atlantic sturgeon is likely to have such a small effect on reproductive output that the loss of these individuals will not change the status or trends of the species; (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 proposed 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

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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 proposed 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 proposed 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 proposed action is likely to result in the capture of Atlantic sturgeon and the mortality of one subadult NYB DPS Atlantic sturgeon; however, as explained above, the loss of this individual 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 proposed 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 proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that NYB DPS can be brought to the point at which they are no longer listed as endangered or threatened. Based on the analysis presented herein, the proposed action, resulting in the mortality of one subadult NYB DPS Atlantic sturgeon, is not likely to appreciably reduce the survival and recovery of this species.

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8.5.4 Chesapeake Bay DPS

Individuals originating from the CB DPS are likely to occur in the action area. The CB DPS has been listed as endangered. 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 329 adults in the James River population. Because the James River is the only river in this DPS known to support spawning, this is also an estimate of the total number of adults in the Chesapeake Bay 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 proposed NJ marine surveys will result in the capture of 109 or fewer Atlantic sturgeon between now and May 31, 2014, of which 20 are expected to be CB DPS Atlantic sturgeon. These captures are likely to occur in the ocean trawl surveys and the striped bass gillnet surveys. We do not anticipate any interactions in the striped bass seine survey or the

Delaware Bay finfish trawl survey. The following analysis applies to anticipated effects on 20 individual from the CB DPS, but given the nature of the effects, it applies equally well to the worst case, the unlikely scenario of all 109 being from the CB DPS. No injury or mortality is anticipated.

Reproductive potential of the CB DPS is not expected to be affected in any way. As no injury or mortality is anticipated and there will be no impacts on fitness of captured individuals, and any captured fish will be released within 45 minutes (20-30 minute tow or gillnet set plus up to 15 minutes of handling time), there will not be any delay or disruption of any essential behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. As the proposed action will occur outside of the rivers where CB DPS fish are expected to spawn (e.g., the James River in Virginia), the proposed 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 proposed 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 in the action area or elsewhere. Any effects to distribution will be minor and temporary and limited to temporary capture in the trawl.

Based on the information provided above, the capture of CB DPS Atlantic sturgeon in the NJ surveys will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of CB DPS Atlantic sturgeon; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the CB DPS of Atlantic sturgeon; (3) and, the action will have only a minor and temporary effect on the distribution of CB DPS Atlantic sturgeon in the action area (related to temporary capture of individuals in the trawl) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the proposed action will not appreciably reduce the likelihood that the CB DPS 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.

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The proposed action is not expected to modify, curtail or destroy the range of the species since it will result in no reduction in the number of CB DPS Atlantic sturgeon and since it will not affect the overall distribution of CB DPS Atlantic sturgeon. The proposed action will not utilize CB DPS Atlantic sturgeon for recreational, scientific or commercial purposes or affect the adequacy of existing regulatory mechanisms to protect this species. The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, there is not expected to affect the persistence of the CB DPS of Atlantic sturgeon. There will not be a change in the status or trend of the CB DPS of Atlantic sturgeon. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the CB DPS of Atlantic sturgeon. The effects of the proposed action will not hasten the extinction timeline or otherwise increase the danger of extinction since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the CB DPS of Atlantic sturgeon can be brought to the point at which they are no longer listed as endangered. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of this species.

8.5.5 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 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. Currently, there are an estimated 343 spawning adults in the Altamaha and 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 598. 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 598 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. 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 proposed NJ marine surveys will result in the capture of 109 or fewer Atlantic sturgeon between now and May 31, 2014, of which 19 are expected to be SA DPS Atlantic sturgeon. These captures are likely to occur in the ocean trawl surveys and the striped bass gillnet surveys. We do not anticipate any interactions in the striped bass seine survey or the Delaware Bay finfish trawl survey. The following analysis applies to anticipated effects on 19

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individual from the SA DPS, but given the nature of the effects, it applies equally well to the worst case, the unlikely scenario of all 109 being from the SA DPS. No injury or mortality is anticipated.

Reproductive potential of the SA DPS is not expected to be affected in any way. As no injury or mortality is anticipated and there will be no impacts on fitness of captured individuals, and any captured fish will be released within 45 minutes (20-30 minute tow or gillnet set plus up to 15 minutes of handling time), there will not be any delay or disruption of any essential behavior including spawning; there will also be no reduction in individual fitness or any future reduction in numbers of individuals. As the proposed action will occur outside of the rivers where SA DPS fish are expected to spawn (e.g., rivers in Georgia and Florida), the proposed 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 proposed 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 in the action area or elsewhere. Any effects to distribution will be minor and temporary and limited to temporary capture in the trawl.

Based on the information provided above, the capture of SA DPS Atlantic sturgeon in the NJ surveys will not appreciably reduce the likelihood of survival of this species (*i.e.*, it will not increase the risk of extinction faced by this species) given that: (1) there will be no mortality and therefore, no reduction in the numbers of SA DPS Atlantic sturgeon; (2) there will be no effect to the fitness of any individuals and no effect on reproductive output of the SA DPS of Atlantic sturgeon; (3) and, the action will have only a minor and temporary effect on the distribution of SA DPS Atlantic sturgeon in the action area (related to temporary capture of individuals in the trawl) and no effect on the distribution of the species throughout its range.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the proposed action will not appreciably reduce the likelihood that the SA DPS 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 proposed action is not expected to modify, curtail or destroy the range of the species since it will result in no reduction in the number of SA DPS Atlantic sturgeon and since it will not affect

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the overall distribution of SA DPS Atlantic sturgeon. The proposed action will not utilize SA DPS Atlantic sturgeon for recreational, scientific or commercial purposes or affect the adequacy of existing regulatory mechanisms to protect this species. The proposed action is not likely to result in any mortality or reductions in fitness or future reproductive output and therefore, there is not expected to affect the persistence of the SA DPS of Atlantic sturgeon. There will not be a change in the status or trend of the SA DPS of Atlantic sturgeon. As there will be no reduction in numbers or future reproduction the action would not cause any reduction in the likelihood of improvement in the status of the SA DPS of Atlantic sturgeon. The effects of the proposed action will not hasten the extinction timeline or otherwise increase the danger of extinction since the action will not cause any mortality or reduction of overall reproductive fitness for the species. The effects of the proposed action will also not reduce the likelihood that the status of the species can improve to the point where it is recovered and could be delisted. Therefore, the proposed action will not appreciably reduce the likelihood that the SA DPS of Atlantic sturgeon can be brought to the point at which they are no longer listed as endangered. Based on the analysis presented herein, the proposed action is not likely to appreciably reduce the survival and recovery of this species.

9.0 CONCLUSION

After reviewing the best available information on the status of endangered and threatened species under NMFS jurisdiction, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects, it is NMFS's biological opinion that the proposed action may adversely affect but is not likely to jeopardize the continued existence of Kemp's ridley, green, or leatherback sea turtles; the NWA DPS of loggerhead sea turtles; or the GOM, NYB, CB or SA DPSs of Atlantic sturgeon. We have also determined that the proposed action is not likely to adversely affect shortnose sturgeon, North Atlantic right whales, humpback whales, fin or sei whales. Because no critical habitat is designated in the action area, none will be affected by the action.

10.0 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA prohibits the take of endangered species of fish and wildlife. "Fish and wildlife" is defined in the ESA "as any member of the animal kingdom, including without limitation any mammal, fish, bird (including any migratory, non-migratory, or endangered bird for which protection is also afforded by treaty or other international agreement), amphibian, reptile, mollusk, crustacean, arthropod or other invertebrate, and includes any part, product, egg, or offspring thereof, or the dead body or parts thereof." 16 U.S.C. 1532(8). "Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by NMFS to include any act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation that actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns including breeding, spawning, rearing, migrating, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. "Otherwise lawful activities" are those actions that meet all State and Federal legal requirements except for the prohibition against taking in ESA Section 9 (51 FR 19936, June

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3, 1986), which would include any state endangered species laws or regulations. Section 9(g) makes it unlawful for any person "to attempt to commit, solicit another to commit, or cause to be committed, any offense defined [in the ESA.]" 16 U.S.C. 1538(g). See also 16 U.S.C. 1532(13)(definition of "person"). Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

The measures described below are non-discretionary, and must be undertaken by USFWS so that they become binding conditions for the exemption in section 7(o)(2) to apply. USFWS has a continuing duty to regulate the activity covered by this Incidental Take Statement. If USFWS (1) fails to assume and implement the terms and conditions or (2) fails to require the State of New Jersey to adhere to the terms and conditions of the Incidental Take Statement through enforceable terms that are added to grants, permits and/or contracts as appropriate, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, USFWS or NJ (as the group carrying out the action) must report the progress of the action and its impact on the species to the NMFS as specified in the Incidental Take Statement [50 CFR §402.14(i)(3)] (See U.S. Fish and Wildlife Service and National Marine Fisheries Service's Joint Endangered Species Act Section 7 Consultation Handbook (1998) at 4-49).

10.1 Anticipated Amount or Extent of Incidental Take

Based on the information presented in the Opinion, we anticipate that the surveys described in this Opinion, to be funded by FWS and carried out by NJ between now and May 31, 2014, will result in the capture of:

- Five NWA DPS loggerhead sea turtles;
- 1 Kemp's ridley sea turtle or 1 green sea turtle or 1 leatherback sea turtle;
- A total of no more than 109 Atlantic sturgeon. Based on mixed stock analyses, we anticipate that 62 of the Atlantic sturgeon will be NYB DPS origin, 20 will be CB DPS origin, 19 will be SA DPS origin and 8 will be GOM DPS origin; and, the mortality of 1 NYB DPS Atlantic sturgeon.

As explained in the "Effects of the Action" section of the Opinion, none of these sea turtles are expected to die, immediately or later, as a result of capture in the trawl gear. Only one of the captured Atlantic sturgeon is likely to die. This level of incidental take is anticipated for the entire period considered in this Opinion. All captures of sea turtles are anticipated to occur in the ocean trawl surveys and all captures of Atlantic sturgeon are anticipated to occur in the ocean trawl survey or the striped bass gillnet survey. The only lethal take anticipated is one Atlantic sturgeon in the striped bass gillnet survey. In the accompanying Opinion, we determined that this level of anticipated take is not likely to result in jeopardy to any species of sea turtle or to any DPS of Atlantic sturgeon.

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10.2 Reasonable and Prudent Measures

In order to effectively monitor the effects of this action, it is necessary to monitor the impacts of the proposed action to document the amount of incidental take (*i.e.*, the number of sea turtles and Atlantic sturgeon captured, collected, injured or killed) and to examine any sea turtles or Atlantic sturgeon that are captured during this monitoring. Monitoring provides information on the characteristics of the turtles and sturgeon encountered and may provide data which will help develop more effective measures to avoid future interactions with listed species. We do not anticipate any additional injury or mortality to be caused by handling and examining sea turtles and sturgeon as required in the RPMs. All live animals are to be released back into the water following the required documentation.

NMFS believes the following reasonable and prudent measures are necessary or appropriate to minimize and monitor impacts of incidental take of listed sea turtles and Atlantic sturgeon:

1. Any listed species caught during the survey must be handled and resuscitated according to established procedures.

2. Any listed species caught and retrieved in the sampling gear must be identified to species.

3. Any listed species caught and retrieved in the sampling gear must be properly documented.

4. NMFS NERO must be notified regarding all interactions with or observations of listed species.

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10.3 Terms and Conditions

In order to be exempt from prohibitions of section 9 of the ESA, USFWS must comply with the following terms and conditions of the Incidental Take Statement, which implement the reasonable and prudent measures described above and outline required reporting/monitoring requirements. These terms and conditions are non-discretionary. Any taking that is in compliance with the terms and conditions specified in this Incidental Take Statement shall not be considered a prohibited taking of the species concerned (ESA Section 7(o)(2)).

1. To implement RPM #1 above, USFWS must ensure that NJ has copies of the sea turtle handling and resuscitation requirements found at 50 CFR 223.206(d)(1) and as reproduced in Attachment A to the vessel operator prior to the commencement of any on-water activity. USFWS must also ensure that NJ carries out these handling and resuscitation procedures as appropriate.

2. To implement RPM#1 above, USFWS must ensure that NJ staff give priority to handling and processing any sea turtles or sturgeon that are captured in the sampling gear. Handling times must be minimized for these species.

3. To implement RPM#1 above, USFWS must ensure that NJ staff resuscitate any Atlantic sturgeon that may appear to be dead by providing a running source of water over the gills.

4. To implement RPM#1 above, USFWS must ensure that NJ has a Passive Integrated Transponder (PIT) tag reader on board all vessels used for the ocean trawl survey and the striped bass gillnet survey and that this reader is used to scan any captured Atlantic sturgeon for tags. Any recorded tags must be reported to the USFWS tagging database. Any untagged sturgeon must be tagged with PIT tags and the tag numbers recorded and reported to the USFWS tagging database.

5. To comply with RPM #2 above, USFWS must ensure that NJ has at least one crew member who is experienced in the identification of western North Atlantic sea turtles and sturgeon on the vessel(s) used for the ocean trawl survey and the striped bass gillnet survey at all times that the on-water survey work is conducted. Experience would include personnel that have received training as a NMFS fisheries observer or who have career experience in the identification of western North Atlantic sea turtles and sturgeon. Information provided as Appendix B can aid in species identification.

6. To comply with RPM #2 above, USFWS must ensure that NJ obtain genetic samples from all captured Atlantic sturgeon. This must be done in accordance with the procedures provided in Appendix C.

7. To comply with RPM #3, all sea turtles and sturgeon must be weighed, measured and photographed. The condition of each animal must be recorded and any injuries documented.

8. To comply with RPM #3, any dead Atlantic sturgeon must be retained and held in cold storage until disposal can be discussed with NMFS.

9. To comply with RPM #4, USFWS must ensure that NJ notifies NMFS PRD within 24 hours of any interaction with a listed species. These reports should be sent by fax (978)281-9394 or e-mail (Incidental.take@noaa.gov). For purposes of monitoring the incidental take of sea turtles and sturgeon during the surveys, reports must be made for any sea turtle or Atlantic sturgeon: (a) found alive, dead, or injured within the sampling gear; (b) found alive, dead, or injured and retained on any portion of the sampling gear outside of the net bag; or (c) interacting with the vessel and gear in any other way must be reported to NMFS.

10. To comply with RPM #4, USFWS must ensure that NJ provides a written report to NMFS NERO within 30 days of any interaction between an ESA-listed sea turtle and the gear and/or vessel used during the survey. The report must include: a clear photograph of the animal (multiple views if possible, including at least one photograph of the head scutes); identification of the animal to the species level; GPS or Loran coordinates describing the location of the interaction; time of interaction; date of interaction; condition of the animal upon retrieval (alive uninjured, alive injured, fresh dead, decomposed, comatose or

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unresponsive); the condition of the animal upon return to the water; GPS or Loran coordinates of the location at which it was released; and a description of the care or handling provided. This report must be sent to the NMFS Northeast Regional Office, Attn: Section 7 Coordinator, 55 Great Republic Drive, Gloucester, MA 01930.

11. To comply with RPM #4, USFWS must ensure that NJ provide a written report to NMFS NERO within 60 days of completion of the on-water work, indicating either that no interactions with ESA-listed species occurred, or providing the total number of interactions that occurred with ESA-listed species. Any reports required by Term and Condition 9 that have not been provided to NMFS NERO must be included in this report. This report must be sent to the NMFS Northeast Regional Office, Attn: Section 7 Coordinator, 55 Great Republic Drive, Gloucester, MA 01930.

The reasonable and prudent measures, with their implementing terms and conditions, are designed to minimize and monitor the impact of incidental take that might otherwise result from the proposed action. Specifically, these RPMs and Terms and Conditions will ensure that NMFS and NJ monitor the impacts of the NJ surveys in a way that allows for the detection, identification and reporting of all interactions with listed species. The discussion below explains why each of these RPMs and Terms and Conditions are necessary or appropriate to minimize or monitor the level of incidental take associated with the proposed action. The RPMs and terms and conditions involve only a minor change to the proposed action.

RPM #1 and the accompanying Term and Condition establish the requirements for handling sea turtles and Atlantic sturgeon captured in gear used in the surveys in order to avoid the likelihood of injury to these species from the hauling, handling, and emptying of the trawl gear.

RPMs #2-4 and the accompanying Terms and Conditions specify the collection of information for any ESA-listed species observed captured in the gear. This is essential for monitoring the level of incidental take associated with the proposed action. 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. NMFS has received no reports of injury or mortality to any sturgeon sampled in this way.

11.0 CONSERVATION RECOMMENDATIONS

In addition to section 7(a)(2), which requires agencies to ensure that proposed actions are not likely to jeopardize the continued existence of listed species, section 7(a)(1) of the ESA places a responsibility on all Federal agencies to utilize their authorities in furtherance of the purposes of the ESA by carrying out programs for the conservation of endangered and threatened species. Conservation Recommendations are discretionary activities designed to minimize or avoid

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adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. The following additional measures are recommended regarding incidental take and sea turtle conservation:

1. USFWS should advise the Principal Investigator for the NJ surveys to provide guidance, before each survey cruise, to the vessel crew members (including scientific crew and vessel operators) to the effect that: (a) all personnel are alert to the possible presence of sea turtles and Atlantic sturgeon in the study area, (b) care must be taken when emptying the trawl gear to avoid damage to sea turtles and Atlantic sturgeon that may be caught in the trawl but are not visible upon retrieval of the gear, and (c) the trawl is emptied as quickly as possible after retrieval in order to determine whether sea turtles or Atlantic sturgeon are present in the gear.

12.0 REINITIATION OF CONSULTATION

This concludes formal consultation on the 2012 NJ marine surveys as funded by USFWS. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In the event that the amount or extent of incidental take is exceeded, Section 7 consultation must be reinitiated immediately.

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13.0 LITERATURE CITED

Aguilar, A. 2002. Fin whale, *Balaenoptera physalus*. Pages 435-438 in W.F. Perrin, B. Würsig, and J.G.M. Thewissen, eds. Encyclopedia of Marine Mammals. San Diego: Academic Press.

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

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

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

ASSRT (Atlantic Sturgeon Status Review Team). 2007. Status review of Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*). National Marine Fisheries Service. February 23, 2007. 188 pp.

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

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

Bain, Mark B., N. Haley, D. L. Peterson, K. K. Arend, K. E. Mills, P. J. Sullivan. 2000. Annual meeting of American fisheries Society. EPRI-AFS Symposium: Biology, Management and Protection of Sturgeon. St. Louis, MO. 23-24 August 2000.

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

Balazs, G.H. 1982. Growth rates of immature green turtles in the Hawaiian Archipelago. Pages 117-125 in K.A. Bjorndal, ed. Biology and conservation of sea turtles. Washington, D.C.: Smithsonian Institution Press.

Balazs, G.H. 1985. Impact of ocean debris on marine turtles: entanglement and ingestion. Pages 387-429 in R.S. Shomura and O. Yoshida, eds. Proceedings of the Workshop on the Fate and Impact of Marine Debris, 27-29 November 1984, Honolulu, Hawaii. NOAA Technical Memorandum NMFS-SWFSC-54.

INACTIVE

Bjorndal, K.A. 1985. Nutritional ecology of sea turtles. Copeia 1985(3):736-751.

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

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

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

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

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

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

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

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

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

Burke, V.J., E.A. Standora, and S.J. Morreale. 1993. Diet of juvenile Kemp's ridley and loggerhead sea turtles from Long Island, New York. Copeia 1993(4):1176-1180.

Burke, V.J., S.J. Morreale, and E.A. Standora. 1994. Diet of the Kemp's ridley sea turtle, *Lepidochelys kempii*, in New York waters. Fishery Bulletin 92:26-32.

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

INACTIVE

Caron, F., D. Hatin, and R. Fortin. 2002. Biological characteristics of adult Atlantic sturgeon (*Acipenser oxyrinchus*) in the Saint Lawrence River estuary and the effectiveness of management rules. Journal of Applied Ichthyology 18: 580-585.

Carr, A.R. 1963. Panspecific reproductive convergence in *Lepidochelys kempi*. Ergebnisse der Biologie 26:298-303.

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

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

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

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

Clapham. P. 2002. Humpback whale, *Megaptera novaeangliae*. Pages 589-592 in W.F. Perrin, B. Würsig, and J.G.M. Thewissen, eds. Encyclopedia of Marine Mammals. San Diego: Academic Press.

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

Collins, M. R., S. G. Rogers, T. I. J. Smith, and M. L. Moser. 2000. Primary factors affecting sturgeon populations in the southeastern United States: fishing mortality and degradation of essential habitats. Bulletin of Marine Science 66: 917-928.

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

Crance, J. H. 1987. Habitat suitability index curves for anadromous fishes. *In*: Common

INACTIVE

Strategies of Anadromous and Catadromous Fishes, M. J. Dadswell (ed.). Bethesda, Maryland, American Fisheries Society. Symposium 1: 554.

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

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

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

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

Dodd, C.K. 1988. Synopsis of the biological data on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758). U.S. Fish and Wildlife Service Biological Report 88(14):1-110.

Doughty, R.W. 1984. Sea turtles in Texas: a forgotten commerce. Southwestern Historical Quarterly 88:43-70.

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

Dunton, K.J., A. Jordaan, K.A. McKown, D.O. Conover, and M.J. Frisk. 2010. Abundance and distribution of Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) within the Northwest Atlantic Ocean, determined from five fishery-independent surveys. Fishery Bulletin 108:450-465.

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

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

Dwyer, K.L., C.E. Ryder, and R. Prescott. 2003. Anthropogenic mortality of leatherback sea turtles in Massachusetts waters. Page 260 in J.A. Seminoff, compiler. Proceedings of the

INACTIVE

Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503.

Eckert, S.A. 1999. Global distribution of juvenile leatherback turtles. Hubbs-Sea World Research Institute Technical Report 99-294:1-13.

Eckert, S.A., and J. Lien. 1999. Recommendations for eliminating incidental capture and mortality of leatherback sea turtles, *Dermochelys coriacea*, by commercial fisheries in Trinidad and Tobago. A report to the Wider Caribbean Sea Turtle Conservation Network (WIDECAST). Hubbs-Sea World Research Institute Technical Report 2000-310:1-7.

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

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

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

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

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

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

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

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

Epperly, S., L. Avens, L. Garrison, T. Henwood, W. Hoggard, J. Mitchell, J. Nance, J. Poffenberger, C. Sasso, E. Scott-Denton, and C. Yeung. 2002. Analysis of sea turtle bycatch

INACTIVE

in the commercial shrimp fisheries of southeast U.S. waters and the Gulf of Mexico. NOAA Technical Memorandum NMFS-SEFSC-490:1-88.

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

Ernst, C.H., and R.W. Barbour. 1972. Turtles of the United States. Lexington: University Press of Kentucky.

Eyler, S., T. Meyer, S. Michaels, and B. Spear. 2007. Review of the fishery management plan in 2006 for horseshoe crab (*Limulus polyphemus*). Prepared by the ASMFC Horseshoe Crab Plan Review Team. 15pp.

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

Fernandes, SJ. Et al. 2010.

Finkbeiner, E.M., B.P. Wallace, J.E. Moore, R.L. Lewison, L.B. Crowder, and A.J. Read. 2011. Cumulative estimates of sea turtle bycatch and mortality in USA fisheries between 1990 and 2007. Biological Conservation 144(11): 2719-2727.

FPL (Florida Power and Light Company) and Quantum Resources. 2005. Florida Power and Light Company, St. Lucie Plant Annual Environmental Operating Report, 2002. 57 pp.

Frazer, N.B., and L.M. Ehrhart. 1985. Preliminary growth models for green, *Chelonia mydas*, and loggerhead, *Caretta caretta*, turtles in the wild. Copeia 1985(1):73-79.

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

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

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

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

INACTIVE

#PRD-2010-10, Revised April 2011, 20p.

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

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

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

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

GMFMC (Gulf of Mexico Fishery Management Council). 2007. Amendment 27 to the Reef Fish FMP and Amendment 14 to the Shrimp FMP to end overfishing and rebuild the red snapper stock. Tampa, Florida: Gulf of Mexico Fishery Management Council. 490 pp. with appendices.

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

Graff, D. 1995. Nesting and hunting survey of the turtles of the island of Sao Tomé. Progress Report July 1995, ECOFAC Componente de Sao Tomé e Príncipe, 33 pp.

Hawkes, L.A., A.C. Broderick, M.H.Godfrey, and B.J. Godley. 2005. Status of nesting loggerhead turtles *Caretta caretta* at Bald Head Island (North Carolina, USA) after 24 years of intensive monitoring and conservation. Oryx 39(1):65-72.

Hawkes, L.A., A.C. Broderick, M.S. Coyne, M.H. Godfrey, L.-F. Lopez-Jurado, P. Lopez-Suárez, S.E. Merino, N. Varo-Cruz, and B.J. Godley. 2006. Phenotypically linked dichotomy in sea turtle foraging requires multiple conservation approaches. Current Biology 16:990-995.

Hawkes, L.A., A.C. Broderick, M.H. Godfrey, and B.J. Godley. 2007. Investigating the potential impacts of climate change on a marine turtle population. Global Change Biology 13:1-10.

Hawkes, L.A., A.C. Broderick, M.H. Godfrey, and B.J. Godley. 2009. Climate change and marine turtles. Endangered Species Research 7:137-154.

INACTIVE

Hays, G.C., A.C. Broderick, F. Glen, B.J. Godley, J.D.R. Houghton, and J.D. Metcalfe. 2002. Water temperature and internesting intervals for loggerhead (*Caretta caretta*) and green (*Chelonia mydas*) sea turtles. Journal of Thermal Biology 27:429-432.

Henwood, T.A., and W. Stuntz. 1987. Analysis of sea turtle captures and mortalities during commercial shrimp trawling. Fishery Bulletin 85(4):813-817.

Hilterman, M.L., and E. Goverse. 2004. Annual report of the 2003 leatherback turtle research and monitoring project in Suriname. World Wildlife Fund - Guianas Forests and Environmental Conservation Project (WWF-GFECP) Technical Report of the Netherlands Committee for IUCN (NC-IUCN), Amsterdam, the Netherlands. 21 pp.

Hirth, H.F. 1971. Synopsis of biological data on the green sea turtle, Chelonia mydas. FAO Fisheries Synopsis 85:1-77.

Hirth, H.F. 1997. Synopsis of the biological data of the green turtle, *Chelonia mydas* (Linnaeus 1758). USFWS Biological Report 97(1):1-120.

Holland, B.F., Jr. and G.F. Yelverton. 1973. Distribution and biological studies of anadromous fishes offshore North Carolina. North Carolina Department of Natural and Economic Resources, Division of Commercial and Sports Fisheries, Morehead City. Special Scientific Report 24:1-132.

Horwood, J. 2002. Sei whale, *Balaenoptera borealis*. Pages 1069-1071 in W.F. Perrin, B. Würsig, and J.G.M. Thewissen, eds. Encyclopedia of Marine Mammals. San Diego: Academic Press.

Hulme, P.E. 2005. Adapting to climate change: is there scope for ecological management in the face of global threat? Journal of Applied Ecology 43: 617-627.

Innis, C., C. Merigo, K. Dodge, M. Tlusty, M. Dodge, B. Sharp, A. Myers, A. McIntosh, D. Wunn, C. Perkins, T.H. Herdt, T. Norton, and M. Lutcavage. 2010. Health Evaluation of Leatherback Turtles (*Dermochelys coriacea*) in the Northwestern Atlantic During Direct Capture and Fisheries Gear Disentanglement. Chelonian Conservation and Biology, 9(2):205-222.

IPCC (Intergovernmental Panel on Climate Change) 2007. Fourth Assessment Report. Valencia, Spain.

Intergovernmental Panel on Climate Change. 2007. Summary for Policymakers. *In* Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (editors). Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom, and New York, New

INACTIVE

York, USA.

James, M.C., C.A. Ottensmeyer, and R.A. Myers. 2005a. Identification of high-use habitat and threats to leatherback sea turtles in northern waters: new directions for conservation. Ecology Letters 8:195-201.

James, M.C., R.A. Myers, and C.A. Ottenmeyer. 2005b. Behaviour of leatherback sea turtles, *Dermochelys coriacea*, during the migratory cycle. Proceedings of the Royal Society of London B 272:1547-1555.

Kahnle, A.W., K.A. Hattala, K.A. McKown. 2007. Status of Atlantic sturgeon of the Hudson River Estuary, New York, USA. American Fisheries Society Symposium. 56:347-363.

Kasparek, M., B.J. Godley, and A.C. Broderick. 2001. Nesting of the green turtle, *Chelonia mydas*, in the Mediterranean: a review of status and conservation needs. Zoology in the Middle East 24:45-74.

Keinath, J.A., J.A. Musick, and R.A. Byles. 1987. Aspects of the biology of Virginia's sea turtles: 1979-1986. Virginia Journal of Science 38(4):329-336.

Kenney, R.D. 2002. North Atlantic, North Pacific and Southern Right Whales. Pages 806-813 in W.F. Perrin, B. Würsig, and J.G.M. Thewissen, eds. Encyclopedia of Marine Mammals. San Diego: Academic Press.

King, T. L., B. A. Lubinski, and A. P. Spidle. 2001. Microsatellite DNA variation in Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) and cross-species amplification in the Acipenseridae. Conservation Genetics 2: 103-119.

Kuller, Z. 1999. Current status and conservation of marine turtles on the Mediterranean coast of Israel. Marine Turtle Newsletter 86:3-5.

Kynard, B. and M. Horgan. 2002. Ontogenetic behavior and migration of Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*, and shortnose sturgeon, *A. brevirostrum*, with notes on social behavior. Environmental Behavior of Fishes 63: 137-150.

Lagueux, C.J. 1998. Demography of marine turtles harvested by Miskitu Indians of Atlantic Nicaragua. Page 90 in R. Byles and Y. Fernandez, compilers. Proceedings of the Sixteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-412.

Laney, R.W., J.E. Hightower, B.R. Versak, M.F. Mangold, W.W. Cole Jr., and S.E. Winslow.

INACTIVE

2007. Distribution, habitat use, and size of Atlantic sturgeon captured during cooperative winter tagging cruises, 1988–2006. Pages 167-182. *In* J. Munro, D. Hatin, J. E. Hightower, K. McKown, K. J. Sulak, A. W. Kahnle, and F. Caron, (eds.) Anadromous sturgeons: habitats, threats, and management. Am. Fish. Soc. Symp. 56, Bethesda, MD.

Laurent, L., P. Casale, M.N. Bradai, B.J. Godley, G. Gerosa, A.C. Broderick, W. Schroth, B. Schierwater, A.M. Levy, D. Freggi, E.M. Abd El-Mawla, D.A. Hadoud, H.E. Gomati, M. Domingo, M. Hadjichristophorou, L. Kornaraki, F. Demirayak, and C. Gautier. 1998. Molecular resolution of the marine turtle stock composition in fishery bycatch: a case study in the Mediterranean. Molecular Ecology 7:1529-1542.

Leland, J. G., III. 1968. A survey of the sturgeon fishery of South Carolina. Bears Bluff Labs. No. 47, 27 pp.

Lewison, R.L., L.B. Crowder, and D.J. Shaver. 2003. The impact of turtle excluder devices and fisheries closures on loggerhead and Kemp's ridley strandings in the western Gulf of Mexico. Conservation Biology 17(4):1089-1097.

Lewison, R.L., S.A. Freeman, and L.B. Crowder. 2004. Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles. Ecology Letters 7:221-231.

Lutcavage, M.E., and P.L. Lutz. 1997. Diving physiology. Pages 277-296 in P.L. Lutz and J.A. Musick, eds. The Biology of Sea Turtles. Boca Raton, Florida: CRC Press.

Lutcavage, M., and J.A. Musick. 1985. Aspects of the biology of sea turtles in Virginia. Copeia 1985(2):449-456.

Lutcavage, M.E., P. Plotkin, B. Witherington, and P.L. Lutz. 1997. Human impacts on sea turtle survival. Pages 387-409 in P.L. Lutz and J.A. Musick, eds. The Biology of Sea Turtles. Boca Raton, Florida: CRC Press.

Maier, P.P., A.L. Segars, M.D. Arendt, J.D. Whitaker, B.W. Stender, L. Parker, R. Vendetti, D.W. Owens, J. Quattro, and S.R. Murphy. 2004. Development of an index of sea turtle abundance based on in-water sampling with trawl gear. Final report to the National Marine Fisheries Service. 86 pp.

Maier, P.P., A.L. Segars, M.D. Arendt, and J.D. Whitaker. 2005. Examination of local movement and migratory behavior of sea turtles during spring and summer along the Atlantic coast off the southeastern United States. Annual report for grant number NA03NMF4720281. 29pp.

Mangin, E. 1964. Croissance en Longueur de Trois Esturgeons d'Amerique du Nord: *Acipenser oxyrhynchus*, Mitchill, *Acipenser fulvescens*, Rafinesque, et *Acipenser brevirostris* LeSueur. Verh. Int. Ver. Limnology 15: 968-974.

INACTIVE

Mansfield, K. L. 2006. Sources of mortality, movements, and behavior of sea turtles in Virginia. Ph.D. dissertation, College of William and Mary. 343 pp.

Mansfield, K.L., J.A. Musick, and R.A. Pemberton. 2001. Characterization of the Chesapeake Bay pound net and whelk pot fisheries and their potential interactions with marine sea turtle species. Final Report to the National Marine Fisheries Service under Contract No. 43EANFO30131. 75 pp.

Marcano, L.A., and J.J. Alio-M. 2000. Incidental capture of sea turtles by the industrial shrimping fleet off northwestern Venezuela. Page 107 in F.A. Abreu-Grobois, R. Briseño-Dueñas, R. Márquez-Millán, and L. Sarti-Martínez, compilers. Proceedings of the Eighteenth International Sea Turtle Symposium. NOAA Technical Memorandum NMFS-SEFSC-436.

Márquez, R. 1990. FAO Species Catalogue, Vol. 11. Sea turtles of the world, an annotated and illustrated catalogue of sea turtle species known to date. FAO Fisheries Synopsis, 125:1-81.

McClellan, C.M., and A.J. Read. 2007. Complexity and variation in loggerhead sea turtle life history. Biology Letters 3:592-594.

Meylan, A., B. Schroeder, and A. Mosier. 1995. Sea turtle nesting activity in the state of Florida. Florida Marine Research Publication 52:1-51.

Mitchell, G.H., R.D. Kenney, A.M. Farak, and R.J. Campbell. 2003. Evaluation of occurrence of endangered and threatened marine species in naval ship trial areas and transit lanes in the Gulf of Maine and offshore of Georges Bank. NUWC-NPT Technical Memo 02-121A. March 2003. 113 pp.

Morreale, S.J., and E.A. Standora. 1993. Occurrence, movement, and behavior of the Kemp's ridley and other sea turtles in New York waters. Okeanos Ocean Research Foundation Final Report April 1988-March 1993. 70 pp.

Morreale, S.J., and E.A. Standora. 1998. Early life stage ecology of sea turtles in northeastern U.S. waters. NOAA Technical Memorandum NMFS-SEFSC-413:1-49.

Morreale, S.J., and E.A. Standora. 2005. Western North Atlantic waters: crucial developmental habitat for Kemp's ridley and loggerhead sea turtles. Chelonian Conservation and Biology 4(4):872-882.

Morreale, S.J., C.F. Smith, K. Durham, R.A. DiGiovanni, Jr., and A.A. Aguirre. 2005. Assessing health, status, and trends in northeastern sea turtle populations. Interim report - Sept. 2002 - Nov. 2004. Gloucester, Massachusetts: National Marine Fisheries Service.

INACTIVE

Mortimer, J.A. 1982. Feeding ecology of sea turtles. Pages 103-109 in K.A. Bjorndal, ed. Biology and conservation of sea turtles. Washington, D.C.: Smithsonian Institution Press.

Mrosovsky, N. 1981. Plastic jellyfish. Marine Turtle Newsletter 17:5-6.

Murawski, S.A. and A.L. Pacheco. 1977. Biological and fisheries data on Atlantic sturgeon, *Acipenser oxyrhynchus* (Mitchill). National Marine Fisheries Service Technical Series Report 10: 1-69.

Murphy, T.M., and S.R. Hopkins. 1984. Aerial and ground surveys of marine turtle nesting beaches in the southeast region. Final Report to the National Marine Fisheries Service. 73pp.

Murphy, T.M., S.R. Murphy, D.B. Griffin, and C. P. Hope. 2006. Recent occurrence, spatial distribution and temporal variability of leatherback turtles (*Dermochelys coriacea*) in nearshore waters of South Carolina, USA. Chelonian Conservation and Biology 5(2):216-224.

Murray, K.T. 2004. Bycatch of sea turtles in the Mid-Atlantic sea scallop (*Placopecten magellanicus*) dredge fishery during 2003. NEFSC Reference Document 04-11; 25 pp.

Murray, K.T. 2006. Estimated average annual bycatch of loggerhead sea turtles (*Caretta caretta*) in U.S. Mid-Atlantic bottom otter trawl gear, 1996-2004. NEFSC Reference Document 06-19:1-26.

Murray, K.T. 2007. Estimated bycatch of loggerhead sea turtles (*Caretta caretta*) in U.S. Mid-Atlantic scallop trawl gear, 2004-2005, and in sea scallop dredge gear, 2005. NEFSC Reference Document 07-04; 30 pp.

Murray, K.T. 2008. Estimated average annual bycatch of loggerhead sea turtles (*Caretta caretta*) in U.S. Mid-Atlantic bottom otter trawl gear, 1996-2004 (2nd edition). NEFSC Reference Document 08-20; 32 pp.

Murray, K.T. 2009a. Proration of estimated bycatch of loggerhead sea turtles in U.S. Mid-Atlantic sink gillnet gear to vessel trip report landed catch, 2002-2006. NEFSC Reference Document 09-19; 7 pp.

Murray, K.T. 2009b. Characteristics and magnitude of sea turtle bycatch in US mid-Atlantic gillnet gear. Endangered Species Research 8:211-224.

Murray, K.T. 2011. Sea turtle bycatch in the U.S. sea scallop (*Placopecten magellanicus*) dredge fishery, 2001–2008. Fish Res. 107:137-146.

INACTIVE

Musick, J.A., and C.J. Limpus. 1997. Habitat utilization and migration in juvenile sea turtles. Pages 137-164 in P.L. Lutz and J.A. Musick, eds. The Biology of Sea Turtles. Boca Raton, Florida: CRC Press.

NAST (National Assessment Synthesis Team). 2000. Climate Change Impacts on the United States: The Potential Consequences of Climate Variability and Change, US Global Change Research Program, Washington DC, 2000 http://www.usgcrp.gov/usgcrp/Library/nationalassessment/1IntroA.pdf

NMFS (National Marine Fisheries Service) NEFSC (Northeast Fisheries Science Center). 2011. Preliminary summer 2010 regional abundance estimate of loggerhead turtles (Caretta caretta) in northwestern Atlantic Ocean continental shelf waters. US Dept Commerce, Northeast Fisheries Science Center Reference Document 11-03; 33 pp.

NEFSC. 2011. Summary of discard estimates for Atlantic sturgeon. Report prepared by Tim Miller and Gary Shepard, NEFSC Population Dynamics Branch, NMFS Northeast Fisheries Science Center. August 19, 2011.

NMFS (National Marine Fisheries Service). 1995. Endangered Species Act Section 7 consultation on United States Coast Guard vessel and aircraft activities along the Atlantic coast. Biological Opinion. September 15, 1995.

NMFS (National Marine Fisheries Service). 1996. Endangered Species Act Section 7 Consultation on the proposed shock testing of the SEAWOLF submarine off the Atlantic Coast of Florida during the summer of 1997. Biological Opinion. December 12, 1996.

NMFS (National Marine Fisheries Service). 1997a. Endangered Species Act Section 7 consultation on Navy activities off the southeastern United States along the Atlantic Coast. Biological Opinion. May 15, 1997.

NMFS (National Marine Fisheries Service). 2007a. Fisheries of the United States 2006. Current Fishery Statistics No. 2006.

NMFS (National Marine Fisheries Service). 2008a. Summary Report of the Workshop on Interactions Between Sea Turtles and Vertical Lines in Fixed-Gear Fisheries. M.L. Schwartz (ed.), Rhode Island Sea Grant, Narragansett, Rhode Island. 54 pp.

NMFS (National Marine Fisheries Service). 2008b. Endangered Species Act Section 7 Consultation on the Atlantic Sea Scallop Fishery Management Plan. Biological Opinion. March 14, 2008.

NMFS (National Marine Fisheries Service). 2008c. Endangered Species Act Section 7 Consultation on the U.S. Navy Atlantic Fleet's conduct of active sonar training along the

INACTIVE

Atlantic Coast of the United States and in the Gulf of Mexico from January 2009 to January 2014. Biological Opinion. January 16, 2008.

NMFS (National Marine Fisheries Service). 2009a. Endangered Species Act Section 7 Consultation on the Proposed Award of Research Set-Aside for the Spring and Fall 2009 Surveys of the NEAMAP Near-Shore Trawl Program. Biological Opinion, April 16.

NMFS (National Marine Fisheries Service). 2009b. Endangered and Threatened Species; Determination of Endangered Status for the Gulf of Maine Distinct Population Segment of Atlantic Salmon. Federal Register Final Rule, 74 FR 117. June 19, 2009.

NMFS (National Marine Fisheries Service). 2009c. Endangered and Threatened Species; Designation of Critical Habitat for Atlantic Salmon (*Salmo salar*) Gulf of Maine Distinct Population Segment. Federal Register Final Rule, 74 FR 117. June 19, 2009.

NMFS (National Marine Fisheries Service). 2009d. Endangered Species Act Section 7 Consultation on U.S. Navy activities in the Virginia Capes, Cherry Point, and Jacksonville Range Complexes from June 2009 to June 2010. Biological Opinion. June 5, 2009.

NMFS (National Marine Fisheries Service). 2011. Biennial Report to Congress on the Recovery Program for Threatened and Endangered Species, October 1, 2008 – September 30, 2010. Washington, D.C.: National Marine Fisheries Service. 194 pp.

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 1991. Recovery plan for U.S. population of loggerhead turtle *Caretta caretta*. Washington, D.C.: National Marine Fisheries Service. 64 pp.

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 1992. Recovery plan for leatherback turtles *Dermochelys coriacea* in the U.S. Caribbean, Atlantic, and Gulf of Mexico. Washington, D.C.: National Marine Fisheries Service. 65 pp.

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 1995. Status reviews for sea turtles listed under the Endangered Species Act of 1973. Silver Spring, Maryland: National Marine Fisheries Service. 139 pp.

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 1998a. Recovery Plan for U.S. Pacific Populations of the Green Turtle (*Chelonia mydas*). Silver Spring, Maryland: National Marine Fisheries Service. 84 pp.

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 1998b. Recovery Plan for U.S. Pacific Populations of the Leatherback Turtle (*Dermochelys coriacea*). Silver Spring, Maryland: National Marine Fisheries Service. 65 pp.

INACTIVE

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 2007a. Loggerhead sea turtle (*Caretta caretta*) 5 year review: summary and evaluation. Silver Spring, Maryland: National Marine Fisheries Service. 65 pp.

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 2007b. Kemp's ridley sea turtle *(Lepidochelys kempii*) 5 year review: summary and evaluation. Silver Spring, Maryland: National Marine Fisheries Service. 50 pp.

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 2007c. Green sea turtle (*Chelonia mydas*) 5 year review: summary and evaluation. Silver Spring, Maryland: National Marine Fisheries Service. 102 pp.

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 2007d. Leatherback sea turtle (*Dermochelys coriacea*) 5 year review: summary and evaluation. Silver Spring, Maryland: National Marine Fisheries Service. 79 pp.

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 2008. Recovery plan for the Northwest Atlantic population of the loggerhead turtle (*Caretta caretta*), Second revision. Washington, D.C.: National Marine Fisheries Service. 325 pp.

NMFS SEFSC (Southeast Fisheries Science Center). 2001. Stock assessments of loggerhead and leatherback sea turtles and an assessment of the impact of the pelagic longline fishery on the loggerhead and leatherback sea turtles of the Western North Atlantic. NOAA Technical Memorandum NMFS-SEFSC-455. 343 pp.

NMFS SEFSC (Southeast Fisheries Science Center). 2009. An assessment of loggerhead sea turtles to estimate impacts of mortality reductions on population dynamics. NMFS SEFSC Contribution PRD-08/09-14. 45 pp.

NRC (National Research Council). 1990. Decline of the Sea Turtles: Causes and Prevention. Washington, D.C.: National Academy Press. 259 pp.

NREFHSC (Northeast Region Essential Fish Habitat Steering Committee). 2002. Workshop on the effects of fishing gear on marine habitat off the northeastern United States. October 23-25, Boston, Massachusetts. NEFSC Reference Document 02-01, 86 pp.

Niklitschek and Secor 2005.

Niklitschek and Secor 2010.

Palka, D. 2000. Abundance and distribution of sea turtles estimated from data collected during cetacean surveys. Pages 71-72 in K.A. Bjorndal and A.B. Bolten, eds. Proceedings of a workshop on assessing abundance and trends for in-water sea turtle populations. NOAA Technical Memorandum NMFS-SEFSC-445.

INACTIVE

Pearce, A.F. 2001. Contrasting population structure of the loggerhead turtle (*Caretta caretta*) using mitochondrial and nuclear DNA markers. Master's thesis, University of Florida. 71 pp.

Pearce, A.F. and B.W. Bowen. 2001. Final report: Identification of loggerhead (*Caretta caretta*) stock structure in the southeastern United States and adjacent regions using nuclear DNA markers. Project number T-99-SEC-04. Submitted to the National Marine Fisheries Service, May 7, 2001. 79 pp.

Perry, S.L., D.P. DeMaster, and G.K. Silber. 1999. The great whales: history and status of six species listed as endangered under the U.S. Endangered Species Act of 1973. Marine Fisheries Review Special Edition. 61(1):59-74.

Pritchard, P.C.H. 1982. Nesting of the leatherback turtle, *Dermochelys coriacea*, in Pacific, Mexico, with a new estimate of the world population status. Copeia 1982(4):741-747.

Pritchard, P.C.H. 2002. Global status of sea turtles: an overview. Document INF-001 prepared for the Inter-American Convention for the Protection and Conservation of Sea Turtles, First Conference of the Parties (COP1IAC), First part August 6-8, 2002.

Rebel, T.P. 1974. Sea turtles and the turtle industry of the West Indies, Florida and the Gulf of Mexico. Coral Gables, Florida: University of Miami Press.

Rees, A.F., A. Saad, and M. Jony. 2005. Marine turtle nesting survey, Syria 2004: discovery of a "major" green turtle nesting area. Page 38 in Book of Abstracts of the Second Mediterranean Conference on Marine Turtles. Antalya, Turkey, 4-7 May 2005.

Ross, J.P. 1996. Caution urged in the interpretation of trends at nesting beaches. Marine Turtle Newsletter 74:9-10.

Ruelle, R., and K.D. Keenlyne. 1993. Contaminants in Missouri River pallid sturgeon. Bull. Environ. Contam. Toxicol. 50: 898-906.

Sarti, L., S.A. Eckert, N. Garcia, and A.R. Barragán. 1996. Decline of the world's largest nesting assemblage of leatherback turtles. Marine Turtle Newsletter 74:2-5.

Sarti, L., S. Eckert, P. Dutton, A. Barragán, and N. García. 2000. The current situation of the leatherback population on the Pacific coast of Mexico and central America, abundance and distribution of the nestings: an update. Pages 85-87 in H. Kalb and T. Wibbels, compilers. Proceedings of the Nineteenth Annual Symposium on Sea Turtle Conservation and Biology. NOAA Technical Memorandum NMFS-SEFSC-443.

INACTIVE

Sarti Martinez, L., A.R. Barragán, D.G. Munoz, N. Garcia, P. Huerta, and F. Vargas. 2007. Conservation and biology of the leatherback turtle in the Mexican Pacific. Chelonian Conservation and Biology 6(1):70-78.

Sasso, C.R., and S.P. Epperly. 2006. Seasonal sea turtle mortality risk from forced submergence in bottom trawls. Fisheries Research 81:86-88.

Schmid, J.R., and W.N. Witzell. 1997. Age and growth of wild Kemp's ridley turtles (*Lepidochelys kempi*): cumulative results of tagging studies in Florida. Chelonian Conservation and Biology 2(4):532-537.

Schueller, P. and D.L. Peterson. 2006. Population status and spawning movements of Atlantic sturgeon in the Altamaha River, Georgia. Presentation to the 14th American Fisheries Society Southern Division Meeting, San Antonio, February 8-12th, 2006.

Schultz, J.P. 1975. Sea turtles nesting in Surinam. Zoologische Verhandelingen (Leiden) 143:3-172.

Scott, W.B. and E.J. Crossman. 1973. Freshwater fishes of Canada. Fisheries Research Board of Canada. Bulletin 184. pp. 80-82.

Secor, D.H. and J.R. Waldman. 1999. Historical abundance of Delaware Bay Atlantic sturgeon and potential rate of recovery. American Fisheries Society Symposium 23: 203-216.

Sears, R. 2002. Blue whale, *Balaenoptera musculus*. Pages 112-116 in W.F. Perrin, B. Würsig, and J.G.M. Thewissen, eds. Encyclopedia of Marine Mammals. San Diego: Academic Press.

Seminoff, J.A. 2004. *Chelonia mydas*. In 2007 IUCN Red List of Threatened Species. Accessed 12 January 2010. http://www.iucnredlist.org/search/details.php/4615/summ.

Seney, E.E., and J.A. Musick. 2007. Historical diet analysis of loggerhead sea turtles (*Caretta caretta*) in Virginia. Copeia 2007(2):478-489.

Shamblin, B.M. 2007. Population structure of loggerhead sea turtles (*Caretta caretta*) nesting in the southeastern United States inferred from mitochondrial DNA sequences and microsatellite loci. Master's thesis, University of Georgia. 59pp.

Shoop, C.R. 1987. The Sea Turtles. Pages 357-358 in R.H. Backus and D.W. Bourne, eds. Georges Bank. Cambridge, Massachusetts: MIT Press.

INACTIVE

Shoop, C.R., and R.D. Kenney. 1992. Seasonal distributions and abundance of loggerhead and leatherback sea turtles in waters of the northeastern United States. Herpetological Monographs 6:43-67.

Shoop, C.R., and C.A. Ruckdeschel. 1989. Long-distance movement of a juvenile loggerhead sea turtle. Marine Turtle Newsletter 47:15.

Smith, T.I.J. 1985. The fishery, biology, and management of Atlantic sturgeon, *Acipenser oxyrhynchus*, in North America. Environmental Biology of Fishes 14(1): 61-72.

Smith, T.I.J. and J.P. Clugston. 1997. Status and management of Atlantic sturgeon, *Acipenser oxyrinchus*, in North America. Environmental Biology of Fishes 48: 335-346.

Snover, M.L., A.A. Hohn, L.B. Crowder, and S.S. Heppell. 2007. Age and growth in Kemp's ridley sea turtles: evidence from mark-recapture and skeletochronology. Pages 89-106 in P.T. Plotkin, ed. Biology and Conservation of Ridley Sea Turtles. Baltimore, Maryland: Johns Hopkins University Press.

Spotila, J.R., A.E. Dunham, A.J. Leslie, A.C. Steyermark, P.T. Plotkin and F.V. Paladino. 1996. Worldwide population decline of *Dermochelys coriacea*: are leatherback turtles going extinct? Chelonian Conservation and Biology 2:209-222.

Spotila, J.R., R.D. Reina, A.C. Steyermark, P.T. Plotkin, and F.V. Paladino. 2000. Pacific leatherback turtles face extinction. Nature 405:529-530.

Squiers et al. 1979.

Squiers et al. 1981

Squiers 2004.

Stabenau, E.K., T.A. Heming, and J.F. Mitchell. 1991. Respiratory, acid-base and ionic status of Kemp's ridley sea turtles (*Lepidochelys kempi*) subjected to trawling. Comparative Biochemistry and Physiology 99A(½):107-111.

Stevenson, J.T., and D.H. Secor. 1999. Age determination and growth of Hudson River Atlantic sturgeon, *Acipenser oxyrinchus*. Fishery Bulletin 97: 153-166.

Stein, A.B., K.D. Friedland, and M. Sutherland. 2004. Atlantic sturgeon marine bycatch and mortality on the continental shelf of the Northeast United States. North American Journal of Fisheries Management 24: 171-183.

INACTIVE

Stewart, K., M. Sims, A. Meylan, B. Witherington, B. Brost, and L.B. Crowder. 2011. Leatherback nests increasing significantly in Florida, USA; trends assessed over 30 years using multilevel modeling. Ecological Applications, 21(1): 263–273.

Suárez, A. 1999. Preliminary data on sea turtle harvest in the Kai Archipelago, Indonesia. Abstract, 2nd ASEAN Symposium and Workshop on Sea Turtle Biology and Conservation, July 15-17, 1999, Sabah, Malaysia.

Suárez, A., P.H. Dutton, and J. Bakarbessy. 2000. Leatherback (*Dermochelys coriacea*) nesting on the North Vogelkop Coast of Irian Jaya, Indonesia. Page 260 in H.J. Kalb and T. Wibbels, compilers. Proceedings of the Nineteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-443.

Taub, S.H. 1990. Interstate fishery management plan for Atlantic sturgeon. Fisheries Management Report No. 17. Atlantic States Marine Fisheries Commission, Washington, D.C. 73 pp.

TEWG (Turtle Expert Working Group). 1998. An assessment of the Kemp's ridley (*Lepidochelys kempii*) and loggerhead (*Caretta caretta*) sea turtle populations in the Western North Atlantic. NOAA Technical Memorandum NMFS-SEFSC-409:1-96.

TEWG (Turtle Expert Working Group). 2000. Assessment update for the Kemp's ridley and loggerhead sea turtle populations in the western North Atlantic. NOAA Technical Memorandum NMFS-SEFSC-444:1-115.

TEWG (Turtle Expert Working Group). 2007. An assessment of the leatherback turtle population in the Atlantic Ocean. NOAA Technical Memorandum NMFS-SEFSC-555:1-116.

TEWG (Turtle Expert Working Group). 2009. An assessment of the loggerhead turtle population in the Western North Atlantic Ocean. NOAA Technical Memorandum NMFS-SEFSC-575:1-131.

Thompson, N.B. 1984. Progress report on estimating density and abundance of marine turtles: results of first year pelagic surveys in the Southeast U.S. Miami, Florida: National Marine Fisheries Service. 64 pp.

Thompson, N.B. 1988. The status of loggerhead, *Caretta caretta*; Kemp's ridley, *Lepidochelys kempi*; and green, *Chelonia mydas* sea turtles in U.S. waters. Marine Fisheries Review 50(3):16-23.

Thompson, N.B., and H. Huang. 1993. Leatherback turtles in southeast U.S. waters. NOAA Technical Memorandum NMFS-SEFSC-318:1-25.

INACTIVE

USFWS (U.S. Fish and Wildlife Service) and NMFS (National Marine Fisheries Service). 1992. Recovery plan for the Kemp's ridley sea turtle (*Lepidochelys kempii*). St. Petersburg, Florida: National Marine Fisheries Service.

Van Eenennaam, J.P., S.I. Doroshov, G.P. Moberg, J.G. Watson, D.S. Moore and J. Linares. 1996. Reproductive conditions of the Atlantic sturgeon (*Acipenser oxyrhynchus*) in the Hudson River. Estuaries 19: 769-777.

Van Eenennaam, J.P., and S.I. Doroshov. 1998. Effects of age and body size on gonadal development of Atlantic sturgeon. Journal of Fish Biology 53: 624-637.

Van Houtan, K.S. and J.M. Halley. 2011. Long-Term Climate Forcing in Loggerhead Sea Turtle Nesting. PLoS ONE 6(4): e19043. doi:10.1371/journal.pone.0019043.

Vargo, S., P. Lutz, D. Odell, E. Van Vleep, and G. Bossart. 1986. Final report: Study of effects of oil on marine turtles. Technical Report. OCS study MMS 86-0070. Volume 2. 181 pp.

VIMS (Virginia Institute of Marine Science). 2010. Environmental Assessment for Data Collection and Analysis in Support of Single and Multispecies Stock Assessments in the Mid-Atlantic: Northeast Area Monitoring and Assessment Program (NEAMAP) Near Shore Trawl Program – 2010-2012. Prepared for the National Marine Fisheries Service, Northeast Region. 102 pp.

Vladykov, V.D. and J.R. Greeley. 1963. Order Acipenseroidea. Pages 24-60 in Fishes of the Western North Atlantic. Memoir Sears Foundation for Marine Research 1(Part III). xxi + 630 pp.

Waldman, J.R., J.T. Hart, and I.I. Wirgin. 1996. Stock composition of the New York Bight Atlantic sturgeon fishery based on analysis of mitochondrial DNA. Transactions of the American Fisheries Society 125: 364-371.

Waldman JR, Grunwald C, Stabile J, Wirgin I. 2002. Impacts of life history and biogeography on genetic stock structure in Atlantic Sturgeon, *Acipenser oxyrinchus oxyrinchus*, Gulf sturgeon *A. oxyrinchus desotoi*, and shortnose sturgeon, *A.brevirostrum.* J Appl Ichthyol 18:509-518

Warden, M.L. 2011a. Modeling loggerhead sea turtle (*Caretta caretta*) interactions with US Mid-Atlantic bottom trawl gear for fish and scallops, 2005-2008. Biological Conservation 144:2202-2212.

Warden, M.L. 2011b. Proration of loggerhead sea turtle (*Caretta caretta*) interactions in U.S. Mid-Atlantic bottom otter trawls for fish and scallops, 2005-2008, by managed species landed. U.S. Department of Commerce, Northeast Fisheries Science Centter Reference Document 11-04. 8 p.

INACTIVE

Warden, M. and K. Bisack 2010. Analysis of Loggerhead Sea Turtle Bycatch in Mid-Atlantic Bottom Trawl Fisheries to Support the Draft Environmental Impact Statement for Sea Turtle Conservation and Recovery in Relation to Atlantic and Gulf of Mexico Bottom Trawl Fisheries. NOAA NMFS NEFSC Ref. Doc.010. 13 pp.

Welsh, Stuart A., Michael F. Mangold, Jorgen E. Skjeveland, and Albert J. Spells. 2002. Distribution and Movement of Shortnose Sturgeon (*Acipenser brevirostrum*) in the Chesapeake Bay. Estuaries Vol. 25 No. 1: 101-104.

Whitehead, H. 2002. Sperm whale, *Physeter macrocephalus*. Pages 1165-1172 in W.F. Perrin, B. Würsig, and J.G.M. Thewissen, eds. Encyclopedia of Marine Mammals. San Diego: Academic Press.

Wirgin, I. et al. In Prep. Stock origin of Atlantic sturgeon in the Minas Basin of the Bay of Fundy. 22 pp.

Wirgin, I. and T. King. 2011. Mixed Stock Analysis of Atlantic sturgeon from coastal locales and a non-spawning river. Presented at February 2011 Atlantic and shortnose sturgeon workshop.

Wirgin, I., J.R. Waldman, J. Rosko, R. Gross, M.R. Collins, S.G. Rogers, and J. Stabile. 2000. Genetic structure of Atlantic sturgeon populations based on mitochondrial DNA control region sequences. Transactions of the American Fisheries Society. 129:476-486.

Witherington, B., P. Kubilis, B. Brost, and A. Meylan. 2009. Decreasing annual nest counts in a globally important loggerhead sea turtle population. Ecological Applications 19:30-54.

Witt, M.J., A.C. Broderick, D.J. Johns, C. Martin, R. Penrose, M.S. Hoogmoed, and B.J. Godley. 2007. Prey landscapes help identify potential foraging habitats for leatherback turtles in the NE Atlantic. Marine Ecology Progress Series 337:231-243.

Witzell, W.N. 2002. Immature Atlantic loggerhead turtles (*Caretta caretta*): suggested changes to the life history model. Herpetological Review 33(4):266-269.

Wynne, K., and M. Schwartz. 1999. Guide to marine mammals and turtles of the U.S. Atlantic and Gulf of Mexico. Narragansett: Rhode Island Sea Grant.

Zug, G.R., and J.F. Parham. 1996. Age and growth in leatherback turtles, *Dermochelys coriacea*: a skeletochronological analysis. Chelonian Conservation and Biology 2(2):244-249.

Zurita, J.C., R. Herrera, A. Arenas, M.E. Torres, C. Calderon, L. Gomez, J.C. Alvarado, and R. Villavicencio. 2003. Nesting loggerhead and green sea turtles in Quintana Roo, Mexico.

INACTIVE

Pages 125-127 in J.A. Seminoff, compiler. Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503.

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APPENDIX A.

Sea turtle and resuscitation measures as found at 50 CFR 223.206(d)(1).

(d) (1) (i) Any specimen taken incidentally during the course of fishing or scientific research activities must be handled with due care to prevent injury to live specimens, observed for activity, and returned to the water according to the following procedures.

(A) Sea turtles that are actively moving or determined to be dead as described in (d)(1)(i)(C) of this section must be released over the stern of the boat. In addition, they must be released only when fishing or scientific collection gear is not in use, when the engine gears are in neutral position, and in areas where they are unlikely to be recaptured or injured by vessels.

(B) Resuscitation must be attempted on sea turtles that are comatose, or inactive, as determined in paragraph (d)(1) of this section by:

(1) placing the turtle on its bottom shell (plastron) so that the turtle is right side up, and elevating its hindquarters at least 6 inches (15.2 cm) for a period of 4 up to 24 hours. The amount of the elevation depends on the size of the turtle; greater elevations are needed 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 (7.6 cm) then alternate to the other side. Gently touch the eye and pinch the tail (reflex test) periodically to see if there is a response.

(2) sea turtles being resuscitated must be shaded and kept damp or moist but under no circumstance be placed into a container holding water. A water-soaked towel placed over the head, neck, and flippers is the most effective method in keeping a turtle moist.

(3) sea turtles that revive and become active must be released over the stern of the boat only when fishing or scientific collection gear is not in use, when the engine gears are in neutral position, and in areas where they are unlikely to be recaptured or injured by vessels. Sea turtles that fail to respond to the reflex test or fail to move within 4 hours (up to 24, if possible) must be returned to the water in the same manner as that for actively moving turtles.

(C) A turtle is determined to be dead if the muscles are stiff (rigor mortis) and/or the flesh has begun to rot; otherwise the turtle is determined to be comatose or inactive and resuscitation attempts are necessary.

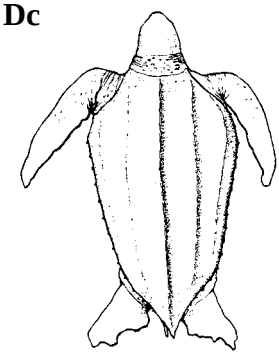
INACTIVE

APPENDIX B

Identification Key for Sea Turtles and Sturgeon Found in Northeast U.S. Waters

SEA TURTLES

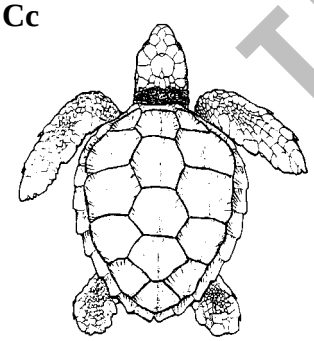
Dc



Leatherback (*Dermocheyls coriacea*)

Found in open water throughout the Northeast from spring through fall. Leathery shell with 5-7 ridges along the back. Largest sea turtle (4-6 feet). Dark green to black; may have white spots on flippers and underside.

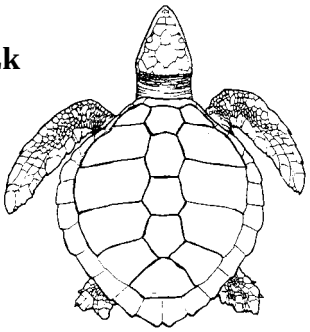
Cc



Loggerhead (*Caretta caretta*)

Bony shell, reddish-brown in color. Mid-sized sea turtle (2-4 feet). Commonly seen from Cape Cod to Hatteras from spring through fall, especially in southern portion of range. Head large in relation to body.

Lk



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Kemp's ridley (*Lepidochelys kempi*)

Most often found in Bays and coastal waters from Cape Cod to Hatteras from summer through fall. Offshore occurrence undetermined. Bony shell, olive green to grey in color. Smallest sea turtle in Northeast (9-24 inches). Width equal to or greater than length.

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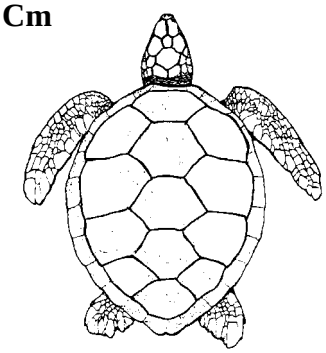
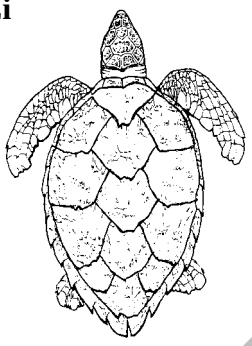
Cm



Green turtle (*Chelonia mydas*)

Uncommon in the Northeast. Occur in Bays and coastal waters from Cape Cod to Hatteras in summer. Bony shell, variably colored; usually dark brown with lighter stripes and spots. Small to mid-sized sea turtle (1-3 feet). Head small in comparison to body size.

Ei



Hawksbill (*Eretmochelys imbricata*)

Rarely seen in Northeast. Elongate bony shell with overlapping scales. Color variable, usually dark brown with yellow streaks and spots (tortoise-shell). Small to mid-sized sea turtle (1-3 feet). Head relatively small, neck long.

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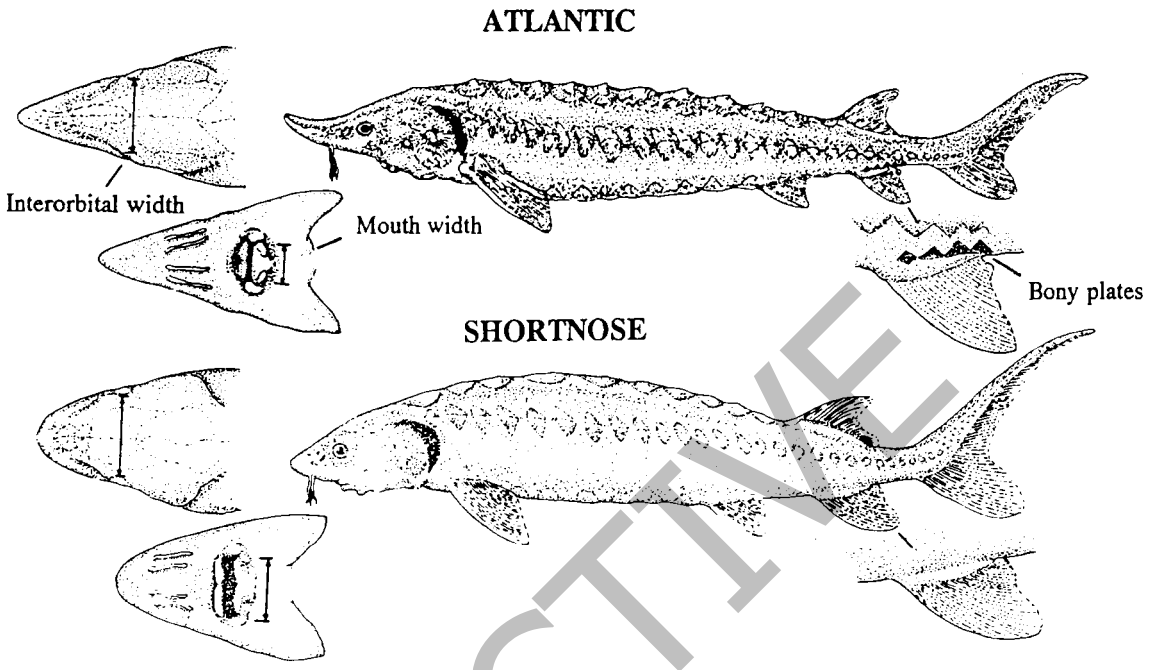
Appendix B continued
Sturgeon Identification



Distinguishing Characteristics of Atlantic and Shortnose Sturgeon

Characteristic	Atlantic Sturgeon, *Acipenser oxyrinchus*	Shortnose Sturgeon, *Acipenser brevirostrum*
Maximum length	> 9 feet/ 274 cm	4 feet/ 122 cm
Mouth	Football shaped and small. Width inside lips < 55% of bony interorbital width	Wide and oval in shape. Width inside lips > 62% of bony interorbital width
*Pre-anal plates	Paired plates posterior to the rectum & anterior to the anal fin.	1-3 pre-anal plates almost always occurring as median structures (occurring singly)
Plates along the anal fin	Rhombic, bony plates found along the lateral base of the anal fin (see diagram below)	No plates along the base of anal fin
Habitat/Range	Anadromous; spawn in freshwater but primarily lead a marine existence	Freshwater amphidromous; found primarily in fresh water but does make some coastal migrations

* From Vecsei and Peterson, 2004

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APPENDIX C

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.

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