ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
BIOLOGICAL OPINION

Action Agency: National Marine Fisheries Service, Northeast Region, through its Sustainable Fisheries Division

Activity: Endangered Species Act Section 7 Consultation on the Atlantic Sea Scallop Fishery Management Plan [Consultation No. F/NER/2012/01461] GARFO-2012-00007

Consulting Agency: National Marine Fisheries Service, Northeast Region, through its Protected Resources Division

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Approved by:

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Section 7(a)(2) of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531 *et seq*.), requires each Federal agency to insure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When the action of a Federal agency may affect a species or critical habitat protected under the ESA, that agency is required to consult with either the NOAA Fisheries Service (NMFS) or U.S. Fish and Wildlife Service (FWS), depending upon the species and/or critical habitat that may be affected. In instances where NMFS or FWS are themselves authorizing, funding, or carrying out an action that may affect listed species, the agency must conduct intra-service consultation. Since the action described in this document is approved and implemented by the NMFS Northeast Region (NERO), this office has requested formal intra-service section 7 consultation.

NMFS NERO has reinitiated formal intra-service consultation, in accordance with section 7(a)(2) of the ESA and 50 CFR 402.16, given the recent listing of five distinct population segments (DPSs) of Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) under the ESA as well as new information on sea turtle interactions that reveals that the continued operation of the Atlantic sea scallop fishery (hereafter referred to as the scallop fishery), which is authorized under the Atlantic Sea Scallop Fishery Management Plan (Scallop FMP), may affect listed species in a manner, or to an extent, not previously considered. This document represents NMFS's biological opinion (Opinion) on the effects of the continued operation of the scallop fishery on ESA-listed species under NMFS jurisdiction, in accordance with section 7 of the ESA.

NMFS NERO reinitiated formal intra-service section 7 consultation on the continued operation of the scallop fishery under the Scallop FMP on February 28, 2012 [Consultation No. F/NER/2012/01461]. For the purposes of this consultation, NMFS NERO, through its Sustainable Fisheries Division (SFD), which administers the scallop fishery through the Scallop FMP, is the Federal action agency and NMFS NERO, through its Protected Resources Division (PRD), is the consulting agency. This Opinion is based on information developed by NMFS NERO as well as other scientific data and reports cited throughout this document. A complete administrative record of this consultation will be kept on file at NMFS NERO.

1.0 CONSULTATION HISTORY

Prior to this formal consultation and Opinion, the continued operation of the scallop fishery under the Scallop FMP was last reviewed via a formal consultation initiated by NMFS NERO on April 3, 2007, and completed on March 14, 2008 (later amended February 5, 2009). The 2008 Opinion issued by NMFS concluded that the continued operation of the scallop fishery under the Scallop FMP would not jeopardize the continued existence of loggerhead (*Caretta caretta*), leatherback (*Dermochelys coriacea*), Kemp's ridley (*Lepidochelys kempii*), or green (*Chelonia mydas*) sea turtles, or any other ESA-listed species under NMFS jurisdiction (NMFS 2008a).

However, the above four species of ESA-listed sea turtles were expected to interact with scallop dredge and trawl gear used in the fishery such that they would come into physical contact with the gear (*i.e.*, be struck by or swim into it) and be potentially captured in the dredge bag of the dredge or the codend of the trawl. An exception is that chain mats equipped to dredge gear would prevent most captures of sea turtles in the dredge bag, and thus prevent subsequent injuries and/or mortalities that would follow either below water or on the deck of the vessel. In accordance with ESA section 7 regulations (50 CFR 402.02), all such interactions with gears used in the fishery are considered "incidental takes." An Incidental Take Statement (ITS) was provided with the 2008 Opinion along with non-discretionary Reasonable and Prudent Measures (RPMs) to minimize the impacts of incidental take. As described in the ITS, the scallop dredge fishery was expected to interact with up to 929 loggerhead sea turtles biennially (595 lethal) as well as one leatherback (non-lethal), two Kemp's ridleys (lethal or non-lethal), and one green sea turtle (lethal or non-lethal) annually. For the scallop trawl fishery, up to 154 loggerheads (20 lethal), one leatherback (lethal or non-lethal), one Kemp's ridley (lethal or non-lethal), and one green sea turtle (lethal or non-lethal) were anticipated to interact with the fishery annually.

Prior to 2008, NMFS also completed formal section 7 consultations on the scallop fishery in 2003, 2004, and 2006. A brief summary of these consultations follows below. Formal consultation on the scallop fishery was first initiated on December 21, 2001, and concluded with the issuance of an Opinion on February 24, 2003. This consultation concluded that the continued operation of the scallop fishery would not jeopardize the continued existence of loggerhead, leatherback, Kemp's ridley, or green sea turtles, or any other ESA-listed species under NMFS jurisdiction (NMFS 2003a). An ITS of 97 sea turtles was estimated based on the annual capture of sea turtles in dredge and trawl gear used in the scallop fishery; 29 of the sea turtles captured in the fishery were expected to die as a result of capture.

Formal section 7 consultation was later reinitiated on November 21, 2003, for two reasons: first, new information on the capture of sea turtles in gear used in the scallop fishery revealed that the continued operation of the scallop fishery may affect listed species or critical habitat in a manner, or to an extent, not previously considered, and second, the agency action was proposed to be modified by Amendment 10 to the Scallop FMP in a manner that caused an effect to listed species or critical habitat not considered in the previous 2003 Opinion. NMFS subsequently modified the proposed action when it initiated an emergency action for the fishery on January 20, 2004. The consultation was, therefore, revised to consider the effects to ESA-listed species from the modified proposed action. The ensuing Opinion concluded on February 23, 2004, that the continued operation of the scallop fishery, including the implementation of Amendment 10 and emergency measures, would not jeopardize the continued existence of loggerhead, leatherback, Kemp's ridley, or green sea turtles, or any other ESA-listed species under NMFS jurisdiction (NMFS 2004a). An ITS was provided for these four sea turtle species along with RPMs.

On September 3, 2004, consultation was again reinitiated to consider new information on the effects of the scallop fishery on sea turtles that was received from the NMFS Northeast Fisheries Science Center (NEFSC). Consultation was completed on December 15, 2004, and concluded that the anticipated capture of 753 sea turtles (752 loggerheads and one leatherback) in the scallop fishery, resulting in death of up to 482 loggerheads and one leatherback, was not expected to result in jeopardy to loggerhead or leatherback sea turtles (NMFS 2004b).

Consultation was again reinitiated on November 1, 2005, based on new information on the number of observed sea turtle interactions in the trawl component of the scallop fishery, as well as new information on the species that interact with scallop fishing gear, and the area(s) where interactions occur. NMFS concluded that consultation on September 18, 2006, with the determination that the continued operation of the fishery was not likely to result in jeopardy to any ESA-listed species under NMFS jurisdiction (NMFS 2006a). Within the ITS of the 2006 Opinion, NMFS anticipated that up to 760 sea turtle interactions (752 in scallop dredge gear and eight in scallop trawl gear) would occur annually as a result of the continued operation of the scallop fishery. Of these, up to 489 interactions (481 in dredge gear and eight in trawl gear) were anticipated to result in death. Nearly all of the interactions (749 of 752 for dredge gear and five of eight for trawl gear) were anticipated to involve loggerhead sea turtles. The remaining anticipated interactions were for leatherback, Kemp's ridley, and green sea turtles.

Aside from these five formal consultations on the Scallop FMP itself, NMFS has also informally reviewed a number of framework adjustments, amendments, research set-aside (RSA) projects, exempted fishing permits (EFPs), and emergency actions associated with the Scallop FMP in regards to their effects on ESA-listed species. These reviews have concluded that either the proposed actions may affect, but were not likely to adversely affect, ESA-listed species under NMFS jurisdiction or that the proposed actions did not trigger reinitiation of formal consultation.

Cause for Reinitiating

As provided in 50 CFR 402.16, reinitiation of formal consultation is required and shall be requested by the Federal agency or by the Service, where discretionary Federal involvement or control over the action has been retained or is authorized by law and if: (1) the amount or extent of incidental take specified in the ITS 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 action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in the Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action.

On February 6, 2012, NMFS published two final rules (77 FR 5880 and 77 FR 5914) listing five DPSs of Atlantic sturgeon as threatened or endangered under the ESA. Four DPSs (New York Bight, Chesapeake Bay, Carolina, and South Atlantic) were listed as endangered and one (Gulf of Maine) was listed as threatened. The ESA listings went into effect on April 6, 2012. The action area for this consultation (see section 2.2) overlaps with the marine range of all five DPSs and Atlantic sturgeon are known to be vulnerable to capture in bottom otter trawl gear (Stein *et al*. 2004; ASMFC TC 2007). Therefore, we have reinitiated formal consultation to consider effects to Atlantic sturgeon, as each of the DPSs may be affected by the proposed action.

We have also reinitiated formal consultation on the scallop fishery to reconsider effects to ESA-listed sea turtles, as several new sources of information on the effects of the scallop fishery on sea turtles have become available since the publication of the last Opinion in 2008. The anticipated incidental take of loggerhead sea turtles in scallop fishing gear exempted by the 2008 Opinion was based on observer data collected for both the dredge and trawl fisheries. The observer data was then analyzed by the NEFSC to provide estimates of the average annual bycatch of loggerheads in scallop fishing gear. In contrast, the anticipated incidental take of leatherback, Kemp's ridley, and green sea turtles in scallop fishing gear exempted by the 2008 Opinion was based on a small sample size of observed captures of these species in the scallop fishery and/or other fisheries using similar gear types or fishing in similar geographic areas.

Several new sources of information on the effects of the scallop fishery on sea turtles have become available since we issued the most recent Opinion in 2008. Reports by Murray (2011) and Warden (2011a) provide new information on the amount of sea turtle interactions occurring annually in both the dredge and trawl components of the fishery. These reports include new estimates of average annual sea turtle bycatch, including unobservable, yet quantifiable interactions (such as a sea turtle interacting with modified gear such as a chain mat equipped dredge). For the scallop dredge fishery, the most recent average annual estimate of hard-shelled sea turtle interactions in the fishery is 125 with a 95% confidence interval (CI) of 88-163 after chain mats were required (*i.e.*, September 26, 2006 through 2008) (Murray 2011). For loggerhead sea turtles, the average annual estimate of interactions in the fishery is 95 with a 95% CI of 63-130 for that same period (Murray 2011). For the scallop trawl fishery, Warden (2011a) estimated that the average annual bycatch of loggerheads in scallop trawl gear during the period of 2005-2008 is 95 with a 95% CI for the four-year annual average of 60-140. These bycatch estimates represent new information on the effects of the scallop fishery on sea turtles. With the issuance of these reports, we also have available to us another useful way to monitor sea turtle bycatch in the scallop fishery over time, even with gear modifications in place (or soon to be required) to reduce serious injuries or mortalities resulting from interactions with the gear.

In addition, there is new information available on the levels of serious injury/mortality to sea turtles in the fishery. In November 2009, NMFS convened a workshop to refine methods to determine the levels of serious injury/mortality to sea turtles interacting with Northeast fisheries. The Sea Turtle Injury Workshop methodology and results (Upite 2011; Memo from C. Upite to the File, March 28, 2012) have recently been made available for application to both the scallop dredge and trawl fisheries and indicate that the serious injury/mortality rates for sea turtles are different than those considered in the 2008 Opinion. Milliken *et al*. (2007), Smolowitz *et al*. (2010), and recent analyses by the Scallop Plan Development Team (PDT) have also assessed the likelihood of serious injury/mortality to sea turtles interacting with a modified, low-profile turtle deflector dredge (TDD) designed to reduce the likelihood of a sea turtle passing under the frame when the dredge fishes on the seafloor.

Finally, new management measures in the fishery have been implemented since we issued the 2008 Opinion. These changes include effort reductions in the Mid-Atlantic implemented through Framework Adjustment 22 (Framework 22) and the requirement in Framework 23 to use the TDD throughout much of the fishery by May 2013. In light of these changes that have been implemented, and the availability of new information on the effects of the fishery on sea turtles, we are reassessing the effects of the scallop fishery on sea turtles in this new Opinion. We will also discuss the change in the ESA listing of loggerhead sea turtles from a single species to separate DPSs, a change we previously determined did not trigger reinitiation on its own (Memo from Patricia A. Kurkul, Regional Administrator, to the Record, November 29, 2011), and will analyze the effects of the scallop fishery on the Northwest Atlantic DPS of loggerheads, which is the only loggerhead DPS likely to occur in the action area for this consultation.

Based upon the information presented above, and in accordance with the regulations at 50 CFR 402.16, formal consultation was reinitiated to reconsider the effects of the continued operation of the scallop fishery under the Scallop FMP, on both ESA-listed sea turtles and Atlantic sturgeon DPSs.

2.0 DESCRIPTION OF THE PROPOSED ACTION

The proposed action is the continued operation of the Atlantic sea scallop (*Placopecten magellanicus*) fishery managed under the Scallop FMP, consistent with all applicable regulations including Amendment 15 and Framework 22, which were implemented in July 2011, as well as Framework 23, which became effective on May 7, 2012. In addition, a temporary rule for emergency action to close the Delmarva (DMV) Scallop Access Area and allocate DMV trips to the Closed Area I (CAI) Scallop Access Area was published on May 14, 2012 (77 FR 28311), is effective from June 13 to November 10, 2012, and will likely be extended until the end of February 2013. As a result, it is also included as part of the proposed action in this Opinion. A summary of the characteristics of the scallop fishery relevant to the analysis of its potential effects on ESA-listed species and critical habitat is presented below.

It is important to note that commercial fishing vessels are often permitted to operate within multiple Federal fisheries at once, and as a result, landings from a particular trip can include multiple species of fish or shellfish managed under multiple FMPs. For the purposes of this Opinion, fishing effort under the Scallop FMP will include actions that result in landings of scallops by federally permitted dredge and bottom trawl vessels operating within the action area as described at the end of this section. In order to identify and analyze fishery impacts on ESA-listed species, ideally, documented interactions with listed species would be linked to the FMP that the vessel was operating under at the time of the interaction (for example, if fishing under the Monkfish FMP, and on a monkfish day at-sea, and a listed species was caught in a net which caught 85% monkfish, 10% groundfish, and 5% scallops, that capture would be attributed to the monkfish fishery, not groundfish or scallops). Alternatively, interactions with listed species could be linked to FMPs proportionally based on the fish catch composition of the fishing trip. As an example, fishing effort and estimated bycatch of listed species for a trip that landed 85% scallops, 10% yellowtail flounder (a species managed under the Northeast Multispecies FMP), and 5% monkfish would be allocated proportionally to the Scallop (85%), Northeast Multispecies (10%), and Monkfish (5%) FMPs. In that example, the overall estimated bycatch for each FMP is the sum of the proportionally allocated bycatch estimates.

At present, not all bycatch estimates for ESA-listed species by FMP completely align with either of the examples above. For loggerhead sea turtles, we have peer-reviewed and published estimates of bycatch in commercial trawl fisheries pertaining to the action area considered in this consultation (see Warden 2011a). The trawl bycatch estimate for loggerhead sea turtles is closely aligned with the second example outlined above, as it proportionally attributes sea turtle interactions consistent with the composition of fish landed on a trip. For leatherback, Kemp's ridley, and green sea turtles, however, observed interactions in trawls are attributed to the FMP that covers the fish species that makes up the majority (by weight) of the landings for the trip during which those sea turtles were caught. These bycatch estimates are more closely aligned with the first example above. It should be noted that the total number, or statistical sample size, of observed non-loggerhead sea turtle interactions attributable to a specific fishery is small.

In regards to the scallop dredge fishery, Murray (2011) describes hard-shelled sea turtle interactions (primarily loggerhead, but also potentially Kemp's ridley and green) in scallop dredge gear from both before and after chain mats were mandated in the fishery. Fisheries observer data were used to develop a model to estimate rates of observable interactions of hard-shelled sea turtles and these rates were then applied to commercial dredge fishing effort to estimate the total number of observable interactions and to infer the number of unobservable, yet quantifiable (*i.e.*, "inferred") interactions after chain mats were implemented. The method, although containing several caveats related to observing and estimating sea turtle interactions in the face of recent gear modifications designed to keep sea turtles out of the dredge (*e.g.*, chain mats, TDDs), provides a way to quantitatively estimate hard-shelled sea turtle interactions in the dredge component of the fishery. However, there are currently no statistical estimates of leatherback sea turtle interactions with the dredge component of the fishery.

For Atlantic sturgeon, an estimate of the number of individuals captured in certain fisheries authorized by NMFS under Federal FMPs in the Northeast is available (NEFSC 2011a). The NEFSC (2011a) report provides a summary of Atlantic sturgeon discard estimates from 2006-2010 for otter trawl and sink gillnet fisheries. Model-based and design-based estimators were explored to try to "assign" these estimated bycatch events to a particular FMP. The design-based ratio estimator was rejected because it relies on the assumption that discards are proportional to the total amount of fish landed (*i.e.*, that the number of Atlantic sturgeon would increase with an increase in total landings of a target species) and this assumption could not be satisfied. The model-based estimator was pursued and a discard estimate for otter trawls and sink gillnets was provided; however, given the high likelihood of inappropriately attributing associations/ responsibilities, the usefulness of the assignments of bycatch to FMP is limited. As a result, we are only able to identify and analyze Atlantic sturgeon interactions by gear type, not by FMP. And since the directed scallop fishery does not use gillnets, we only assessed data for trawls.

2.1 Description of the Current Fishery for Atlantic Sea Scallops

The current management measures for the scallop fishery can be found in documents prepared in accordance with the National Environmental Policy Act (NEPA) for Amendment 15 and Frameworks 22 and 23 to the Scallop FMP, as well as the recent emergency action to close DMV (NEFMC 2010, 2011a, 2011b, 2012). The history of the fishery and the general distribution and habitat characteristics of scallops are described in *Status of Fishery Resources off the Northeastern US – Atlantic Sea Scallop* (Hart 2006) and the *50th Northeast Regional Stock Assessment Workshop (SAW) Assessment Report* (NEFSC 2010). Additional information on the distribution and habitat characteristics of scallops can also be found in the Essential Fish Habitat source documents for the species (Packer *et al*. 1999; Hart and Chute 2004). A summary of the current fishery and its management history based on these sources is provided below.

The fishing year (FY) for the scallop fishery is defined for management purposes as March 1 through the last day of February (50 CFR 648.53(b)(5)). The commercial fishery operates year-round in U.S. waters (Hart 2001), although seasonal peaks in scallop landings are evident. These peaks may be influenced by management measures, market conditions, weather, and scallop spawning, among other factors. Recreational fishing for scallops is insignificant (Hart 2006).

Scallops are found in the Northwest Atlantic Ocean from North Carolina to Newfoundland, along the continental shelf, typically on sand and gravel bottoms (Packer *et al.* 1999; Hart and Chute 2004). However, scallops are not evenly distributed throughout this area and they often occur in aggregations called beds (Hart and Chute 2004). Major aggregations of scallops in U.S. waters occur in the Mid-Atlantic region from Virginia to Long Island, on Georges Bank, in the Great South Channel, and in the Gulf of Maine (Hart and Rago 2006). For the purposes of this Opinion, the Mid-Atlantic region refers to the Mid-Atlantic Bight, which is defined as the coastal ocean area between Cape Hatteras, North Carolina and Long Island, New York. In the Mid-Atlantic region and Georges Bank, scallops are harvested primarily at depths of 30-100 meters, while the bulk of landings from the Gulf of Maine are from nearshore, relatively shallow waters (<40 meters) (NEFSC 2010). Landings from Georges Bank and the Mid-Atlantic region have dominated the fishery since 1964 (NEFSC 2010). Recreational diver harvesting of scallops in shallow coastal waters of the U.S. Atlantic accounts for an extremely small amount of landings.

The commercial harvest of scallops has occurred along the continental shelf from the Gulf of St. Lawrence to Cape Hatteras since the late 1880s (NEFMC 1982). Scallop landings in the U.S. increased substantially after the mid-1940s, with peaks occurring around 1960, 1978, 1990, and 2004 (NEFSC 2010). Maximum U.S. landings were 29,109 metric tons of meats in 2004. Scallop fishing effort reached its maximum in 1991, and then declined during the 1990s so that effort in 1999 was less than half that in 1991 (NEFSC 2010). Effort in the most recent period has been fairly stable. Landings per unit effort (LPUE) showed general declines from the mid-1960s through the mid-1990s, with brief occasional increases due to strong recruitment (NEFSC 2010). LPUE more than quadrupled between 1998 and 2001, and remained high during 2001-2009 (NEFSC 2010). LPUE has been especially high in the Mid-Atlantic and Georges Bank access areas (areas that had been closed and are now under special management; Figure 1).

U.S. Georges Bank landings had peaks during the early 1960s, around 1980 and 1990, but declined precipitously during 1993 and remained low through 1998 (NEFSC 2010). Landings on Georges Bank during 1999-2004 were fairly steady, averaging almost 5,000 metric tons annually, and then increased in 2005-2006, primarily due to reopening of portions of the groundfish closed areas to scallop fishing. Roughly one-half of the productive scallop grounds on Georges Bank and Nantucket Shoals were closed to both groundfish and scallop gear during most of the time since December 1994. Limited openings to allow scallop fishing in closed areas contributed more than half of Georges Bank landings during 1999-2000 and 2004-2006. Poor recruitment in the mid-2000s and the reduction of biomass in the Georges Bank access areas have led to reductions in landings in the most recent years (NEFSC 2010). There are currently two limited access areas on Georges Bank: CAI and Closed Area II (CAII) (Figure 1). A third access area, Nantucket Lightship (NLS) is in Southern New England waters but is generally considered part of the Georges Bank component of the rotational area program (Figure 1). Each of these areas has been routinely closed under the area rotation program to allow for the scallop resource within the area to grow to a harvestable level. In addition, closures of the access areas within these areas have occurred as a result of high yellowtail flounder bycatch. Most recently, NLS was closed in 2011 to prevent a level of scallop catch that would be too high relative to the abundance of scallops at the time, and to prevent excessive yellowtail flounder bycatch.

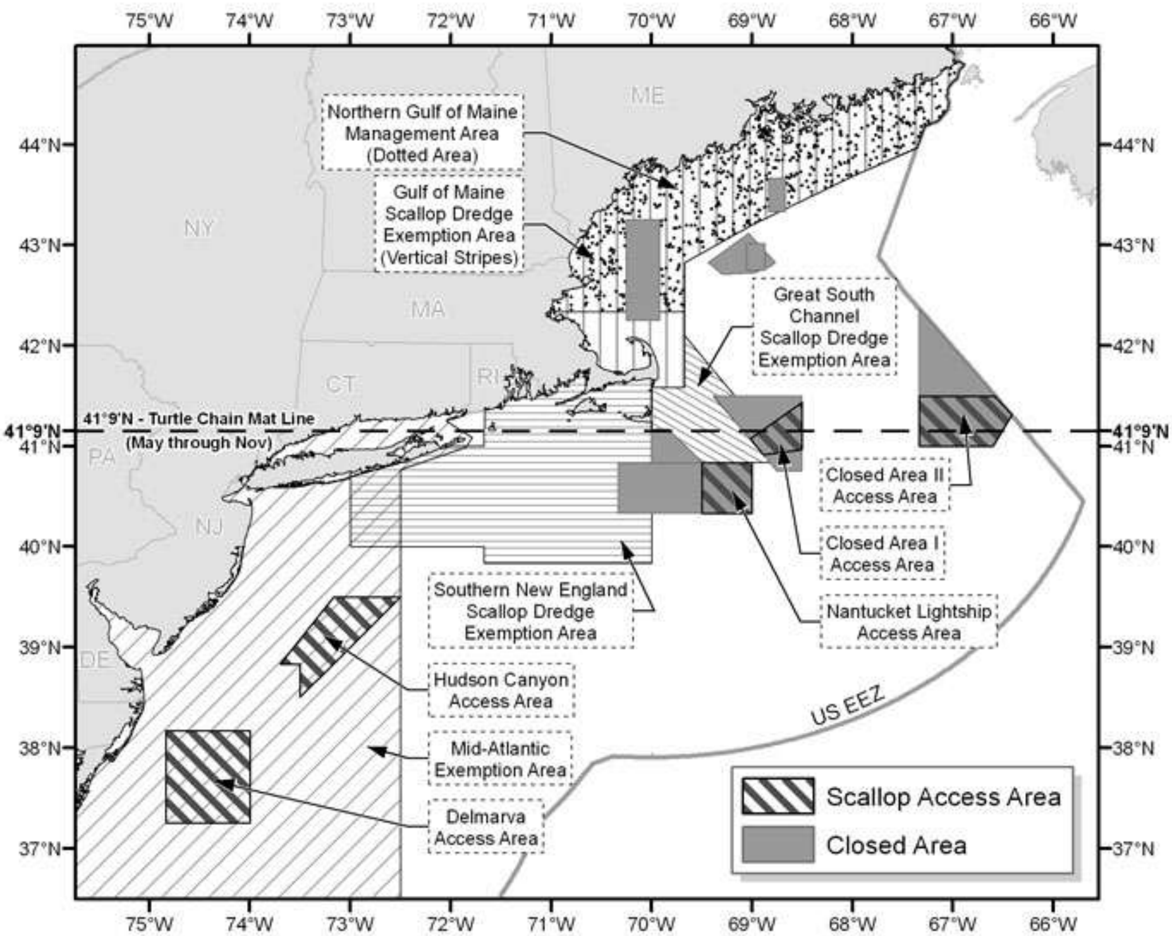

Figure 1. Scallop management areas (including the five current rotational access areas) in New England and the Mid-Atlantic.

Until recently, Mid-Atlantic landings were lower than those on Georges Bank. Mid-Atlantic landings during 1962-1982 averaged less than 1,800 metric tons per year. An upward trend in both recruitment and landings has been evident in the Mid-Atlantic since the mid-1980s. Landings peaked in 2004 at 24,494 metric tons. In the Mid-Atlantic, there have been four rotational scallop closure areas. Two areas (Hudson Canyon [HC] and Virginia Beach [VB]) were closed in 1998 and then reopened in 2001. Although the small VB closure was unsuccessful, scallop biomass built up in HC while it was closed, and substantial landings were obtained from HC during 2001-2007. This area was again closed in 2008, but reopened in 2011. A third rotational closure area, the Elephant Trunk (ET) area east of Delaware Bay, was closed in 2004, after extremely high densities of small scallops were observed in surveys during 2002 and 2003. About 30,000 metric tons of scallops have been landed from that area since it reopened in 2007. In August 2011, the ET area was re-designated as an open area (*i.e.*, it is no longer a rotational closure area with limited access). A fourth closure area (DMV), directly south of the ET area, was closed in 2007, reopened in 2009, and is now scheduled to be closed again for the remainder of the 2012 fishing year due to low biomass of harvestable scallops and a recently discovered high abundance of small scallops.

Landings from other areas (Gulf of Maine and Southern New England) are minor in comparison to Georges Bank and the Mid-Atlantic. Most of the Gulf of Maine scallop population is assessed and managed by the State of Maine because it is primarily in state waters. Gulf of Maine landings in 2009 were less than 1% of the total U.S. scallop landings. Maximum landings in the Gulf of Maine were 1,614 metric tons during 1980 (NEFSC 2010).

At present, the U.S. scallop fishery operates primarily through vessels with "limited access" (LA) permits. Two types of allocation are given to each vessel. The first are trips (with a trip limit, typically of 18,000 pounds of meats) to rotational access areas that had been closed to scallop fishing in the past. The second are days at-sea (DAS), which can be used in areas outside the closed and access areas. Vessels fishing under DAS with either a standard 15-foot New Bedford scallop dredge or with scallop trawl gear are restricted to a seven-man crew in order to limit their shucking power. Another set of vessels that are required to use one, 10.5-foot dredge have a more restrictive five-man crew restriction. The percentage of landings from the access area trips have increased since the access area program began in 1999; in recent years, about 60% of landings are from the access areas.

The remainder of U.S. scallop landings comes from vessels operating under "general category" (GC) permits that are restricted to 400 pounds per trip, with a maximum of one trip per day. Landings from these vessels were less than 1% of total landings in the late 1990s, but increased dramatically after 2000 to more than 10% in 2005 and 2006. After the limited access program was developed for this fleet of scallop vessels in 2008 through Amendment 11 (73 FR 20090, April 14, 2008; and 73 FR 23386, April 30, 2008), the fishery was capped at 10% of the total scallop catch in 2008 and 2009 (about five million pounds). This type of permit had been open access, but was converted to an individual transferable quota (ITQ) fishery in March 2010 with a cap of 5% of the total scallop catch (about three million pounds).

Principal ports in the U.S. scallop fishery are New Bedford, Massachusetts; Cape May, New Jersey; and Newport News, Virginia. In 2010, these three ports combined for over 75% of all landed value for scallops in the U.S., with New Bedford alone accounting for 52% of all landed value for scallops. New Bedford style scallop dredges are the main gear type in all regions, although some scallop vessels use otter trawls in the Mid-Atlantic. Recreational catch of scallops is negligible, although a small amount of catch in the Gulf of Maine via hand and net bags is due to recreational divers.

The Scallop FMP was implemented on May 15, 1982. From 1982 to 1994, the primary management control was a minimum average meat weight requirement for landings. Amendment 4 to the FMP, implemented in 1994, changed the management strategy from meat count regulation to limited access, effort control, and gear regulations for the entire U.S. Exclusive Economic Zone (EEZ). In 1994, three areas of Georges Bank were closed to scallop fishing under the Northeast Multispecies FMP in order to protect regulated groundfish stocks: CAI, CAII, and NLS (NEFSC 2010). Incremental restrictions were made on DAS, minimum ring size, and crew limits. Subsequent amendments and framework adjustments to the FMP during the 1990s added new management measures or revised existing measures such as the establishment of two closed areas in the Mid-Atlantic, changes to the DAS reduction schedule, and vessel upgrade/replacement provisions.

The limited access program and DAS allocations, first established under Amendment 4, remain the basic effort control measures for the scallop fishery. There are eight different types of scallop limited access permits. Depending on the type of limited access permit for which a vessel qualified, the owner of a vessel with a scallop limited access permit may have the option of fishing with dredge gear, a small dredge, or trawl nets. The permit categories are:

- Full-time dredge gear (Category 2)
- Part-time dredge gear (Category 3)
- Occasional dredge gear (Category 4)
- Full-time small dredge (Category 5)
- Part-time small dredge (Category 6)
- Full-time trawl (Category 7)
- Part-time trawl (Category 8)
- Occasional trawl (Category 9)

Open area DAS and scallop access area trip allocations to a scallop vessel vary depending on whether the vessel qualifies in the full-time, part-time, or occasional permit category. The greatest number of DAS access area trips are allocated to vessels that qualify in the full-time permit category.

Limited access vessels assigned to either the part-time or occasional categories can increase their DAS allocation by opting into the small dredge program, which effectively places them one category higher (*e.g.*, a part-time limited access vessel becomes a full-time limited access vessel in the small dredge program, and an occasional limited access vessel becomes a part-time limited access vessel in the small dredge program). The small dredge program requires participating vessels to: (1) fish exclusively with one dredge no more than 10.5 feet in width; (2) have only one dredge on board or in use; and (3) have no more than five people (versus seven for limited access vessels not in the small dredge program), including the operator, on board (NEFMC 2003). Crew limits affect how fast a haul of scallops can be shucked and, as a result, how quickly subsequent hauls can be made. However, crew limits do not apply in access areas because of the limitations on the amount of scallops that can be harvested per trip and the limit on the number of trips in each access area.

After fishing year 2000, fishing effort started to increase as more limited access vessels began to participate in the scallop fishery. The increase in total effort was mostly due to the increase in the number of vessels because total DAS allocations (mostly less than 120 days) were lower than the DAS allocations in the mid-1990s (over 142 days). The recovery of the scallop resource and the dramatic increase in fishable abundance after 1999 increased profits in the scallop fishery, thus leading to an increase in participation by limited access vessels that had been inactive during the previous years. Georges Bank closed areas were opened to scallop fishing starting in 1999 by Framework 11 (CAII) and later by Framework 13 (CAII, CAI, NLS), encouraging many vessel owners to take the opportunity to fish in those lucrative areas. Frameworks 14 and 15 provided controlled access to the Hudson Canyon and Virginia Beach access areas. As a result, 45 new limited access vessels became active in the scallop fishery after 2000 during the next four fishing years. The total number of full-time equivalent vessels reached 310 in 2003 and total fishing effort by the fleet increased to 31,864 days in 2003 from about 22,627 days in 2000.

There has been a steady decline in total DAS used by limited access scallop vessels from 1994-2010 as a result of the effort reduction measures since Amendment 4. Total fishing effort (DAS used) declined after 2003 even though the number of active vessels increased to 343 vessels in 2006 from 310 vessels in 2003. For 1999 through 2003, DAS totals include time fished in open and access areas, which accrued as DAS fished. With the implementation of Amendment 10 in 2004, the limited access vessels were allocated DAS for open areas and a number of trips for the specific access areas with no open area trade-offs. The open area allocations were reduced to 42 DAS in 2004 whereas full-time vessels were allocated seven access area trips in the same year via Framework 16. Even though total DAS equivalent allocations remained around the same levels during 2005-2007 (at about 110 equivalent days), fishing effort (*i.e.*, fleet DAS used) increased in the 2007 fishing year as many vessels took their unused 2005 Hudson Canyon access area trips in that year. If not for those trips, total effort in the scallop fishery would probably have stayed constant during 2005-2007 with almost all qualified limited access vessels participating in the fishery.

Total DAS (including days spent fishing in access areas) used declined further in 2008 to 24,121 days as the open area DAS allocations were reduced by 30% from 51 days to 35 days per full-time vessel, but increased to 26,300 in 2009 as the limited access vessels received access area trips (five trips per vessel). Open area DAS allocations were slightly higher in 2010 (38 DAS versus 37 DAS in 2009). Total DAS-used by the limited access vessels were slightly higher in FY 2010 despite lower number of access area trips (four trips per vessel). The impact of the decline in effort below 30,000 DAS since 2005 (with the exception of 2007) on scallop revenue per vessel was small, however, due to the increase in LPUE from about 1,600 pounds per DAS in 2007 to over 2,000 pounds per DAS since 2010.

The limited access scallop fishery consists of 347 vessels. It is primarily full-time, with 250 full-time dredge, 52 full-time small dredge vessels, and 11 full-time net boats. No occasional permits are left in the fishery because those 32 were converted to part-time small dredge in 2010. Similarly, there are only two part-time permits because most were converted into full-time dredge vessels after 2000. Since 2001, there has been considerable growth in fishing effort and landings by vessels with general category permits, primarily as a result of resource recovery and higher scallop prices.

Amendment 11 implemented a limited entry program for the general category fishery allocating 5% of the total projected scallop catch to the general category vessels qualified for limited access. The main objective of the action was to control capacity and mortality in the general category scallop fishery. There is also a separate limited entry program for general category fishing in the Northern Gulf of Maine (NGOM). In addition, a separate limited entry incidental catch permit was adopted that permits vessels to land and sell up to 40 pounds of scallop meat per trip while fishing for other species. During the transition period to the full-implementation of Amendment 11, the general category vessels were allocated 10% of the scallop TAC.

Since the full implementation of Amendment 11 provisions did not occur until March 2010, it is too early to assess the impacts of this amendment on the ownership patterns in the general category vessels. However, the number of general category permits declined considerably after 2007 as a result of the Amendment 11 provisions. Although not all vessels with general category permits were active in the years preceding 2008, there is no question that the number of vessels (and owners) that hold a limited access general category permit under the Amendment 11 regulations is less than the number of general category vessels that were active prior to 2008.

Most limited access category effort is from vessels using scallop dredges, including small dredges. The number of vessels using scallop trawl gear has decreased continuously and has been at 11 full-time trawl vessels since 2006. According to 2009-2010 Vessel Trip Report (VTR) data, the majority of these vessels (10 out of 11 in 2010) landed scallops using dredge gear even though they had a trawl permit. In comparison, there has been an increase in the numbers of full-time and part-time small dredge vessels after 2002. About 80% of the scallop pounds are landed by full-time dredge and about 13% landed by full-time small dredge vessels since the 2007 fishing year. Most general category effort is, and has been, from vessels using scallop dredge and other trawl gear. The percentages of scallop landings show that landings made with a scallop dredge in 2011 continue to be the highest compared to other general category gear types. The percentages of scallop landings with otter trawl gear in 2008 and 2009 were the highest they have been since 2001, but are still significantly less than dredge landings.

In the fishing years 2009 and 2010, the landings from the scallop fishery stayed above 56 million pounds, surpassing the levels observed historically. The recovery of the scallop resource and consequent increase in landings and revenues is striking given that average scallop landings per year were below 16 million pounds during the 1994-1998 fishing years, less than one-third of the present level of landings. The landings by the general category vessels declined, however, in 2010 as a result of the Amendment 11 implementation that restricts TAC for the limited access general category (LAGC) fishery to 5.5% of the total catch, which is now specified as the annual catch limit (ACL) under Amendment 15.

In July 2011, NMFS finalized both Amendment 15 and Framework 22 to the Scallop FMP. Amendment 15 established a mechanism to set ACLs and accountability measures (AMs) in order for the Scallop FMP to comply with the provisions of the Magnuson-Stevens Fishery Conservation and Management Act (MSA). The primary need for Framework 22 was to set management measures for the scallop fishery for the 2011 and 2012 fishing years, as well as to set default measures for 2013 in case the action that would set the 2013 and 2014 measures was delayed past the start of the 2013 fishing year. Framework 22 also addressed other issues such as updated RSA priorities and agency/industry compliance with RPM #1 of the 2008 Opinion regarding effort reduction in the Mid-Atlantic. The Framework 22 measures as submitted to NMFS also included an acceptable biological catch (ABC) level as required by the reauthorized MSA of 2007. Excluding discards and incidental mortality, the ABC was set at 27,269 metric tons for 2011 and 28,961 metric tons for 2012. The ABC level was not set for 2013, as that will be done in the next biennial scallop framework action when final specifications are set for that fishing year. However, a default ABC level of 28,700 metric tons was recommended for 2013 and included in the final rule for the action in the event that the rulemaking for the 2013-2014 specifications was delayed past the start of the 2013 fishing year.

In regards to and in fulfillment of RPM #1 from the 2008 Opinion, Framework 22 permitted a maximum of one access area trip in the Mid-Atlantic from June 15 to October 31 with no seasonal closures of Mid-Atlantic access areas. This differed from the previous measures that were in place under Framework 21 to respond to RPM #1, in which the DMV access area was closed from September 1 to October 31 and vessels were restricted to taking two of the three allocated Mid-Atlantic access area trips from June 15 to October 31. A caveat was also included in Framework 22 such that if a vessel traded for two additional Mid-Atlantic access area trips (to have four total), that vessel would be permitted to take up to two trips during the sea turtle window instead of one.

Framework 23 to the Scallop FMP took effect on May 7, 2012, and among other measures, requires all limited access scallop vessels (regardless of permit category or dredge size), as well as LAGC IFQ vessels that have a dredge with a width of 10.5 feet or greater, to use a TDD in the Mid-Atlantic (west of 71° W longitude) from May through October beginning in 2013. This new dredge requirement is delayed until May 1, 2013, to allow for gear manufacturers to produce dredges for the whole fleet by that time. The TDD is expected to provide a conservation benefit to sea turtles by reducing the severity of interactions on the ocean bottom. By deflecting sea turtles over the dredge rather than under the cutting bar, the TDD is expected to reduce sea turtle injuries due to contact with the dredge frame on the ocean bottom (including being crushed under the dredge frame). When combined with the effects of chain mats, which decrease captures in the dredge bag, the TDD should provide greater sea turtle benefits, by reducing serious injury/mortality, than a standard New Bedford dredge.

The measures included in Framework 23 meet the requirements of RPM #2 of the 2008 Opinion because they incorporate a gear modification that is expected to reduce the severity of sea turtle interactions, but do not result in more than a minor change in the fishery. Studies have shown that the TDD is just as efficient, and may be more efficient, at catching scallops and can also significantly decrease the unwanted bycatch of flounders and skates (Smolowitz *et al*. 2012). Therefore, this requirement is not expected to impact fishing behavior significantly, and may in fact lead to less effort expended to land a given quota of scallops. It is possible that some vessels will choose to fish in areas and seasons outside of the TDD requirement, but some of the limited access fleet is already using this dredge inside the TDD area, and more vessels are expected to switch to this dredge when fishing in the TDD area prior to the required date due to reports of increased scallop catch and reduced finfish bycatch compared to the standard commercial dredge. Furthermore, if vessels do fish outside the area and season in which the TDD is required, they will be fishing in areas and at times when sea turtles are much less abundant. It is for these reasons that NMFS will be requiring the use of the TDD per the conditions described above starting in May 2013; these requirements are considered as part of the proposed action.

Operation of the scallop fishery has also been modified as a result of measures implemented under the ESA. 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 (referred to as a “chain mat”) between the sweep and the cutting bar when fishing in Mid-Atlantic waters south of 41° 9.0’ N from the shoreline to the outer boundary of the EEZ during the period of May 1 through November 30 each year. The requirement was modified by emergency rule in November 2006 (71 FR 66466). In November 2007, NMFS re-proposed the chain-mat modified dredge requirements in the sea scallop fishery, with some modifications (72 FR 63537). That action added a transiting provision and clarified the regulatory text regarding the chain mat-modified gear including that the spaces formed by the intersecting chains must have no more than four sides and the length of each side of the opening must be less than or equal to 14 inches (73 FR 18984, April 8, 2008). In 2009, the chain mat regulations were further modified by NMFS in a rule that (a) clarified where on the dredge the chain mat should be hung, (b) excluded the sweep from the requirement that the side of each opening in the chain mat be less than or equal to 14 inches, and (c) added definitions of the sweep and the diamonds, which are terms used to describe parts of the scallop dredge gear (74 FR 46930; September 14, 2009).

Both chain mats and TDDs are expected to reduce the severity (*e.g.*, mortality and serious injury) of sea turtle interactions with scallop dredge gear. However, the gear modifications are not expected to reduce the actual number of sea turtle interactions with scallop dredge gear. Based on the condition of sea turtles observed captured in the dredge bag of scallop dredge gear as well as the configuration of the gear and fishing method, interactions are likely occurring both on or near the bottom and in the water column. The chain mat is intended to keep sea turtles out of the dredge bag thus preventing injuries that occur to them once they are in the bag (*e.g.*, crushing in the dredge bag, crushing on deck). Use of the chain mat on scallop dredges is not expected to eliminate or reduce injuries to sea turtles that occur as a result of the sea turtle coming into contact with that part of the scallop dredge gear forward of the chain mat (*e.g.*, the frame and the cutting bar) when the gear is fishing on or near the bottom. However, a TDD is expected to ameliorate this risk to some degree. Additional information on the use of gear modifications in the fishery is presented in section 5.2, in which the effects of the continued operation of the fishery, including TDDs equipped with chain mats, are analyzed.

2.1.1 Exempted Fishing Permits and Scientific Research under the Scallop FMP

Regulations at 50 CFR 600.745 allow the Northeast Regional Administrator to authorize the targeting or incidental harvest of species managed under an FMP or fishing activities that would otherwise be prohibited for scientific research, limited testing, public display, data collection, exploration, health and safety, environmental cleanup, hazardous waste removal purposes, or for educational activities. Every year, NMFS NERO may issue a small number of EFPs and/or exempted educational activity authorizations (EEAAs) exempting the collection of a limited number of scallops from Northeast Federal waters from regulations implementing the Scallop FMP. For example, between 2007 and 2011, NERO issued five EFPs and one EEAA relative to the scallop fishery. The EFPs and EEAA involved fishing by commercial or research vessels using methods that were similar or identical to those of the scallop fishery, which is the primary subject of this Opinion. The only differences involved (a) the use of modified gear (*e.g.*, dual mesh twine tops, low profile dredges), which was not authorized under the FMP at the time, or (b) requests for additional DAS or trips to scallop access areas beyond what the annual specifications for the fishery allowed. Nearly all the permitted fishing effort occurred in waters off southern New England and the Mid-Atlantic.

For the five EFPs and one EEAA examined between 2007 and 2011, we were able to conclude that in all cases, the types and rates of interactions with listed species from those activities would be similar to those analyzed in the most recent Opinion. Given our past experience with and knowledge of the usual applicants (and when and where they fish), we expect that future EFPs and/or EEAAs would propose fishing types and associated fishing effort similar to previous EFPs/EEAAs and, therefore, not introduce a significant increase in effort levels for the overall fishery considered in this Opinion. As a result, the issuance of those EFPs and EEAAs would be expected to fall within the level of effort and impacts considered in this Opinion. For example, the issuance of an EFP to an active commercial vessel that is similar to the ones described above likely does not add additional effects compared to those that would otherwise accrue from the vessel's normal commercial activities (unless, for example, that vessel was looking for exemptions from the TDD requirements in areas with high concentrations of sea turtles). Similarly, issuance of an EFP or EEAA to a vessel to conduct a minimal number of scallop tows/trips with dredge or bottom trawl gear likely would not add sufficient fishing effort to produce a detectable change in the overall amount of fishing effort in a given year. Therefore, we consider the future issuance of most EFPs and EEAAs by NMFS NERO to be within the scope of this Opinion. If an EFP or EEAA is proposed which modifies this agency action in a manner that causes an effect to listed species or critical habitat not considered in this Opinion, then consultation will be reinitiated.

Each year approximately 3% of the total allowable scallop catch is used to fund the Scallop RSA Program. In 1998, the scallop industry opted to set aside a portion of their total annual scallop catch in order to promote greater industry involvement in scientific research. Now, each year when NMFS NERO and the New England Fishery Management Council (NEFMC) set the annual catch limits for the fishery, a portion is reserved for cooperative research projects. From 2000-2011, 119 research projects were supported through RSA allocations. In FY 2012, 13 cooperative research projects between fishermen and scientists were selected for funding. Approximately 60 vessels will participate in the program in FY 2012, and share in the sale proceeds of an estimated 1.2 million pounds of scallops. As is the case with EFPs and EEAAs, we consider the future issuance of most RSA grants to be within the scope of this Opinion, as that level of scallop fishing effort has already been accounted for and analyzed here. If we determine that the distribution of effort or the types of gear utilized in an RSA project are not within the scope of this Opinion, additional section 7 consultation would be necessary.

2.1.2 Scallop Fishery Observer Program

Fisheries observer programs for listed species in the Northeast cover nearly all fisheries for which there is a Federal FMP and some state fisheries as well (*e.g.*, North Carolina southern flounder fishery). Observer coverage is typically allocated in proportion to fishing effort, by month and port, with vessels selected randomly for coverage (Murray 2009a). Levels of observer coverage in these fisheries may also vary depending on the amount of funding available to offset the cost of observers and the likelihood of bycatch of non-target species (including listed species) during normal fishing operations. In the Northeast Region, there are two important fisheries observer programs: the Northeast Fisheries Observer Program (NEFOP) and the At-Sea Monitoring Program (ASM), both of which are overseen by the NEFSC Fisheries Sampling Branch (FSB). Fisheries observers undergo an extensive three-week training class, led by the NEFSC; the sea turtle and sturgeon components include classroom training, hands on workshops, and exams on species identification, measuring, tagging, and handling (among other things), and typically last one full day. Ultimately, the data collected by fisheries observer programs can be used to estimate the amount and extent of bycatch of listed species in commercial fisheries and to track and monitor the ITSs of FMP Opinions.

The scallop fishery utilizes an industry-funded observer program which requires scallop vessel owners to pay for the cost of carrying observers. A portion of the scallop resource (1% of the total catch) is set-aside to help offset the cost of observers. Information on the industry-funded scallop fishery observer program under the NEFOP can be found at: http://www.nefsc.noaa.gov/fsb/scallop/. For FY 2012, observer coverage target rates for the scallop fishery are approximately 15% for open areas, 8% for DMV, 23% for CAI, 23% for CAII, 20% for HC, and 22% for NLS. Regulations implementing the scallop fishery observer program can be found at 50 CFR 648.11(g).

2.1.3 Summary of the Fishery

In Georges Bank and the Mid-Atlantic, scallops are harvested primarily at depths of 30-100 meters, while the bulk of landings from the Gulf of Maine are from nearshore relatively shallow waters (<40 meters) (Murray 2004b; 2005; NEFSC 2010). Landings from Georges Bank and the Mid-Atlantic dominate the fishery (NEFSC 2010). Scallop biomass increased considerably in both the Georges Bank and Mid-Atlantic areas since the mid-1990s (Hart and Rago 2006; NEFMC 2007). In Georges Bank, biomass and abundance increased during 1995-2000 after implementation of closures and effort reduction measures (NEFSC 2010). Scallop abundance and biomass have been modestly declining during recent years due to poor recruitment and reopening of portions of the groundfish closed areas (NEFSC 2010). In the Mid-Atlantic, abundance and biomass were at low levels during 1975-1997, and then increased rapidly during 1998-2003 due to area closures, reduced fishing mortality, changes in fish selectivity, and strong recruitment (NEFSC 2010). Biomass was relatively stable during 2003-2006 (NEFSC 2010). LPUE in the fishery more than quadrupled between 1998 and 2001, and remained high during 2001-2009 (NEFSC 2010). Data from observed (open area) trips indicates that the number of hours actually fished during a day absent from port dropped from around 18 hours in the mid-1990s to 14 hours or less during the most recent years (NEFSC 2010). The number of hours fished during trips to formerly closed areas is considerably less (NEFSC 2010). Overfishing is not occurring in the scallop fishery, and the stock is not overfished (NEFSC 2010).

Currently, an emergency action has been issued to close DMV from June 13 through November 10, 2012, and to instead allocate those limited access, full-time vessel trips into CAI on Georges Bank. Survey results from DMV in FY 2011 (March 1, 2011, through February 29, 2012) recently became available and indicate that the overall scallop biomass in DMV is substantially lower than expected for FY 2012 (March 1, 2012, through February 28, 2013). The results also indicate that DMV is one of the few areas in the Mid-Atlantic where recruitment (*i.e.*, evidence of young scallops) was noticeable. Although Framework 22 allocated DMV FY 2012 trips to many scallop vessels, these recent survey results represent the best scientific information to-date regarding the status of the scallop resource in DMV and indicate that it should be closed in FY 2012. As a result of this closure, and its likely extension until the end of February 2013, the proposed action will include the closure of DMV throughout the remainder of FY 2012, and the potential increase in effort in other areas of the fishery including in CAI on Georges Bank.

2.2 Action Area

The management unit for the Scallop FMP is defined as the range of the scallop resource along the U.S. Atlantic coast. Scallops range from Newfoundland to North Carolina along the continental shelf of North America. The direct and indirect effects of the scallop fishery managed under the Scallop FMP have been summarized as impacts resulting from the fishing gear coming in contact with and disturbing the sea bed, and the removal of various species from the environment (some of which are discarded as unwanted or regulatory discards) (NEFMC 2003). For the purposes of this Opinion, the area to be directly and indirectly affected by the scallop fishery (the action area) is the area in which the scallop fishery operates, broadly defined as all EEZ waters from Maine through the Virginia/North Carolina scallop stock area (which ends at the southern boundaries of NMFS statistical areas 635, 636, 637, 638, and 639, at 35° N latitude). The action area also includes adjoining state waters that are affected through the regulation of activities of Federal scallop permit holders fishing in those waters as well as intracoastal waters traversed by scallop fishing vessels as they make their way to/from port.

3.0 STATUS OF LISTED SPECIES AND CRITICAL HABITAT

We have determined that the action being considered in this Opinion may affect the following ESA-listed species in a manner that will likely result in adverse effects:

Common name	**Scientific name**	**ESA Status**
Loggerhead sea turtle - NWA DPS[1]	*Caretta caretta*	Threatened
Leatherback sea turtle	*Dermochelys coriacea*	Endangered
Kemp's ridley sea turtle	*Lepidochelys kempii*	Endangered
Green sea turtle	*Chelonia mydas*	Endangered[2]
Atlantic sturgeon	*Acipenser oxyrinchus oxyrinchus*	
Gulf of Maine (GOM) DPS		Threatened
New York Bight (NYB) DPS		Endangered
Chesapeake Bay (CB) DPS		Endangered
Carolina DPS		Endangered
South Atlantic (SA) DPS		Endangered

We have determined that the action being considered in this Opinion will not affect shortnose sturgeon (*Acipenser brevirostrum*), the Gulf of Maine DPS of Atlantic salmon (*Salmo salar*), and hawksbill sea turtles (*Eretmochelys imbricata*), and is not likely to adversely affect North Atlantic right whales (*Eubalaena glacialis*), humpback whales (*Megaptera novaengliae*), fin whales (*Balaenoptera physalus*), sei whales (*Balaenoptera borealis*), blue whales (*Balaenoptera musculus*), and sperm whales (*Physeter macrocephalus*), all of which are listed as endangered under the ESA. The following discussions are our rationale for these determinations.

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. The species is anadromous in the southern portion of its range (*i.e.*, south of Chesapeake Bay), while some northern populations are amphidromous (NMFS 1998a). Given the range of the species (remaining mostly in the river systems, with some coastal migrations between rivers), and the proposed action occurring in more offshore ocean areas, shortnose sturgeon are not expected to

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[1] NWA DPS = Northwest Atlantic DPS, the only loggerhead DPS expected to occur in the action area

[2] 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.

be present in areas where the scallop fishery operates. In addition, effects are not expected since interactions with shortnose sturgeon have never been documented from the scallop fishery. We have reviewed all available observer records and there have been no observed captures of shortnose sturgeon in scallop dredge gear or any other gear when the primary trip or haul target was scallops (NEFOP database). Because there are no proposed changes to the scallop fishery that would increase the likelihood of interactions between shortnose sturgeon and this fishery, we do not anticipate any future interactions. Because of this, we do not expect any effects to shortnose sturgeon from this fishery.

The naturally spawned and conservation hatchery populations of anadromous Atlantic salmon whose freshwater range occurs in the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River, including those that were already listed in November 2000, are listed as endangered under the ESA (NMFS 2009a, 2009b). These populations include those in the Dennys, East Machias, Machias, Pleasant, Narraguagus, Ducktrap, Sheepscot, Penobscot, Androscoggin, and Kennebec Rivers as well as Cove Brook. Juvenile salmon in New England rivers typically migrate to sea in May after a two- to three-year period of development in freshwater streams, and remain at sea for two winters before returning to their U.S. natal rivers to spawn (Reddin 2006). The preferred habitat of post-smolt salmon in the open ocean is principally the upper ten meters of the water column, although there is evidence of forays into deeper water for shorter periods. In contrast, adult Atlantic salmon demonstrate a wider depth profile (ICES 2005). Results from a 2001-2003 post-smolt trawl survey in the nearshore waters of the Gulf of Maine indicate that Atlantic salmon post-smolts are prevalent in the upper water column throughout this area in mid to late May (Lacroix and Knox 2005). Therefore, fishing close to the bottom with dredge and trawl gear, as practiced throughout the scallop fishery, reduces the potential for catching Atlantic salmon as either post-smolts or adults.

In its report on salmon bycatch, the Working Group for North Atlantic Salmon (WGNAS) concluded that bycatch of Atlantic salmon in Northeast Atlantic commercial fisheries was not an obvious concern. The 2006 WGNAS report also discussed potential salmon bycatch implications from these fisheries and indicated there was insufficient information to quantify bycatch, although based on information reviewed so far, there was no evidence of major bycatch of salmon in Northeast fisheries (ICES 2006). We find it is highly unlikely that the action being considered in this Opinion will affect the Gulf of Maine DPS of Atlantic salmon given that operation of the scallop fishery does not occur in or near the rivers where concentrations of Atlantic salmon are likely to be found. We have reviewed all available observer records from the NEFOP and there have been no observed captures of Atlantic salmon in scallop dredge gear or any other gear when the primary trip or haul target was scallop. Because there are no proposed changes to the scallop fishery that would increase the likelihood of interactions between Atlantic salmon and this fishery, we do not anticipate any future interactions. Because of this, we do not expect any effects to Atlantic salmon from this fishery. Thus, neither this species nor its designated critical habitat will be considered further in this Opinion.

The hawksbill sea turtle is uncommon in the waters of the continental U.S. Hawksbills prefer tropical, coral reef habitats, such as those found in the Caribbean and Central America. The waters surrounding Mona and Monito Islands (Puerto Rico) are designated as critical habitat for the species, and Buck Island (St. Croix, U.S. Virgin Islands) also contains 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 in these areas is rare. Hawksbills have been recorded from all U.S. states adjacent to the Gulf of Mexico and along the east coast of the U.S. as far north as Massachusetts, although 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 the scallop fishery does not operate in waters that are typically used by hawksbill sea turtles, it is highly unlikely that the fishery will adversely affect this sea turtle species. We have reviewed all available observer records from the NEFOP and there have been no observed captures of hawksbill sea turtles in scallop dredge gear or any other gear when the primary trip or haul target was scallop. Because there are no proposed changes to the scallop fishery that would increase the likelihood of interactions between hawksbills and this fishery, we do not anticipate any future interactions. Because of this, we do not expect any effects to hawksbill sea turtles from this fishery.

Right, humpback, and fin whales occur in Mid-Atlantic and New England waters over the continental shelf. Sei whales are also observed over the continental shelf although they typically occur over the continental slope or in basins situated between banks (NMFS 2011a). All four of these species follow a similar, general pattern of foraging at high latitudes (*e.g.*, southern New England and Canadian waters) in the spring and summer months and calving in lower latitudes (*i.e.*, off of Florida for right whales and in the West Indies for humpback whales) in the winter months (CeTAP 1982; Hain *et al.* 1992; Clark 1995; Perry *et al.* 1999; Horwood 2002; Kenney 2002). Based on this information, operation of the scallop fishery may overlap with the distribution of these four large whale species during part of each year, particularly in Mid-Atlantic waters in the early spring and fall, and in southern New England waters in the spring and summer. One interaction between a large whale and scallop fishing gear is known to have occurred. In 1983, a humpback whale became entangled in the cables of scallop dredge gear off of Chatham, Massachusetts. Nevertheless, we have determined that this was a unique and very rare event that is extremely unlikely to reoccur given that these large whales have the speed and maneuverability to get out of the way of oncoming scallop fishing gear. Also, observer coverage of many fishing trips using mobile gear (*e.g.*, dredge, trawl gear) has shown that these gear types do not pose a reasonable risk of entanglement or capture for large whales. Therefore, we believe that these four large whales are not likely to interact with gear used in the scallop fishery.

We have also determined that any effects of the continued operation of the scallop fishery on the availability of prey for humpback, fin, and sei whales will be insignificant and discountable. Like right whales, sei whales feed on copepods (Perry *et al.* 1999). As indicated above, the scallop fishery will not affect the availability of copepods for foraging sei whales because copepods are very small organisms that will pass through scallop fishing gear rather than being captured in it. Dense aggregations of late stage and diapausing *Calanus finmarchicus* in the Gulf of Maine and Georges Bank region will not be affected by the scallop fishery. In addition, the physical and biological conditions and structures of the Gulf of Maine and Georges Bank region and the oceanographic conditions in Jordan, Wilkinson, and Georges Basin that aggregate and distribute *Calanus finmarchicus* are not affected by the scallop fishery. Humpback and fin whales feed on krill as well as small schooling fish (*e.g.*, sand lance, herring, mackerel) (Aguilar 2002; Clapham 2002). Scallop fishing gear operates on or very near the bottom. Fish species caught in scallop gear are species that live in benthic habitat (on or very near the bottom) such as flounders versus schooling fish such as herring and mackerel that occur within the water column. Therefore, the continued operation of the scallop fishery will not affect the availability of prey for foraging humpback or fin whales. In addition, the scallop fishery does not operate in low latitude waters where the overwhelming majority of calving and nursing occurs for these large whale species (Aguilar 2002; Clapham 2002; Horwood 2002; Kenney 2002; Sears 2002). Therefore, the continued operation of the scallop fishery will not affect the oceanographic conditions that are conducive for calving and nursing. Based on this analysis, the continued operation of the scallop fishery is not likely to adversely affect right, humpback or fin whales.

Blue whales do not regularly occur in waters of the U.S. EEZ (Waring *et al.* 2012). In the North Atlantic, blue whales are most frequently sighted in the St. Lawrence from April to January (Sears 2002). No blue whales were observed during the Cetacean and Turtle Assessment Program (CeTAP) surveys of the Mid- and North Atlantic areas of the outer continental shelf (CeTAP 1982). Calving for the species occurs in low latitude waters outside of the area where the scallop fishery operates. Blue whales feed on euphausiids (krill) (Sears 2002) which are too small to be captured in scallop fishing gear. Given that the species is unlikely to occur in areas where the scallop fishery operates, and given that the operation of the fishery will not affect the availability of blue whale prey or areas where calving and nursing of young occurs, we have determined that the continued operation of the scallop fishery is not likely to adversely affect blue whales.

Unlike blue whales, sperm whales do regularly occur in waters of the U.S. EEZ. However, the distribution of the sperm whale in the U.S. EEZ occurs on the continental shelf edge, over the continental slope, and into mid-ocean regions (Waring *et al.* 2012). In contrast, the scallop fishery operates in continental shelf waters. The average depth of sperm whale sightings observed during the CeTAP surveys was 1,792 meters (CeTAP 1982). Female sperm whales and young males almost always inhabit waters deeper than 1,000 meters and at latitudes less than 40° N (Whitehead 2002). Sperm whales feed on larger organisms that inhabit the deeper ocean regions (Whitehead 2002). Calving for the species occurs in low latitude waters outside of the area where the scallop fishery operates. Given that sperm whales are unlikely to occur in areas (based on water depth) where the scallop fishery operates, and given that the operation of the fishery will not affect the availability of sperm whale prey or areas where calving and nursing of young occurs, we have determined that the continued operation of the scallop fishery is not likely to adversely affect sperm whales.

We have determined that the action being considered in the Opinion is not likely to adversely affect designated critical habitat for right whales in the Northwest Atlantic. This determination is based on the action's effects on the conservation value of the habitat that has been designated. Specifically, we considered whether the action was likely to affect the physical or biological features that afford the designated area value for the conservation of right whales. Critical habitat for right whales has been designated in the Atlantic Ocean in Cape Cod Bay, Great South Channel, and in nearshore waters off Georgia and Florida (50 CFR 226.13). Cape Cod Bay and Great South Channel, which are located within the action area, were designated as critical habitat for right whales due to their importance as spring/summer foraging grounds for the species. What makes these two areas so critical is the presence of dense concentrations of copepods. The scallop fishery will not affect the availability of copepods for foraging right whales because copepods are very small organisms that will pass through scallop fishing gear rather than being captured in it. Since the action being considered in this Opinion is not likely to affect the availability of copepods and these were the biological feature that characterized feeding habitat, this action is not likely to adversely affect designated critical habitat for right whales and, therefore, right whale critical habitat will not be considered further in this Opinion.

3.1 Status of Sea Turtles

Sea turtles continue to be affected by many factors occurring on the nesting beaches and in the water. Poaching, habitat loss, and nesting predation by introduced species affect hatchlings and nesting females while on land. Fishery interactions, vessel interactions, and channel dredging operations, for example, affect sea turtles in the neritic zone (defined as the marine environment extending from mean low water down to 200 meter (660 foot) depths, generally corresponding to the continental shelf (Lalli and Parsons 1997; Encyclopedia Britannica 2012)). Fishery interactions also affect sea turtles when these species and the fisheries co-occur in the oceanic zone (defined as the open ocean environment where bottom depths are greater than 200 meters (Lalli and Parsons 1997))[3]. As a result, sea turtles still face many of the original threats that were the cause of their listing under the ESA.

With the exception of loggerheads, sea turtles are listed under the ESA at the species level rather than as subspecies or DPSs. 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), 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).

[3] As described in Bolten (2003), oceanographic terms have frequently been used incorrectly to describe sea turtle life stages. In sea turtle literature the terms benthic and pelagic were used incorrectly to refer to the neritic and oceanic zones, respectively. The term benthic refers to occurring on the bottom of a body of water, whereas the term pelagic refers to in the water column. Sea turtles can be "benthic" or pelagic" in either the neritic or oceanic zones.

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

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

As noted above, a thorough assessment of the long-term effects of the spill on sea turtles has not yet been completed. However, the spill resulted in the direct mortality of many sea turtles and may have had 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.

3.1.1 Loggerhead sea turtle

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

Listing History
Loggerhead sea turtles were listed as threatened throughout their global range on July 28, 1978. Since that time, several status reviews have been conducted to review the status of the species and make recommendations regarding its ESA listing status. Based on a 2007 five-year status review of the species, which discussed a variety of threats to loggerheads including climate change, NMFS and FWS determined that loggerhead sea turtles should not be delisted or reclassified as endangered. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPSs should be identified for the

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

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

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

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

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

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

<u>Presence of Loggerhead Sea Turtles in the Action Area</u>
The effects of this proposed action are only experienced within the Atlantic Ocean. NMFS has considered the available information on the distribution of the nine DPSs to determine the origin of any loggerhead sea turtles that may occur in the action area. As noted in Conant *et al*. (2009), the range of the four DPSs occurring in the Atlantic Ocean are as follows: NWA DPS – north of the equator, south of 60° N latitude, and west of 40° W longitude; Northeast Atlantic Ocean (NEA) DPS – north of the equator, south of 60° N latitude, east of 40° W longitude, and west of 5° 36’ W longitude; South Atlantic DPS – south of the equator, north of 60° S latitude, west of 20° E longitude, and east of 60° W longitude; Mediterranean DPS – the Mediterranean Sea east of 5° 36’ W longitude. These boundaries were determined based on oceanographic features, loggerhead sightings, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. While adults are highly structured with no overlap, there may be some degree of overlap by juveniles of the NWA, NEA, and Mediterranean DPSs on oceanic foraging grounds (Laurent *et al*. 1993, 1998; Bolten *et al*. 1998; LaCasella *et al*. 2005; Carreras *et al*. 2006, Monzón-Argüello *et al*. 2006; Revelles *et al*. 2007). Previous literature (Bowen *et al*. 2004) has suggested that there is the potential, albeit small, for some juveniles from the Mediterranean DPS to be present in U.S. Atlantic coastal foraging grounds. These conclusions must be interpreted with caution however, as they may be

representing a shared common haplotype and lack of representative sampling at Eastern Atlantic rookeries rather than an actual presence of Mediterranean DPS turtles in US Atlantic coastal waters. A re-analysis of the data by the Atlantic loggerhead Turtle Expert Working Group has found that that it is unlikely that U.S. fishing fleets are interacting with either the Northeast Atlantic loggerhead DPS or the Mediterranean loggerhead DPS (Peter Dutton, NMFS, Marine Turtle Genetics Program, Program Leader, personal communication, September 10, 2011). Given that the action area is a subset of the area fished by U.S. fleets, it is reasonable to assume that based on this new analysis, no individuals from the Mediterranean DPS or Northeast Atlantic DPS would be present in the action area. Sea turtles of the South Atlantic DPS do not inhabit the action area of this consultation (Conant *et al*. 2009). 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 five-year status review for loggerheads (NMFS and USFWS 2007a), the TEWG report (2009), and the final revised recovery plan for loggerheads in the Northwest Atlantic Ocean (NMFS and USFWS 2008), 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 22-49 meters 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, 2011; McClellan and Read 2007; Mansfield *et al*. 2009).

Loggerhead sea turtles occur year round in ocean waters off North Carolina, South Carolina, Georgia, and Florida. In these areas of the South Atlantic Bight, water temperature is influenced by the proximity of the Gulf Stream. As coastal water temperatures warm in the spring, loggerheads begin to migrate to inshore waters of the Southeast United States (*e.g.*, Pamlico and Core Sounds) and also move up the U.S. Atlantic Coast (Epperly *et al.* 1995a, 1995b, 1995c; Braun-McNeill and Epperly 2004), occurring in Virginia foraging areas as early as April/May and on the most northern foraging grounds in the Gulf of Maine in June (Shoop and Kenney

1992). The trend is reversed in the fall as water temperatures cool. The large majority leave the Gulf of Maine by mid-September but some turtles may remain in Mid-Atlantic and Northeast areas until late fall. By December, loggerheads have migrated from inshore and more northern coastal waters to waters offshore of North Carolina, particularly off of Cape Hatteras, and waters further south where the influence of the Gulf Stream provides temperatures favorable to sea turtles (Shoop and Kenney 1992; Epperly *et al.* 1995b).

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

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

As presented on the next page, Table 1 (taken directly from the 2008 loggerhead recovery plan) highlights the key life history parameters for loggerheads nesting in the U.S.

Population Dynamics and Status

By far, the majority of Atlantic nesting occurs on beaches of the southeastern United States (NMFS and USFWS 2007a). For the past decade or so, the scientific literature has recognized five distinct nesting groups, or subpopulations, of loggerhead sea turtles in the Northwest Atlantic, divided geographically as follows: (1) a northern group of nesting females that nest from North Carolina to northeast Florida at about 29° N latitude; (2) a south Florida group of nesting females that nest from 29° N latitude on the East Coast to Sarasota on the West Coast; (3) a Florida Panhandle group of nesting females that nest around Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán group of nesting females that nest on beaches of the eastern Yucatán Peninsula, Mexico; and (5) a Dry Tortugas group that nests on beaches of the islands of the Dry Tortugas, near Key West, Florida and on Cal Sal Bank (TEWG 2009). Genetic analyses of mitochondrial DNA, which a sea turtle inherits from its mother, indicate that there are genetic differences between loggerheads that nest at and originate from the beaches used by each of the five identified nesting groups of females (TEWG 2009). However, analyses of microsatellite loci from nuclear DNA, which represents the genetic contribution from both

Table 1: 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).

parents, indicates little to no genetic differences between loggerheads originating from nesting beaches of the five Northwest Atlantic nesting groups (Pearce and Bowen 2001; Bowen 2003; Bowen *et al.* 2005; Shamblin 2007). These results suggest that female loggerheads have site fidelity to nesting beaches within a particular area, while males provide an avenue of gene flow between nesting groups by mating with females that originate from different nesting groups (Bowen 2003; Bowen *et al.* 2005). The extent of such gene flow, however, is unclear (Shamblin 2007).

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

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

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

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

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

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

Genetic studies of juvenile and a few adult loggerhead sea turtles collected from Northwest Atlantic foraging areas (beach strandings, a power plant in Florida, and North Carolina fisheries) show that the loggerheads that occupy East Coast U.S. waters originate from these Northwest

Atlantic nesting groups; primarily from the nearby nesting beaches of southern Florida, as well as the northern Florida to North Carolina beaches, and finally from the beaches of the Yucatán Peninsula, Mexico (Rankin-Baransky *et al*. 2001; Witzell *et al*. 2002; Bass *et al.* 2004; Bowen *et al*. 2004). The contribution of these three nesting assemblages varies somewhat among the foraging habitats and age classes surveyed along the east coast. The distribution is not random and bears a significant relationship to the proximity and size of adjacent nesting colonies (Bowen *et al*. 2004). Bass *et al*. (2004) attribute the variety in the proportions of sea turtles from loggerhead turtle nesting assemblages documented in different East Coast foraging habitats to a complex interplay of currents and the relative size and proximity of nesting beaches.

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

Maier *et al.* (2004) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the Southeast Coast of the U.S. (Winyah Bay, South Carolina to St. Augustine, Florida) during the period 2000-2003. A comparison of loggerhead catch data from this study with historical values suggested that in-water populations of loggerhead sea turtles along the southeast U.S. coast are larger, possibly an order of magnitude higher than they were 25 years ago, but the authors caution against 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 study of loggerhead abundance in the Indian River Lagoon System of Florida found a significant increase in the relative abundance of loggerheads over the last four years of the study (Ehrhart *et al.* 2007). However, there was no discernible trend in loggerhead abundance during the 24-year time period of the study (1982-2006) (Ehrhart *et al.* 2007). At St. Lucie Power Plant, data collected from 1977-2004 show an increasing trend of loggerheads at the power plant intake structures (FPL and Quantum Resources 2005).

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

As with other sea 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 (SEFSC 2009). The model results for population trajectory suggest that the population is most likely declining, but this result was very sensitive to the choice of the position of the parameters within their range and hypothesized distributions. The pelagic stage survival parameter had the largest effect on the model results. As a result of the large uncertainty in our knowledge of loggerhead life history, at this point predicting the future populations or population trajectories of loggerhead sea turtles with precision is very difficult. 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 sea turtle telemetry studies were conducted along the U.S. 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 NEFSC (2011b), 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 (NEFSC 2011b). 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 2012-2014, depending on available funds.

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

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

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

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

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

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

In addition to improvements in TED designs and TED enforcement, interactions between loggerheads and the shrimp fishery have also been declining because of reductions in fishing effort unrelated to fisheries management actions. The 2002 Opinion take estimates were based in part on fishery effort levels. In recent years, low shrimp prices, rising fuel costs, competition with imported products, and the impacts of recent hurricanes in the Gulf of Mexico have all impacted the shrimp fleets; in some cases reducing fishing effort by as much as 50% for offshore waters of the Gulf of Mexico (GMFMC 2007). As a result, loggerhead interactions and mortalities in the Gulf of Mexico have been substantially less than were projected in the 2002 Opinion. In 2008, the NMFS Southeast Fisheries Science Center (SEFSC) estimated annual number of interactions between loggerheads and shrimp trawls in the Gulf of Mexico shrimp fishery to be 23,336, with 647 (2.8%) of those interactions resulting in mortality (Memo from Dr. B. Ponwith, Southeast Fisheries Science Center to Dr. R. Crabtree, Southeast Region, PRD, December 2008). However, the most recent section 7 consultation on the shrimp fishery, completed in May 2012, was unable to estimate the total annual level of loggerhead interactions at present. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in at least thousands and possibly tens of thousands of interactions annually, of which at least hundreds and possibly thousands are expected to be lethal (NMFS 2012a).

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

There have been several published estimates of the number of loggerheads interacting annually with 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 SST, 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 interacting annually with U.S. Mid-Atlantic gillnet fisheries has also recently been published (Murray 2009a, 2009b). 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, SST, 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 2004c). NMFS has mandated gear changes for the HMS fishery to reduce sea turtle bycatch and the likelihood of death from those incidental takes that would still occur (Garrison and Stokes 2012). In 2010, there were 40 observed interactions between loggerhead sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2012). All of the loggerheads were released alive, with 29 out of 40 (72.5%) released with all gear removed. A total of 344.4 (95% CI: 236.6-501.3) loggerhead sea turtles were estimated to have interacted with the longline fisheries managed under the HMS FMP in 2010 based on the observed bycatch events (Garrison and Stokes 2012). The 2010 estimate is considerably lower than those in 2006 and 2007 and is well below the historical highs that occurred in the mid-1990s (Garrison and Stokes 2012). This fishery represents just one of several longline fisheries operating in the Atlantic Ocean. Lewison *et al.* (2004) estimated that 150,000-200,000 loggerheads were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries as well as others).

Documented interactions also occur in other fishery gear types and by non-fishery mortality sources (*e.g.*, hopper dredges, power plants, vessel collisions), although quantitative/qualitative estimates are only available for activities on which NMFS has consulted.

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

Climate change also has the potential to result in changes at nesting beaches that may affect loggerhead sex ratios. Loggerhead sea turtles exhibit temperature-dependent sex determination. Rapidly increasing global temperatures may result in warmer incubation temperatures and highly female-biased sex ratios (*e.g.*, Glen and Mrosovsky 2004; Hawkes *et al*. 2009); however, to the extent that nesting can occur at beaches further north where sand temperatures are not as warm, these effects may be partially offset. The BRT specifically identified climate change as a threat to loggerhead sea turtles in the neritic/oceanic zone where climate change may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution. In the threats matrix analysis, climate change was considered for oceanic juveniles and adults as well as for 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.

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). Based on the BRT report, it is unlikely that impacts from climate change will have a significant effect on the status of loggerheads over the scope of the action assessed in this Opinion, which, as explained later on in sections 5.0 and 6.0, is the next ten years. This is because significant changes to biological trajectories resulting from climate change are expected to occur gradually over time (on a century scale), rather than immediately (Parmesan and Yohe 2003). However, significant impacts from climate change in the future beyond ten years are to be expected, but the severity of and rate at which these impacts will occur is currently unknown. It is likely that once climate change impacts get to a certain level, there will be feedback loops that may cause indications of climate change (*e.g.*, increases in greenhouse gas concentrations, rising global temperatures, and sea level rise) to get much worse much more quickly (Torn and Harte 2006).

In terms of "climate forcing" (which is different from what we are defining as "climate change," in that it also factors in the effects of cyclical climate patterns such as the North Atlantic and Pacific Decadal Oscillations in addition to ongoing effects from anthropogenically-induced changes in climate under IPCC projections), Van Houtan and Halley (2011) recently developed climate-based models to investigate loggerhead nesting in the Northwest Atlantic and North Pacific. These models, which considered juvenile recruitment and breeding remigration, found that climate conditions/oceanographic influences explain loggerhead nesting variability, with climate models alone explaining an average of 60% (range 18%-88%) of the observed nesting changes over the past several decades. Hindcasts indicate that climatic conditions may have been a factor in past nesting declines in both the Atlantic and Pacific. However, in terms of future nesting projections, modeled climate data show a future positive trend for Atlantic nesting in Florida, with substantial increases through 2040 as a result of the Atlantic Multidecadal Oscillation signal (Van Houton and Halley 2011). Thus, independent of any dramatic losses of sea turtle nesting habitat in the Northwest Atlantic due to climate change, NWA DPS loggerheads are expected to increase their nesting output over the next few decades. Van Houton and Halley (2011) did not project nesting trends in the Northwest Atlantic beyond 2040 as forecasting beyond that point was not deemed possible given their methods. Much like our analyses of climate change, climate forcing analyses can only predict so far into the future.

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

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

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

Based on the information presented above, for purposes of this Opinion, we consider that the status of NWA DPS of loggerheads over the next ten years will be no worse than it is currently. Actions have been taken to reduce anthropogenic impacts to loggerhead sea turtles from various sources, particularly since the early 1990s. These include lighting ordinances, predation control, and nest relocations to help increase hatchling survival, as well as measures to reduce the mortality of pelagic immatures, benthic immatures, and sexually mature age classes from various fisheries and other marine activities (Conant *et al*. 2009). Recent actions have taken significant steps towards reducing the recurring sources of mortality and improving the status of all nesting stocks. For example, TED and chain mat regulations represent a significant improvement in the baseline effects of trawl and dredge fisheries on loggerheads in the Northwest Atlantic, although shrimp trawling is still considered to be one of the largest sources of anthropogenic mortality on loggerheads (SEFSC 2009). Nevertheless, loggerhead nesting has been on the rise since 2008 and Van Houton and Halley (2011) indicate that nesting in Florida, which contains by far the largest loggerhead rookery in the DPS, could substantially increase over the next few decades.

3.1.2 Leatherback sea turtle

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

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

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

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

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

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

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

Leatherbacks in the eastern Pacific face a number of threats to their survival. For example, commercial and artisanal swordfish fisheries off Chile, 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. Given the declines in leatherback nesting in the Pacific, some researchers have concluded that the leatherback is on the verge of extinction in the Pacific Ocean (*e.g.,* Spotila *et al.* 1996, 2000).

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

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

Atlantic Ocean
<u>Distribution and Life History</u>
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 (Sea Turtle Stranding and Salvage Network [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 one to 4,151 meters, but 84.4% of sightings were in waters less than 180 meters (Shoop and Kenney 1992). Leatherbacks were sighted in waters within a SST range similar to that observed for loggerheads: from 7° to

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

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

Leatherbacks are a long lived species (>30 years). They were originally believed to mature at a younger age than loggerhead sea turtles, with a previous estimated age at sexual maturity of about 13-14 years for females with nine years reported as a likely minimum (Zug and Parham 1996) and 19 years as a likely maximum (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 U.S. and Caribbean, female leatherbacks nest from March through July. In the Atlantic, most nesting females average between 150-160 centimeters curved carapace length (CCL), although smaller (<145 centimeters CCL) and larger nesters are observed (Stewart *et al*. 2007; TEWG 2007). They nest frequently (up to seven nests per year) during a nesting season and nest about every two to three years. They produce 100 eggs or more in each clutch and can produce 700 eggs or more per nesting season (Schultz 1975). However, a significant portion (up to approximately 30%) of the eggs can be infertile. 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 centimeters CCL, Eckert (1999) found that leatherback juveniles remain in waters warmer than 26°C until they exceed 100 centimeters CCL.

<u>Population Dynamics and Status</u>
As described earlier, sea turtle nesting survey data is important because it provides information on the relative abundance of nesting, and the contribution of each population/subpopulation to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually, and as an indicator of the trend in the number of nesting females in the nesting group. The five-year review for leatherback sea turtles (NMFS and USFWS 2007b) compiled the most recent information on mean number of leatherback nests

per year for each of the seven leatherback populations or groups of populations that were identified by the Leatherback TEWG as occurring within the Atlantic. These are: Florida, North Caribbean, Western Caribbean, Southern Caribbean, West Africa, South Africa, and Brazil (TEWG 2007).

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

The CETAP aerial survey conducted from 1978-1982 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 (CV=0.38) and 1,174 turtles (CV=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).

<u>Threats</u>

The five-year status review (NMFS and USFWS 2007b) and TEWG (2007) reports both provide summaries of natural as well as anthropogenic threats to leatherback sea turtles. Of the Atlantic sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear, particularly trap/pot gear. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), their diving and foraging behavior, their distributional overlap with the gear, their possible attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, and perhaps to the lightsticks used to attract target species in longline fisheries. Leatherbacks entangled in fishing gear generally have a reduced ability to feed, dive, surface to breathe, or perform any other behavior essential to survival (Balazs 1985). In addition to drowning from forced submergence, they may be more susceptible to boat strikes if forced to remain at the surface, and entangling lines can constrict blood flow resulting in tissue necrosis. The long-term impacts of entanglement on leatherback health remain unclear. Innis *et al*. (2010) conducted a health evaluation of leatherback sea turtles during direct capture (n=12) and disentanglement (n=7). They found no significant difference in many of the measured health parameters between entangled and directly captured turtles. However, blood parameters, including but not limited to sodium, chloride, and blood urea nitrogen, for entangled turtles showed several key differences that were most likely due to reduced foraging 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.*, Opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, occurred annually (since implementation of bycatch mitigation measures). Kemp's ridleys interacted with fisheries most frequently, with the highest level of mean annual mortality (2,700), followed by loggerheads (1,400), greens (300), and leatherbacks (40). The Southeast/Gulf of Mexico shrimp trawl fishery was responsible for the vast majority of U.S. interactions (up to 98%) and mortalities (more than 80%). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations. The most recent section 7 consultation on the shrimp fishery, completed in May 2012, was unable to estimate the total annual level of take for leatherbacks at present. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in a few hundred interactions annually, of which a subset are expected to be lethal (NMFS 2012a).

Leatherbacks have been documented interacting with longline, trap/pot, trawl, and gillnet fishing gear. For instance, an estimated 6,363 leatherback sea turtles were caught by the U.S. Atlantic tuna and swordfish longline fisheries between 1992 and 1999 (SEFSC 2001). Currently, the U.S. tuna and swordfish longline fisheries managed under the HMS FMP are estimated to capture 1,764 leatherbacks (no more than 252 mortalities) for each three-year period starting in 2007 (NMFS 2004c). In 2010, there were 26 observed interactions between leatherback sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2012). All leatherbacks were released alive, with all gear removed in 14 (53.8%) of the 26 captures. A total of 170.9 (95% CI: 104.3-280.2) leatherback sea turtles are estimated to have interacted with the longline fisheries managed under the HMS FMP in 2010 based on the observed bycatch events (Garrison and Stokes 2012). The 2010 estimate continues a downward trend since 2007 and remains well below the average prior to implementation of gear regulations (Garrison and Stokes 2012). Since the U.S. fleet accounts for only 5-8% of the longline hooks fished in the Atlantic Ocean, adding up the under-represented observed takes of the other 23 countries actively fishing in the area would likely result in annual take estimates of thousands of leatherbacks (SEFSC 2001). Lewison *et al.* (2004) estimated that 30,000-60,000 leatherbacks were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries).

Leatherbacks are susceptible to entanglement in the lines associated with trap/pot gear used in several fisheries. From 1990-2000, 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 2008b). Of the 128 confirmed events during this period, 117 events involved leatherbacks. NMFS identified the gear type and fishery for 72 of the 117 confirmed events, which included lobster (42[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 2002a). 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 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 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 NEFOP from 1994-1998 (excluding 1997) indicate that a total of 37 leatherbacks were

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

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 (SEFSC 2001). In addition to these, in September 1995, two dead leatherbacks were removed from an 11-inch (28.2-centimeter) monofilament shark gillnet set in the nearshore waters off of Cape Hatteras (STSSN unpublished data reported in 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, including in Canadian waters. Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland/Labrador were entangled in salmon nets, herring nets, gillnets, trawl lines, and crab pot lines. Leatherbacks are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo *et al.* 1994; Graff 1995). Gillnets are one of the suspected causes for the decline 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 Trinidad and Tobago with mortality estimated to be between 50% and 95% (Eckert and Lien 1999). Many of the sea turtles do not die as a result of drowning, but rather because the fishermen butcher them to get them out of their nets (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 turtle's 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 (Mrosovsky *et al*. 1984 and Hawkes *et al*. 2007 in NMFS and USFWS 2007b). However, due to the tendency of leatherbacks to have individual nest placement preferences and deposit some clutches in the cooler tide zone of beaches, the effects of long-term climate on sex ratios may be mitigated (Kamel and Mrosovsky 2004 in NMFS and USFWS 2007b). Additional potential effects of climate change on leatherbacks include range expansion and changes in migration routes as increasing ocean temperatures shift range-limiting isotherms north (Robinson *et al*. 2008). Leatherbacks have expanded their range in the Atlantic north by 330 kilometers in the last 17 years as warming has caused the northerly migration of the 15°C SST isotherm, the lower limit of thermal tolerance for leatherbacks (McMahon and Hays 2006). Leatherbacks are speculated to be the best able to cope with climate change of all the sea turtle species due to their wide geographic distribution and relatively weak beach fidelity. Leatherback sea turtles may be most affected by any changes in the distribution of their primary jellyfish prey, which may affect leatherback distribution and foraging behavior (NMFS and USFWS 2007b). Jellyfish populations may increase due to ocean warming and other factors (Brodeur *et al*. 1999; Attrill *et al*. 2007; Richardson *et al*. 2009). However, any increase in jellyfish populations may or may not impact leatherbacks as there is no evidence that any leatherback populations are currently food-limited.

As discussed for loggerheads, increasing temperatures are expected to result in rising sea levels (Titus and Narayanan 1995 in Conant *et al*. 2009), which could result in increased erosion rates along nesting beaches. Sea level rise could result in the inundation of nesting sites and decrease available nesting habitat (Fish *et al*. 2005). This effect would potentially be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents. While there is a reasonable degree of certainty that climate change related effects will be experienced globally (*e.g.*, rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects of climate change on this species are not predictable or quantifiable at this time (Hawkes *et al*. 2009). Based on the most recent five-year status review (NMFS and USFWS 2007b), and following from the climate change discussion in the previous section on NWA DPS loggerheads, it is unlikely that impacts from climate change will have a significant effect on the status of leatherbacks over the scope of the action assessed in this Opinion, which is the next ten years. However, significant impacts from climate change in the future beyond ten years are to be expected, but the severity of and rate at which these impacts will occur is currently unknown.

Summary of Status for Leatherback Sea Turtles

In the Pacific Ocean, the abundance of leatherback sea turtles on nesting beaches has declined dramatically during the past 10 to 20 years. Nesting groups throughout the eastern and western Pacific Ocean have been reduced to a fraction of their former abundance due to human activities that have reduced the number of nesting females and reduced the reproductive success of females (for example, by 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 in this region (NMFS and USFWS 2007b). The species as a whole continues to face numerous threats in nesting and marine habitats. As with the other sea turtle species, mortality due to fisheries interactions accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like pollution and habitat destruction account for an unknown level of other anthropogenic mortality. The long term recovery potential of this species may be further threatened by observed low genetic diversity, even in the largest nesting groups (NMFS and USFWS 2007b).

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

3.1.3 Kemp's ridley sea turtle

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

Kemp's ridleys mature at 10-17 years (Caillouet *et al.* 1995; Schmid and Witzell 1997; Snover *et al.* 2007; NMFS and USFWS 2007c). Nesting occurs from April through July each year, with hatchlings emerging after 45-58 days (NMFS *et al*. 2011). Females lay an average of 2.5 clutches within a season (TEWG 1998, 2000) and the mean remigration interval for adult females is two years (Márquez *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 with resource quality and quantity (TEWG 2000). Developmental habitats are defined by several characteristics, including sheltered coastal areas such as embayments and estuaries, and nearshore temperate waters shallower than 50 meters (NMFS and USFWS 2007c). The suitability of these habitats depends on resource availability, with optimal environments providing rich sources of crabs and other invertebrates. Kemp's ridleys consume a variety of crab species, including *Callinectes*, *Ovalipes*, *Libinia*, and *Cancer* species. Mollusks, shrimp, and fish are consumed less frequently (Bjorndal 1997). A wide variety of substrates have been documented to provide good foraging habitat, including seagrass beds, oyster reefs, sandy and mud bottoms, and rock outcroppings (NMFS and USFWS 2007c).

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

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

Population Dynamics and Status

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

Threats

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

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

Although modifications to shrimp trawls have helped to reduce mortality of Kemp's ridleys, a recent assessment found that the Southeast/Gulf of Mexico shrimp trawl fishery remained responsible for the vast majority of U.S. fishery interactions (up to 98%) and mortalities (more than 80%). Finkbeiner *et al*. (2011) compiled cumulative sea turtle bycatch information in U.S. fisheries from 1990 through 2007, before and after implementation of bycatch mitigation measures. Information was obtained from peer reviewed publications and NMFS documents (*e.g.*, 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. The most recent section 7 consultation on the shrimp fishery, completed in May 2012, was unable to estimate the total annual level of take for Kemp's ridleys at present. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in at least tens of thousands and possibly hundreds of thousands of interactions annually, of which at least thousands and possibly tens of thousands are expected to be lethal (NMFS 2012a).

This species is also affected by other sources of anthropogenic impact (fishery and non-fishery related), similar to those discussed above. Three Kemp's ridley captures in Mid-Atlantic trawl fisheries were documented by NMFS observers between 1994 and 2008 (Warden and Bisack 2010), and eight Kemp's ridleys were documented by NMFS observers in mid-Atlantic sink gillnet fisheries between 1995 and 2006 (Murray 2009a). Additionally, in the spring of 2000, 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 NEFSC 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 2006b).

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

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

Another potential impact from global climate change is sea level rise, which may result in increased beach erosion at nesting sites. Beach erosion may be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents. In the case of the Kemp's ridley where most of the critical nesting beaches are undeveloped, beaches may shift landward and still be available for nesting. The Padre Island National Seashore (PAIS) shoreline is accreting, unlike much of the Texas coast, and with nesting increasing and 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). Based on the most recent five-year status review (NMFS and USFWS 2007c), and following from the climate change discussions on loggerheads and leatherbacks, it is unlikely that impacts from climate change will have a significant effect on the status of Kemp's ridleys over the scope of the action assessed in this Opinion, which is the next ten years. However, significant impacts from climate change in the future beyond ten years are to be expected, but the severity of and rate at which these impacts will occur is currently unknown.

Summary of Status for Kemp's Ridley Sea Turtles

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

As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like dredging, pollution, and habitat destruction also contribute to annual human caused mortality, but the levels are unknown. Based on their five-year status review of the species, NMFS and USFWS (2007c) determined that Kemp's ridley sea turtles should not be reclassified as threatened under the ESA. A revised bi-national recovery plan was published for public comment in 2010, and in September 2011, NMFS, USFWS, and the Services and the Secretary of Environment and Natural Resources, Mexico (SEMARNAT) released the second revision to the Kemp's ridley recovery plan. Based on the information presented above, for purposes of this Opinion, we consider that the status of Kemp's ridleys over the next ten years will be no worse than it is currently and that the species may actually be in the early stages of recovery, although this should be viewed in the context of a much larger population in the mid-20th century.

3.1.4 Green sea turtle

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

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

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

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 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 six 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 Syrian coast was surveyed in 1991, but nesting activity was attributed to loggerheads) bodes well for the speculation that the unsurveyed coast of Libya may also host substantial nesting.

Atlantic Ocean

<u>Distribution and Life History</u>

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

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

Some of the principal feeding areas in the western Atlantic Ocean include the upper west coast of Florida, the Florida Keys, and the northwestern coast of the Yucatán Peninsula. Additional important foraging areas in the western Atlantic include the Mosquito and Indian River Lagoon systems and nearshore wormrock reefs between Sebastian and Ft. Pierce Inlets in Florida, Florida Bay, the Culebra archipelago and other Puerto Rico coastal waters, the south coast of

Cuba, the Mosquito Coast of Nicaragua, the Caribbean coast of Panama, and scattered areas along Colombia and Brazil (Hirth 1971). The waters surrounding the island of Culebra, Puerto Rico, and its outlying keys are designated critical habitat for the green sea turtle.

Age at maturity for green sea turtles is estimated to be 20-50 years (Balazs 1982; Frazer and Ehrhart 1985; Seminoff 2004). Adult females may nest multiple times in a season (average three nests/season with approximately 100 eggs/nest) and typically do not nest in successive years (NMFS and USFWS 1991; Hirth 1997).

Population Dynamics and Status

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

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

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

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

The statewide Florida surveys (2000-2006) 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.

<u>Threats</u>
Green sea turtles face many of the same natural threats as loggerhead and Kemp's ridley sea turtles. In addition, green sea turtles appear to be particularly susceptible to fibropapillomatosis, an epizootic disease producing lobe-shaped tumors on the soft portion of a turtle's body. Juveniles appear to have the highest incidence of disease and the most extensive lesions, whereas lesions in nesting adults are rare. Also, green sea turtles frequenting nearshore waters, areas adjacent to large human populations, and 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.*, Opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, occurred annually (since implementation of bycatch mitigation measures). Kemp's ridleys interacted with fisheries most frequently, with the highest level of mean annual mortality (2,700), followed by loggerheads (1,400), greens (300), and leatherbacks (40). The Southeast/Gulf of Mexico shrimp trawl fishery was responsible for the vast majority of U.S. interactions (up to 98%) and mortalities (more than 80%). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations. The most recent section 7 consultation on the shrimp fishery, completed in May 2012, was unable to estimate the total annual level of take for green sea turtles at present. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in at least hundreds and possibly low thousands of interactions annually, of which hundreds are expected to be lethal (NMFS 2012a).

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

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

As noted above, the increasing female bias in green sea turtle hatchlings is thought to be at least partially linked to increases in temperatures at nesting beaches. However, due to a lack of scientific data, the specific future effects of climate change on green sea turtles species are not predictable or quantifiable to any degree at this time (Hawkes *et al*. 2009). For example, information is not available to predict the extent and rate to which sand temperatures at the nesting beaches used by green sea turtles may increase in the short-term future and the extent to which green sea turtles may be able to cope with this change by selecting cooler areas of the beach or shifting their nesting distribution to other beaches at which increases in sand temperature may not be experienced. Based on the most recent five-year status review (NMFS and USFWS 2007d), and following from the climate change discussions on the other three species, it is unlikely that impacts from climate change will have a significant effect on the status of green sea turtles over the scope of the action assessed in this Opinion, which is the next ten years. However, significant impacts from climate change in the future beyond ten years are to be expected, but the severity of and rate at which these impacts will occur is currently unknown.

Summary of Status of Green Sea Turtles

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

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

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

As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like hopper dredging, pollution, and habitat destruction also contribute to human caused mortality, though the level is unknown. Based on its five-year status review of the species, NMFS and USFWS (2007d) determined that the listing classification for green sea turtles should not be changed. However, it was also determined that an analysis and review of the species should be conducted to determine whether DPSs should be identified (NMFS and USFWS 2007d). Based on the information presented above, for purposes of this Opinion, we consider that the status of green sea turtles over the next ten years will be no worse than it is currently and that the status of the species in the Atlantic Ocean may actually be stable or improving due to increased nesting.

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

3.2 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 likely to occur in the action area.

The Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) is a subspecies of sturgeon distributed along the eastern coast of North America from Hamilton Inlet, Labrador, Canada to Cape Canaveral, Florida, USA (Scott and Scott 1988; ASSRT 2007; T. Savoy, CT DEP, pers. comm.). We have delineated U.S. populations of Atlantic sturgeon into five DPSs[6] (77 FR 5880 and 77 FR 5914). They are: the Gulf of Maine (GOM), New York Bight (NYB), Chesapeake Bay (CB), Carolina, and South Atlantic (SA) DPSs (Figure 2). 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 Atlantic sturgeon from each DPS and Canada occur throughout the full range of the subspecies. Therefore, Atlantic sturgeon originating from any of the five DPSs can be affected by threats in the marine, estuarine, and riverine environment that occur far from natal spawning rivers.

On February 6, 2012, we published a notice in the *Federal Register* that we were listing the NYB, CB, Carolina, and SA DPSs as "endangered," and the GOM DPS as "threatened" (77 FR 5880 and 77 FR 5914). The effective date of the listings is April 6, 2012. The DPSs do not include Atlantic sturgeon that are spawned in Canadian rivers. Therefore, Canadian spawned fish are not included in the listings. As described below, individuals originating from all five of the 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.

Atlantic sturgeon life history

Atlantic sturgeon are long lived (approximately 60 years), late maturing, estuarine dependent, anadromous[7] fish (Bigelow and Schroeder 1953; Vladykov and Greeley 1963; Mangin 1964; Pikitch *et al.* 2005; Dadswell 2006; ASSRT 2007). They are a relatively large fish, even amongst sturgeon species (Pikitch *et al.* 2005). Atlantic sturgeon are bottom feeders that suck food into a ventrally-located protruding mouth (Bigelow and Schroeder 1953). Four barbels in front of the mouth assist the sturgeon in locating prey (Bigelow and Schroeder 1953). Diets of adult and migrant subadult Atlantic sturgeon include mollusks, gastropods, amphipods, annelids, decapods, isopods, and fish such as sand lance (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007; Savoy 2007). Juvenile Atlantic sturgeon feed on aquatic insects, insect larvae, and other invertebrates (Bigelow and Schroeder 1953; ASSRT 2007; Guilbard *et al.* 2007).

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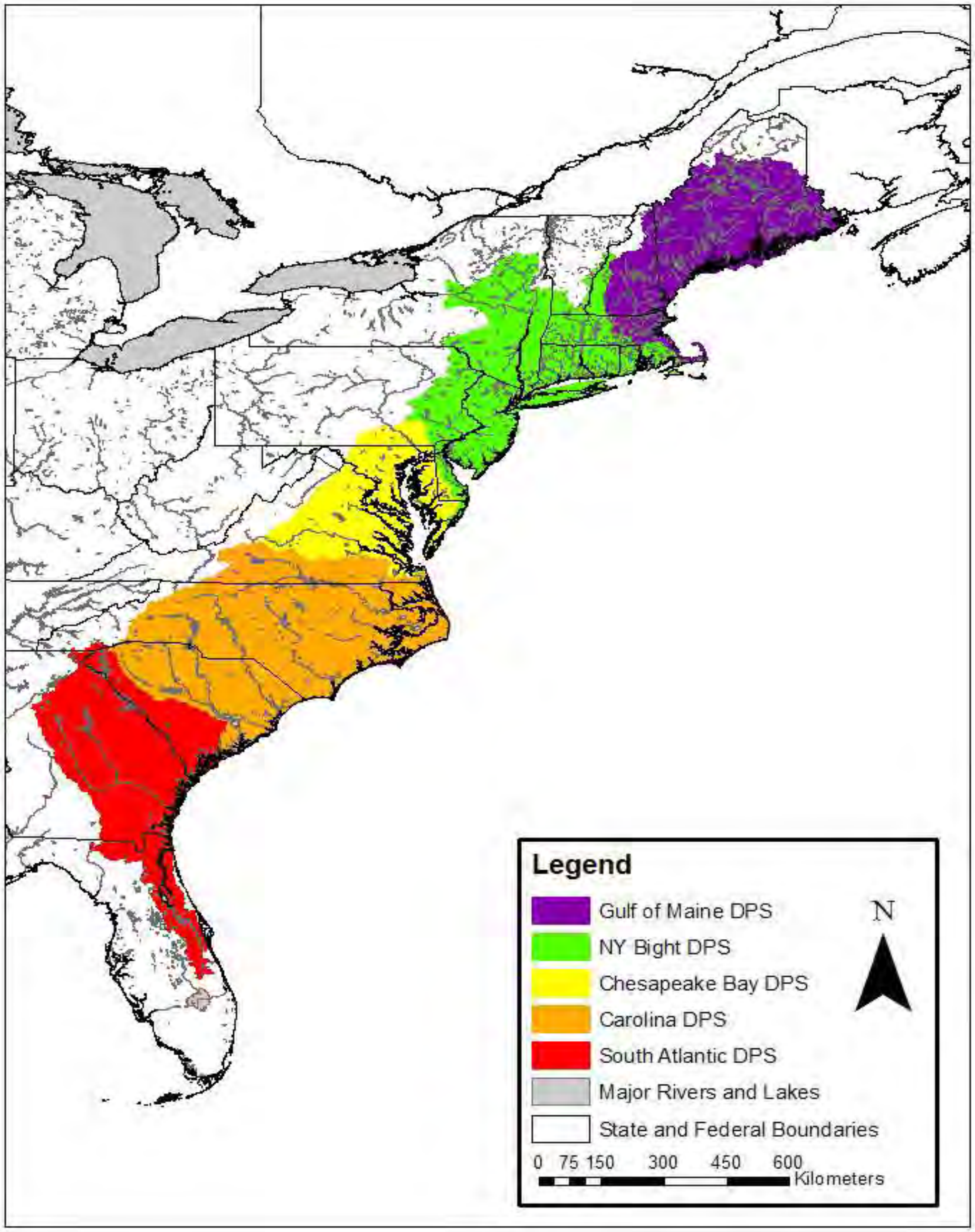
[6] To be considered for listing under the ESA, a group of organisms must constitute a "species." A "species" is defined in section 3 of the ESA to include "any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds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

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

Figure 2. Map depicting the boundaries of the five Atlantic sturgeon DPSs



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

Water temperature plays a primary role in triggering the timing of spawning migrations (Greene *et al*. 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; Greene *et al*. 2009), and remain on the spawning grounds throughout the spawning season (Bain 1997). Females begin spawning migrations when temperatures are closer to 12° to 13°C (54° to 55°F) (Dovel and Berggren 1983; Smith 1985; Collins *et al.* 2000), make rapid spawning migrations upstream, and quickly depart following spawning (Bain 1997).

The spawning areas in most U.S. rivers have not been well defined. However, the habitat characteristics of spawning areas have been identified based on historical accounts of where fisheries occurred, tracking and tagging studies of spawning sturgeon, and physiological needs of early life stages. Spawning is believed to occur in flowing water between the salt front of estuaries and the fall line of large rivers, when and where optimal flows are 46-76 centimeters/second and depths are 3-27 meters (Borodin 1925; Dees 1961; Leland 1968; Scott and Crossman 1973; Crance 1987; Shirey *et al.* 1999; Bain *et al.* 2000; Collins *et al.* 2000; Caron *et al.* 2002; Hatin *et al.* 2002; Greene *et al*. 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; Greene *et al*. 2009), and become adhesive shortly after fertilization (Murawski and Pacheco 1977; Van den Avyle 1984; Mohler 2003). Incubation time for the eggs increases as water temperature decreases (Mohler 2003). At temperatures of 20° and 18°C, hatching occurs approximately 94 and 140 hours, respectively, after egg deposition (ASSRT 2007).

Larval Atlantic sturgeon (*i.e.*, less than four weeks old, with total lengths (TL) less than 30 millimeters; Van Eenennaam *et al.* 1996) are assumed to 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; Greene *et al*. 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 meters in depth, using coastal bays, sounds, and ocean waters (Vladykov and Greeley 1963; Murawski and Pacheco 1977; Dovel and Berggren 1983; Smith 1985; Collins and Smith 1997; Welsh *et al.* 2002; Savoy and Pacileo 2003; Stein *et al.* 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 meters during winter and spring, and in the northern portion of the Mid-Atlantic Bight at depths less than 20 meters in summer and fall (Erickson *et al.* 2011). Shirey (Delaware Department of Fish and Wildlife, unpublished data reviewed in Greene *et al*. 2009) found a similar movement pattern for juvenile Atlantic sturgeon based on recaptures of fish originally tagged in the Delaware River. After leaving the Delaware River estuary during the fall, juvenile Atlantic sturgeon were recaptured by commercial fishermen in nearshore waters along the Atlantic coast as far south as Cape Hatteras, North Carolina from November through early March. In the spring, a portion of the tagged fish re-entered the Delaware River estuary. However, many fish continued a northerly coastal migration through the Mid-Atlantic as well as into southern New England waters where they were recovered throughout the summer months. Movements as far north as Maine were documented. A southerly coastal migration was apparent from tag returns reported in the fall. The majority of these tag returns were reported from relatively shallow nearshore fisheries with few fish reported from waters in excess of 25 meters (C. Shirey, Delaware Department of Fish and Wildlife, unpublished data reviewed in Greene *et al*. 2009). Areas where migratory Atlantic sturgeon commonly aggregate include the Bay of Fundy (*e.g.*, Minas and Cumberland Basins), Massachusetts Bay, Connecticut River estuary, Long Island Sound, New York Bight, Delaware Bay, Chesapeake Bay, and waters off of North Carolina from the Virginia/North Carolina border to Cape Hatteras at depths up to 24 meters (Dovel and Berggren 1983; Dadswell *et al.* 1984; Johnson *et al.* 1997; Rochard *et al.* 1997; Kynard *et al.* 2000; Eyler *et al.* 2004; Stein *et al.* 2004;

Wehrell 2005; Dadswell 2006; ASSRT 2007; Laney *et al.* 2007). These sites may be used as foraging sites and/or thermal refuge.

Determination of DPS Composition in the Action Area

As explained above, the range of all five DPSs overlaps and extends from Canada through Cape Canaveral, Florida. We have considered the best available information to determine from which DPSs individuals in the action area are likely to have originated. We have determined that Atlantic sturgeon in the action area likely originate from the five DPSs at the following frequencies: NYB 46%; SA 29%; CB 16%; GOM 8%; and Carolina 0.5%. These percentages are largely based on genetic sampling of individuals (n=89) sampled in commercial fisheries by NEFOP. This covers captures from the Gulf of Maine to Cape Hatteras and is generally aligned with the action area for this consultation. Therefore, this represents the best available information on the likely genetic makeup of individuals occurring in the action area. Carolina DPS origin fish have rarely been detected in samples taken in the Northeast; however, mixed stock analysis from some sampling efforts (*e.g.*, Long Island Sound, n=275), indicates that approximately 0.5% of the fish sampled were Carolina DPS origin. Because any Carolina origin Atlantic sturgeon that were sampled in Long Island Sound would have swam through the action area, it is reasonable to expect that 0.5% of the Atlantic sturgeon captured in the action area could originate from the Carolina DPS. The genetic assignments have a plus/minus 5% confidence interval; however, for purposes of section 7 consultation we have selected the reported values above, which approximate the mid-point of the range, as a reasonable indication of the likely genetic makeup of Atlantic sturgeon in the action area. These assignments and the data from which they are derived are described in detail in Damon-Randall *et al*. (2012a).

Distribution and Abundance

Atlantic sturgeon underwent significant range-wide declines from historical abundance levels due to overfishing in the mid to late 19th century when a caviar market was established (Scott and Crossman 1973; Taub 1990; Kennebec River Resource Management Plan 1993; Smith and Clugston 1997; Dadswell 2006; ASSRT 2007). Abundance of spawning-aged females prior to this period of exploitation was predicted to be greater than 100,000 for the Delaware, and at least 10,000 females for other spawning stocks (Secor and Waldman 1999; Secor 2002). Historical records suggest that Atlantic sturgeon spawned in at least 35 rivers prior to this period. Currently, only 16 U.S. rivers are known to support spawning based on available evidence (*i.e.*, presence of young-of-year or gravid Atlantic sturgeon documented within the past 15 years) (ASSRT 2007). While there may be other rivers supporting spawning for which definitive evidence has not been obtained (*e.g.*, in the Penobscot and York Rivers), the number of rivers supporting spawning of Atlantic sturgeon are approximately half of what they were historically. In addition, only four rivers (Kennebec, Hudson, Delaware, and 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, Georgia, 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 *et al.* (2012b) and is summarized briefly below. We used this method because for those four DPSs, there are: (1) no total population estimates available; (2) with the exception of the Hudson River, no estimates of the number of mature adults; and, (3) no information from directed population surveys which could be used to generate an estimate of the number of spawning adults, total adult population or total DPS population.

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

We were able to use this estimate of the adult population in the Hudson River and the rate at which Atlantic sturgeon from the Hudson River are intercepted in certain Northeast commercial fisheries[8] 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 *et al.* (2012b).

[8] Bycatch information was obtained from a report prepared by NEFSC (2011a).

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 2011a). This report indicates that based on observed interactions with Atlantic sturgeon in sink gillnet and otter trawl fisheries from 2006-2010, on average 3,118 Atlantic sturgeon are captured in these fisheries each year. Information in the NEFOP database indicates that 25% of captured Atlantic sturgeon are adults (determined as length greater than 150 centimeters) and 75% are subadults (determined as length less than 150 centimeters). By applying the mixed stock genetic analysis of individuals[9] sampled by the NEFOP and ASM 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 four of the DPSs (GOM, NYB, CB, and SA) as follows in Table 2. 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 Rivers are being captured in Northeast fisheries considered in the NEFSC (2011a) bycatch report. Spawning is known to occur in other rivers in the SA DPS, including the Altamaha (estimate of 343 adult spawners per year).

Table 2: Summary of Calculated Population Estimates for the five Atlantic Sturgeon DPSs

DPS	**Estimated Mature 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
Carolina*	496	1,488
SA*	598	1,794

*see note regarding the Carolina and SA DPS population sizes in paragraphs below

[9] 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).

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 observer dataset and based on this, as well as genetics information on fish captured in other coastal sampling programs in the Northeast[10] are assumed to be rarely intercepted in Northeast fisheries.

Currently, there are an estimated 343 spawning adults in the Altamaha River 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 SA 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 indicates that only individuals from the Savannah and Ogeechee Rivers are being captured in Northeast fisheries considered in the NEFSC (2011a) 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.

While we are unable to calculate a population estimate for the Carolina DPS using the above methodology, we do have an estimate of 1,500 adult spawners/year (5 spawning rivers x 300 spawning adults per river) described in the Atlantic sturgeon status review report. As noted above, for the SA DPS, using this method, the estimated number of fish in the SA DPS would be 1,800 spawning adults (6 spawning rivers x 300 spawning adults per river). Therefore, the Carolina DPS has approximately 17% less fish than the SA DPS. Based on the methodology described above, the estimated number of mean annual mature adults for the SA DPS is 598 fish. Using the proportion of Carolina DPS fish to SA DPS fish, we estimate that the mean number of annual mature adults in the Carolina DPS is 496 (17% less than 598).

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 (Table 2). 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.

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

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[10] 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.

declines from historical abundance levels due to overfishing (for caviar and meat) and impacts to habitat in the 19th and 20th centuries (Taub 1990; Smith and Clugston 1997; Secor and Waldman 1999).

Based on the best available information, we have 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, Florida, 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 EEZ in the course of a commercial fishing activity.

Commercial fisheries for Atlantic sturgeon still exist in Canadian waters (DFO 2011). Sturgeon belonging to one or more of the DPSs may be harvested in the Canadian fisheries. In particular, the Bay of Fundy fishery in the Saint John estuary may capture sturgeon of U.S. origin given that sturgeon from the GOM and the NYB DPSs have been incidentally captured in other Bay of Fundy fisheries (DFO 2011; 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 GOM DPS, with a smaller percentage from the NYB DPS.

Fisheries bycatch in U.S. waters is one of the primary threats faced by all five DPSs (ASSRT 2007; 77 FR 5880 and 77 FR 5914). 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 (NEFSC 2011a) in the Northeast Region, but we do not have a similar estimate for Southeast fisheries. We also do not have an estimate of the number of Atlantic sturgeon captured or killed in state fisheries. At this time, we are not able to quantify the effects of other significant threats (*e.g.*, vessel strikes, poor water quality, water availability, dams, and dredging) in terms of habitat impacts or loss of individuals. While we have some information on the number of mortalities that have occurred in the past in association with certain activities (*e.g.*, mortalities in the Delaware and James Rivers that are thought to be due to vessel strikes), we are not able to use those numbers to extrapolate effects throughout one or more DPS. This is because of the small number of data points and 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 2011a). The analysis prepared by the NEFSC estimates that from 2006 through 2010 there were 2,250 to 3,862 encounters per year in observed gillnet and trawl fisheries, with an average of 3,118 encounters. Mortality rates in gillnet gear are approximately 20%. Mortality rates in otter trawl gear are believed to be lower at approximately 5%. Comparing the estimated annual average mortalities to the adult population estimates for each of the five 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.

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

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

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

Implications of climate change to the Atlantic sturgeon DPSs can been speculated, yet no scientific data are available on past trends related to climate effects on these species, and current scientific methods are not able to reliably predict the future magnitude of climate change and associated impacts or the adaptive capacity of these species. 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), any prediction of effects is made more difficult by a lack of information on the rate of expected change in conditions and a lack of information on the adaptive capacity of the species (*i.e.*, its ability to evolve to cope with a changing environment). Further analysis on potential effects of climate change on Atlantic sturgeon in the action area is included in section 6.0.

3.2.1 Gulf of Maine (GOM) DPS

The GOM DPS includes the following: all anadromous Atlantic sturgeon 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, Massachusetts. Within this range, Atlantic sturgeon historically spawned in the Androscoggin, Kennebec, Merrimack, Penobscot, and Sheepscot Rivers (ASSRT 2007). Spawning still occurs in the Kennebec and Androscoggin Rivers, and it is possible that it still occurs in the Penobscot River as well. Spawning in the Androscoggin River may also be occurring. Maine Department of Marine Resources reported the capture of a larval Atlantic sturgeon during the 2011 spawning season below the Brunswick Dam; this suggests that spawning may be occurring in this area. There is no evidence of recent spawning in the remaining rivers. In the 1800s, construction of the Essex Dam on the Merrimack River at river kilometer (rkm) 49 blocked access to 58% of Atlantic sturgeon habitat in the river (Oakley 2003; ASSRT 2007). However, the accessible portions of the Merrimack seem to be suitable habitat for Atlantic sturgeon spawning and rearing (*i.e.*, nursery habitat) (Kieffer and Kynard 1993). Therefore, the availability of spawning habitat does not appear to be the reason for the lack of observed spawning in the Merrimack River. Studies are on-going to determine whether Atlantic sturgeon are spawning in these rivers. Atlantic sturgeon that are spawned elsewhere continue to use habitats within all of these rivers as part of their overall marine range (ASSRT 2007). The movement of subadult and adult sturgeon between rivers, including to and from the Kennebec River and the Penobscot River, demonstrates that coastal and marine migrations are key elements of Atlantic sturgeon life history for the GOM DPS as well as likely throughout the entire range (ASSRT 2007; Fernandes *et al.* 2010).

Several threats play a role in shaping the current status of GOM DPS Atlantic sturgeon. Historical records provide evidence of commercial fisheries for Atlantic sturgeon in the Kennebec and Androscoggin Rivers dating back to the 17th century (Squiers and Smith 1979). In 1849, 160 tons of sturgeon was caught in the Kennebec River by local fishermen (Squiers and Smith 1979). Following the 1880s, the sturgeon fishery was almost non-existent due to a collapse of the sturgeon stocks. All directed Atlantic sturgeon fishing as well as retention of Atlantic sturgeon bycatch has been prohibited since 1998. Nevertheless, mortalities associated with bycatch in fisheries occurring in state and Federal waters still occurs. In the marine range, GOM DPS Atlantic sturgeon are incidentally captured in Federal and state managed fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (Stein *et al.* 2004; ASMFC TC 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 due to commercial fisheries bycatch are likely two of the primary concerns facing this DPS (ASSRT 2007; 77 FR 5880).

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

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

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

There are no empirical abundance estimates for the GOM DPS. The Atlantic Sturgeon SRT (2007) presumed that the GOM DPS was comprised of less than 300 spawning adults per year, based on abundance estimates for the Hudson and Altamaha River riverine populations of Atlantic sturgeon. Surveys of the Kennebec River over two time periods, 1977-1981 and 1998-2000, resulted in the capture of nine adult Atlantic sturgeon (Squiers 2004). However, since the surveys were primarily directed at capture of shortnose sturgeon, the capture gear used may not have been selective for the larger-sized, adult Atlantic sturgeon; several hundred subadult Atlantic sturgeon were caught in the Kennebec River during these studies. As explained above, we have estimated that there is an annual mean of 166 mature adult Atlantic sturgeon in the GOM DPS.

Summary of the Gulf of Maine DPS

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

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

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

3.2.2 New York Bight (NYB) DPS

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

The abundance of the Hudson River Atlantic sturgeon riverine population prior to the onset of expanded exploitation in the 1800s is unknown but, has been conservatively estimated at 10,000 adult females (Secor 2002). Current abundance is likely at least one order of magnitude smaller than historical levels (Secor 2002; ASSRT 2007; Kahnle *et al.* 2007). As described above, an estimate of the mean annual number of mature adults (863 total; 596 males and 267 females) was calculated for the Hudson River riverine population based on fishery-dependent data collected from 1985-1995 (Kahnle *et al.* 2007). Kahnle *et al.* (1998, 2007) also showed that the level of fishing mortality from the Hudson River Atlantic sturgeon fishery during the period of 1985-1995 exceeded the estimated sustainable level of fishing mortality for the riverine population and may have led to reduced recruitment. All available data on abundance of juvenile Atlantic sturgeon in the Hudson River Estuary indicate a substantial drop in production of young since the mid-1970s (Kahnle *et al.* 1998). A decline appeared to occur in the mid to late 1970s followed by a secondary drop in the late 1980s (Kahnle *et al.* 1998; Sweka *et al.* 2007; ASMFC 2010). Catch-per-unit-effort (CPUE) data suggests that recruitment has remained depressed relative to catches of juvenile Atlantic sturgeon in the estuary during the mid-late 1980s (Sweka *et al.* 2007; ASMFC 2010). 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 1800s indicate that this was historically a large population with an estimated 180,000 adult females prior to 1890 (Secor and Waldman 1999; Secor 2002). Sampling in 2009 to target young-of-the-year (YOY) Atlantic sturgeon in the Delaware River (*i.e.*, natal sturgeon) resulted in the capture of 34 YOY, ranging in size from 178 to 349 millimeters TL (Fisher 2009) and the collection of 32 YOY Atlantic sturgeon in a separate study (Brundage and O'Herron in Calvo *et al.* 2010). Genetics information collected from 33 of the 2009 year class YOY indicates that at least three females successfully contributed to the 2009 year class (Fisher 2011). Therefore, while the capture of YOY in 2009 provides evidence that successful spawning is still occurring in the Delaware River, the relatively low numbers suggest the existing riverine population is 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 NYB DPS. Similar to the Hudson River, there is currently not enough information to determine a trend for the Delaware River population.

Summary of the New York Bight DPS

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

In the marine range, NYB DPS Atlantic sturgeon are incidentally captured in Federal and state managed fisheries, reducing survivorship of subadult and adult Atlantic sturgeon (Stein *et al.* 2004; ASMFC TC 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% of the Atlantic sturgeon bycatch interactions in the Mid Atlantic Bight region were sturgeon from the NYB DPS. Individual-based assignment and mixed stock analysis of samples collected from sturgeon captured in Canadian fisheries in the Bay of Fundy indicated that approximately 1%-2% were from the NYB DPS. At this time, we are not able to quantify the impacts from other threats or estimate the number of individuals killed as a result of other anthropogenic threats.

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

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

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

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

Studies have shown that to rebuild, Atlantic sturgeon can only sustain low levels of anthropogenic mortality (Boreman 1997; ASMFC TC 2007; Kahnle *et al.* 2007; Brown and Murphy 2010). There are no empirical abundance estimates of the number of Atlantic sturgeon in the NYB DPS. As explained above, we have estimated that there are an annual mean total of 950 mature adult Atlantic sturgeon in the NYB DPS. NMFS has determined that the NYB DPS is currently at risk of extinction due to: (1) precipitous declines in population sizes and the protracted period in which sturgeon populations have been depressed; (2) the limited amount of current spawning; and (3) the impacts and threats that have and will continue to affect population recovery.

3.2.3 Chesapeake Bay (CB) DPS

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

Several threats play a role in shaping the current status of CB DPS Atlantic sturgeon. Historical records provide evidence of the large-scale commercial exploitation of Atlantic sturgeon from the James River and Chesapeake Bay in the 19th century (Hildebrand and Schroeder 1928; Vladykov and Greeley 1963; ASMFC 1998; Secor 2002; Bushnoe *et al.* 2005; ASSRT 2007) as well as subsistence fishing and attempts at commercial fisheries as early as the 17th century (Secor 2002; Bushnoe *et al.* 2005; ASSRT 2007; Balazik *et al.* 2010). Habitat disturbance caused by in-river work such as dredging for navigational purposes is thought to have reduced available spawning habitat in the James River (Holton and Walsh 1995; Bushnoe *et al.* 2005;

ASSRT 2007). At this time, we do not have information to quantify this loss of spawning habitat.

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

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

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

Summary of the Chesapeake Bay DPS

Spawning for the CB DPS is known to occur in only the James River. Spawning may be occurring in other rivers, such as the York, but has not been confirmed. There are anecdotal reports of increased sightings and captures of Atlantic sturgeon in the James River. However, this information has not been comprehensive enough to develop a population estimate for the James River or to provide sufficient evidence to confirm increased abundance. Some of the impact from the threats that facilitated the decline of the CB DPS have been removed (*e.g.*, directed fishing) or reduced as a result of improvements in water quality since passage of the CWA. As explained above, we have estimated that there is an annual mean of 329 mature adult Atlantic sturgeon in the CB DPS. We do not currently have enough information about any life stage to establish a trend for this DPS. Areas with persistent, degraded water quality, habitat impacts from dredging, continued bycatch in U.S. state and federally-managed fisheries, Canadian fisheries, and vessel strikes remain significant threats to the CB DPS of Atlantic sturgeon. Studies have shown that Atlantic sturgeon can only sustain low levels of bycatch mortality (Boreman 1997; ASMFC TC 2007; Kahnle *et al.* 2007). The CB DPS is currently at risk of extinction given (1) precipitous declines in population sizes and the protracted period in which sturgeon populations have been depressed; (2) the limited amount of current spawning; and (3) the impacts and threats that have and will continue to affect the potential for population recovery.

3.2.4 Carolina DPS

Distribution and Abundance
The Carolina DPS includes all Atlantic sturgeon that spawn or are spawned in the watersheds (including all rivers and tributaries) from Albemarle Sound southward along the southern Virginia, North Carolina, and South Carolina coastal areas to Charleston Harbor. The marine range of Atlantic sturgeon from the Carolina DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida. The riverine range of the Carolina DPS and the adjacent portion of the marine range are shown in Figure 2. Sturgeon are commonly captured 40 miles offshore (D. Fox, DSU, pers. comm.). Records providing fishery bycatch data by depth show the vast majority of Atlantic sturgeon bycatch via gillnets is observed in waters less than 50 meters deep (Stein *et al*. 2004; ASMFC TC 2007), but Atlantic sturgeon are recorded as bycatch out to 500 fathoms.

Rivers known to have current spawning populations within the range of the Carolina DPS include the Roanoke, Tar-Pamlico, Cape Fear, Waccamaw, and Pee Dee Rivers. We determined spawning was occurring if 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

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

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

be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development. There may also be spawning populations in the Neuse, Santee, and Cooper Rivers, though it is uncertain. Historically, both the Sampit and Ashley Rivers were documented to have spawning populations at one time. However, the spawning population in the Sampit River is believed to be extirpated and the current status of the spawning population in the Ashley River is unknown. Both rivers may be used as nursery habitat by young Atlantic sturgeon originating from other spawning populations. This represents our current knowledge of the river systems utilized by the Carolina DPS for specific life functions, such as spawning, nursery habitat, and foraging. However, fish from the Carolina DPS likely use other river systems than those listed here for their specific life functions.

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

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

Threats

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

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

Overutilization of Atlantic sturgeon from directed fishing caused initial severe declines in Atlantic sturgeon populations in the Southeast, from which they have never rebounded. Further, continued overutilization of Atlantic sturgeon as bycatch in commercial fisheries is an ongoing impact to the Carolina DPS. Atlantic sturgeon are more sensitive to bycatch mortality because they are a long-lived species, have an older age at maturity, have lower maximum fecundity values, and a large percentage of egg production occurs later in life. Based on these life history traits, Boreman (1997) calculated that Atlantic sturgeon can only withstand the annual loss of up to 5% of their population to bycatch mortality without suffering population declines. Mortality rates of Atlantic sturgeon taken as bycatch in various types of fishing gear range between 0% and 51%, with the greatest mortality occurring in sturgeon caught by sink gillnets. Atlantic sturgeon are particularly vulnerable to being caught in sink gillnets, therefore fisheries using this type of gear account for a high percentage of Atlantic sturgeon bycatch. Little data exists on bycatch in the Southeast and high levels of bycatch underreporting are suspected. Further, a total population abundance for the DPS is not available, and it is therefore not possible to calculate the percentage of the DPS subject to bycatch mortality based on the available bycatch mortality rates for individual fisheries. However, fisheries known to incidentally catch Atlantic sturgeon occur throughout the marine range of the species and in some riverine waters as well. Because Atlantic sturgeon mix extensively in marine waters and may access multiple river systems, they are subject to being caught in multiple fisheries throughout their range. In addition, stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (*e.g.*, exposure to toxins and low DO). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even post-capture mortality.

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

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

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

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

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

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

3.2.5 South Atlantic (SA) DPS

Distribution and Abundance
The SA DPS includes all Atlantic sturgeon that spawn or are spawned in the watersheds (including all rivers and tributaries) of the Ashepoo, Combahee, and Edisto Rivers (ACE) Basin southward along the South Carolina, Georgia, and Florida coastal areas to the St. Johns River, Florida. The marine range of Atlantic sturgeon from the SA DPS extends from the Hamilton Inlet, Labrador, Canada, to Cape Canaveral, Florida. The riverine range of the SA DPS and the adjacent portion of the marine range are shown in Figure 2. Sturgeon are commonly captured 40 miles offshore (D. Fox, DSU, pers. comm.). Records providing fishery bycatch data by depth show the vast majority of Atlantic sturgeon bycatch via gillnets is observed in waters less than 50 meters deep (Stein *et al*. 2004, ASMFC TC 2007), but Atlantic sturgeon are recorded as bycatch out to 500 fathoms.

Rivers known to have current spawning populations within the range of the SA DPS include the Combahee, Edisto, Savannah, Ogeechee, Altamaha, and Satilla Rivers. We determined spawning was occurring if YOY were observed, or mature adults were present, in freshwater portions of a system (Table 4). However, in some rivers, spawning by Atlantic sturgeon may not be contributing to population growth because of lack of suitable habitat and the presence of other stressors on juvenile survival and development. Historically, both the Broad-Coosawatchie and St. Marys Rivers were documented to have spawning populations at one time; there is also evidence that spawning may have occurred in the St. Johns River or one of its tributaries. However, the spawning population in the St. Marys River, as well as any historical spawning population present in the St. Johns, is believed to be extirpated, and the status of the spawning population in the Broad-Coosawatchie is unknown. Both the St. Marys and St. Johns Rivers are used as nursery habitat by young Atlantic sturgeon originating from other spawning populations.

The use of the Broad-Coosawatchie by sturgeon from other spawning populations is unknown at this time. The presence of historical and current spawning populations in the Ashepoo River has not been documented; however, this river may currently be used for nursery habitat by young Atlantic sturgeon originating from other spawning populations. This represents our current knowledge of the river systems utilized by the SA DPS for specific life functions, such as spawning, nursery habitat, and foraging. However, fish from the SA DPS likely use other river systems than those listed here for their specific life functions.

The riverine spawning habitat of the SA DPS occurs within the South Atlantic Coastal Plain ecoregion (TNC 2002b), which includes fall-line sandhills, rolling longleaf pine uplands, wet pine flatwoods, isolated depression wetlands, small streams, large river systems, and estuaries. Other ecological systems in the ecoregion include maritime forests on barrier islands, pitcher

Table 4. Major rivers, tributaries, and sounds within the range of the SA 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	

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 SA DPS's spawning rivers, located in the South Atlantic Coastal Plain, are primarily of two types: brownwater (with headwaters north of the Fall Line, silt-laden) and blackwater (with headwaters in the coastal plain, stained by tannic acids).

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

Threats

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

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

Overutilization of Atlantic sturgeon from directed fishing caused initial severe declines in Atlantic sturgeon populations in the Southeast, from which they have never rebounded. Further, continued overutilization of Atlantic sturgeon as bycatch in commercial fisheries is an ongoing impact to the SA DPS. Atlantic sturgeon are more sensitive to bycatch mortality because they are a long-lived species, have an older age at maturity, have lower maximum fecundity values, and a large percentage of egg production occurs later in life. Based on these life history traits, Boreman (1997) calculated that Atlantic sturgeon can only withstand the annual loss of up to 5% of their population to bycatch mortality without suffering population declines. Mortality rates of Atlantic sturgeon taken as bycatch in various types of fishing gear range between 0% and 51%, with the greatest mortality occurring in sturgeon caught by sink gillnets (ASSRT 2007). Atlantic sturgeon are particularly vulnerable to being caught in sink gillnets, therefore fisheries using this type of gear account for a high percentage of Atlantic sturgeon bycatch. Little data exists on bycatch in the Southeast and high levels of bycatch underreporting are suspected. Further, a total population abundance for the DPS is not available, and it is therefore not possible to calculate the percentage of the DPS subject to bycatch mortality based on the available bycatch mortality rates for individual fisheries. However, fisheries known to incidentally catch Atlantic sturgeon occur throughout the marine range of the species and in some riverine waters as well. Because Atlantic sturgeon mix extensively in marine waters and may access multiple river systems, they are subject to being caught in multiple fisheries throughout their range. In addition, stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (*e.g.*, exposure to toxins and low DO). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even post-capture mortality.

As a wide-ranging anadromous species, Atlantic sturgeon are subject to numerous Federal (U.S. and Canadian), state and provincial, and inter-jurisdictional laws, regulations, and agency activities. While these mechanisms have addressed impacts to Atlantic sturgeon through directed fisheries, there are currently no mechanisms in place to address the significant risk posed to Atlantic sturgeon from commercial bycatch. Though statutory and regulatory mechanisms exist that authorize reducing the impact of dams on riverine and anadromous species, such as Atlantic sturgeon, and their habitat, these mechanisms have proven inadequate for preventing dams from blocking access to habitat upstream and degrading habitat downstream. Further, water quality continues to be a problem in the SA DPS, even with existing controls on some pollution sources. Current regulatory regimes are not necessarily effective in controlling water allocation issues (*e.g.*, no permit requirements for water withdrawals under 100,000 gallons per day 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 U.S. Atlantic Coast, especially in areas where habitat is limited and water quality is severely degraded, will require improvements in the following areas: (1) elimination of barriers to spawning habitat either through dam removal, breaching, or installation of successful fish passage facilities; (2) operation of water control structures to provide appropriate flows, especially during spawning season; (3) imposition of dredging restrictions including seasonal moratoriums and avoidance of spawning/nursery habitat; and, (4) mitigation of water quality parameters that are restricting sturgeon use of a river (*i.e.*, DO). Additional data regarding sturgeon use of riverine and estuarine environments is needed.

The concept of a viable population able to adapt to changing environmental conditions is critical to Atlantic sturgeon, and the low population size of every river population in the SA DPS puts them in danger of extinction throughout their range. Although the largest impact that caused the precipitous decline of the species has been curtailed (directed fishing), the population sizes within the SA DPS have remained relatively constant at greatly reduced levels (approximately 6% of historical population sizes in the Altamaha River, and 1% of historical population sizes in the remainder of the DPS) (ASSRT 2007). Small numbers of individuals resulting from drastic reductions in populations, such as occurred with Atlantic sturgeon due to the commercial fishery, can remove the buffer against natural demographic and environmental variability provided by large populations (Berry 1971; Shaffer 1981; Soulé 1980). Recovery of depleted populations is an inherently slow process for a late-maturing species such as Atlantic sturgeon, and they continue to face a variety of other threats that contribute to their risk of extinction. Their late age at maturity provides more opportunities for individual Atlantic sturgeon to be removed from the population before reproducing. While a long life-span also allows multiple opportunities to contribute to future generations, it also results increases the timeframe over which exposure to the multitude of threats facing the SA DPS can occur.

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

Summary of the Status of the SA DPS of Atlantic Sturgeon

The SA DPS is estimated to number fewer than 6% of its historical population size, with all river populations except the Altamaha estimated to be less than 1% 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 SA DPS by habitat alteration, bycatch, and from the inadequacy of existing regulatory mechanisms to address and reduce habitat alterations and bycatch.

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

4.0 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 ESA-listed species in the action area. The impacts of all these activities are reflected in the status and trends of sea turtles and Atlantic sturgeon DPSs contained in the *Status of the Species* section above.

4.1 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.

4.1.1 Authorization of Fisheries through Fishery Management Plans

NMFS authorizes the operation of several fisheries in the action area under the authority of the MSA and through FMPs 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 (Federal waters), Atlantic bluefish, Atlantic mackerel/squid/butterfish, Atlantic sea scallop, monkfish, northeast skate complex, 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 Opinions on the operations of these fisheries, which overlap at least in part with the action area for the scallop fishery (NMFS 2001, 2002b, 2008a, 2010a, 2010b, 2010c, 2010d, 2010e, 2010f, 2010g, 2010h). 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 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 between sea turtles and fishing vessels (*i.e.*, vessel strikes) 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.

NMFS's Southeast Regional Office (SERO) has carried out formal ESA section 7 consultations for several FMPs that at least partially overlap with the action area considered in this Opinion. These include: coastal migratory pelagics, swordfish/tuna/shark/billfish (*i.e.*, HMS), snapper/grouper, dolphin/wahoo, and the Southeast shrimp trawl fisheries (NMFS 2003b, 2004c, 2006c, 2007a, 2008c, 2012). The ITSs provided with these Opinions are included in Table 5 below.

In 2004, NMFS SERO conducted a formal consultation on the pelagic longline component of the Atlantic HMS 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. However, NMFS concluded that the action was likely to jeopardize the continued existence of leatherback sea turtles (NMFS 2004c). This Opinion included a Reasonable and Prudent Alternative (RPA) 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 Table 5. Unless specifically noted, all numbers denote an annual number of captures that may be lethal or non-lethal.

In addition to this consultation, we are in the process of reinitiating seven other formal consultations that consider fisheries actions (*i.e.*, FMPs) that may affect Atlantic sturgeon: (1) Atlantic bluefish, (2) Atlantic mackerel/squid/butterfish, (3) monkfish, (4) northeast multispecies, (5) northeast skate complex, (6) spiny dogfish, and (7) summer flounder/scup/black sea bass. Atlantic sturgeon originating from the five DPSs considered in this consultation are either known or are likely to be captured in these fisheries that operate in the action area. 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 (NEFSC 2011a).

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. At this time, there is only one Southeast fishery, the Southeast shrimp trawl fishery, which has a bycatch estimate for Atlantic sturgeon. In their May 8, 2012, Opinion on the fishery, NMFS SERO estimated that a total of 1,731 total interactions, including 243 captures (of which 27 are expected to be lethal), are likely to occur every three years as a result of the Southeast shrimptrawl fishery. The level of interactions and mortality were expected to be greatest within the SA DPS, followed by the Carolina, NYB, CB, and GOM DPSs. Other fisheries in the Southeast that operate with sink gillnets or otter trawls are also likely to interact with Atlantic sturgeon and be an additional source of mortality in the action area. Consultation on the smooth dogfish fishery is currently being conducted by NMFS SERO in coordination with the NMFS HMS Division and may soon provide an additional bycatch estimate for Atlantic sturgeon in that fishery.

Table 5. Dates of the most recent Opinions prepared by NMFS NERO and SERO for federally managed fisheries in the action area (excluding Atlantic Sea Scallops) and their respective ITSs for sea turtles. Unless noted, levels of incidental take exempted are on an annual basis.

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
Multispecies	October 29, 2010	46 (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 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
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
South Atlantic snapper-grouper	June 7, 2006	202 (67 lethal) every 3 years	19 (8 lethal) every 3 years	39 (14 lethal) every 3 years	25 (15 lethal) every 3 years
Pelagic longline fishery under the HMS FMP (per the RPA)	June 1, 2004	1,905 (339 lethal) every 3 years	*105 (18 lethal) every 3 years	*105 (18 lethal) every 3 years	1764 (252 lethal) every 3 years
South-Atlantic dolphin-wahoo**	August 27, 2003	12 (2 lethal) every 3 years	2 (1 lethal) every 3 years	2 (1 lethal) every 3 years	12 (1 lethal) every 3 years
Southeastern shrimp trawling***	May 8, 2012	Not able to be estimated	Not able to be estimated	Not able to be estimated	Not able to be estimated
Tilefish	March 13, 2001	6 (3 lethal)			1

**combination of 105 (18 lethal) Kemp's ridley, green, hawksbill, or olive ridley*
***combination of 16 turtles total every 3 years with 2 lethal (Kemp's ridley, green, hawksbill, leatherback)*
****although the ITS in this Opinion does not provide actual estimates of incidental take for any sea turtle species, the effects section provides a qualitative assessment of likely impacts based on orders of magnitude (e.g., for Kemp's ridleys, at least tens of thousands and possibly hundreds of thousands of interactions are expected annually; of those interactions, thousands and possibly tens of thousands are expected to be lethal)*

4.1.2 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 or the Wilmington District. Hopper dredging projects in this area have resulted in the recorded mortality of approximately 87 loggerheads, four greens, nine Kemp's ridleys, and four unidentified hard-shelled sea 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 three records documenting interactions between hopper dredges and Atlantic sturgeon in the action area (two in Virginia near the Chesapeake Bay entrance, and one in New York Bight). NMFS NERO and SERO have completed several ESA section 7 consultations with the Army Corps of Engineers, as well as one with the National Aeronautics and Space Administration (NASA), to consider effects of hopper dredging projects on listed sea turtles. Several of these consultations have recently been reinitiated to consider effects to Atlantic sturgeon. Table 6 below provides information on Opinions prepared for dredging projects in the action area and the associated ITSs for sea turtles.

Table 6. Information on formal consultations conducted by NMFS for dredging projects that occur in the action area. Unless otherwise noted, take estimates are per dredge cycle.

Project	Date of Opinion	Loggerhead	Kemp's ridley	Green	Leatherback	Notes
USCOE - Continued Hopper Dredging of Channels and Borrow Areas in the SE U.S.	9/25/1997	24	7	7	0	Annual Estimate
Dredging of Sandbridge Shoals, VA	4/2/1993	5	1 Kemp's ridley or green		0	
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
MD Coastal Beach Protection Project (includes several projects with different ITSs)	4/6/1998	10	1	2	0	total takes over 25 year Assateague Island project
		6	1	1	0	takes per dredge cycle for MD shoreline protection project
Thimble Shoals and Atlantic Ocean Channels Dredging	4/25/2002	4 (≤1 million cy) 10 (>1 to ≤3 million cy) 18 (>3 to ≤5 million cy)	1 (≤1 million cy) 2 (>1 to ≤3 million cy 4 (>3 to ≤5 million cy)	0	0	
Ambrose Channel, NJ Sand Mining	10/11/2002	2	1	1	1	1 leatherback OR Kemp's
Cape Henry, York Spit, York River Entrance, and Rappahannock Shoal Channels - Maintenance Dredging	7/24/2003	4 (≤1 million cy); 10 (>1 to ≤3 million cy); 18 (>3 to ≤5 million cy)	1 (≤1 million cy); 2 (>1 to ≤3 million cy); 4 (>3 to ≤5 million cy)	0	0	
		Relocation Trawling: 120 non-lethal takes for any combination of the four species.				
Dam Neck Naval Facility Beach Dredging and Beach Nourishment	12/12/2003	4	1 green or Kemp's ridley		0	
VA Beach Hurricane Protection Project	12/2/2005	4	0	0	1	
		Relocation Trawling: Up to 45 takes in any combination of loggerheads, greens, leatherbacks, and Kemps ridleys. 1 lethal take of a loggerhead, green, leatherback OR Kemps ridley.				

Atlantic Coast of Maryland Shoreline Protection Project	11/30/2006	1 (≤0.5 million cy); 2 (>0.5 to ≤1 million cy); 3 (>1 to ≤1.5 million cy); 4 (>1.5 to ≤1.6 million cy)			2	Over life of project (through 2044), ~ 10-12 million cy will be dredged with an anticipated total of 24 turtles killed (2 Kemp's, 22 loggerheads)
NASA's Wallops Is. Shoreline Restoration/ Infrastructure Protection Program	7/22/2010	9			1	total over 50 year project life

4.1.3 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, and NOAA. 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 Bureau of Ocean Energy Management (BOEM), Federal Energy Regulatory Commission (FERC), and Maritime Administration (MARAD) on vessel traffic related to energy projects in the Northeast Region and these agencies have 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. 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 primarily 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, 2008d, 2009e) and USCG (NMFS 1995, 1998b) contain details on the scope of vessel operations for these agencies and the conservation measures that are being implemented as standard operating procedures. In the U.S. Atlantic, the operation of USCG boats and cutters is estimated to take no more than one individual sea turtle, of any species, per year (NMFS 1995).

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

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

Similarly, operations of vessels by other Federal agencies within the action area (NOAA, EPA, and Army Corps of Engineers) may adversely affect sea turtles, as well as Atlantic sturgeon. 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. For example, NOAA research vessels conducting fisheries surveys for the NEFSC are estimated to capture no more than 11 sea turtles and nine Atlantic sturgeon per year. This includes up to seven NWA DPS loggerheads, one leatherback, two Kemp's ridleys, and one green sea turtle, as well as four NYB, two SA, one GOM, one CB, and one Carolina DPS origin Atlantic sturgeon per year (NMFS 2012b).

In addition to the NEFSC surveys which occur throughout the year, NMFS also funds the Northeast Area Monitoring and Assessment Program (NEAMAP) nearshore trawl surveys which are conducted for one month every spring and fall by the Virginia Institute of Marine Science in shallow, nearshore waters (up to 120 feet) from Cape Hatteras, NC to Montauk, NY. The 2012 surveys conducted by VIMS, and funded by NMFS through the Mid-Atlantic RSA Program, are expected to result in the annual capture of six NWA DPS loggerhead sea turtles, four Kemp's ridley sea turtles, one green sea turtle, one leatherback sea turtle, and no more than 32 Atlantic sturgeon. Based on mixed stock analyses, NMFS anticipates that up to 15 of the interactions will involve fish of NYB DPS origin, five of CB DPS origin, nine of SA DPS origin, and three of GOM DPS origin. No mortalities of any ESA-listed species are expected (NMFS 2012c).

4.2 Non-federally regulated fisheries

Several fisheries for species that are not managed by a Federal FMP occur in both state and Federal waters of the action area. The amount of gear contributed to the environment by these fisheries is often unknown. In most cases, there is limited observer coverage of these fisheries and the extent of interactions with ESA-listed species is difficult to estimate. Sea turtles and Atlantic sturgeon may be vulnerable to capture, injury, and mortality in a number of these fisheries. Captures of both sea turtles (SEFSC 2001; Murray 2009a; Warden 2011a) and Atlantic sturgeon (ASSRT 2007; NMFS Sturgeon Workshop 2011) in these fisheries have been reported.

The available bycatch data for FMP fisheries indicate that sink gillnets and otter trawl gear pose the greatest risk to Atlantic sturgeon (ASMFC TC 2007), although Atlantic sturgeon are occasionally caught by hook and line, fyke nets, and crab pots as well (NMFS Sturgeon Workshop 2011). It is likely that this vulnerability to these types of gear is similar for non-Federal fisheries, although there is little data available to support this. Information on the number of Atlantic sturgeon captured or killed in non-Federal fisheries, which primarily occur in state waters, is extremely limited. An Atlantic sturgeon "reward program," where commercial fishermen were provided monetary rewards for reporting captures of Atlantic sturgeon in Chesapeake Bay, operated from 1996 to 2012 in Maryland (Mangold *et al.* 2007). The data from this program show that Atlantic sturgeon have been caught in a wide variety of gear types, including hook and line, pound nets, gillnets, crab pots, eel pots, hoop nets, trawls, and fyke nets. Pound nets (58.9%) and gillnets (40.7%) accounted for the vast majority of captures. Of the more than 2,000 Atlantic sturgeon reported in the reward program during 11 years (1996-2006), biologists counted ten individuals that died as a result of their capture. No information on post-release mortality is available.

Efforts are currently underway to obtain more information on the numbers of Atlantic sturgeon captured and killed in state-water fisheries and a handful of states (*e.g.*, Delaware, New Jersey, New York, and North Carolina) are in the process of applying for ESA section 10 permits to cover the incidental capture of Atlantic sturgeon in their state fisheries. Preliminary and anecdotal information suggests the numbers of Atlantic sturgeon captured or killed in state-water fisheries is small. Atlantic sturgeon are also vulnerable to capture in state-water fisheries occurring in rivers, such as shad fisheries; however, these riverine areas are outside the action area under consideration in this Opinion. Where available, state-specific information on sea turtle and Atlantic sturgeon interactions in non-Federal fisheries is provided below.

Atlantic croaker fishery

An Atlantic croaker fishery using trawl and gillnet gear also occurs within the action area and sea turtle interactions 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 2011a). 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). These estimates encompass the bycatch of loggerheads in the Atlantic croaker fishery in both state and Federal waters.

Atlantic sturgeon interactions have also been observed in the Atlantic croaker fishery, but a quantitative assessment of the number of Atlantic sturgeon captured in the croaker fishery is not available. A mortality rate of Atlantic sturgeon in commercial trawls has been estimated at 5%. A review of the NEFOP 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 trips that included a NEFOP observer onboard.

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 gillnets, pound nets, haul seines, flynets, and trawls, with the majority of landings occurring in the fall and winter months (ASMFC 2002). Weakfish landings were dominated by the trawl fishery through the mid-1980s, after which gillnet 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). Sea turtle bycatch in the weakfish fishery has occurred (Murray 2009a, 2009b; Warden 2011a) and NMFS originally assessed the impacts of the fishery on sea turtles in an Opinion back in 1997 (NMFS 1997b). Currently, the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the weakfish fishery is estimated to be one (Warden 2011a). Additional information on loggerhead 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 per year with a 95% CI of 0-1 (Murray 2009b). These estimates encompass the bycatch of loggerheads in the weakfish fishery in both state and Federal waters.

A quantitative assessment of the number of Atlantic sturgeon captured in the weakfish fishery is not available. A mortality rate 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, and most inshore fisheries are not observed. An earlier review of bycatch rates and landings for the weakfish fishery reported that the weakfish-striped bass fishery had an Atlantic sturgeon bycatch rate of 16% from 1989-2000; the weakfish-Atlantic croaker fishery had an Atlantic sturgeon bycatch rate of 0.02%, and the weakfish fishery had an Atlantic sturgeon bycatch rate of 1.0% (ASSRT 2007).

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, Massachusetts, Connecticut, New York, New Jersey, 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). Loggerhead, leatherback, and green sea turtles are known to become entangled in lines associated with pot/trap gear used in several fisheries including lobster, whelk, and crab species (SEFSC 2001; Dwyer *et al*. 2002: NMFS 2007b). Whelk fisheries in Massachusetts, New York, New Jersey, and Virginia were verified as the fisheries involved in 18 sea turtle entanglements from 2001 to 2010. Twelve entanglement events involved a leatherback sea turtle, five involved a loggerhead sea turtle, and one involved a green sea turtle (Northeast Region Sea Turtle Disentanglement Network [STDN] database). 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. Loggerhead, leatherback, and green sea turtles are known to become entangled in lines associated with pot/trap gear used in several fisheries including lobster, whelk, and crab species (SEFSC 2001; Dwyer *et al*. 2002: NMFS 2007b). The Virginia blue crab fishery was verified as the fishery involved in four sea turtle entanglements from 2001 to 2010. Two entanglement events involved a leatherback sea turtle and two involved a loggerhead sea turtle (Northeast Region STDN database).

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 to 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 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), coincident 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.

Atlantic sturgeon are known to be caught in state water horseshoe crab fisheries, which currently operate in all action area states except New Jersey. Along the U.S. East Coast, hand, trawl, and dredge fisheries account for more than 85% of the commercial horseshoe crab landings in the bait fishery. Other methods used are gillnets, pound nets, and traps (ASMFC 2011a). State waters from Delaware to Virginia are closed to horseshoe crab harvest and landing from January 1 to June 7 (ASMFC 2011a). The majority of horseshoe crab landings in 2010 came from Massachusetts, Virginia, and Delaware. Stein *et al*. (2004) examined bycatch of Atlantic sturgeon using the NMFS sea-sampling/observer database (1989-2000) and found that the bycatch rate for horseshoe crabs was low, at 0.05%. An Atlantic sturgeon "reward program," where commercial fishermen were provided monetary rewards for reporting captures of Atlantic sturgeon in the Maryland waters of Chesapeake Bay, operated from 1996 to 2012 (Mangold *et al*. 2007).[11] The data from this program during the 11-year period of 1996-2006 show that one of 1,395 wild Atlantic sturgeon was found caught in a crab pot (Mangold *et al*. 2007).

Virginia pound net fishery
Sea turtle have been observed to interact with the Virginia pound net fishery, which is contiguous to the action area at the mouth of Chesapeake Bay. Pound nets with large-mesh and stringer leaders set in Virginia waters of Chesapeake Bay have been implicated in sea turtle mortalities as a result of entanglement in the pound net leader, and live sea turtles have also been found in the pounds. As described in section 4.4.4 below, NMFS has taken regulatory action to address sea turtle bycatch in the Virginia pound net fishery. Atlantic sturgeon are also captured in pound nets; however, mortality rates are thought to be very low. No estimate of the number of Atlantic sturgeon caught in pound nets in the action area is currently available.

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 Interstate Fishery Management Plan (ISFMP). Like the Federal waters component of the fishery mentioned in section 4.1, 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 2001 and 2010, lobster trap gear traced back to a fisherman possessing a state permit was verified as the gear involved in 33 leatherback entanglements in the Northeast Region. Of those, 28 were state-permitted only (*i.e.*, they had to have occurred in state waters). The other five could have potentially occurred in Federal waters, as the fisherman either had both state and Federal permits or it was not known if they had a Federal permit. All entanglements involved the vertical line of the gear. These verified/confirmed entanglements occurred in waters off Maine, Massachusetts, Rhode Island, and Connecticut from June through October; the vast majority (27 of the 33) were documented in waters off Massachusetts (Northeast Region STDN database). Atlantic sturgeon are not known to interact with lobster trap gear.

Fish trap, seine, and channel net fisheries
Incidental captures of loggerheads in fish traps have been reported from several states along the U.S. Atlantic coast (Shoop and Ruckdeschel 1989; W. Teas, NMFS, pers. comm.), while leatherbacks have been documented as entangled in the buoy line systems of conch and sea bass traps off Massachusetts (Northeast Region STDN database). Long haul seines, purse seines, and

[11] The program was terminated in February 2012, with the listing of Atlantic sturgeon under the ESA.

channel nets are also known to incidentally capture sea turtles in sounds and other inshore waters along the U.S. Atlantic coast, although no lethal interactions have been reported (SEFSC 2001). No information on interactions between Atlantic sturgeon and fish traps, long haul seines, purse 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.

Northern shrimp fishery
A Northern shrimp fishery also occurs in state waters of Maine, New Hampshire, and Massachusetts, and is managed under the ASMFC's ISFMP. In 2010, the ISFMP implemented a 126-day season, from December 1 to April 15, but the shrimp fishery has exceeded its TAC and closed early ever year, ending on February 17 in 2012. The majority of northern shrimp are caught with otter trawls, which must be equipped with Nordmore grates (ASMFC NSTC 2011). Otter trawls in this fishery are known to interact with Atlantic sturgeon, but exact numbers are not available (NMFS Sturgeon Workshop 2011). A significant majority (84%) of Atlantic sturgeon bycatch in otter trawls occurs at depths <20 meters, with 90% occurring at depths of <30 meters (Miller 2007). During the spring and fall inshore trawl surveys, northern shrimp are most commonly found in tows with depths of >64 meters (ASFMC NSTC 2011), which is well below the depths at which most Atlantic sturgeon bycatch is occurring. Atlantic sturgeon are known to interact with shrimp trawls, but mortality is low: NEFOP data from 2002-2004 showed 0.2% Atlantic sturgeon mortality in shrimp and otter trawls; Stein *et al*. (2004) reported no immediate Atlantic sturgeon mortality in trawls from 1989-2000 from North Carolina to Maine; and Cooperative Winter Tagging Cruises captured 146 Atlantic sturgeon from 1988-2006, of which none died (Laney *et al*. 2007; ASSRT 2007).

American shad fishery
An American shad fishery also occurs in state waters of New England and the Mid-Atlantic and is managed under the ASMFC's ISFMP. The directed commercial and recreational shad fisheries were closed in all Atlantic coastal states in 2005, with exceptions for sustainable systems as determined through state-specific management programs. Presently, only Connecticut has a directed commercial shad fishery that may occur in the action area, while Maine, New Hampshire, Massachusetts, New York, Rhode Island, Connecticut, New Jersey, and Delaware have limited recreational fisheries that may occur in the action area. New York's commercial shad fishery had been a problem in the past, but the fishery is now closed.

About 40-500 Atlantic sturgeon were reportedly captured in the spring shad fishery in the past, primarily from the Delaware Bay, with only 2% caught in the river. Effort has more recently switched to striped bass, however. The fishery uses five-inch mesh gillnets left overnight to soak, but, based on the available information, there is little bycatch mortality. Unreported mortality may be occurring in the recreational shad fishery, but the extent is unknown (NMFS Sturgeon Workshop 2011).

Recreational hook and line shad fisheries are known to capture Atlantic sturgeon, particularly in southern Maine, where it is considered to be an "acute" problem (NMFS Sturgeon Workshop 2011). Data from the Atlantic Coast Sturgeon Tagging Database (2000-2004) shows that the shad fishery accounted for 8% of Atlantic sturgeon recaptures. The shad fishery also had one of the highest bycatch rates of 30 directed fisheries according to NMFS Observer Program data from 1989-2000 (ASSRT 2007). However, greater rates of bycatch do not necessarily translate into high mortality rates. Other factors, such as gear, season, and soak times, may be important variables in understanding Atlantic sturgeon mortality.

Striped bass fishery
The striped bass fishery occurs in only in state waters, as Federal waters have been closed to the harvest and possession of striped bass since 1990, except that possession is allowed in a defined area around Block Island, Rhode Island (ASMFC 2011b). The ASMFC has managed striped bass since 1981, and provides guidance to states from Maine to North Carolina through an ISFMP. All states are required to have recreational and commercial size limits, recreational creel limits, and commercial quotas. The commercial striped bass fishery is closed in Maine, New Hampshire, and Connecticut, but open in Massachusetts (hook and line only), Rhode Island, New Jersey (hook and line only), Delaware, Maryland, Virginia, and North Carolina. Recreational striped bass fishing occurs all along the U.S. East Coast.

Several states have reported incidental catch of Atlantic sturgeon (NMFS Sturgeon Workshop 2011). In southern Maine, the recreational striped bass fishery is known to catch Atlantic sturgeon and in New Hampshire, live bait recreational fisheries are also known to catch Atlantic sturgeon, although numbers are not available. The hook and line striped bass fishery along the south shore of Long Island has reports of Atlantic sturgeon bycatch, with hundreds of reports of sturgeon caught or snagged in recreational gear particularly around Fire Island and Far Rockaway. Atlantic sturgeon bycatch is occurring in the Delaware Bay and River, but little bycatch mortality has been reported. Unreported mortality is likely occurring. And in North Carolina, the Winter Beach seine fishery for striped bass is known to capture Atlantic sturgeon (adults and subadults), but has not reported mortalities.

Data from the Atlantic Coast Sturgeon Tagging Database (2000-2004) shows that the striped bass fishery accounted for 43% of Atlantic sturgeon recaptures (ASSRT 2007). The striped bass-weakfish fishery also had one of the highest bycatch rates of 30 directed fisheries according to NMFS Observer Program data from 1989-2000 (ASSRT 2007). However, greater rates of bycatch do not necessarily translate into high mortality rates. Other factors, such as gear, season, and soak times, may be important variables in understanding Atlantic sturgeon mortality.

State gillnet fisheries
Two 10- to 14-inch (25.6- to 35.9-centimeter) mesh gillnet fisheries, the black drum and sandbar shark gillnet fisheries, occur in Virginia state waters along the tip of the eastern shore. Given the gear type, these fisheries may capture or entangle sea turtles. Entanglements of sea turtles in gillnet sets targeting and/or landing both species have been recorded in the NEFOP database. Similarly, sea turtles are thought to be vulnerable to capture in small mesh gillnet fisheries occurring in Virginia state waters. During May-June 2001, NMFS observed 2% of the Atlantic croaker fishery and 12% of the dogfish fishery (which represent approximately 82% of Virginia's total small mesh gillnet landings from offshore and inshore waters during this time),

yet no sea turtle captures were observed (NMFS 2004e). Based on gear type (*i.e.*, gillnets), it is likely that Atlantic sturgeon would be vulnerable to capture in these fisheries. An Atlantic sturgeon "reward program" where fishermen were provided monetary rewards for reporting captures of Atlantic sturgeon, operated in the late 1990s in Virginia. The majority of reports of Atlantic sturgeon captures were in drift gillnets and pound nets. No quantitative information on the number of Atlantic sturgeon captured or killed in Virginia fisheries is currently available.

In North Carolina, a large-mesh gillnet fishery for southern flounder in the southern portion of Pamlico Sound is known to incidentally capture sea turtles. ESA section 10 incidental take permits have been issued by NMFS to the state for this fishery in 2000, 2001, 2002, and 2005 (76 FR 61670). The section 10 permit was most recently renewed for the 2005-2010 fishing years with incidental take estimates derived from the 2001-2004 at-sea monitoring program. The 2005-2010 incidental take permit exempted the 'estimated' capture of 41 Kemp's ridley (14 lethal), 168 green (48 lethal), and 41 loggerhead sea turtles (three lethal) over sequential three-year periods (2005-2007, 2008-2010). It also exempted the 'observed' capture of two leatherbacks, two hawksbills, and six Kemp's ridley/green/loggerhead sea turtles (any combination of the three species) over those same time periods. The state of North Carolina is currently reapplying for incidental take coverage for sea turtles for three more years. During 2004, 42 Atlantic sturgeon were observed captured in gillnet fisheries operating in Albemarle and Pamlico Sounds. Of these observed Atlantic sturgeon, five mortalities were reported. A quantitative assessment of the number of Atlantic sturgeon captured or killed in North Carolina state fisheries that occur in the action area is not currently available. The state is currently applying for ESA section 10 coverage of Atlantic sturgeon captures in this fishery.

State recreational fisheries

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 (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. Stranding data also provide some evidence of interactions between recreational hook-and-line gear and sea turtles, but assigning the gear to a specific fishery is rarely, if ever, possible. Atlantic sturgeon have also been observed captured in hook-and-line gear, yet the number of interactions that occur annually is unknown. While most Atlantic sturgeon are likely to be released alive, we currently have no information on post-release survival. NMFS is currently working on a pilot project to assess the extent of sea turtle interactions that occur in recreational fisheries of the Southeast (North Carolina to Florida) and believes that the survey platform and questionnaire may also be applicable for determining the amount of Atlantic sturgeon interactions as well.

4.3 Other Activities

4.3.1 Maritime Industry

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

4.3.2 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 studies (Vargo *et al.* 1986; NOAA 2010).

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.

4.3.3 Coastal development

Beachfront development, lighting, and beach erosion control all are ongoing activities along the U.S. Atlantic coastline. These activities could reduce or degrade potential sea turtle nesting habitats in the Mid-Atlantic (from North Carolina to as far north as New Jersey) 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.

4.4 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.

4.4.1 Large-Mesh Gillnet Requirements in the Mid-Atlantic

Since 2002, NMFS has regulated the use of large mesh gillnets in Federal waters off North Carolina and Virginia (67 FR 13098, March 21, 2002) to reduce the impact of these fisheries on ESA-listed sea turtles. These restrictions were revised in 2006 (73 FR 24776, April 26, 2006). Currently, gillnets with stretched mesh size 7 inches (17.8-centimeters) or larger are prohibited in the EEZ (as defined in 50 CFR 600.10) during the following times and in the following areas: (1) north of the North Carolina/South Carolina border to Oregon Inlet at all times, (2) north of Oregon Inlet to Currituck Beach Light, North Carolina from March 16 through January 14, (3) north of Currituck Beach Light, North Carolina to Wachapreague Inlet, Virginia from April 1 through January 14, and (4) north of Wachapreague Inlet, Virginia to Chincoteague, Virginia from April 16 through January 14. Federal waters north of Chincoteague, Virginia remain unaffected by the large-mesh gillnet restrictions. These measures are in addition to Harbor Porpoise Take Reduction Plan measures that prohibit the use of large-mesh gillnets in southern Mid-Atlantic waters (territorial and Federal waters from Delaware through North Carolina out to 72° 30' W longitude) from February 15 through March 15, annually. The measures are also in addition to comparable North Carolina and Virginia regulations for large-mesh gillnet fisheries in their respective state waters that were enacted in 2005.

NMFS has also issued regulations to address interactions of sea turtles in gillnet gear fished in Pamlico Sound, North Carolina. Waters of Pamlico Sound are closed to fishing with gillnets with a stretched mesh size larger than 4¼ inches (10.8 centimeters) from September 1 through December 15 each year to protect sea turtles. The closed area includes all inshore waters of Pamlico Sound, and all contiguous tidal waters, south of 35° 46.3' N latitude, north of 35° 00' N latitude, and east of 76° 30' W longitude.

4.4.2 TEDs requirements in trawl fisheries

TEDs are required in the shrimp and summer flounder fisheries. TEDs allow sea turtles to escape the trawl net, reducing injury and mortality resulting from capture in the net. Approved

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

TEDs are also required for summer flounder trawlers in the summer flounder fishery-sea turtle protection area. This area is bounded on the north by a line extending along 37° 05' N (Cape Charles, Virginia) and on the south by a line extending out from the North Carolina/South Carolina border. Vessels north of Oregon Inlet, North Carolina are exempt from the TED requirement from January 15 through March 15 each year (50 CFR 223.206). The TED requirements for the summer flounder trawl fishery do not require the use of the larger escape opening. NMFS is considering increasing the size of the TED escape opening currently required in the summer flounder fishery and implementing sea turtle conservation requirements in other trawl fisheries and in other areas (72 FR 7382, February 15, 2007; 74 FR 21630, May 8, 2009).

Currently, NMFS is proposing to withdraw the alternative tow time restriction and require all skimmer trawls, pusher-head trawls, and wing nets (butterfly trawls) rigged for shrimp fishing to use TEDs in their nets. A draft EIS for this rule has been prepared and is currently undergoing public comment. The intent of this proposed rule is to further reduce incidental bycatch and mortality of sea turtles in the southeastern U.S. shrimp fisheries (77 FR 27411, May 10, 2012).

4.4.3 Sea Turtle Conservation Requirements in the Virginia Pound Net Fishery

NMFS has issued several regulations to help protect sea turtles from entanglement in and impingement on Virginia pound net gear (66 FR 33489, June 22 2001; 67 FR 41196; June 17, 2002; 68 FR 41942, July 16, 2003; 69 FR 24997, May 5, 2004). Currently, all offshore pound leaders in Pound Net Regulated Area I (see Figure 3 below) must meet the definition of a modified pound net leader from May 6 through July 15. The modified leader has been found to be effective in reducing sea turtle interactions. Nearshore pound net leaders in Pound Net Regulated Area I and all pound net leaders in Pound Net Regulated Area II (Figure 3) must have mesh size less than 12 inches (30.5 centimeters) stretched mesh and may not employ stringers (50 CFR 223.206) from May 6 through July 15 each year. A pound net leader is exempt from these measures only if it meets the definition of a modified pound net leader. In addition, there are monitoring and reporting requirements in this fishery (50 CFR 223.206). Since the 2010 fishing season, the state of Virginia has required modified pound net leaders (as defined by Federal regulations) east of the Chesapeake Bay Bridge year round, and in offshore leaders in

Regulated Area I (also as defined by Federal regulations) from May 6 to July 31. This is a 16-day extension of the Federal regulations in this area.

Figure 3. Management Areas in the Virginia Pound Net Fishery

4.4.4 Sea Turtle Conservation Requirements in the HMS Fishery

NMFS completed the most recent Opinion on the FMP for the Atlantic HMS fisheries for swordfish, tunas, and sharks on June 1, 2004, and concluded that the Atlantic HMS fisheries, particularly the pelagic longline fisheries, were likely to jeopardize the continued existence of leatherback sea turtles. An RPA was provided to avoid jeopardy to leatherback sea turtles as a result of operation of the HMS fisheries. Although the Opinion did not conclude jeopardy for loggerhead sea turtles, the RPA is also expected to benefit this species by reducing mortalities resulting from interactions with the gear. A number of requirements have been put in place as a result of the Opinion and subsequent research. These include measures related to the fishing gear, bait, disentanglement gear and training. Since 2004, bycatch estimates for both loggerheads and leatherbacks in pelagic longline gear have been well below the average prior to implementation of gear regulations under the RPA (Garrison and Stokes 2012).

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

NMFS requires the use of specific gears and release equipment in the pelagic longline component of the HMS fishery in order to minimize lethal impacts to sea turtles. Sea turtle handling and release protocols for the HMS fishery are described in detail in SEFSC (2008). Sea turtle handling and release placards are required to be posted in the wheelhouse of certain commercial fishing vessels. NMFS has also initiated an extensive outreach and education program for commercial fishermen that engage in these fisheries in order to minimize the impacts of this fishery on sea turtles. As part of the grogram, NMFS has distributed sea turtle identification and resuscitation guidelines to HMS fishermen who may incidentally hook, entangle, or capture sea turtles during their fishing activities and has also conducted hands on workshops on safe handling, release, and identification of sea turtles.

4.4.5 Sea Turtle Handling and Resuscitation Techniques

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

4.4.6 Exception for Injured, Dead, or Stranded Specimens

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

4.4.7 Education and Outreach Activities

Education and outreach activities are considered some of the primary tools we can use to reduce the threats to all protected species. For example, NMFS has been active in public outreach to educate commercial fishermen regarding sea turtle handling and resuscitation techniques and has issued guidelines for recreational fishermen and boaters on how to avoid the likelihood of interactions with sea turtles. NMFS intends to continue these outreach efforts in an attempt to reduce interactions with protected species, and to reduce the likelihood of injury to protected species when interactions do occur.

4.4.8 Sea Turtle Stranding and Salvage Network (STSSN)

There is an extensive network of STSSN participants along the U.S. Atlantic and Gulf of Mexico coasts that collects data on dead sea turtles and rescues and rehabilitates live stranded sea turtles, reducing mortality of injured or sick animals. Data collected by the STSSN are used to monitor stranding levels and identify areas where unusual or elevated mortality is occurring, and to identify sources of mortality. These data are also used to monitor incidence of disease, study toxicology and contaminants, and conduct genetic studies to determine population structure. All of the states that participate in the STSSN tag live sea turtles when encountered (either via the stranding network through incidental takes or in-water studies). Tagging studies help improve our understanding of sea turtle movements, longevity, and reproductive patterns, all of which contribute to our ability to reach recovery goals for the species.

4.4.9 Sea Turtle Disentanglement Network (STDN)

NMFS NERO established the Northeast Region STDN in 2002 in response to the high number of leatherback sea turtles found entangled in pot gear along the U.S. Northeast Atlantic coast. The STDN is considered a component of the larger STSSN program and operates in all states in the region. The STDN responds to entangled sea turtles in order to disentangle and release live animals, thereby reducing serious injury and mortality. In addition, the STDN collects data on these events, providing valuable information for management purposes. The NMFS NERO oversees the STDN program and manages the STDN database.

4.5 Reducing Threats to Atlantic Sturgeon

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

4.6 Magnuson-Stevens Fishery Conservation and Management Act

In addition to the measures described in sections 4.4 and 4.5, there are numerous regulations mandated by the Magnuson-Stevens Fishery Conservation and Management Act that benefit ESA-listed species. Many fisheries are subject to different time and area closures. These area closures can be seasonal or year-round. Closure areas benefit ESA-listed species due to elimination of active gear in areas where sea turtles and/or Atlantic sturgeon are present. However, if closures shift effort to areas with a comparable or higher density of sea turtles or Atlantic sturgeon, then risk of interaction could actually increase. Fishing effort reduction measures (*i.e.*, landing/possession limits or trap allocations) also benefit ESA-listed species by limiting the amount of time that gear is present in the species environment. Additionally, gear restrictions and modifications required for fishing regulations also decrease the risk of entanglement with endangered species, as described in the sections above. A complete listing of fishery regulations in the action area is located at http://www.nero.noaa.gov/nero/regs/info.html.

5.0 CLIMATE CHANGE

In addition to the information on climate change presented in the *Status of the Species* section for sea turtles and Atlantic sturgeon, the discussion below presents further background information on global climate change as well as past and predicted future effects of global climate change throughout the range of the ESA-listed species considered here. Additionally, we present the available information on predicted effects of climate change in the action area and how listed sea turtles and Atlantic sturgeon may be affected by those predicted environmental changes over a time span of the proposed action for which we can realistically analyze impacts. Climate change is also relevant to the *Environmental Baseline* and *Cumulative Effects* sections of this Opinion, but rather than include partial discussions in several sections of this Opinion, we are synthesizing this additional information into one discussion.

The global mean temperature has risen 0.76ºC (1.36°F) over the last 150 years, and the linear trend over the last 50 years is nearly twice that for the last 100 years (IPCC 2007). Precipitation has increased nationally by 5%-10%, mostly due to an increase in heavy downpours (NAST 2000). There is a high confidence, based on substantial new evidence, that observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. Ocean acidification resulting from massive amounts of carbon dioxide and other pollutants released into the air can have major adverse impacts on the calcium balance in the oceans. Changes to the marine ecosystem due to climate change include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007); these trends 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 (GHGs), indicate that temperatures in the U.S. will rise by about 3º-5ºC (5º-9ºF) on average in the next 100 years which is more than the projected global increase (NAST 2000). A warming of about 0.2ºC (0.4°F) per decade is projected for the next two decades over a range of emission scenarios (IPCC 2007). This temperature increase will very likely be associated with more extreme precipitation and faster evaporation of water, leading to greater frequency of both very wet and very dry conditions. Climate warming has resulted in increased precipitation, river discharge, and glacial and sea-ice melting (Greene *et al.* 2008).

The past three decades have witnessed major changes in ocean circulation patterns in the Arctic, and these were accompanied by climate associated changes as well (Greene *et al.* 2008). Shifts in atmospheric conditions have altered Arctic Ocean circulation patterns and the export of freshwater to the North Atlantic (IPCC 2007; Greene *et al.* 2008). 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 2007). The NAO impacts climate variability throughout the northern hemisphere (IPCC 2007). 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 2007). This warming extends over 1,000 meters (0.62 miles) deep and is deeper than anywhere in the world oceans and is particularly evident under the Gulf Stream/North Atlantic Current system (IPCC 2007). 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 (IPCC 2007; Greene *et al.* 2008). There is evidence that the NADW has already freshened significantly (IPCC 2007). 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 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 U.S. 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. over the next 25 to 50 years regardless of reduction in GHGs, due to emissions that have already occurred (NAST 2000). It is very likely that the magnitude and frequency of ecosystem changes will continue to increase in the next 25 to 50 years, and it is possible that 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. Information below on impacts to rivers is generally relevant to Atlantic sturgeon, given they inhabit rivers for early development, foraging, seeking refuge, and spawning, and to sea turtles to the extent rivers affect conditions in estuaries, bays, and coastal areas where sea turtles forage, seek refuge, and use for other purposes. 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 DO 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, where the Carolina and South Atlantic DPSs of Atlantic sturgeon spawn in some rivers, 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 0.2ºC (0.4°F) per decade; and 3) a rise in sea level (NAST 2000). A warmer and drier climate will reduce stream flows and increase water temperature resulting in a decrease of DO and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing. Sea level is expected to continue rising: during the 20th century global sea level has increased 15 to 20 centimeters (6 to 8 inches).

Effects of climate change in the action area
As there is significant uncertainty in the rate and timing of change, as well as the effect of any changes that may be experienced in the action area due to climate change, it is difficult to predict the impact of these changes on sea turtles and Atlantic sturgeon. Generally speaking, the scallop fishery is expected to continue in the near and mid-term future in similar areas, at similar times, and with similar levels of effort, but there is no way to predict at this point in time whether the scallop resource and other environmental conditions will support a fishery that is similar to the proposed action in the long-term future or indefinitely.

Recently, stock assessments and essential fish habitat analyses for the scallop fishery have been conducted at five-year intervals. Due to frequent changes in the fishery, habitat, and status of the scallop resource, using stock and EFH assessments to inform management decisions beyond five years is not realistic. Due to the availability of staff resources, our time frames for producing new bycatch estimates for loggerheads and Atlantic sturgeon in trawl, gillnet, and dredge fisheries are also proposed to occur on staggered five-year cycles, with additional periods of time to assess whether there have been significant changes in bycatch rates from one time period to the next. Therefore, taking into account the different timelines for all these assessments, we expect that we will have to evaluate whether there is a need reinitiate consultation on the fishery at some point in the next ten years, and that beyond ten years the effects of the fishery in combination with environmental changes on ESA-listed species may be completely different than they are currently.

Given the timeframes related to the data on which management of the fishery are based, we do not believe that it is possible to analyze reliably effects of the action far into the future. Anticipating that the scallop fishery will operate the same way for more than ten years is not only speculative, but the history and pace of change in the fishery described in sections 1.0 and 2.0 suggests that it is not reasonable to expect the fishery to continue to operate as it is currently beyond ten years from now. As mentioned in NEFMC (2012), in general, scallop biomass has increased over the last 2-3 years on Georges Bank while it has decreased in the Mid-Atlantic. This led to an emergency closure of DMV in FY 2012 and reapportionment of effort onto Georges Bank. Since the distribution of effort in the fishery and the status of the resource can change over just a few years, we will primarily consider the effects of climate change over the next ten years. Longer-term effects of the fishery and climate change on ESA-listed species, whatever they may be, are much more difficult to pinpoint and extrapolate beyond ten years.

Sea turtles 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 been a problem for sea turtles or Atlantic sturgeon. 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. Recent studies suggest that up to half of the current available sea turtle nesting areas globally could be lost with predicted sea level rise (Fish *et al*. 2008; Mazaris *et al*. 2009; Witt *et al*. 2010), particularly at islands where no retreat options exist (Baker *et al*. 2006) or where anthropogenic coastal fortification causes 'coastal squeeze' (Fish *et al*. 2008). However, translocation, artificial shading, and watering of sea turtle nests have been offered up as a few stop-gap ways to help ameliorate the effects of climate change on sea turtles when it comes to nesting (Witt *et al*. 2010; Patino-Martinez *et al*. 2012). Studies into the success of these measures are ongoing. 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.

In the action area, it is possible that changing seasonal temperature regimes could result in changes in the timing of seasonal migrations through the area as sea turtles and Atlantic sturgeon move amongst nesting/spawning areas, summer foraging areas, and overwintering grounds. There could be shifts in the timing of nesting/spawning; presumably, if water temperatures warm earlier in the spring, nesting/spawning migrations and nesting/spawning events could occur earlier in the year (as water temperature is a primary nesting/spawning cue). For loggerhead sea turtles, warmer sea surface temperatures in the spring have been correlated to an earlier onset of nesting (Weishampel *et al*. 2004; Hawkes *et al*. 2007), shorter internesting intervals (Hays *et al*. 2002), and a decrease in the length of the nesting season (Pike *et al*. 2006). Green sea turtles also exhibited shorter internesting intervals in response to warming water temperatures (Hays *et al*. 2002). However, because nesting/spawning is not triggered solely by water temperature, it is difficult to predict how any change in water temperature alone will affect the seasonal movements of sea turtles and Atlantic sturgeon through the action area.

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

As described above, over the long term, global climate change may affect sea turtles and Atlantic sturgeon by affecting nesting/spawning patterns, distribution of prey, and water temperature. However, there is significant uncertainty, due to a lack of scientific data, on the degree to which these effects may be experienced and the degree to which sea turtles or Atlantic sturgeon will be able to successfully adapt to any such changes. Any activities occurring within and outside the action area that contribute to global climate change are also expected to affect sea turtles and Atlantic sturgeon in the action area. While we can make some predictions on the likely effects of climate change on these species, without modeling and additional scientific data, a high degree of uncertainty characterizes these predictions. Additionally, these predictions do not take into account the adaptive capacity of these species which may allow them to deal with change better than predicted. We do believe, however, that there will not be any new effects of climate change in the action area over the time frame assessed in this Opinion (*i.e.*, the next ten years) that may affect any of these species in a manner that was not already considered in the *Status of the Species* sections above.

6.0 EFFECTS OF THE ACTION

Pursuant to section 7(a)(2) of the ESA (16 U.S.C. 1536), Federal agencies are directed to ensure that activities or programs they authorize, fund, or carry out are not likely to jeopardize the continued existence of any listed species or result in the destruction or adverse modification of critical habitat. This Opinion examines the likely effects of the proposed action on ESA-listed species within the action area to determine if the continued operation of the scallop fishery under the Scallop FMP is likely to jeopardize the continued existence of those species. This analysis is done after careful review of the listed species' status and environmental baseline, as described above, as well as the effects of the action, cumulative effects, and the factors that affect the survival and recovery of those species. Since the proposed action is not expected to affect designated critical habitat, this Opinion will focus only on the jeopardy analysis.

In this section of the Opinion, we assess the direct and indirect effects of the proposed action on ESA-listed sea turtles and the five DPSs of Atlantic sturgeon. The purpose of the assessment is to determine if it is reasonable to conclude that the fishery is likely to have direct or indirect effects that appreciably reduce their likelihood of surviving and recovering in the wild by reducing their reproduction, numbers, or distribution.

As described in Section 3.0, we have determined that ESA-listed loggerhead, leatherback, Kemp's ridley, and green sea turtles, as well as the GOM, NYB, CB, Carolina, and SA DPSs of Atlantic sturgeon may be adversely affected by the continued operation of the scallop fishery as a result of interactions with gear used in the fishery. Our assessment of the effects of ESA-listed species interactions with scallop gear is provided below in order for us to make a determination as to whether the proposed action is likely to jeopardize the continued existence of these species.

6.1 Approach to the Assessment

We generally approach a jeopardy analysis in three steps. The first step identifies the probable direct and indirect effects of an action on the physical, chemical, and biotic environment of the action area, including the effects on individuals of threatened or endangered species. The second step determines the reasonableness of expecting threatened or endangered species to experience reductions in reproduction, numbers, or distribution in response to these effects. The third step determines if any reductions in a listed species' reproduction, numbers, or distribution (identified in the second step of the analysis) will appreciably reduce its likelihood of surviving and recovering in the wild.

The final step of the analysis - relating reductions in a species' reproduction, numbers, or distribution to reductions in the species' likelihood of surviving and recovering in the wild - is the most difficult step because (a) the relationship is not linear; (b) to persist over geologic time, most species have evolved to withstand some level of variation in their birth and death rates without a corresponding change in their likelihood of surviving and recovering in the wild; and (c) our knowledge of the population dynamics of other species and their response to human perturbation is usually too limited to support anything more than rough estimates. Nevertheless, our analysis must distinguish between anthropogenic reductions in a species' reproduction, numbers, and distribution that can reasonably be expected to affect the species' likelihood of survival and recovery in the wild and other (natural) declines. Consistent with statements from the U.S. Congress to provide the "benefit of the doubt" to threatened and endangered species [House of Representatives Conference Report No.697, 96th Congress, Second Session,12 (1979)], jeopardy analyses are designed to avoid concluding that actions have no effect on listed species or critical habitat when, in fact, there would be an effect.

In order to identify, describe, and assess the effects to ESA-listed sea turtles and Atlantic sturgeon DPSs resulting from fishing gear used in the scallop fishery, we are using: (1) information on interactions of loggerhead sea turtles with dredge and trawl gear where effort in the scallop fishery and sea turtle distribution overlap (Murray 2011; Warden 2011a), (2) information on the interactions of other sea turtle species in dredge and trawl gear in the scallop and other fisheries using similar gear types, (3) life history information for sea turtles and Atlantic sturgeon, and (4) the effects of fishing gear interactions with sea turtles and Atlantic sturgeon that has been published in a number of documents. These sources include status reviews and biological reports (TEWG 2000, 2007, 2009; SEFSC 2001; Stein *et al*. 2004; ASMFC TC 2007; ASSRT 2007; NMFS and USFWS 2007a, 2007b, 2007c, 2007d; Conant *et al*. 2009; NEFSC 2011a; Damon-Randall *et al*. 2012a), recovery plans (NMFS and USFWS 1991, 1992, 2008; NMFS *et al*. 2011), fisheries observer databases (*e.g.*, NEFOP and ASM), and numerous other sources of information from the published literature as cited within this Opinion.

6.1.1 Description of the Gear

The components of a commercial scallop dredge have been described in several documents, which are summarized as follows. The dredge frame keeps the dredge bag spread wide and on the bottom (NEFMC 2003). The cutting bar, which is located on the bottom aft part of the frame, rides about four inches off the seabed (Smolowitz 1998). In a flat area, it remains off the bottom, but in areas of sand waves, for example, the cutting bar hits the top of the sand waves and tends to knock them down (Smolowitz 1998). Shoes on the cutting bar are in contact with and ride along the substrate surface (NREFHSC 2002; NEFMC 2003). A sweep chain in the form of an arc is attached to each shoe and the bottom of the ring bag (Smolowitz 1998). The bag, which drags on the substrate when fished, is made up of metal rings with twine mesh on the top and, sometimes, chafing gear on the bottom (NEFMC 2003). The very end of the ring bag is the club stick, which is responsible for maintaining the shape of the ring bag, especially while dumping the catch on deck (Smolowitz 1998). For scalloping on hard bottoms, rock chains running front to back from the frame to the ring bag, are used in addition to tickler chains, which run from side to side between the frame and the ring bag (Smolowitz 1998). Fishermen use rock chains when fishing on rocky bottoms to prevent boulders from getting into the ring bag, which would cause damage to the gear or to the scallops in the bag (Smolowitz 1998). The number and configuration of rock chains depends on the size of rocks the fishermen wish to exclude, which varies by area (NEFSC pers. comm.) Underwater video of dredges being towed at speeds of five knots show that the chains do not dig into the bottom (Smolowitz 1998). Instead they tend to skip over the bottom, hitting it periodically and bouncing up organisms like starfish that are on the bottom (Smolowitz 1998). Dredges also have a twine top, which allows for reduced bycatch of groundfish and other finfish (NEFMC 2003). A standard 15-foot dredge frame weighs approximately 4,500 pounds (Memo to the File, E. Keane, March 2008). Vessels travel at speeds of 4-5 knots when towing dredge gear (NREFHSC 2002; Murray 2004b, 2005), although the speed of the gear moving through the water column during haulback is usually slower, approximately 1-4 miles per hour (0.9-3.5 knots) (NMFS 2006a).

As described in section 2.1, NMFS has published a final rule that requires federally-permitted scallop vessels fishing with dredge gear to modify their gear by adding a chain mat between the frame and the ring bag when fishing in Mid-Atlantic waters south of 41° 9.0’ N from the shoreline to the outer boundary of the EEZ during the period of May 1 through November 30 each year. Although rock chains and the chain mat are rigged differently, they are both designed to act as a barrier to prevent the capture of objects (rocks or sea turtles respectively) in the ring bag. The chain mat is designed to have more consistently sized openings which, excluding the side created by the sweep, must be 14 inches or less on each side.

The TDD, effective May 1, 2013, via Framework 23, requires the following low-profile design:

(1) The cutting bar must be located in front of the depressor plate.
(2) The angle between the front edge of the cutting bar and the top of the dredge frame must be less than or equal to 45 degrees.
(3) All bale bars must be removed, except the outer bale (single or double) bars and the center support beam, leaving an otherwise unobstructed space between the cutting bar and forward bale wheels, if present. The center support beam must be less than 6 inches wide. For the purpose of flaring and safe handling of the dredge, a minor appendage, not to exceed 12 inches in length, may be attached to the outer bale bar.
(4) Struts must be spaced no more than 12 inches apart from each other.
(5) For all dredges with widths of 10 feet, 6 inches or greater, the TDD must include a straight extension (“bump out”) connecting the outer bale bars to the dredge frame. This “bump out” must exceed 12 inches in length.

The characteristics of trawl gear vary based on the species targeted. An overview of bottom otter trawl gear and the components of the gear, in general, is provided in the Supplemental Environmental Impact Statement for Amendment 10 to the Scallop FMP (NEFMC 2003). Briefly, bottom otter trawls are comprised of a net to catch the target species (NEFMC 2003).

Doors attached to two cables are used to keep the mouth of the net open while deployed. A sweep runs along the bottom of the net mouth (NEFMC 2003). Depending on the bottom type and species targeted, the sweep may be configured with chains, "cookies" (small rubber disks), or larger rubber disks (rock-hoppers or roller gear) that help to prevent the net from snagging on bottom that contains rocks or other structures (NREFHSC 2002; NEFMC 2003). A scallop trawl is a type of bottom otter trawl that is modified to catch scallops (Murray 2007). Scallop trawls differ from the general bottom otter trawl in that scallop trawls generally have no overhang in the net (the floatline, or headline, and the groundrope at the opening of the net are parallel to each other), and the doors are closer to the wings of the trawl (H. Milliken, NEFSC, pers. comm. in Murray 2007). Tickler chains are sometimes used ahead of the trawl to help move scallops off of the sea bed (NEFMC 2003; Murray 2007). NMFS is considering additional bycatch reduction measures in Atlantic trawl fisheries (74 FR 21627, May 8, 2009).

6.1.2 Factors Affecting Sea Turtle Interactions with Scallop Fishing Gear

As described in section 3.1, the occurrence of loggerhead, leatherback, Kemp's ridley, and green sea turtles in New England and Mid-Atlantic waters is primarily temperature dependent (Keinath *et al*. 1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998, 2005; Mitchell *et al*. 2003; Braun-McNeill and Epperly 2004; James *et al*. 2005a; Braun-McNeill *et al*. 2008). 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, 2005; Mitchell *et al*. 2003; Braun-McNeill and Epperly 2004; James *et al*. 2005a; Braun-McNeill *et al*. 2008). 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, 2005; Mitchell *et al*. 2003; Braun-McNeill and Epperly 2004; James *et al*. 2005a). Recreational anglers have reported sightings of sea turtles in inshore waters (bays, inlets, rivers, or sounds) 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 the southern Mid-Atlantic (Virginia and North Carolina) from May through November (Mansfield *et al*. 2009) and in inshore, nearshore, and offshore waters of the northern Mid-Atlantic (New York and New Jersey) 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 waters generally south of Cape Cod (Morreale and Standora 1998). Leatherback sea turtles have a similar seasonal distribution, but have a more extensive range in the Gulf of Maine compared to the hard-shelled sea turtle 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 revealed that loggerheads were observed at the surface in waters from the beach to waters with bottom depths of up to 4,481 meters (CeTAP 1982). However, they were generally found in waters where bottom depths ranged from 22-49 meters deep (the median value was 36.6 meters; Shoop and Kenney 1992). Leatherbacks were sighted at the surface in waters with bottom depths ranging from 1-4,151 meters deep (Shoop and Kenney 1992). However, 84.4% of leatherback sightings occurred in waters where the bottom depth was less than 180 meters (Shoop and Kenney 1992), whereas 84.5% of loggerhead sightings occurred in waters where the bottom depth was less than 80 meters (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 and identifying these smaller sea turtle species (CeTAP 1982).

In the summer of 2010, as part of the AMAPPS project, the NEFSC and SEFSC estimated the abundance of juvenile and adult loggerhead sea turtles in the portion of the northwestern Atlantic continental shelf between Cape Canaveral, Florida and the mouth of the Gulf of St. Lawrence, Canada. The abundance estimates were based on data collected from an aerial line-transect sighting survey as well as satellite tagged loggerheads. The preliminary regional abundance estimate was about 588,000 individuals (approximate inter-quartile range of 382,000-817,000) based on only the positively identified loggerhead sightings, and about 801,000 individuals (approximate inter-quartile range of 521,000-1,111,000) when based on the positively identified loggerheads and a portion of the unidentified sea turtle sightings (NEFSC 2011b). The satellite tracks of loggerheads studied as part of the AMAPPS program can be found at http://www.seaturtle.org/tracking/?project_id=537&dyn=1324309895 (accessed July 6, 2012). Satellite tag locations of approximately 40 loggerheads tagged by Coonamessett Farm (through the Scallop RSA program) and the AMAPPS project from 2009-2011 can be found in Figures 37-40 of the Environmental Assessment (EA) for Framework 23 (NEFMC 2011b). In addition to data on observed fishery interactions, the results from these satellite tagging studies were used by NMFS in the development of regulations regarding the times and areas where TDDs will be required.

Starting in 2007, Coonamessett Farm also began a series of research projects to assess and implement the use of a remotely operated vehicle (ROV) to observe sea turtles in the water column and on the sea floor in the Mid-Atlantic. The ROV studies focused on the scallop grounds with water depths of 40-80 meters during the months of June (2008, 2009), July (2009), August, (2008) and September (2007, 2009) (Smolowitz and Weeks 2009, 2010; Weeks *et al*. 2010). In addition to the ROV, visual observation and recordings from the masthead were obtained. In 2007, no sea turtles were recorded on video using the ROV cameras. Subsequent to that trip, the ROV and techniques were refined for future studies. During the subsequent studies, over 50 sea turtles were tracked by ROV for periods ranging from two minutes to over eight hours (Smolowitz and Weeks 2009; Weeks *et al*. 2010). A range of loggerhead behaviors were observed, including feeding, diving, swimming, and social behaviors. Loggerheads were observed feeding on jellyfish within the top ten meters of the surface and on crabs and scallops on the ocean bottom (Smolowitz and Weeks 2009; Weeks *et al*. 2010). A number of sea turtles were recorded on the ocean bottom at depths of 49-70 meters, and water temperatures of 7.5°-11.5°C (Smolowitz and Weeks 2009, 2010; Weeks *et al*. 2010). Bottom times in excess of 30 minutes were recorded (Weeks *et al*. 2010). Diving and surface behaviors were also documented and are described in detail by Smolowitz and Weeks (2009, 2010) and Weeks *et al*. (2010). In addition, these reports detail social behaviors of sea turtles that were observed.

We have also considered other factors that might affect the likelihood that ESA-listed sea turtles will be incidentally taken in scallop fishing gear. These other factors include the behavior of the animals in the presence of fishing gear, as well as the effect of certain oceanographic features and fishery practices on population distributions and abundances. For example, video footage recorded by the SEFSC's Pascagoula Laboratory showed that loggerhead sea turtles will keep swimming in front of an advancing shrimp trawl, rather than deviating to the side, until the turtles become fatigued and are caught by the trawl or the trawl is hauled up (NMFS 2002a). However, it was later determined that the data collected by the SEFSC were inconclusive and that sometimes sea turtles remained on the bottom, while others shot to the top with bottom disturbance from trawl gear (J. Mitchell pers. comm. in DeAlteris 2010). There was also additional discussion about whether sea turtle behavior in front of approaching trawl gear was more indicative of how long it had been since the turtle had last surfaced for air.

Intensity of biological activity in the Gulf of Maine has been associated with oceanographic fronts, including nutrient fluxes and biological productivity. Particular oceanographic features and processes that influence biological activity are vertical mixing by tides; the seasonal cycle of heating and cooling that leads to winter convection and vertical stratification in summer; pressure gradients from density contrasts set up by deep water inflows and lower salinity waters; and influxes of the cold, but fresher waters associated with Scotian Shelf Water (Townsend *et al.* 2006). Such oceanographic features occurring in the same area as the operation of scallop gear may increase the risk of interactions between scallop gear and ESA-listed sea turtles that would be attracted to these areas for feeding. However, at present there is no information to clearly indicate any of these as influencing ESA-listed sea turtle interactions with scallop fishing gear.

Given the seasonal distribution of sea turtles and the times and areas when the scallop fishery operates, all four species of sea turtles are likely to overlap with operation of the fishery from May through November in Mid-Atlantic waters and along the southern edge of Georges Bank. Based on the best, currently available information, sea turtle interactions with scallop gear are likely at times when and in areas where their distribution overlaps with operation of the fishery.

6.1.3 Description of Sea Turtles Interacting with Scallop Fishing Gear

Sea turtles incidentally captured or entangled in fishing gear must be reported to NMFS on VTRs that are required for the scallop fishery and other Federal fisheries. At present, compliance with the requirement for federally permitted fishermen to report sea turtle interactions on their VTRs is believed to be very low (as evidenced by the lack of reported interactions that have been documented on vessels with observers in recent years). Without reliable VTR reporting of sea turtle interactions, we are using information collected through the NEFOP and ASM Programs, which are managed through the NEFSC FSB. Both of these programs collect, process, and manage data and biological samples obtained by trained observers during commercial fishing trips throughout the New England and the Mid-Atlantic regions. Target observer coverage rates for the scallop fishery in FY 2011 ranged from 3% to 13%, depending upon the month, area (access or open), vessel permit category, and available industry funding (NEFSC 2012).

The discussion of sea turtle interactions with scallop fishing gear that follows will focus on dredge and trawl gear. Past observed interactions of sea turtles in dredge and trawl gear were reviewed in the 2008 Opinion for the scallop fishery. Updated information is provided herein. Important to note is that the reported interactions are likely a fraction of the total amount occurring, which is unknown. However, in the case of loggerhead sea turtles, there are annual estimates of bycatch available for both the scallop dredge and trawl fisheries in the Mid-Atlantic (Murray 2011; Warden 2011a). These analyses only encompass the Mid-Atlantic because there were only two observed Kemp's ridley interactions with the scallop dredge fishery and only a single observed loggerhead interaction with the scallop trawl fishery that occurred in the Gulf of Maine/Georges Bank area during the time periods chosen for both analyses. With only three records outside the Mid-Atlantic, too little information was available to support a robust model-based analysis for the entire action area. Similarly, too few interactions were observed with non-loggerhead sea turtle species throughout both the Gulf of Maine/Georges Bank and Mid-Atlantic to support bycatch estimates for those species in scallop trawl gear (Warden 2011b), although Murray (2011) does provide an estimate of unidentified hard-shelled sea turtle interactions (which includes Kemp's ridley and green sea turtles, in addition to loggerheads) in the scallop dredge fishery. In regards to bottom trawl fisheries for fish species[12], there have been three observed sea turtle interactions with bottom trawls (fish) on Georges Bank. This includes two loggerheads (2005 and 2009) and one unidentified turtle (2008). However, these records are also too few to support a trawl bycatch estimate for any sea turtle species north of the Mid-Atlantic.

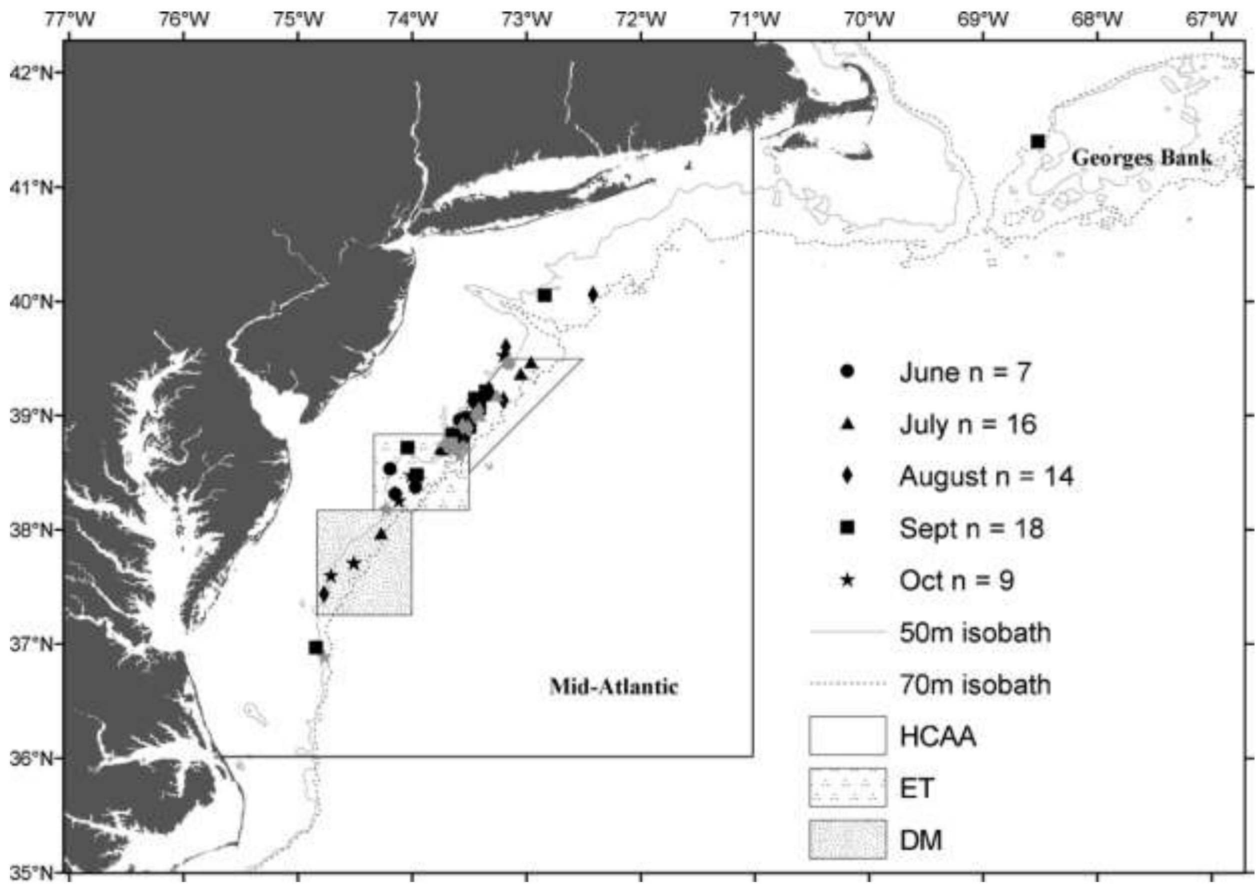
The majority of interactions between sea turtles and fisheries off the U.S. Atlantic coast have occurred south of the New England region since the distribution of sea turtles correlates with warmer water temperatures, resulting in greater densities of sea turtles south of Cape Cod. The spatial distribution of sea turtles in southern New England and the Mid-Atlantic is coincident with several fisheries which may either target or incidentally land scallops. As indicated above, the vast majority of sea turtle interactions with the scallop fishery (both dredge and trawl components) involve loggerheads (Haas *et al*. 2008; Murray 2011; Warden 2011a).

The first NMFS-approved observer records of sea turtle captures in scallop dredge gear occurred in 1996 (loggerhead) and 1997 (green) (71 FR 50361, August 25, 2006). The most recent NMFS-approved observer record of a sea turtle interaction with scallop dredge gear was of a loggerhead in DMV in December 2011 (Appendix A). Although NMFS-approved observers have observed some portion of scallop dredge trips taken in every month in recent years, sea turtles interactions with scallop dredge gear have primarily been observed in the months of June through October (with the exception of the unusual December interaction noted above). This is consistent with the time of year when sea turtles are present in the action area. In terms of depth and distribution, observed sea turtle interactions in the scallop dredge fishery have primarily occurred in waters between 40-80 meters deep off the coast from New Jersey to Virginia (NEFMC 2011b).

[12] NEFOP breaks down bottom trawl fisheries into bottom trawl (fish) and bottom trawl (scallops)

The following paragraphs are a summary of findings from Murray (2011), the most recent peer-reviewed analysis of interactions between sea turtles and the scallop dredge fishery. From 2001-2008, a total of 64 sea turtle captures in scallop dredge gear were reported by NMFS trained observers that were "on-watch" (Murray 2011; Figure 4). In addition, 15 sea turtle interactions occurred on hauls when an observer was "off-watch" and were excluded from the rate analysis in Murray (2011). Lastly, eight severely decomposed sea turtles were caught in scallop dredge gear from 2001-2008, though these sea turtles were also excluded from Murray's (2011) analysis because the state of decomposition suggested they died prior to interacting with the gear. Sea turtle interactions with the scallop dredge fishery as reported by the observers can include: (a) sea turtles that are observed to be captured in the gear (either in the dredge bag or on parts of the dredge frame such as the sweep or chains), (b) sea turtles lying on top of the gear without being physically caught on the gear, and (c) sea turtles observed to swim into the gear or that are bumped by the gear when they are at the water surface (Haas *et al*. 2008; Murray 2011).

Figure 4. Distribution of observed sea turtle interactions with scallop dredge gear in the Mid-Atlantic during on-watch hauls from 2001-2008. Unidentified sea turtles are in gray and the sea turtle outside of the study area is a Kemp's ridley. Taken from Murray (2011).



The majority of sea turtles observed to be captured in the scallop fishery are loggerheads. Of the 48 sea turtles identified to species during "on-watch" tows between 2001 and 2008, 47 were loggerheads and one was a Kemp's ridley. Sixteen were unidentified to species. "Off-watch" observed sea turtles included nine loggerheads, one Kemp's ridley, and 5 unidentified sea turtles (Murray 2011). Additional training of observers since 2001 has greatly reduced the number of sea turtles that are not identified to species by observers. However, unknowns are still likely to be reported because the observer does not always have the opportunity to identify the sea turtle to species (*e.g.*, when a sea turtle drops or swims out of the gear before the dredge can be brought on deck). Unidentified sea turtles described in Murray (2011) are assumed to be either loggerheads, Kemp's ridleys, or greens as these species are very similar looking, whereas leatherbacks are more distinctive and can in most cases be accurately identified by an observer.

During 2001-2008, 88% (n=49) of observed loggerheads interacting with dredge gear during on and off-watch hauls were alive (with or without injuries), and 12% (n=7) were dead. One Kemp's ridley was alive and the other was dead. All of the unidentified species were alive. Seventy-eight percent (n=18) of the benthic immature loggerheads were alive, and 100% of the adults were alive (Murray 2011).

In regards to sea turtle captures in scallop dredge gear, Haas *et al*. (2008) described a number of locations that were recorded by fishery observers prior to the requirement of chain mats including: in the dredge (generic), in the bag, on top of the catch, in the sweep or chains, in the frame, atop of the dredge, and other. Out of 74 sea turtle captures recorded from 1996-2005, the most frequent occurred in the dredge (n=27), in the dredge bag (n=11), or on top of the catch (n=7). Only a few sea turtles were reported in the sweep (n=2), in the dredge frame (n=4), or atop of the dredge (n=1). About 75% of the sea turtles were brought aboard the fishing vessel. Of the sea turtles not brought aboard, some were recorded as being bumped by the gear, being in the dredge but swimming out, swimming from the gear while it was being rinsed, being washed off the bail, being atop of the dredge, falling from the sweep area, or falling "from" or "out of" the dredge (Haas *et al*. 2008). Based on this information, predicting where on the dredge the majority of sea turtle interactions will occur has been and will continue to be difficult, especially given recent gear modifications such as chain mats and TDDs that are designed to keep sea turtles from being captured in the gear. However, sea turtles may continue to be captured in dredge gear if the gear is not designed properly, if it malfunctions, or if the sea turtle is small enough such that the gear modifications are not successful in excluding or deflecting the turtle from the dredge.

Loggerhead sea turtles also represent the majority of sea turtles species observed incidentally captured in trawl gear in the action area. Observers reported 112 loggerhead sea turtle interactions with non-TED bottom otter trawl gear fished in the Mid-Atlantic from 1994-2008 (Warden 2011b). Bottom trawls for fish were involved in 99 of the interactions, while bottom trawls for scallops were involved in the other 13. Additional observed sea turtle interactions not included in the Warden (2011b) analysis included one loggerhead outside of the Mid-Atlantic, as well as three Kemp's ridleys, two leatherbacks, and six unidentified sea turtles. Thirteen moderately or severely decomposed carcasses (four loggerheads and nine unidentified) were also excluded as those mortalities were not likely due to the gear interaction.

The estimate of loggerhead sea turtle bycatch in bottom otter trawl gear published in Warden (2011a, 2011b) represents the best available information for and analysis of loggerhead bycatch in Mid-Atlantic trawl fisheries. This estimate is described further in Section 5.2.2. Such estimates for trawl gear are not available for leatherback, Kemp's ridley, and green sea turtles. Therefore, fisheries observer data for these species represent the best available information.

The NEFSC FSB documents the most landed commercial species (by weight) per trip when an interaction occurs (among many other variables), and that information has been used to look at the relative frequency that individual commercial fish species are associated with the incidental bycatch of leatherback, Kemp's ridley, and green sea turtles. From 2001-2010, only one unidentified sea turtle was captured in bottom trawl gear targeting scallops (NEFSC FSB 2011).

While it may be informative to look at the number of leatherback, Kemp's ridley, and green sea turtles observed to have been captured on bottom trawl (scallop) trips or on bottom trawl (fish) trips when the majority of the landings were scallops, using this number as the estimated number of interactions would be an underestimate in two ways. First, sea turtles could have been captured on trips where scallops were part of the catch, but constituted less than the majority of the catch. Second, these captures are only observed captures and we are not currently able to extrapolate this number to generate an estimate of total bycatch. In order to partially compensate for this underestimate, for the purposes of estimating interactions of leatherback, Kemp's ridley, and green sea turtles with fishing gear authorized under the Scallop FMP, we look at interactions by gear type as illustrated in the table below (Table 7).

Observations of sea turtle interactions in bottom trawls indicate that fisheries using this gear type are capable of incidentally capturing sea turtles and that some of these interactions are lethal. Sea turtles have been observed to dive to the bottom and hunker down when alarmed by loud noise or gear (L. Lankshear, Memorandum to the File, December 4, 2007; DeAlteris 2010), which could place them in the path of bottom gear such as a trawl. However, others may instead continue to swim in front of an advancing trawl or swim above it. Benthic immature and adult loggerhead and Kemp's ridley sea turtles are known to feed on benthic organisms such as crabs, whelks, and other invertebrates including bivalves (Keinath *et al.* 1987; Lutcavage and Musick 1985; Dodd 1988; Burke *et al.* 1993, 1994; Morreale and Standora 2005; Seney and Musick 2005, 2007). We anticipate that the same life stages of green sea turtles will interact with trawl gear in the same manner as loggerhead and Kemp's ridley sea turtles (*i.e.*, both on the bottom and in the water column). Therefore, if loggerhead, Kemp's ridley, and green sea turtles are foraging in areas where the scallop fishery operates, the sea turtles would be at risk.

Tagging studies have shown that leatherback sea turtles, which occur seasonally in western North Atlantic continental shelf waters where the scallop fishery operates, stay within the water column rather than near the bottom (James *et al.* 2005a). Given the largely pelagic life history of leatherbacks (Rebel 1974; CeTAP 1982; NMFS and USFWS 1992), and the dive-depth information on leatherback use of western North Atlantic continental shelf waters (James *et al.* 2005a, 2005b), they are likely to spend more time in the water column than on the bottom. Given that leatherbacks forage primarily within the water column rather than on the bottom, interactions between leatherbacks and scallop gear are expected to occur when the gear is traveling through the water column versus on the bottom. Given that leatherback interactions have been observed in gear used or consistent with that used in the scallop fishery (Table 7), as well as known distribution patterns of leatherbacks in the water column along the U.S. Atlantic coast, interactions with leatherbacks are expected to occur in both the dredge and trawl fishery.

Table 7. Documented incidental captures of leatherback, Kemp's ridley, green, and unidentified sea turtles (excluding moderately and severely decomposed sea turtles) in bottom otter trawl gear (BOT: fish and scallops) from 2001-2010. Source: NEFSC FSB (2011).

Species	**Documented # of incidental captures in BOT gear**	**Average # of documented incidental captures/year in BOT gear**
Leatherback sea turtle	3	0.3
Kemp's ridley sea turtle	2	0.2
Green sea turtle	1	0.1
Unidentified sea turtle	6	0.6

6.1.4 Factors Affecting Atlantic Sturgeon Interactions with Scallop Fishing Gear

While in the ocean, Atlantic sturgeon feed primarily on small benthic invertebrates such as mollusks, gastropods, amphipods, annelids, decapods, and isopods. Because of the benthic nature of their inverterate prey, it is likely that feeding Atlantic sturgeon could occur in the path of a dredge or bottom trawl vessel operating in the action area. While migrating, Atlantic sturgeon may be present throughout the water column and could also interact with the gear while it is moving through the water column. However, scallop dredge gear is much more rigid, has a lower profile while on being fished on the ocean bottom, and is hauled up more vertically than trawl gear. As a result, dredge gear does not pose a threat of bycatch to Atlantic sturgeon on the bottom or in the water column as trawl gear. Like sea turtles, Atlantic sturgeon interactions with trawl gear are likely at times when and in areas where their distribution overlaps with operation of the fishery.

6.1.5 Description of Atlantic Sturgeon Interacting with Scallop Fishing Gear

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 waters less than 50 meters deep. Some information suggests that captures in trawl gear are most likely to occur in waters with depths less than 30 meters (ASMFC TC 2007). Atlantic sturgeon captures in Northeast fisheries have been documented and recorded by the NEFOP. We have reviewed the available information and no Atlantic sturgeon have been reported as caught in scallop dredge gear or in trawl gear where the haul target or trip target is scallop. However, given the known capture of Atlantic sturgeon in trawl fisheries operating in the action area (Stein *et al*. 2004; ASMFC TC 2007; NEFSC 2011a), it is reasonable to anticipate that some small level of bycatch may occur in the scallop trawl fishery. Given the way that scallop dredges operate, we believe that the lack of documented interactions is likely reflective of a true lack of captures of Atlantic sturgeon in scallop dredge gear. As described above, we expect that Atlantic sturgeon in the action area will originate from the five DPSs in the following proportions: NYB (46%); SA (29%); CB (16%); GOM (8%), and Carolina (0.5%). It is also possible that a small fraction (<1%) of Atlantic sturgeon in the action area may be of Canadian origin (*i.e.*, from the St. John River).

6.2 Anticipated Effects of the Proposed Action

The proposed action is likely to adversely affect the four species of sea turtles and five DPSs of Atlantic sturgeon whenever they come into physical contact with scallop fishing gear (dredges and trawls for sea turtles; trawls only for Atlantic sturgeon). Interactions with sea turtles, some of which have resulted in serious injuries and mortalities, have occurred in both gear types used in this fishery, while interactions with Atlantic sturgeon have occurred in other fisheries utilizing trawl gear. Other effects to sea turtles and Atlantic sturgeon as a result of the proposed action, including the effects of vessel strikes and impacts on the availability of prey, are expected to be insignificant or discountable.

In this section of the Opinion, we will determine, given the currently available information, the anticipated number of sea turtles and Atlantic sturgeon, by species, that will be adversely affected by the continued operation of the scallop fishery.

6.2.1 Anticipated interactions of sea turtles with scallop gear

As described earlier in this Opinion, the Murray (2011) and Warden (2011a) reports analyze fishery observer data and VTR data from fishermen in order to estimate the average annual number of sea turtle interactions in scallop dredge and trawl gear in the Mid-Atlantic. Unfortunately, due to small sample sizes of observer records, these reports only compute estimates for loggerheads and, in the case of Murray (2011), hard-shelled sea turtles (loggerheads and unidentified hard-shelled sea turtles pooled). For loggerheads, both reports estimate the average number of turtles that interact with or are captured in each gear type annually. These reports on Mid-Atlantic interactions represent the most accurate predictor of annual loggerhead sea turtle interactions in the scallop fishery, as interactions on Georges Bank and in the Gulf of Maine are infrequent and have not been able to be assessed statistically.

Scallop dredge fishery
As described above, no method has yet been identified for comprehensively determining the actual level of sea turtle interactions in the scallop dredge fishery. The extent of loggerhead bycatch has been estimated for some years based on data collected by fishery observers. Based on data collected by observers for reported sea turtle captures in or retention upon scallop dredge gear, the NEFSC estimated loggerhead bycatch in the scallop dredge fishery for 2001, 2002, 2003, 2004, 2005, and for the entire period between 2001 and 2008 (Murray 2004a, 2004b, 2005, 2007, 2011). These estimates were only applicable to portions of the scallop dredge fishery operating in Mid-Atlantic waters in those years. The estimates of both loggerhead and all hard-shelled sea turtle interactions with the scallop dredge fishery as presented in Murray (2011) provide the best available information for determining the anticipated number of loggerhead, Kemp's ridley, and green sea turtle interactions in the dredge fishery. For the purposes of this Opinion, we are using the annual estimate of both loggerhead and hard-shelled sea turtle interactions for the period after chain mats were required through the end of 2008. This method allows us to account for and estimate not only observed interactions, but also unobserved yet quantifiable (*i.e.*, inferred) interactions such as a sea turtle interacting with a chain mat below the water surface and not entering the dredge bag or ending up on deck (Warden and Murray 2011).

As presented in Murray (2011), the Mid-Atlantic scallop dredge fishery was estimated to interact with an average of 125 hard-shelled sea turtles per year, with a 95% CI of 88-163, from September 26, 2006 (the date chain mats were required), through 2008. This estimate includes both observable and unobservable, quantifiable interactions. For loggerhead sea turtles, Murray (2011) estimated an annual average of 95 interactions with a 95% CI of 63-130 over that time period. For the purposes of this Opinion, we are assuming that the upper end of the 95% CI is the best available information for, and most conservative estimate of, the anticipated amount of annual hard-shelled and confirmed loggerhead sea turtle interactions in the dredge component of the fishery. Thus, we anticipate an annual average of 130 interactions that can be confirmed to be loggerheads and an additional 33 interactions of hard-shelled sea turtles for which the species cannot be identified. Of those 33, it is expected that the vast majority of those would be loggerheads as well. Only a few would be Kemp's ridley or green sea turtles, based on: (1) the observer data presented in Murray (2011), (2) additional observer data on sea turtle interactions with the scallop dredge fishery as recorded by the NEFOP, and (3) the fact that loggerheads are by far the most abundant sea turtles in the action area.

Using the "on watch" observer data presented in Murray (2011), only 2% (1 out of 48) of sea turtle interactions in the dredge fishery from 2001-2008 for which the species was able to be confirmed were Kemp's ridleys. Although there were no confirmed green sea turtle interactions in dredge gear from 2001-2008, one occurred back in 1997 and we anticipated one interaction annually in the 2008 Opinion (NMFS 2008a). Green sea turtle interactions in the fishery, although likely to be rare, cannot be ruled out. Based on this information, we assume that of the 33 additional hard-shelled sea turtles interacting with the scallop dredge fishery annually, one will be a Kemp's ridley (2% of 33 is 0.66, which is rounded up to one), one will be a green (based on the 1997 interaction), and the remaining 31 will be loggerheads. Due to the one Kemp's ridley sea turtle observed "off-watch" (Murray 2011), we expect one additional Kemp's ridley to interact with the scallop dredge fishery annually, bringing the total to two.

In summary, we expect the annual interaction rate for loggerheads in scallop dredge gear to be 161 turtles. For the other two hard-shelled species, we expect a total of two Kemp's ridley and one green sea turtle interactions per year. These represent the total numbers of hard-shelled sea turtle interactions we expect to occur annually, with any unobservable, unquantifiable interactions as well as any infrequent interactions that occur outside of the Mid-Atlantic subsumed within the estimate (which is the upper end of the 95% CI rather than the mean). Again, we believe that hard-shelled sea turtle interactions with the scallop dredge fishery on Georges Bank and in the Gulf of Maine will be low and subsumed within the Mid-Atlantic estimate since the distribution of these three species in U.S. waters of the Northwest Atlantic is primarily temperature-dependent and often restricted to waters south of Cape Cod (Shoop and Kenney 1992; Morreale and Standora 1998; Mitchell *et al*. 2003). Hard-shelled sea turtles that make their way this far north will likely be migrating to/from and/or foraging seasonally in the protected waters of Cape Cod Bay (as evidenced by numerous Kemp's ridley and loggerhead as well as occasional green sea turtle strandings due to cold-stunning that are recorded along the bay each fall and winter) rather than occupying deeper, offshore waters of Georges Bank or cooler waters of the Gulf of Maine.

There have been no confirmed interactions between leatherback sea turtles and scallop dredge gear recorded by the NEFOP. Tagging studies have shown that leatherbacks, which forage seasonally in western North Atlantic continental shelf waters where the scallop dredge fishery operates, stay within the water column rather than near the bottom (James *et al*. 2005a). Given the largely pelagic life history of leatherback sea turtles (Rebel 1974; CeTAP 1982; NMFS and USFWS 1992), and the more recent dive-depth information on leatherback use of western North Atlantic continental shelf waters (James *et al*. 2005a, 2005b), it is unlikely that a leatherback would occur on the bottom in the action area. Therefore, leatherback sea turtles are not likely to be struck by or captured in scallop dredge gear when the gear is being towed along the bottom. Based on observations of loggerhead sea turtles captured in scallop dredge gear, we believe some sea turtle interactions with scallop dredge gear occur within the water column (NMFS 2008a). Given the large size of the dredge bag and the presence of leatherback sea turtles in areas where the scallop dredge fishery occurs, we believe that leatherback sea turtles can interact with scallop dredge gear when the gear is in the water column. Based on the lack of observer records, interactions between leatherback sea turtles and any mobile gear operating within the action area, including scallop dredge gear, would be rare. However, given the low level of observer coverage in the scallop dredge fishery as well as the fact that chain mats are designed to prevent large sea turtles like leatherbacks from becoming entrained in the dredge, it is likely that some interactions with leatherback sea turtles have occurred but were not observed. Therefore, we believe that up to one leatherback sea turtle annually will interact with dredge gear operating in the action area for this Opinion. This represents the total number of leatherback interactions we expect to occur annually and not just the number observed.

Scallop trawl fishery

The trawl estimate method in Warden (2011a) assigned trips (and associated bycatch) to multiple FMPs/individual species landed based on the distribution of landings for that trip. For example, trips in a certain time and area using trawls were estimated to have a certain bycatch rate of loggerhead sea turtles (based on the observed interactions). In the estimate, the trip and its associated interactions (calculated using the bycatch rate), were assigned to several fisheries in a ratio that reflected the catch composition of that trip by weight. This method is meant to reflect the multispecies nature of many of the fisheries that operate in the Mid-Atlantic region.

Based on data collected by observers for reported sea turtle captures in bottom otter trawl gear, the NEFSC estimated the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear for trips primarily landing scallops during 2005-2008 as 95 loggerheads with a 95% CI for the four-year annual average of 60-140 (Warden 2011a). This estimate of loggerhead sea turtle bycatch in bottom otter trawl gear provides the best available information for determining the anticipated number of loggerhead sea turtle interactions per year in that component of the fishery. For the purposes of this Opinion, we consider the annual average of 140 loggerheads per year (the upper end of the 95% CI) to be the best available information for the anticipated number of loggerhead sea turtle interactions in the trawl component of the fishery. This represents the total number of loggerhead interactions we are expecting annually in the trawl component of the fishery and not just the number observed.

There are no total bycatch estimates for leatherback, Kemp's ridley, or green sea turtles in trawl gear. The very low number of observed non-loggerhead interactions in trawl gear used in the scallop fishery suggests that interactions with these species within the action area are rare events. However, given the fact that observer coverage in the fishery is low, it is likely that some interactions with non-loggerhead sea turtles have occurred but were not observed or reported. Given effort in the fishery as a whole, and the seasonal overlap in distribution of these species with operation of scallop gear, leatherback, Kemp's ridley, and green sea turtles are likely to interact with trawl gear.

As summarized in Table 7, the annual average number of documented leatherback captures in bottom otter trawl gear in the action area is 0.3. Since the capture of a partial sea turtle is not possible, we anticipate the annual capture of one leatherback sea turtle in bottom otter trawl gear used in the scallop fishery. The annual average number of documented Kemp's ridley captures in bottom otter trawl gear in the action area is 0.2. Adding 0.6 to that to account for the possibility that the unidentified sea turtles captured in trawl gear could all be Kemp's ridleys gives a value of 0.8 captures in trawl gear annually. Again, since the capture of a partial sea turtle is not possible, we anticipate the annual capture of one Kemp's ridley sea turtle in bottom otter trawl gear used in the scallop fishery. The annual average number of documented green sea turtle captures in bottom otter trawl gear in the action area is 0.1. Adding 0.6 to that to account for the possibility that the unidentified sea turtles captured in trawl gear could all be green sea turtles gives a value of 0.7 captures in trawl gear annually. Rounding up, we anticipate the annual capture of one green sea turtle in bottom otter trawl gear used in the scallop fishery.

Summary

Annually, we expect the scallop dredge fishery to interact with 161 loggerheads and the scallop trawl fishery to interact with 140 loggerheads. That amounts to a total of 301 loggerhead interactions with the scallop fishery each year. Adding together the interactions expected annually for both dredge and trawl gear for the other three sea turtle species results in a total of two leatherback sea turtle, three Kemp's ridley, and two green sea turtle interactions in the scallop fishery annually. These estimates of annual sea turtle interactions encompass those expected to occur throughout the entire action area, from the Mid-Atlantic through the Gulf of Maine. However, based on records of interactions over the past decade, the vast majority are expected to occur in the Mid-Atlantic (Murray 2004a, 2004b, 2005, 2007, 2011; Warden 2011).

6.2.1.1 Age classes of sea turtles anticipated to interact with the scallop fishery

Loggerhead sea turtles. The 2008 recovery plan identifies five life stages for loggerhead sea turtles: (1) hatchling: 4 centimeters CCL, 1-5 days; (2) post-hatchling: 4-6 centimeters CCL, <6 months; (3) oceanic juvenile: 8.5-64 centimeters CCL, 7-11.5 years; (4) neritic juvenile: 46-87 centimeters CCL, 13-20 years; and (5) adult male/female: >83 centimeters CCL and >87 centimeters CCL (respectively), >25 years for females (NMFS and USFWS 2008). Both Haas *et al*. (2008) and Murray (2011) presented data on loggerhead sea turtles interacting with scallop fishing gear that we can use to determine estimated sizes of future interactions. The mean CCL of sea turtles incidentally captured in the scallop dredge fishery from 1996-2005 (which included 34 loggerheads, one Kemp's ridley, and one unidentified sea turtle, and excluded moderately and heavily decomposed sea turtles) was 78.1 centimeters (95% CI: 72.9-83.4 centimeters) (Haas *et al*. 2008). Observed loggerheads incidentally captured in the scallop dredge fishery from 2001-2008 ranged from 62 to 107 centimeters CCL (Murray 2011). These ranges correspond to the benthic juvenile and adult life stages. Based on these observer measurements and the known distribution of loggerhead sea turtles captured in other U.S. Atlantic coastal fisheries, we expect that both benthic juvenile and adult loggerheads may be captured in scallop gear as a result of the continued operation of the fishery because both life stages are present within the action area.

Leatherback sea turtles. We believe that leatherback sea turtles may interact with scallop fishing gear given the presence of leatherbacks in areas where the fishery occurs. Sighting and stranding records suggest that both juvenile and adult leatherbacks occur within the action area where the scallop fishery operates (NMFS and USFWS 1992; SEFSC 2001). Satellite-tracking of tagged leatherbacks also demonstrates the movement of sexually mature leatherbacks over U.S. continental shelf waters (James *et al*. 2005a, 2005b). Therefore, both juveniles and adults could interact with scallop gear since both age classes occur in areas where the fishery operates.

Kemp's ridley sea turtles. The post-hatchling stage for Kemp's ridley sea turtles was defined by the TEWG as Kemp's ridleys of 5-20 centimeters standard carapace length (SCL), while turtles 20-60 centimeters SCL were considered to be benthic immature (TEWG 2000). The latter stage is described as sea turtles that have recruited to coastal benthic habitat. Mid-Atlantic and coastal New England waters (as far north as approximately Cape Cod) are known to be developmental foraging habitat for immature Kemp's ridley sea turtles, while adults have been documented from waters and nesting beaches along the South Atlantic coast of the U.S. and in the Gulf of Mexico (Musick and Limpus 1997; TEWG 2000; Morreale and Standora 2005). One Kemp's ridley incidentally captured in scallop dredge gear was 24.3 centimeters CCL while the other was not measured (Haas *et al*. 2008; Murray 2011). Given the life history of the species and the small size of the individual that was incidentally captured in dredge gear, we expect that only immature Kemp's ridley sea turtles are likely to interact with scallop gear as a result of the continued operation of the fishery.

Green sea turtles. Hirth (1997) defined a juvenile green sea turtle as a post-hatchling up to 40 centimeters SCL. A subadult was defined as green sea turtles from 41 centimeters through the onset of sexual maturity (Hirth 1997). Sexual maturity was defined as green sea turtles greater than 70-100 centimeters SCL (Hirth 1997). Like Kemp's ridleys, Mid-Atlantic waters are recognized as developmental habitat for juvenile green sea turtles after they enter the benthic environment (Musick and Limpus 1997; Morreale and Standora 2005). However, nesting individuals are also known to occur and feed in the Mid-Atlantic on occasion. A green sea turtle nest was documented in Delaware in 2011 and nests have also been recorded previously in North Carolina and Virginia (Peterson *et al*. 1985; Hawkes *et al*. 2005). The one green sea turtle captured in the scallop dredge fishery in 1997 had an estimated length of 70 centimeters (Haas *et al*. 2008). Thus, we expect that both benthic immature and sexually mature green sea turtles are likely to interact with scallop fishing gear as a result of the continued operation of the fishery.

6.2.1.2 Estimated mortality of sea turtles captured in scallop fishing gear

Sea turtle interactions with scallop dredge and trawl gear likely result in a higher level of sea turtle mortality than is evident based on the number of sea turtles returned to the water alive. Injuries suffered by sea turtles interacting with scallop fishing gear fall into two main categories: (1) submergence injuries characterized by an absence or obvious reduction in breathing and consciousness with no other apparent injury, and (2) contact injuries resulting from collisions with the gear or entanglement of flippers and/or other body parts in the gear. Contact injuries can be characterized by scrapes to soft tissue, cracks to the carapace and/or plastron, missing or damaged scutes, and/or bleeding from one or more orifice. The following information is provided as an assessment of the extent of these types of injuries likely to occur to sea turtles affected by the continued operation of the scallop fishery. It should be noted that the severity of sea turtle injuries as a result of scallop dredge interactions will be less if the turtle is interacting with a TDD or dredge equipped with chain mats as compared to a standard dredge.

Sea turtles forcibly submerged in any type of restrictive gear 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.

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, however, 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 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 reanalyzed (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).

Tows by scallop dredge vessels are usually around an hour or less, while tows by bottom otter trawl vessels are usually around one to two hours in duration. However, Murray (2008) found that tow times of bottom otter trawl gear that resulted in sea turtle bycatch ranged from 0.5 to over 5 hours. Shortened tow durations in the dredge fishery, which have been used to limit yellowtail flounder bycatch (NEFMC 2011b), should help to reduce the risk of death from forced submergence for sea turtles caught in dredges (primarily those without chain mats), but they do not eliminate the risk. For the trawl fishery, assuming that the mortality rate for sea turtles from forced submergence in scallop trawl gear is comparable to that measured for the shrimp fishery by Epperly *et al*. (2002) and Sasso and Epperly (2006), sea turtles may die as a result of capture and forced submergence in trawl gear used in the scallop fishery, especially if they are caught at the beginning of long tows.

Contact injuries involving damage to the carapace and/or plastron of sea turtles have been frequently observed in the scallop dredge fishery, most often in the case of dredges not equipped with chain mats. However, fishery observers often cannot assess whether dredge-related injuries occurred on the bottom, in the water column, or on the deck of the vessel; they can only determine whether injuries occurred before or after the turtle was brought aboard the vessel (Haas *et al*. 2008). As stated in section 5.1 above, no underwater interactions of living sea turtles with scallop dredge gear have been observed or photographed; although studies by Milliken *et al*. (2007) and Smolowitz *et al*. (2010) used video monitoring of sea turtle carcasses to assess the effects of a TDD on sea turtles. Given the current knowledge of sea turtle life history, the condition of sea turtles captured in or upon dredge gear as described by observers (Haas *et al*. 2008; Murray 2011), and an understanding of the gear and how it is fished, there are several ways that a sea turtle might suffer injuries during interactions with dredge gear. Scallop dredge gear is heavy and fishes with part of the gear in contact with the bottom. Mid-Atlantic waters are known to be foraging areas for sea turtles in the spring through fall (Keinath *et al.* 1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998; Braun-McNeill and Epperly 2004; James *et al*. 2005b; Morreale and Standora 2005). Loggerhead and Kemp's ridley sea turtles are known to feed on benthic organisms such as crabs, whelks, and other invertebrates including bivalves and scallops (Keinath *et al*. 1987; Lutcavage and Musick 1985; Dodd 1988; Burke *et al*. 1993; Burke *et al*. 1994; Morreale and Standora 2005; Seney and Musick 2005; Smolowitz and Weeks 2009; Weeks *et al*. 2010), while green sea turtles are known to feed on seagrasses and benthic algae. The scallop dredge fishery is known to capture crabs, whelks, and other organisms as bycatch while catching scallops on the ocean bottom (NEFMC 2003). Therefore, if loggerhead, Kemp's ridley, and green sea turtles are foraging in areas where scallop dredging occurs, they will likely be spending some of their time on or near the bottom where they would be at risk of being struck or captured by scallop dredge gear.

Given that the cutting bar of a standard dredge rides only a few inches off the seabed (Smolowitz 1998), and the gear weighs approximately 4,500 pounds (Memo to the File, E. Keane, March 2008), it is reasonable to believe that a sea turtle struck by a dredge on or very near the bottom would suffer cracks to the shell (carapace and/or plastron) as a result of being struck by the dredge and passing under the gear that is forward of the dredge bag opening before passing into the dredge bag. If a sea turtle enters the dredge bag, it may be injured by large rocks that are also caught in the dredge bag. It is reasonable to believe that sea turtles caught in scallop dredge gear may also be injured during one or more steps that are necessary to empty the dredge bag. Under typical fishing operations, the dredge is hauled to the surface at the end of each tow alongside the vessel, lifted above the deck of the vessel and emptied by turning the bag over. After the bag is dumped, the dredge frame is often dropped on top of the catch. Contact between the dredge bag and the side of the vessel as the bag is hauled out of the water, as well as the dumping of the catch and the sudden lowering of the gear onto the deck are times when sea turtles captured in or upon the gear could reasonably be injured as a result of hitting against the side of the vessel, falling onto the deck, or being hit by the dredge contents and/or the dredge itself. Again, it is expected that most of these injuries, with a few exceptions, will occur due to interactions with non-TDD, non-chain mat equipped dredges.

Some observers have reported sea turtles that are found within the dredge bag upon hauling of the gear that have no apparent injuries. Given the weight of the dredge frame, the presence of the cutting bar forward of the dredge opening, and the typical shallow height of the cutting bar above the seabed while the dredge is fished, it seems improbable that a sea turtle on or very near the bottom in the path of the dredge could be passed over by the dredge frame and cutting bar, swept into the dredge bag, tumbled around or hit by debris inside the dredge bag as the gear is towed on the bottom, and not suffer any apparent injury. However, during haulback of the dredge, it is possible that a sea turtle in the water column could pass into the dredge bag with little or no contact with the cutting bar and the dredge frame in front of the opening to the dredge bag. Thus, the sea turtle would have no observable severe injuries (*i.e.*, cracks to the carapace and/or plastron) upon hauling of the dredge. For these reasons, we believe that some sea turtles may interact with or be captured in non-chain mat equipped dredge gear when the dredge is in the water column. In regards to leatherback sea turtles, all dredge interactions are expected to occur in the water column and are expected to be non-lethal for those equipped with chain mats.

As described in section 2.1, NMFS requires scallop dredge gear to be equipped with chain mats when fished in Mid-Atlantic waters south of 41° 9.0'N latitude from the shoreline to the outer boundary of the EEZ during the period of May 1 through November 30 each year. NMFS will also require all limited access and certain LAGC scallop vessels to utilize a TDD in Mid-Atlantic waters west of 71° W longitude from May through October starting May 1, 2013. The effects of the proposed action (the continued operation of the scallop fishery) include the effects of the fishery using both chain mats and TDDs (once they are required). Since sea turtles, no matter how initially captured, can suffer injuries following capture in or upon the dredge (*e.g.*, from being tumbled around or hit by debris in the dredge while the gear is fishing on the bottom, from the dredge hitting into the side of the vessel during haulback, or from falling and crushing injuries suffered during emptying of the dredge bag on deck), keeping sea turtles from going underneath the dredge and keeping them out of the dredge bag is expected to reduce the severity of some interactions that occur.

Installing a chain mat over the opening of the dredge bag and/or utilization of a TDD will not increase or decrease the number of sea turtles that will come into contact with dredge gear used in the fishery. The chain mat simply prevents a sea turtle encountering the gear from entering the dredge bag where it would be at further risk of injury, while a TDD is designed to deflect sea turtles over the dredge rather than underneath it. In 2008, the TDD was evaluated in Cape Cod Bay, Massachusetts. Seven frozen sea turtle carcasses were placed in the path of the modified dredge, interactions were videoed, and five recovered carcasses were evaluated for injuries. The only observed damage to the carcasses were superficial scratches and chips, and in the nine video recorded interactions, all carcasses hit the dredge at some point and passed over the dredge frame (Smolowitz *et al*. 2010). In a TDD, the placement of the cutting bar forward of the dredge frame allows a sea turtle to be directed up and over dredge. In a standard dredge, the cutting bar is behind and under the depressor plate, preventing a sea turtle from rising above the dredge. Sea turtles are also not expected to suffer injuries as a result of swimming into or being hit by the chain mat, only, during a water column interaction. During haulback, a dredge travels through the water column at speeds of one to four miles per hour. Sea turtles that are struck by the chain mat portion of the dredge during haulback are not expected to sustain serious injury leading to death, given the slow speed of the vessel during haulback (NMFS 2008a) and given that contact is made in the water column (a fluid environment) rather than against the bottom.

Although many sea turtles caught in or retained upon scallop dredge gear have some type of obvious injury when first observed, regulations require that fishermen return all sea turtles (regardless of the level of injury) to the water as soon as possible unless they require resuscitation.

Serious injury/mortality calculation - dredge gear

Based on the descriptions provided by fisheries observers, it seems probable that some injured sea turtles observed captured in commercial fishing gear and that were returned to the water alive would have subsequently died as a result of those injuries. In 2004, we developed and defined three categories for making serious injury determinations for sea turtles captured in scallop dredge gear (Memorandum from Mary Colligan to Patricia A. Kurkul, September 23, 2004). These categories were based on the advice of a panel of experts with experience in the treatment and care of sea turtles after their review of information on the types of injuries that NMFS-trained observers documented on sea turtle interactions with scallop dredge gear. To more fully assess the effects of the scallop fishery on sea turtles, the final working guidance also assigned a rate of survival for Category II injuries as 50%. In the 2008 Opinion, we assigned a 0% chance of survival to Category I injuries, and a 100% chance of survival for Category III injuries (NMFS 2008a). Based on the final working guidance and the information obtained from observer reports of loggerhead sea turtles captured in scallop dredge gear during the 2003 scallop fishing year, we determined that the sea turtle mortality rate for the scallop dredge fishery was 64% (NMFS 2004b).

For other gear types, in previous Opinions, we used the number of dead loggerhead sea turtles documented by the NEFOP and reported in the bycatch estimates (Murray 2008, 2009a) to estimate the number of loggerheads that survive interactions with bottom otter trawl gear. While the best available information at the time, it became apparent that injury criteria (like developed for scallop dredge gear) should be relevant to all other fishing gear and sea turtle injury types. We recognized the need to expand guidance developed for the scallop dredge fishery to attempt to encompass other Northeast Region gear types (*e.g.*, trawl) and a wide range of sea turtle injuries, and to use a consistent approach for assessing post-release survival.

In November 2009, NMFS NERO and NEFSC hosted a workshop to discuss sea turtle injuries in Northeast Region fishing gear and associated post-release survival. The workshop convened various experts in sea turtle veterinary medicine, health assessment, anatomy, and/or rehabilitation. The information gathered by individual participants at this workshop was then used by NMFS to develop technical guidelines for assessing sea turtle injuries in Northeast fishing gear (Upite 2011). The Technical Guidelines consist of a variety of injury descriptions that may be found from sea turtles captured in fishing gear, organized by those injuries with a resulting low probability of mortality (Category I), an intermediate probability of mortality (Category II), and a high probability of mortality (Category III). Animals exhibiting the injuries found in Category I were considered to have a 20% probability of post-release mortality based upon their capture condition and assessment, animals with injury descriptions in Category II had a 50% probability of post-release mortality, and animals with the injuries listed in Category III had a 80% probability of post-release mortality. Turtles believed to be dead or released into the water in an unresponsive state were given a 100% mortality rate. These injury percentages were based upon discussions at the workshop and expert opinion. Based upon the best available information, we believe that the Technical Guidelines are reasonable measures of what to expect for sea turtles captured by fishing gear and associated post-release survival.

After the workshop report was published, the NMFS Northeast sea turtle injury workgroup developed a plan to implement the Technical Guidelines and review observer records to assess post-release survival. The scope of the review was determined to be five years (2006 to 2010), for a resulting total of 145 observer records. The workgroup members reviewed each observer record and first determined if the injury was a result of the fishery interaction (haul/set/tow), interpreted as a "fresh" injury, using the guidance in Upite (2011) and expert opinion. If fresh, then the members used the Technical Guidelines to place the turtle into one of the three categories with the identified post-release mortality rates, or provided justification for a 100% mortality determination.

After the determinations were finalized, the records were separated by gear type. Based upon the percent probability of mortality and numbers of turtles in each category (of the Technical Guidelines), turtle mortalities were calculated for each category by each gear type. The number of dead turtles was then combined to obtain an overall mortality number by gear type, and the mortality percentage (number of dead turtles/number of total observations) was calculated.

The majority of the observed fishery interactions from 2006 to 2010 involved loggerheads. For non-loggerheads, the sample size would be too small to develop valid mortality rates for each species by gear type. The decision was made to combine all species in order to develop one mortality rate by gear type. Further, the associated mortality rates (20%, 50%, 80%) for the three categories factor in any potential variations in species differences. The Technical Guidelines and resulting mortality percentages apply to all sea turtle species.

After the review of observer records, the Northeast sea turtle injury workgroup calculated a resulting mortality rate for scallop dredge gear of 80% (11 records reviewed; C. Upite, Memorandum to the File, March 28, 2012). The time period of review was from 2006-2010. Chain mats were required in this fishery on September 25, 2006 (71 FR 50361). Besides one loggerhead capture which was before the September 2006 requirement and one Kemp's ridley capture which was north of 41° 9' N, chain mats were used on all of the dredges with observed sea turtle interactions. However, it should be noted that in several instances, the chain mats were improperly configured.

The post-release mortality rate of 80% in scallop dredge gear is higher than the previous percentage (64%) used in the 2008 Opinion. The new rate uses more comprehensive and updated injury guidelines and considers more recent and a longer time series of take information, which may better reflect the current fishery. The mortality rate of 80% for scallop dredge gear represents the best available information, pre-TDD. Thus, for the 2012 fishing year (before the TDD regulations are fully in effect), 129 of the 161 loggerheads interactions with scallop dredge gear are anticipated to result in serious injury/mortality. However, the Mid-Atlantic effort reduction measures still in place through Framework 22 may help to offset the effect of the 80% mortality rate in FY 2012 (NEFMC 2011a).

In conjunction with the NEFMC, we have implemented a requirement that all limited access vessels (regardless of permit category or dredge size), and limited access general category vessels that fish with a dredge with a width of 10.5 feet or greater, use a TDD in the Mid-Atlantic (west of 71° W longitude) from May 1 through October 31. Observations of interactions between sea turtle carcasses and the TDD suggest that the serious injury rate of the TDD is much lower than a traditional dredge (Smolowitz *et al*. 2010). Smolowitz *et al*. (2010) observed nine interactions between a loggerhead carcass and a TDD, and in all cases the carcasses hit the dredge at some point and passed over the dredge frame. Assuming a binomial probability distribution, in nine trials it was concluded with 95% confidence that a minimum of 72% of sea turtles interacting with a TDD will go over the dredge and a maximum of 28% will go under the dredge. If all sea turtles that are deflected over the dredge do not sustain serious injuries (0% serious injury rate), and if all sea turtles that go under the dredge have serious injuries (100% serious injury rate), then the maximum serious injury rate for sea turtles interacting with a TDD would be 28%. It is reasonable to assume that all sea turtles sent over the dredge will not sustain serious injuries as scallop dredges generally move slowly along the bottom and likely would not cause serious trauma or dragging/crushing injuries during interactions in which a turtle is deflected upward. In the Smolowitz *et al*. (2010) study, none of the damage observed on the recovered carcasses that went over the dredge was consistent with categories of injury indicating low or medium chances of survival (see NMFS 2004b). Plus, there is no evidence to suggest that live turtle interactions with the TDD would be more severe than indicated by the damage observed to the recovered carcasses. Smolowitz *et al*. (2010) indicate that using sea turtle carcasses may represent the worst-case scenario on effects of the TDD on sea turtles because live sea turtles could exhibit escape behavior and may be structurally stronger than a decomposing carcass. Finally, the TDD eliminates a number of sources of potential entrapment at the front and on top of the dredge frame (*e.g.*, sloping face of the forward cutting bar, reduced number of bale support bars, reduced spacing of struts). Based on this theoretical injury rate, we will assume in this Opinion that up to 28% of sea turtles interacting with a TDD will experience a serious injury/mortality while a minimum of 72% will survive.

The use of chain mats and TDDs are not expected to reduce the number of sea turtles that come into contact with scallop dredge gear, but are anticipated to reduce the likelihood of serious injury or mortality from interactions. However, in the 2008 Opinion, we stated that we could not quantify the reduction in mortality rate from chain mats. At that time, the 64% mortality rate remained the best available information for defining the number of sea turtle interactions with scallop dredge gear (with chain mats) that are likely to result in death. In the EA that evaluated the TDD measures, the combined benefit of chain mats and TDDs was estimated, because both measures will soon be in effect, not just the TDD without chain mats. In addition, since the conservation benefit of chain mats was not previously quantified, it is appropriate to compare the combined benefits (TDD and chain mats) to the standard dredge (no TDD and no chain mats). It was estimated that the TDD dredge with chain mats has a maximum estimated serious injury rate of 28% (Smolowitz *et al*. 2010; NEFMC 2011b). The 28% rate will be subsequently used as the post-release mortality rate from scallop dredge gear, for vessels using both a chain mat and TDD.

It should be noted that the area and seasonality of the chain mat and TDD requirements are different. Chain mats are required south of 41° 9' N latitude from May 1-November 30, and TDDs will be required west of 71° W longitude from May 1-October 31. As such, there is an area east of 71° W and south of 41° 9' N that only has a chain mat requirement. Also, for the month of November, only chain mats are required (south of 41° 9' N). Given that only chain mats would be used, the injury rate in the aforementioned area and in the month of November may be higher than the calculated 28%, although a separate injury rate for just chain mats has not been quantified. As a result, applying the pre-TDD implementation injury rate of 80% to all anticipated sea turtle interactions outside the Mid-Atlantic as well as those in the Mid-Atlantic in November is appropriate and should be considered the best available information to apply to the anticipated interactions. However, since very few sea turtle interactions have previously occurred either in November or in waters outside the Mid-Atlantic (Haas *et al*. 2008; Murray 2011), we believe that 46 (28%) of the 161 annual loggerheads interactions with scallop dredge gear for the 2013 fishing year and beyond are anticipated to result in serious injury/mortality.

As the serious injury/mortality rates for dredge gear can also be applied to the other three sea turtle species, it is anticipated that the all of the leatherback, Kemp's ridley, and green sea turtle interactions with scallop dredge gear in 2012 may result in serious injury/mortality (leatherbacks and greens: 1 x 80% = 0.80, which is rounded up to one; Kemp's ridleys: 2 x 80% = 1.6, which is rounded up to two). However, from 2013 on, only one of the two Kemp's ridley interactions annually may result in serious injury/mortality (2 x 28% = 0.56, which is rounded up to one).

<u>Serious injury/mortality calculation - trawl gear</u>

The 2009 workshop and resulting Technical Guidelines apply to other Northeast Region fishing gears besides just scallop dredge gear. The same approach outlined above for scallop dredge gear was taken to review and determine the injury rate for trawl gear. After the review of observer records from 2006-2010, the Northeast sea turtle injury workgroup calculated a resulting mortality rate for trawl gear of 47% (97 records reviewed; C. Upite, Memorandum to the File, March 28, 2012). Thus, of the 140 loggerhead interactions expected to occur annually in the scallop trawl fishery, 66 of those are expected to result in serious injury/mortality. As the serious injury/mortality rate for trawls can also be applied to the other three sea turtle species, it is anticipated that the one leatherback, one Kemp's ridley, and one green sea turtle interaction annually with scallop trawl gear may also result in serious injury/mortality.

6.2.2 Anticipated interactions of Atlantic sturgeon with scallop gear

As noted above, we have reviewed incidental bycatch data of Atlantic sturgeon recorded by the NEFOP and there have been no observed captures in scallop dredge gear. Further, Stein *et al*.

(2004) and ASMFC TC (2007) do not include this gear type in their analyses of gears fished in the Northeast that are likely to result in the capture of Atlantic sturgeon. As such, we do not anticipate any future interactions between Atlantic sturgeon and scallop dredge gear.

The capture of Atlantic sturgeon in bottom otter trawls used in commercial fisheries of New England and the Mid-Atlantic is well documented (Stein *et al.* 2004; ASMFC TC 2007). But, as noted above, we have reviewed the NEFOP data and there are no observed captures of Atlantic sturgeon in otter trawls where the trip target or haul target was recorded as scallops. In the Atlantic sturgeon bycatch report prepared by the NEFSC (2011a), Table 10 gives FMP weights (as a percentage of total landings) for total estimated captures (derived from the model-based estimator) of Atlantic sturgeon in otter trawl gear. For scallops, a weight of 0.013 is given based on 2006-2010 observer data. This equates to a total of about 20 estimated captures per year in the trawl fishery for scallops. However, based on the report's indication that "partitioning of discard encounters to FMPs is not a particularly informative exercise because of the high likelihood of inappropriately assigning associations/responsibilities" (NEFSC 2011a), we believe that reliance on this weight to estimate future interactions associated with the Scallop FMP is not advisable. Oftentimes scallops are landed as bycatch in other trawl fisheries that are known to incidentally capture Atlantic sturgeon (*e.g.*, summer flounder/scup/black sea bass, multispecies, skate, monkfish), and it is likely because of this that the weight for scallops is as high as it is.

Since we know that Atlantic sturgeon can be captured in bottom trawl gear, we expect some level of interaction between Atlantic sturgeon and trawl gear fishing for scallops. As there have been no reported or observed interactions between scallop trawl gear and Atlantic sturgeon, we expect the incidence rate to be very low. Therefore, we anticipate that no more than one Atlantic sturgeon will be captured in trawl gear fishing for scallops annually. This is the best estimate of annual interactions we can provide at this time given the lack of data on documented bycatch in bottom trawls targeting scallops.

The mortality rate for Atlantic sturgeon in commercial bottom otter trawls is estimated at approximately 5% (NEFSC 2011a). Based on this mortality rate, we anticipate one Atlantic sturgeon mortality for every 20 Atlantic sturgeon captured. Given that we anticipate no more than one capture per year, we anticipate no more than one mortality of an Atlantic sturgeon every 20 years. We expect that these interactions could be with Atlantic sturgeon from any of the five DPSs, but are likely to occur in this proportion: NYB 46%; SA 29%; CB 16%; GOM 8%; and Carolina 0.5%.

6.3 Summary of anticipated interactions of ESA-listed species in the scallop fishery

The primary gear types used in the scallop fishery are dredges and bottom trawls. The greatest amount of effort and landings for scallops are accounted for by dredge vessels, which may interact with sea turtles. Atlantic sturgeon are not known to interact with scallop dredge gear. Trawl vessels may interact with sea turtles and Atlantic sturgeon. Based on the analyses above in this Opinion, including analysis of observer data and comparison to similar fisheries, the scallop fishery primarily affects sea turtles in the Mid-Atlantic from May through November, with the majority of interactions occurring between June and October (Haas *et al*. 2008; Murray 2011; Warden 2011). Sea turtle interactions outside the Mid-Atlantic (*i.e.*, on Georges Bank and in the Gulf of Maine) and outside the above months are considered to be rare. Individuals from all five Atlantic sturgeon DPSs can occur throughout the action area at any time.

Based on the best available information, we anticipate up to 161 loggerhead interactions in scallop dredge gear annually as a result of the continued operation of the scallop dredge fishery. For fishing year 2012 (pre-TDD), 129 of those interactions are expected to result in serious injuries or mortality. For fishing year 2013 and beyond (post-TDD), 46 of those interactions are expected to result in serious injuries or mortalities each year. That represents a 64% reduction in serious injury/mortality from 2012 to 2013 and beyond. These are estimates of total observed plus unobserved, but quantifiable, interactions in the dredge fishery annually.

As indicated above, gear modifications including chain mats and the impending requirement of the TDD are expected to reduce the number of lethal interactions (including serious injuries) for all sea turtles that interact with scallop dredge gear. Murray (2011) estimated an average of 105 hard-shelled sea turtles per year (125 turtles reduced to 20) were not captured in dredge gear from September 26, 2006, through 2008 because chain mats were utilized. These 105 turtles represent the unobserved, quantifiable interactions estimated in the fishery since chain mats were implemented (Murray 2011; Warden and Murray 2011). Thus, the estimated maximum conservation benefit of chain mats could be expressed as 105 sea turtles per year (if all the turtles captured in the dredge suffered serious injuries/mortalities and all those excluded from the dredge did not). If all of those 105 turtles survived the interaction with the chain mat, and would not have survived had they been captured in the bag, then the maximum conservation benefit of chain mats alone could be viewed as an 84% ((125-20)/125) reduction in serious injury and mortality. However, there was not enough information in the Murray (2011) analysis to evaluate how the chain mat affected the injury and mortality rate of sea turtles in the gear, though by design the chain mat is intended to reduce the likelihood of a sea turtle's capture in the dredge bag. There is no evidence suggesting that the injury rate of a chain mat equipped dredge is higher than that of a traditional dredge for sea turtles captured in the dredge. As stated in Murray (2011), the realized conservation benefit could be better quantified if mortality and injury rates in traditional gear were refined, and serious injury/mortality rates in chain mat gear were known.

The continued operation of the scallop fishery is also expected to result in the total annual capture of one leatherback, two Kemp's ridleys, and one green sea turtle in dredge gear. Interactions of Kemp's ridley and green sea turtles with scallop dredge gear are expected to be either lethal or non-lethal. Interactions of leatherback sea turtles with scallop dredge gear are expected to be non-lethal given the use of chain mats (which should exclude all leatherbacks from the dredge bag) and the likelihood that all leatherback interactions will occur within the water column rather than on the bottom. However, if an interaction occurs with a dredge not equipped with chain mats, the interaction could be lethal.

Scallop trawl gear is expected to result in the estimated annual average capture of up to 140 loggerhead sea turtles, of which up to 66 are expected to be lethal. Scallop trawl gear is also expected to result in the total annual capture of one leatherback, one Kemp's ridley, and one green sea turtle annually. These interactions may be either lethal or non-lethal. Loggerhead, leatherback, and green sea turtles interacting with scallop dredge and trawl gear are expected to include both juvenile and adult sea turtles, while Kemp's ridley sea turtles interacting with scallop fishing gear are expected to include only benthic immature individuals.

Finally, the continued operation of the scallop fishery is expected to result in the capture of one Atlantic sturgeon annually, which may come from any of the five DPSs which are assessed above. Given an estimated mortality rate of 5% in commercially fished bottom otter trawl gear and a capture rate of one Atlantic sturgeon per year, we anticipate one Atlantic sturgeon mortality in scallop trawl gear every 20 years. We expect that these interactions could be with Atlantic sturgeon from any of the five DPSs, but are likely to occur in this proportion: NYB 46%; SA 29%; CB 16%; GOM 8%; and Carolina 0.5%.

7.0 CUMULATIVE EFFECTS

Cumulative effects as defined in 50 CFR 402.02 include the effects of future State, tribal, local, or private actions that are reasonably certain to occur within the action area considered in this 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. For that reason, future effects of other Federal fisheries are not considered in this section of the document; all Federal fisheries that may affect listed species are the subject of formal section 7 consultations. Effects of ongoing Federal activities, including other fisheries, are considered in the *Environmental Baseline* and *Status of the Species* sections above and are also factored into the *Integration and Synthesis of Effects* section below.

Sources of human-induced mortality, injury, and/or harassment of sea turtles and Atlantic sturgeon in the action area that are reasonably certain to occur in the future include interactions in state-regulated and recreational fishing activities, vessel collisions, ingestion of plastic debris, pollution, global climate change, coastal development, and catastrophic events. While the combination of these activities may affect sea turtles and Atlantic sturgeon, 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 sea turtles and Atlantic sturgeon. 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. 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. 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%). Fishing gear in state waters, including bottom trawls, gillnets, trap/pot gear, and pound nets, interacts with sea turtles each year. NMFS is working with state agencies to address the bycatch of sea turtles in state water fisheries within the action area of this consultation where information exists to show that these fisheries capture sea turtles. Action has been taken by some states to reduce or remove the likelihood of sea turtle bycatch and/or the likelihood of serious injury or mortality in one or more gear types. However, given that state managed commercial and recreational fisheries along the U.S. Atlantic coast are reasonably certain to occur within the action area in the foreseeable future, additional interactions of sea turtles with these fisheries are anticipated. There is insufficient information 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 bycatch in some state water fisheries, the overall effect of these actions 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's STSSN data indicate that vessel interactions are responsible for a number of sea turtle strandings within the action area each year. In the U.S. Atlantic from 1997-2005, 14.9% of all stranded loggerheads were documented as having sustained some type of propeller or collision injuries (NMFS and USFWS 2007a). The incidence of propeller wounds rose from approximately 10% in the late 1980s to a record high of 20.5% in 2004 (STSSN database). Such collisions are reasonably certain to continue into the future. Collisions with boats can stun, injure, or kill sea turtles, and many live-captured and stranded sea 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 vessel interactions with sea turtles will continue in the future. An estimate of the number of sea turtles that will likely be killed by vessels is not available 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, 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 effects on listed species' reproduction and survival. Excessive turbidity due to coastal development and/or construction sites could influence sea turtle or sturgeon foraging ability. Marine debris (*e.g.*, discarded fishing line or lines from boats, plastics) also has the potential to entangle sea turtles in the water or to be fed upon by them. Sea turtles commonly ingest plastic or mistake debris for food and sometimes this may lead to asphyxiation. 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.

8.0 INTEGRATION AND SYNTHESIS OF EFFECTS

The analyses conducted in the previous sections of this Opinion serve to provide a basis to determine whether the proposed action would be likely to jeopardize the continued existence of any ESA-listed sea turtles or Atlantic sturgeon DPSs. In Section 6.0, we outlined how the proposed action would affect these species at the individual level and the extent of those effects in terms of the number of associated interactions and serious injuries/mortalities of each species to the extent possible with the best available data. Now we assess each of these species' response to this impact, in terms of overall population effects, and whether the effects of the proposed action, in the context of the status of the species (Section 3.0), the environmental baseline (Section 4.0), climate change (Section 5.0), and cumulative effects (Section 7.0), will jeopardize their continued existence.

"Jeopardize the continued existence of" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR 402.02). Thus, in making this conclusion for each species, we typically first look at whether there will be a reduction in the reproduction, numbers, or distribution. Then, if there is a reduction in one or more of these elements, we explore whether it will cause an appreciable reduction in the likelihood of both the survival and the recovery of the species.

The ESA Section 7 Handbook (USFWS and NMFS 1998) defines survival and recovery as they apply to the ESA's jeopardy standard. Survival is defined as, "the species' persistence … beyond the conditions leading to its endangerment, with sufficient resilience to allow for the potential recovery from endangerment." 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 ESA." Section 4(a)(1) 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. Recovery is the process by which species' ecosystems are restored and/or threats to the species are removed so self-sustaining and self-regulating populations of listed species can be supported as persistent members of native biotic communities.

Below, for each of the listed species that may be affected by the proposed action, we summarize the status of the species, environmental baseline, effects of climate change, and cumulative effects and then consider the effects of the action in that context. In considering the effects of the action, we look to whether the proposed action will result in reductions in reproduction, numbers, or distribution of that species and then consider 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 ESA.

8.1 Integration and Synthesis of Effects on Sea Turtles and Atlantic Sturgeon

This Opinion has identified in Section 5 (*Effects of the Action*) that the proposed action, the continued operation of the scallop fishery under the Scallop FMP, may adversely affect loggerhead, leatherback, Kemp's ridley, and green sea turtles as a result of interactions with dredge and trawl gear used in the fishery. No other direct or indirect effects to ESA-listed sea turtles are expected as a result of this activity. This Opinion has also identified that the proposed action may adversely affect five Atlantic sturgeon DPSs as a result of capture in trawl gear used in the fishery. No other direct or indirect effects to ESA-listed Atlantic sturgeon are expected as a result of this activity. The following discussions in Sections 8.1.1 through 8.1.5 below provide our determinations of whether there is a reasonable expectation loggerhead, leatherback, Kemp's ridley, and green sea turtles as well as Atlantic sturgeon DPSs will experience reductions in reproduction, numbers, or distribution in response to these effects, and whether any reductions in the reproduction, numbers, or distribution of these species can be expected to appreciably reduce the species' likelihood of surviving and recovering in the wild.

8.1.1 Loggerhead sea turtle – NWA DPS

Based on information from Murray (2011) and Warden (2011a), we anticipate up to 301 loggerhead sea turtles from the NWA DPS will interact annually with gear utilized in the scallop fishery. Using Murray (2011), we believe that an annual average of up to 161 loggerhead sea turtles are expected to interact with scallop dredge gear based on the average number of annual interactions for hard-shelled sea turtles for the years after the chain mat requirement went into effect. In addition, an average of up to 140 loggerheads are expected to be captured annually in scallop trawl gear, based on upper end of the 95% CI for the bycatch estimate in Warden (2011a). Eighty percent (129) of the annual interactions in dredge gear in 2012, 28% (46) of the annual interactions in dredge gear in 2013 and beyond, and 47% (66) of the annual interactions in trawl gear are expected to lead to serious injury or mortality. Therefore, up to 195 loggerhead sea turtles that interact with the scallop fishery in 2012 and up to 112 loggerheads that interact with the fishery in 2013 and each subsequent year after that are expected to die or sustain serious injuries leading to death or failure to reproduce.

Loggerhead sea turtle interactions with scallop dredge and bottom trawl gear could result in death due to forced submergence, given that there are no regulatory controls on tow times in the fishery. The towing of dredge and trawl gear on benthic habitat, and the temporary removal of loggerhead prey from the environment (which may be returned to the water alive or dead) as a result of these fishing activities, will have an insignificant effect on loggerhead sea turtles. No other direct or indirect effects to loggerheads are expected as a result of the proposed actions.

The lethal removal of up to 195 loggerhead sea turtles from the NWA DPS in 2012 and up to 112 loggerheads every year after will reduce the number of loggerheads as compared to the number that would have been present in the absence of the proposed action (assuming all other variables remained the same). These lethal interactions would also result in a future reduction in reproduction as a result of lost reproductive potential, as some of these individuals would be females who would have survived other threats and reproduced in the future, thus eliminating each female individual's contribution to future generations. For example, an adult female loggerhead sea turtle can lay three or four clutches of eggs every two to four years, with 100 to 130 eggs per clutch. The annual loss of adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. A reduction in the distribution of loggerhead sea turtles is not expected from lethal interactions attributed to the proposed action. Because all the potential interactions are expected to occur at random throughout the action area and loggerheads generally have large ranges in which they disperse, the distribution of loggerhead sea turtles in the action area is expected to be unaffected.

Whether or not the reductions in NWA DPS loggerhead numbers and reproduction attributed to the proposed action would appreciably reduce the likelihood of survival for loggerheads depends on what effect these reductions in numbers and reproduction would have on overall population sizes and trends (*i.e.*, whether the estimated reductions, when viewed within the context of the *Status of the Species*, *Environmental Baseline*, *Climate Change*, and *Cumulative Effects* and are to such an extent that adverse effects on population dynamics are appreciable). Loggerhead sea turtles are a slow growing, late-maturing species. Because of their longevity, loggerheads require high survival rates throughout their life to maintain a population. In other words, late-maturing species cannot tolerate much anthropogenic mortality without going into decline. Conant *et al*. (2009) concluded that loggerhead natural growth rates are small, natural survival needs to be high, and even low to moderate mortality can drive the population into decline. Because recruitment to the adult population is slow, population modeling studies suggest even small increased mortality rates in adults and sub-adults could substantially impact population numbers and viability (Crouse *et al*. 1987; Crowder *et al*. 1994; Heppell *et al*. 1995; Chaloupka and Musick 1997).

With multiple sources of mortality, there need to be broad-based reductions in mortality across these multiple sources. Actions have been taken to reduce anthropogenic impacts to loggerhead sea turtles from various sources, particularly since the early 1990s. These include lighting ordinances, predation control, and nest relocations to help increase hatchling survival, as well as measures to reduce the mortality of juveniles and adults in various fisheries and other marine activities. Conant *et al*. (2009) concluded that the results of their models (*i.e.*, predicted continued declines) are largely driven by mortality of juvenile and adult loggerheads from fishery bycatch that occurs throughout the Northwest Atlantic. While significant progress has been made to reduce bycatch in some fisheries in certain parts of the loggerhead's range (including throughout the scallop fishery), and the results of new nesting trend analyses may indicate the positive effects of those efforts, serious bycatch problems still remain unaddressed. The question we are left with for this analysis is whether the effects of the proposed action are too much, given the current status of the species and predicted population trajectories, the many natural and human-caused impacts on sea turtles, including the impacts of the Deepwater Horizon oil release event and climate change, which may be causing long-term effects on the population status and trends of loggerheads which may not be seen until several years from now.

The SEFSC (2009) report estimated that the loggerhead adult female population for the Northwest Atlantic in the 2004-2008 time frame ranged from 20,000 to 40,000 or more individuals (median 30,050), with a large range of uncertainty in total population size. Estimates were based on the following equation: adult females = (nests/(nests per female)) x remigration interval. The estimate of Northwest Atlantic adult loggerhead females was considered conservative for several reasons. The number of nests used for the Northwest Atlantic was based primarily on U.S. nesting beaches. Thus, the results are a slight underestimate of total nests because of the inability to collect complete nest counts for many non-U.S. nesting beaches within the DPS. In estimating the current population size for adult nesting female loggerhead sea turtles, the SEFSC (2009) report simplified the number of assumptions and reduced uncertainty by using the minimum total annual nest count over the relevant five year period (2004-2008) (*i.e.*, 48,252 nests). This was a particularly conservative assumption considering how the number of nests and nesting females can vary widely from year to year (*e.g.*, the 2008 nest count was 69,668 nests, which would have increased the adult female estimate proportionately to between 30,000 and 60,000). Also, minimal assumptions were made about the distribution of remigration intervals and nests per female parameters, which are fairly robust and well known.

Although not in the SEFSC (2009) report, a much less robust estimate for total benthic females in the Northwest Atlantic was produced by the SEFSC that ranges from approximately 60,000 to 700,000 individuals, and possibly up to a little less than one million. The estimate of overall benthic females is considered less robust because it is model-derived, assumes a stable age/stage distribution, and is highly dependent upon the life history input parameters. Relative to the more robust estimate of adult females, this estimate of the total benthic female population is consistent with our knowledge of loggerhead life history and the relative abundance of adults and benthic juveniles: the benthic juvenile population is an order of magnitude larger than adults. Therefore, we believe female benthic loggerheads number in the hundreds of thousands, and therefore smaller pelagic stage individuals would occur in similar or even greater numbers.

As described in the *Status of the Species*, we believe that the Deepwater Horizon oil release event had an adverse impact on loggerhead sea turtles, and resulted in mortalities to an unquantified number of individuals, along with unknown lingering impacts outside the action area resulting from nest relocations, non-lethal exposure, and foraging resource impacts. However, there is no information to indicate, or basis to believe, that a significant population-level impact has occurred that would have changed the species' status to an extent that the expected interactions from the scallop fishery would result in a detectable change in the population status of the NWA DPS of loggerhead turtles. This is especially true given the size of the population and that, unlike Kemp's ridleys, the NWA DPS of loggerheads is proportionally much less intrinsically linked with the Gulf of Mexico.

It is possible that the Deepwater Horizon oil release event reduced the survival rate of all age classes to varying degrees, and may continue to do so for some undetermined time into the future. However, there is no information at this time that it has, or should be expected to have, substantially altered the long-term survival rates in a manner that would significantly change the population dynamics compared to the conservative estimates used in this Opinion. Any impacts are not thought to alter the population status to a degree in which the number of mortalities from the proposed actions could be seen as reducing the likelihood of survival of the species.

We believe that the effects on loggerhead sea turtles associated with the proposed action are not reasonably expected to cause an appreciable reduction in the likelihood of survival of the NWA loggerhead DPS, even in light of the impacts of the Deepwater Horizon oil release event and climate change. We believe the currently large population is still under the threat of possible future decline until large mortality reductions in all fisheries and other sources of mortality (including impacts outside U.S. jurisdiction) are achieved and/or the impacts of past efforts are realized within the population. However, over the next ten years, we expect the Northwest Atlantic population of adult females to remain large (tens or hundreds of thousands of individuals) and to retain the potential for recovery, as explained below. The effects of the proposed action will most directly affect the overall size of the population, which we believe will remain sufficiently large for several decades to come, even if the population were still in a minor decline, such that the action will not cause the population to lose genetic heterogeneity, broad demographic representation, or successful reproduction, nor affect loggerheads' ability to meet their life cycle requirements, including reproduction, sustenance, and shelter.

The final revised recovery plan for loggerhead sea turtles in the Northwest Atlantic includes several objective and measurable recovery criteria which, when met, would result in a determination that the species be removed from the List of Endangered and Threatened Wildlife (NMFS and USFWS 2008). Recovery criteria can be viewed as targets, or values, by which progress toward achievement of recovery objectives can be measured. Recovery criteria may include such things as population numbers and sizes, management or elimination of threats by specific mechanisms, and specific habitat conditions. As a result, there is a need to frame recovery criteria in terms of both population parameters (Demographic Recovery Criteria) and the five listing factors (Listing Factor Recovery Criteria). The nesting beach Demographic Recovery Criteria are specific to recovery units. The remaining criteria cannot be delineated by recovery unit because individuals in the recovery units mix in the marine environment; therefore, these criteria are applicable to all recovery units. Recovery criteria must be met for all recovery units in order for the species to be de-listed (NMFS and USFWS 2008). The Demographic Criteria for nests and nesting females were based on a time frame of one generation for U.S. loggerheads, defined as 50 years. To be considered for delisting, each recovery unit will have recovered to a viable level and each recovery unit will have increased for at least one generation. The rate of increase used for each recovery unit was dependent upon the level of vulnerability of each recovery unit. The minimum statistical level of detection (based on annual variability in nest counts over a generation time of 50 years) of 1% per year was used for the PFRU, the least vulnerable recovery unit. A higher rate of increase of 3% per year was used for the NGMRU and DTRU, the most vulnerable recovery units. A rate of increase of 2% per year was used for the NRU, a moderately vulnerable recovery unit (NMFS and USFWS 2008).

A fundamental problem with restricting population analyses to nesting beach surveys is that they are unlikely to reflect changes in the entire population. This is because of the long time lag to maturity and the relatively small proportion of females that are reproducing for the first time on a nesting beach, at least in populations with high adult survival rates. A decrease in oceanic juvenile or neritic juvenile survival rates may be masked by the natural variability in nesting female numbers and the slow response of adult abundance to changes in recruitment to the adult population (Chaloupka and Limpus 2001). In light of this, two additional Demographic Criteria were developed to ensure a more representative measure of population status was achieved. The first of these additional Demographic Criteria assesses trends in abundance on foraging grounds, and the other assesses age-specific trends in strandings relative to age-specific trends in abundance on foraging grounds. For the foraging grounds, a network of index in-water sites, both oceanic and neritic, distributed across the foraging range must be established and monitored to measure abundance. Recovery can be achieved if there is statistical confidence (95%) that a composite estimate of relative abundance from these sites is increasing for at least one generation. For trends in strandings relative to in-water abundance, recovery can be achieved if stranding trends are not increasing at a rate greater than the trends in in-water relative abundance for similar age classes for at least one generation. These latter two demographic criteria are not specific to recovery units because progeny from the various recovery units mix on the foraging grounds. As a result, in-water trends were not developed for the individual recovery units (NMFS and USFWS 2008).

As mentioned above, assuming some or all of the loggerhead sea turtles killed annually through interactions with the scallop fishery are females, the loss of female loggerhead sea turtles as a result of the proposed action is expected to reduce the reproduction of loggerheads in the NWA DPS compared to the reproductive output of NWA DPS loggerheads in the absence of the proposed action. In addition to being linked to survival, these losses are relevant to the Demographic Recovery Criteria for nests and nesting females. NMFS and USFWS (2008), Witherington *et al.* (2009), and TEWG (2009) provide comprehensive analyses of the status of the nesting assemblages within the NWA DPS using standardized data collected over 10-23 years. The results of these analyses, using different analytical approaches, were consistent—there had been a significant, overall nesting decline within this DPS. However, with the addition of nesting data from 2008-2010, which was not available at the time those analyses were conducted, the nesting trend from 1989-2010 is slightly negative, but the rate of decline is not statistically different from zero (76 FR 58868, September 22, 2011). Additionally, the range from the statistical analysis of the nesting trend includes both negative and positive growth (NMFS and USFWS 2010). The 2010 Florida index nesting number was the largest since 2000. Nesting in the Northwest Atlantic in 2011 was on par with 2010, providing further evidence that the nesting trend may have stabilized. It is important to note, however, that even if the trend has stabilized, overall numbers have a long way to go to meet the goals of the recovery plan.

As previously stated, loggerheads exist as five subpopulations in the western Atlantic (recognized as recovery units in the 2008 recovery plan for the species) and show limited evidence of interbreeding. The 2008 recovery plan compiled the most recent information on the 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 nests per year with approximately 1,272 females nesting per year; (2) for the PFRU, a mean of 64,513 nests per year with approximately 15,735 females nesting per year; (3) for the DTRU, a mean of 246 nests per year with approximately 60 females nesting per year; and (4) for the NGMRU, a mean of 906 nests per year with approximately 221 females nesting per year. For the GCRU, the only estimate available for the number of loggerhead nests per year is from Quintana Roo, Yucatán, Mexico, where a range of 903-2,331 nests per year was estimated from 1987-2001 (NMFS and USFWS 2007a). There are no annual nest estimates available for the Yucatán since 2001 or for any other regions in the GCRU, nor are there any estimates of the number of nesting females per year for any nesting assemblage in this recovery unit. However, the 2008 recovery plan indicates that the Yucatán nesting aggregation has at least 1,000 nesting females annually. It should be noted here, and it is explained further below, that the above numbers only include nesting females (*i.e.*, do not include non-nesting adult females, adult males, or juvenile males or females in the population).

Although limited information is available on the genetic makeup of loggerheads in an area as extensive as the action area, it is likely that loggerheads interacting with the scallop fishery originate from several, if not all of the recovery units. Cohorts from each of the five Northwest Atlantic nesting stocks have been documented to occur in the action area. Genetic analysis of samples collected from immature loggerheads captured in pound nets in the Pamlico-Albemarle Estuarine Complex in North Carolina between 1995-1997 indicated that 80% of the juveniles and sub-adults utilizing the foraging habitat originated from the south Florida nesting stock, 12% from the northern nesting stock, 6% from the Yucatán nesting stock, and 2% from other rookeries (including the Florida Panhandle, Dry Tortugas, Brazil, Greece, and Turkey nesting stocks) (Bass *et al*. 2004). In a separate study, genetic analysis of samples collected from loggerheads from Massachusetts to Florida also found that all five western Atlantic loggerhead stocks were represented (Bowen *et al.* 2004). However, earlier studies by Rankin-Baransky *et al*. (2001) and Witzell *et al*. (2002) indicated that only a few nesting stocks were represented along the U.S. Atlantic coast: south Florida (59% and 69% of the loggerheads sampled, respectively), northern (25% and 10%, respectively), and Mexico (16% and 20%, respectively). Most recently, Haas *et al.* (2008) found that 89% of the loggerheads captured in the U.S. Atlantic scallop fishery from 1996-2005 originated from the south Florida nesting stock, 4% were from the Mexican stock, 3% were from the northern (northeast Florida to North Carolina) stock, 1% were from the northwest Florida stock, and 0% were from the Dry Tortugas stock. The remaining 3% of loggerheads sampled were attributed to nesting stocks in Greece. However, a re-analysis of loggerhead genetics data by the Atlantic Loggerhead TEWG has found that it is unlikely that U.S. fishing fleets are interacting with the Mediterranean DPS (Peter Dutton, NMFS, pers. comm.). Given that updated, more refined analyses are ongoing and the occurrence of Mediterranean DPS juveniles in U.S. Atlantic waters is rare and uncertain, if even occurring at all, it is unlikely that individuals from the Mediterranean DPS would be present in the action area (Memorandum from Patricia A. Kurkul, Regional Administrator, to the Record, November 29, 2011). As a result, those records are excluded from our analysis and are reapportioned to the five Northwest Atlantic stocks which are expected to contribute to individuals in the action area.

The previously defined loggerhead nesting stocks do not share the exact delineations of the recovery units identified in the 2008 recovery plan. However, the PFRU encompasses the south Florida stock, the NRU is roughly equivalent to the northern nesting stock, the northwest Florida stock is included in the NGMRU, the Mexico stock is included in the GCRU, and the DTRU encompasses the Dry Tortugas stock. Based on the genetic analysis presented in Haas *et al.* (2008), which is the most recent and one of the most comprehensive (in terms of the area from which samples were acquired) of the loggerhead genetics studies referenced above, and is in fact based on captures from the scallop fishery itself, the vast majority of the up to 195 loggerheads that are anticipated to be seriously injured or killed due to scallop fishing operations in 2012 (and up to 112 in 2013 and beyond) are likely to originate from the PFRU, with the remainder originating from the NRU, GCRU, NGMRU, and DTRU. Using the mean percent contributions in Haas *et al*. (2008) and then reapportioning the extra 3% attributable to nesting stocks in Greece, we expect that 175 of the loggerheads will be from the PFRU, 7 from the NRU, 9 from the GCRU, 3 from the NGMRU, and 1 from the DTRU in 2012. In 2013 and beyond, 100 of the loggerhead serious injuries/mortalities are expected to be from the PFRU, 4 from the NRU, 5 from the GCRU, 2 from the NGMRU, and 1 from the DTRU annually. The best available information indicates that the proportion of the interactions from each recovery unit are consistent with the relative sizes of the recovery units, and we conclude, based on the available evidence, that none of the recovery units will be disproportionately impacted by interactions in the scallop fishery. Thus, genetic heterogeneity should be maintained in the species even in the face of this level of annual serious injury/mortality as a result of the proposed action.

In the 2008 Opinion on the scallop fishery (NMFS 2008a), we determined, based on the data results of a population viability analysis (PVA) (Merrick and Haas 2008), that the continued operation of the scallop fishery would not reduce appreciably the likelihood of survival and recovery of loggerhead sea turtles. We believe that it is appropriate to use the results of the 2008 PVA as a benchmark to assess whether the fishery as it currently operates will result in jeopardy for the NWA DPS of loggerhead sea turtles.

The PVA was used to estimate quasi-extinction (the point at which so few animals remain that the species/population will inevitably become extinct) likelihoods under conditions with and without fishery effects (Merrick and Haas 2008). Since the PVA was count-based, Merrick and Haas (2008) used the only relatively complete and available population time series at the time—index nesting beach counts for 1998-2005 for the analysis. As such, the analysis focused on the viability of the adult females and did not model the viability of the entire loggerhead population (Merrick and Haas 2008).

The PVA is described in detail in Merrick and Haas (2008) (Appendix B). Briefly, to conduct the PVA, the authors used:

- an estimate of loggerhead nests in 2005 in the southeastern U.S. (North Carolina to Alabama) representing the northern and peninsular Florida nesting stocks (*i.e.*, the NRU and PFRU, respectively) to estimate the number of adult females;
- quasi-extinction (the point at which so few animals remain that the species/population will inevitably become extinct) rather than extinction (the point at which no animals of that species/population are alive) as the reference point for survival;
- measures to assess the likelihood of quasi-extinction, which include the probability of quasi-extinction (at 25, 50, 75, and 100 years) and the number of simulations with quasi-extinction probabilities at 25, 50, 75, or 100 years greater than 0.05.

In short, the PVA established a baseline using the rate of change of the adult female population (which implicitly included the mortalities from the scallop fishery up to that time), and the 2005 count of adult females estimated from all beaches in the Southeast U.S. based on an extrapolation from nest counts (Merrick and Haas 2008). The rate of change was then adjusted by adding back the scallop fishery interactions (converted to adult female equivalents), and re-running the PVA. The results of these two analyses were then compared. Merrick and Haas (2008) determined that both the baseline and adjusted baseline (adding back the fishery interactions) had quasi-extinction probabilities of zero (0) at periods of 25, 50, and 75 years and a probability of 1% at 100 years.

Based on the PVA results, we determined in 2008 that the continued operation of the Atlantic sea scallop fishery, resulting in mortalities of loggerhead sea turtles, would not have an appreciable effect on the number of adult female loggerhead sea turtles in the Northwest Atlantic over a future 100 years. While a statistically significant difference was detected in the number of iterations out of 1,000 with quasi-extinction probabilities at 100 years greater than 5%, the differences smoothed out over the 1,000 iterations and, taken together, the probability of quasi-extinction at 100 years was the same (1%) under both baseline conditions, and when the baseline was adjusted by removing interactions as a result of the scallop fishery. Therefore, we concluded that the continued operation of the scallop fishery was not likely to appreciably reduce the likelihood of survival and recovery for loggerhead sea turtles in the Northwest Atlantic within the future 100 years (NMFS 2008a).

Although the PVA is four years old, it used data from 1989-2005, and it modeled different effects of the fishery on loggerheads than what may occur presently, we can still use it as a standard for comparison in this Opinion as the current levels of loggerhead nesting in the Southeast U.S. (*i.e.*, the NRU and PFRU) are believed to be on the same trend and scale as they were during the time period assessed in the PVA, while loggerhead mortality in the scallop fishery is likely to be much lower in the future (post-TDD implementation specifically) than when the PVA was originally run. In light of the substantially lower number of anticipated mortalities as a result of the fishery expected in 2013 and beyond, as well as the effects of activities in the *Status of the Species*, *Environmental Baseline*, *Climate Change*, and *Cumulative Effects* sections over the next ten years, we can confidently say that if quasi-extinction in the next 100 years as a result of the scallop fishery was not likely back in 2008, it is still not likely today.

The PVA analysis done for the 2008 Opinion and our comparison of its results to the current status and trends of the NWA DPS of loggerheads (in light of effects from the scallop fishery, other baseline activities, and climate change) supports the conclusion that continued operation of the scallop fishery will neither affect the number of nests and nesting females (Demographic Criteria #1) nor the trends in abundance on foraging grounds (Demographic Criteria #2) to the point where there is an appreciable reduction in the species' likelihood of recovery. This is a NMFS determination based on the PVA results, it is not a determination of the PVA itself. Recovery is the process of removing threats so self-sustaining populations persist in the wild. The required use of chain mats and TDDs throughout much of the area where sea turtle interactions with the fishery have been known to occur supports and implements the Services' recovery plan developed for the NWA DPS (NMFS and USFWS 2008). The proposed action would not impede progress on carrying out any aspect of the recovery program or achieving the overall recovery strategy. The recovery plan estimates that the population will reach recovery in 50 to 150 years, as recovery actions are implemented. The minimum end of the range assumes a rapid reversal of the current declining trends; the higher end assumes that additional time will be needed for recovery actions to bring about population growth.

Even amidst ongoing threats to the species such as fishery mortality and climate change, the potential loss of 195 loggerheads in 2012 and 112 loggerheads annually in 2013 and beyond from the Atlantic over the next ten years (and potentially beyond) is not likely to result in any additional threat of endangerment to the NWA DPS within the foreseeable future throughout all or a significant portion of its range. This is due to the large size of the current nesting population, the fact that 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. The level of annual loggerhead serious injury and mortality anticipated as a result of the scallop fishery is lower than it was when it was last assessed in the 2008 Opinion, while the status of the species, environmental baseline, effects of climate change, and cumulative effects have not changed significantly to the point where loggerheads are much worse off than they were in 2008, when the last PVA on the effects of the scallop fishery was run.

8.1.2 Leatherback sea turtle

NMFS-approved observers have not recorded any interactions between leatherback sea turtles and scallop dredge or trawl gear. However, leatherback sea turtles may interact with the scallop fishery given that their distribution overlaps with operation of scallop gear and leatherbacks have been observed captured in bottom trawl gear similar to that used in the scallop fishery. From 2001-2010, there were three confirmed captures of leatherback sea turtles in bottom otter trawl (fish) gear in the action area (NEFSC FSB 2011). Based on these data, the bycatch of leatherback sea turtles in any mobile gears operating within the action area, including scallop dredge and trawl gear, is expected to occur, but likely at low levels.

Captures of leatherback sea turtles in dredge and bottom trawl gear could result in death due to forced submergence, given that there are no regulatory controls on tow times in this fishery. Since leatherbacks forage within the water column rather than on the bottom, interactions with dredge and bottom trawl gear are expected to occur when the gear is traveling through the water column versus on the bottom. The use of chain mats at times and in parts of the action area where leatherbacks are most abundant is expected to prevent most leatherback captures in scallop dredge gear since the chains form a pattern of openings across the mouth of the dredge bag that are typically too small for a leatherback to pass through. Since a chain mat will likely prevent a leatherback from entering the dredge bag, it is less likely to suffer injuries as a result of forced submergence or those that would otherwise occur from capture in the dredge bag (*e.g.*, injuries as a result of falls or crushing during the emptying of the dredge bag). Interactions of leatherback sea turtles with scallop dredge gear are still expected to involve physical contact between the turtle and the gear, which could result in serious injury or mortality. However, since dredge gear is hauled through the water column at a relatively slow speed and contact between the turtle and the gear would most often occur in a fluid environment versus on the bottom, leatherbacks occurring within the water column are not expected to be as susceptible to injury or death as a result of physical contact with a scallop dredge as are the other three hard-shelled sea turtle species which spend much more time on the ocean bottom. However, we are not ruling out the possibility for serious injuries or mortality due to an interaction with either a dredge or bottom trawl. Thus, as described in Section 6.2, we anticipate up to two annual lethal leatherback sea turtle interactions with the fishery.

Lethal interactions of leatherback sea turtles, whether male or female, immature or mature, would reduce their respective populations compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. The lethal interactions could also result in a potential reduction in future reproduction, assuming one or more of these individuals would be female and would have otherwise survived to reproduce in the future. For example, an adult female leatherback sea turtle can produce up to 700 eggs or more per nesting season (Schultz 1975). Although a significant portion (up to approximately 30%) of the eggs can be infertile, the annual loss of adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. Thus, the death of any female leatherbacks that would have otherwise survived to reproduce would eliminate the individual's and its future offspring's contribution to future generations. The anticipated lethal interactions are expected to occur anywhere in the action area where the turtles may be present, which is primarily in Mid-Atlantic waters from May through November. Given that leatherbacks generally have large ranges in which they disperse, no reduction in the distribution of leatherback sea turtles is expected from the proposed action. Whether the estimated reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

The Leatherback TEWG estimated that there are between 34,000-94,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) in the North Atlantic (TEWG 2007). Of the five leatherback populations or groups of populations in the North Atlantic, three show an increasing or stable trend (Florida, Northern Caribbean, and Southern Caribbean). This includes the largest nesting population, located in the Southern Caribbean at Suriname and French Guiana. In 2001, the number of nests for Suriname and French Guiana was 60,000; this was one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). Of the remaining two populations, there was not enough information available on the West African population to conduct a trend analysis, while for the Western Caribbean, a slight decline in annual population growth rate was detected (TEWG 2007). An annual growth rate of 1.0 is considered a stable population; the growth rates of two nesting populations in the Western Caribbean were 0.98 and 0.96 (TEWG 2007). A stable trend in nesting suggests that leatherbacks are able to maintain current levels of nesting as well as current numbers of adult females despite on-going activities as described in the *Status of the Species*, *Environmental Baseline*, *Climate Change*, and *Cumulative Effects* sections. An increasing trend in nesting suggests that the combined impact to Atlantic leatherbacks from these on-going activities is less than what has occurred in the past. The result of which is that more female leatherbacks are maturing and subsequently nesting, and/or are surviving to an older age and producing more nests across their lifetime.

We believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of leatherback sea turtles in the wild. Although the anticipated mortalities would result in a reduction in absolute population numbers, it is not likely this reduction would appreciably reduce the likelihood of survival of this species. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be replaced through recruitment of new breeding individuals from successful reproduction of sea turtles unaffected by the proposed action. Considering that nesting trends for the Florida and Northern Caribbean populations as well as the largest nesting population, the Southern Caribbean, are all either stable or increasing, we believe the proposed action is not likely to have any measurable effect on overall population trends. These trends already reflect the past impact of fisheries occurring in the action area and the proposed action is expected to control those impacts by maintaining effort levels consistent with or lower than those that have occurred in previous years. As explained in the *Environmental Baseline*, although no direct leatherback impacts (*i.e.*, oiled sea turtles or nests) from the Deepwater Horizon oil spill in the northern Gulf of Mexico were observed, some impacts from that event may be expected. However, there is no information to indicate, or basis to believe, that a significant population-level impact has occurred that would change the species' status to an extent that the expected interactions from these fishery would result in a detectable change in the population status of leatherback sea turtles. Any impacts are not thought to alter the population status to a degree in which the number of mortalities from the proposed action could be seen as reducing the likelihood of survival and recovery of the species.

As described in the *Environmental Baseline*, regulatory actions have been taken to reduce anthropogenic effects to Atlantic leatherbacks. These include measures to reduce the number and severity of leatherback interactions in the U.S. Atlantic longline fisheries, the U.S. South Atlantic and Gulf of Mexico shrimp fisheries, and the scallop dredge fishery. Reducing the number of leatherback sea turtles injured and killed as a result of these activities is expected to increase the number of Atlantic leatherbacks, and increase leatherback reproduction in the Atlantic. Since most of these regulatory measures have been in place for several years now, it is likely that current nesting trends reflect the benefit of these measures to Atlantic leatherback sea turtles. Therefore, the current nesting trends for leatherback sea turtles in the Atlantic are likely to continue to improve as a result of the regulatory actions taken for these and other fisheries. There are no new known sources of serious injury or mortality for leatherback sea turtles in the Atlantic other than potential impacts from the Deepwater Horizon oil spill, which would likely be localized to a small number of individuals foraging in or migrating through that portion of the northern Gulf of Mexico.

The recovery plan for Atlantic leatherback sea turtles (NMFS and USFWS 1992) lists the following recovery objectives which are relevant to the proposed action in this Opinion:

- The adult female population increases over the next 25 years, as evidenced by a statistically significant trend in the number of nests at Culebra, Puerto Rico; St. Croix, U.S. Virgin Islands; and along the east coast of Florida.

We believe the proposed action is not likely to impede the recovery objectives above and will not result in an appreciable reduction in the likelihood of leatherback sea turtles' recovery in the wild. In Puerto Rico, the main nesting areas are at Fajardo on the main island of Puerto Rico and on the island of Culebra. Between 1978 and 2005, nesting increased in Puerto Rico from a minimum of nine nests recorded in 1978 to 469-882 nests recorded each year between 2000 and 2005. Annual growth rate was estimated to be 1.1 with a growth rate interval between 1.04 and 1.12, using nest numbers between 1978 and 2005 (NMFS and USFWS 2007b). In the U.S. Virgin Islands, researchers estimated a population growth of approximately 13% per year at Sandy Point National Wildlife Refuge from 1994-2001. Between 1990 and 2005, the number of nests recorded has ranged from 143 (1990) to 1,008 (2001). The average annual growth rate was calculated as approximately 1.10 (with an estimated interval of 1.07 to 1.13) (NMFS and USFWS 2007b). In Florida, a Statewide Nesting Beach Survey program has documented an increase in leatherback nesting numbers from 98 (1989) to 800-900 (early 2000s). Based on standardized nest counts made at Index Nesting Beach Survey sites surveyed with constant effort over time, there has been a substantial increase in leatherback nesting in Florida since 1989. The estimated annual growth rate was approximately 1.18 (with an estimated 95% CI of 1.1 to 1.21) (NMFS and USFWS 2007b).

Based on the information provided above, the loss of up to two leatherback sea turtles annually in the Atlantic as a result of the continued operation of the fishery will not appreciably reduce the likelihood of survival for leatherbacks in the Atlantic given the increased, stable, or nearly stable nesting trend at all Atlantic nesting sites, and given measures that reduce the number of Atlantic leatherback sea turtles injured and killed in the Atlantic (which should result in increases to the numbers of leatherbacks in the Atlantic that would otherwise have not occurred in the absence of those regulatory measures). The scallop fishery has no effects on leatherback sea turtles that occur outside of the Atlantic. Therefore, in light of other ongoing actions affecting leatherback sea turtles in the action area (including climate change), the continued operation of the fishery over the next ten years will not appreciably reduce the likelihood of survival for leatherbacks in the Atlantic. As a result, the proposed action will not appreciably reduce the likelihood of survival of the species.

The annual loss of up to two leatherback sea turtles, together with an increase in nesting, is not expected to affect the positive growth rate in the female population of leatherback sea turtles nesting in Puerto Rico, the U.S. Virgin Islands, and Florida. Therefore, the continued operation of the fishery under the Scallop FMP will not appreciably reduce the likelihood of recovery for leatherback sea turtles in the Atlantic. Since the fishery has no effects on leatherback sea turtles that occur outside of the Atlantic, its continued operation will not appreciably reduce the likelihood of recovery for the species.

Even amidst an ongoing decline in the overall number of leatherbacks in the Pacific and the threat of climate change on the species as a whole, the potential loss of two leatherbacks annually from the Atlantic over the next ten years (and potentially beyond) is not likely to result in any additional threat of extinction within the foreseeable future throughout all or a significant portion of its range. The potential loss of two leatherbacks annually due to the scallop fishery only represents a decline in the North Atlantic adult leatherback population by 0.006% at the greatest (two out of 34,000). Taking into account the number of Atlantic leatherbacks in other life stages as well as all those occurring in the Pacific (regardless of life stage) indicates how minor this level of annual mortality is in regards to the species achieving its recovery objectives.

8.1.3 Kemp's ridley sea turtle

Kemp's ridley sea turtles have been documented to interact with both dredge and bottom trawl gear in the action area, although there have been no known captures of Kemp's ridleys in trawl gear targeting scallops. From 2001-2010 there were two confirmed captures of Kemp's ridley sea turtles in bottom otter trawl (fish) gear in the action area (NEFSC FSB 2011). The distribution of Kemp's ridleys overlaps seasonally with the use of both gears and they are known to be captured by dredge gear used in the fishery, albeit at low levels (Murray 2011).

There have been two confirmed captures of Kemp's ridley sea turtles in scallop dredge gear since 2001. One of these was killed as a result of the interaction. One of the turtles was confirmed to be an immature based on its size of 24.3 centimeters CCL (Murray 2011). This is not unexpected since Mid-Atlantic and southern New England waters are recognized as developmental habitat for Kemp's ridley sea turtles after they enter the benthic environment (Musick and Limpus 1997; Morreale and Standora 2005). Given the relatively small size of this species of sea turtle, the use of chain mat modified scallop dredge gear is not expected to prevent a small Kemp's ridley sea turtle struck by the gear from entering the dredge bag. Therefore, Kemp's ridley interactions with scallop dredge gear may result in serious injuries and/or mortality as a result of forced submergence in the gear, other injuries suffered as a result of capture in the dredge bag, and/or injuries suffered upon hauling and emptying of the dredge bag. If the turtle encountered the gear when on the bottom versus when swimming in the water column, physical contact with the dredge against the bottom would also be expected to result in serious injury and/or death to the turtle if a TDD was not being used at the time.

Based on the data in Murray (2011) on the observed captures of Kemp's ridley sea turtles in scallop dredge gear, as well as the expectation that one interaction will occur annually with bottom trawl gear given the overlap of the fishery with the species, the continued operation of the scallop fishery (dredge and trawl components combined) is anticipated to result in up to three lethal interactions with Kemp's ridleys in 2012 and up to two lethal interactions annually in 2013 and beyond. It is assumed that there is an equal chance of lethally capturing a male or female Kemp's ridley since available information suggests that both sexes occur in the action area. All Kemp's ridleys interacting with the fishery in the action area are expected to be immatures.

The proposed action would reduce the species' population compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. The proposed action could also result in a potential reduction in future reproduction, assuming at least some of these individuals would be female and would have survived to reproduce in the future. The annual loss of adult females could preclude the production of thousands of eggs and hatchlings, of which a small percentage is expected to survive to sexual maturity. Thus, the death of any females would eliminate their contribution to future generations, and result in a reduction in sea turtle reproduction. The anticipated lethal interactions are expected to occur anywhere in the action area where the turtles may be present, which is primarily in Mid-Atlantic waters from May through November. Since Kemp's ridleys generally have large ranges in which they disperse, no reduction in the distribution of Kemp's ridley sea turtles is expected from these fishery interactions. Whether the reductions in numbers and reproduction of Kemp's ridley sea turtles would appreciably reduce their likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

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. The total annual number of nests recorded at Rancho Nuevo and adjacent camps has exceeded 10,000 in recent years. Over 20,000 nests were recorded in 2009 at Rancho Nuevo and adjacent camps (J. Pena, GPZ, pers. comm.). From 2002-2009, a total of 771 Kemp's ridley nests were documented on the Texas coast. This is more than nine times greater than the 81 nests recorded over the previous 54 years from 1948-2001 (Shaver and Caillouet 1998; Shaver 2005), indicating an increasing nesting population in Texas. From 2005-2009, the number of nests from all monitored beaches indicate approximately 5,500 females are nesting each season in the Gulf of Mexico (NMFS *et al*. 2011). The observed increase in nesting of Kemp's ridley sea turtles suggests that the combined impact to Kemp's ridley sea turtles from on-going activities as described in the *Status of the Species*, *Environmental Baseline*, *Climate Change*, and *Cumulative Effects* sections are less than what has occurred in the past. The result of which is that more female Kemp's ridley sea turtles are maturing and subsequently nesting, and/or are surviving to an older age and producing more nests across their lifetime.

Heppell *et al*. (2005) predicted in a population model that the Kemp's ridley sea turtle population is expected to increase at least 12%-16% per year and that the population could attain at least 10,000 females nesting on Mexico beaches by 2015. NMFS *et al*. (2011) contains an updated model which predicts that the population is expected to increase 19% per year and that the population could attain at least 10,000 females nesting on Mexico beaches by 2011. Approximately 25,000 nests would be needed for an estimate of 10,000 nesters on the beach, based on an average 2.5 nests/nesting female. In 2009 the population was on track with 21,144 nests, but an unexpected and as yet unexplained drop in nesting occurred in 2010 (13,302), deviating from the NMFS *et al*. (2011) model prediction. A subsequent increase to 20,570 nests occurred in 2011, but we will not know if the population is continuing the trajectory predicted by the model until future nesting data is available. Of course, this updated model assumes that current survival rates within each life stage remain constant. The recent increases in Kemp's ridley sea turtle nesting seen in the last two decades is likely due to a combination of management measures including elimination of direct harvest, nest protection, the use of TEDs, reduced trawling effort in Mexico and the U.S., and possibly other changes in vital rates (TEWG 1998, 2000). While these results are encouraging, the species' limited range and low global abundance makes it particularly vulnerable to new sources of mortality as well as demographic and environmental stochasticity, all of which are often difficult to predict with any certainty.

It is likely that the Kemp's ridley sea turtle was the sea turtle species most affected by the Deepwater Horizon oil spill on a population level. In addition, the sea turtle strandings documented in 2011 in Alabama, Louisiana, and Mississippi primarily involved Kemp's ridley sea turtles. Nevertheless, the effects on Kemp's ridley sea turtles from the proposed action are not likely to appreciably reduce overall population numbers over time due to current population sizes, expected recruitment, and continuing strong nesting numbers (including, based on preliminary information, in 2011), even in light of the adverse impacts expected to have occurred from the Deepwater Horizon oil spill and the strandings documented in 2011.

As described in the *Environmental Baseline*, regulatory actions have been taken to reduce anthropogenic effects to Kemp's ridley sea turtles. These include measures implemented to reduce the number and severity of Kemp's ridley sea turtle interactions in the U.S. South

Atlantic and Gulf of Mexico shrimp fisheries and the scallop dredge fishery. Since some of these regulatory measures have been in place for a number of years now, it is likely that current nesting trends reflect the benefit of these measures to Kemp's ridley sea turtles. Therefore, the current nesting trends for Kemp's ridleys are likely to continue to improve as a result of regulatory actions taken for these and other fisheries. There are no new known sources of serious injury or mortality for Kemp's ridley sea turtles other than potential impacts from the Deepwater Horizon oil spill, which would likely be localized to individuals foraging in or migrating through that portion of the northern Gulf of Mexico.

The recovery plan for the Kemp's ridley sea turtle (NMFS *et al*. 2011) lists the following recovery objectives for downlisting that are relevant to the scallop fishery:

- Demographic: A population of at least 10,000 nesting females in a season (as measured by clutch frequency per female per season) distributed at the primary nesting beaches (Rancho Nuevo, Tepehuajes, and Playa Dos) in Mexico is attained. Methodology and capacity to implement and ensure accurate nesting female counts have been developed.
- Listing factor: TED regulations, or other equally protective measures, are maintained and enforced in U.S. and Mexican trawl fisheries (*e.g.*, shrimp, summer flounder, whelk) that are known to have an adverse impact on Kemp's ridleys in the Gulf of Mexico and Northwest Atlantic Ocean.

Based upon the NMFS *et al*. (2011) projection that the population could attain at least 10,000 females nesting on Mexico beaches by 2011, the species appears to be on course for achieving the above demographic recovery criterion for downlisting. Plus, Kemp's ridleys mature and nest at an age of 7-15 years, which is earlier than other sea turtles. A younger age at maturity may be a factor in the positive response of this species to recovery actions. In regards to the listing factor recovery criterion, NMFS *et al*. (2011) states "the highest priority needs for Kemp's ridley recovery are to maintain and strengthen the conservation efforts that have proven successful. In the water, successful conservation efforts include maintaining the use of … TEDs … in fisheries currently required to use them, expanding TED-use to all trawl fisheries of concern, and reducing mortality in gillnet fisheries. Adequate enforcement in both the terrestrial and marine environment also is also noted essential to meeting recovery goals." We are currently undertaking several of these initiatives which should aid in the recovery of the species. The required use of TEDs in shrimp trawls in the U.S. under sea turtle conservation regulations and in Mexican waters has had dramatic effects on the recovery of Kemp's ridley sea turtles.

Based on the information provided above, the loss of up to three Kemp's ridley sea turtles in 2012 and two annually in 2012 and beyond as a result of the continued operation of the scallop fishery will not appreciably reduce the likelihood of survival for Kemp's ridley sea turtles given both the increased nesting trend and ongoing measures that reduce the number of Kemp's ridley sea turtles injured and killed (which should result in increases to the numbers of Kemp's ridley sea turtles that would not have occurred in the absence of those regulatory measures). The scallop fishery has no effects on Kemp's ridley sea turtles that occur outside of the Atlantic. Therefore, since the continued operation of the fishery will not appreciably reduce the likelihood of survival of Kemp's ridley sea turtles in the Atlantic, the proposed action will not appreciably reduce the likelihood of survival for the species.

The loss of up to three Kemp's ridleys in 2012 and up to three annually in 2013 and beyond is not expected to significantly change the trend in increased nesting, especially if the Kemp's ridleys killed in the fishery are male, and will not compromise the continued existence of the species. Based on what we know about historical shrimp trawling effort (*i.e.*, that there has been much higher effort in the recent past), it is likely that large numbers of turtles were being impacted by shrimp trawls for the past decade or more. Despite this fact, the estimated population size of Kemp's ridleys has continued to increase. Therefore, in light of other ongoing actions affecting Kemp's ridley sea turtles in the action area (including climate change), the continued operation of the fishery under the Scallop FMP over the next ten years will not appreciably reduce the likelihood of recovery for the species.

Even amidst the impacts of the Deepwater Horizon oil release event and the threat of climate change on the species as a whole, the potential loss of two or three Kemp's ridleys annually from the Atlantic over the next ten years (and potentially beyond) is not likely to result in any additional threat of extinction within the foreseeable future throughout all or a significant portion of its range. The potential loss of two or three Kemp's ridleys annually due to the scallop fishery only represents a small loss to the growing Kemp's ridley population in the Northwest Atlantic and would only involve immature turtles that are not yet part of the breeding population.

8.1.4 Green sea turtle

Green sea turtles have been documented to interact with both dredge and bottom trawl gear in the action area, although there have been no known captures of green sea turtles in trawl gear targeting scallops. From 2001-2010 there was one confirmed capture of a green sea turtle in bottom otter trawl (fish) gear in the action area (NEFSC FSB 2011). The distribution of green sea turtles overlaps seasonally with the use of both gears and they are known to be captured by dredge gear used in the fishery, albeit at low levels.

The one confirmed capture of a green sea turtle in scallop dredge gear occurred in 1997. It is difficult to determine to which age class the green sea turtle captured in scallop dredge gear might have belonged given that its size was estimated rather than measured. However, at 70 centimeters, it is likely the individual was either a large juvenile or an adult. Atlantic and southern New England waters are recognized as developmental habitat for green sea turtles after they enter the benthic environment (Musick and Limpus 1997; Morreale and Standora 2005). Therefore, it would seem more likely that the green sea turtle observed captured in scallop dredge gear was an immature turtle. However, given the uncertainty of the size of the turtle observed captured in scallop dredge gear, it is reasonable to expect that benthic immature and/or sexually mature green sea turtles will be captured in scallop dredge gear as a result of the continued operation of the scallop fishery. Chain mats may or may not prevent green sea turtles from entering the dredge bag depending on the size of the animal encountered. If the turtle is small enough to pass between the chains and into the dredge bag, then the turtle may be killed as a result of forced submergence in the gear, injured as a result of capture in the dredge bag, or injured upon hauling and emptying of the dredge bag. If the turtle encountered the gear when on the bottom versus when swimming in the water column, then physical contact with the dredge against the bottom would also be expected to result in injury to the turtle if a TDD was not being used at the time. If the turtle was large enough to be prevented from entering the dredge bag by the chain mat, then the turtle would not be subject to injuries that can occur as a result of forced submergence, capture in the dredge bag, and hauling and emptying of the dredge. The turtle would still be expected to be injured if it made physical contact with the dredge gear when both the turtle and the gear were on the bottom. Regardless of their size or age class, green sea turtles in the water column are not expected to be injured as a result of physical contact, alone, (without subsequent capture) with the dredge gear when the gear is also in the water column given the relatively slow speed at which the gear is hauled through the water column and contact between the turtle and the gear would occur in a fluid environment.

Based on the observed captures of green sea turtles in both dredge and trawl gear in the waters of the action area, the continued operation of the scallop fishery (dredge and trawl gear components combined) is anticipated to result in the annual serious injury or mortality of up to two green sea turtles. It is assumed that there is an equal chance of lethally interacting with a male or female green sea turtle since available information suggests that both sexes occur in the action area. Shallow, coastal waters of the Mid-Atlantic and southern New England are recognized as developmental habitat for green sea turtles after they enter the benthic environment (Musick and Limpus 1997; Morreale and Standora 2005; Makowski *et al*. 2006). In addition, nesting females have been documented to occur in action area waters as far north as Delaware, and nest in large numbers along the southeast coast of Florida. Thus, it is reasonable to expect that both benthic immature and sexually mature green sea turtles may interact with dredge and bottom trawl gear as a result of the continued operation of the scallop fishery.

Lethal interactions would reduce the number of green sea turtles, compared to their numbers in the absence of the proposed actions, assuming all other variables remained the same. Lethal interactions would also result in a potential reduction in future reproduction, assuming some individuals would be females and would have otherwise survived to reproduce. For example, an adult female green sea turtle can lay 1-7 clutches (usually 2-3) of eggs every two to four years with 110-115 eggs/nest, of which a small percentage is expected to survive to sexual maturity. A lethal capture of a female green sea turtle in dredge or bottom trawl gear would likely remove this level of reproductive output from the species. The anticipated lethal interactions are expected to occur anywhere in the action area where the turtles may be present, which is primarily in Mid-Atlantic waters from May through November. Since green sea turtles generally have large ranges in which they disperse, no reduction in the distribution of green sea turtles is expected from these interactions. Whether the reductions in numbers and reproduction of this species would appreciably reduce its likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

The five-year status review for green sea turtles states that of the seven green sea turtle nesting concentrations in the Atlantic Basin for which abundance trend information is available, all were determined to be either stable or increasing (NMFS and USFWS 2007d). That review also states that the annual nesting female population in the Atlantic basin ranges from 29,243-50,539 individuals. Additionally, the pattern of green sea turtle nesting shows mostly biennial peaks in abundance, with a generally positive trend during the ten years of regular monitoring since the establishment of index beaches in Florida in 1989. An average of 5,039 green sea turtle nests were laid annually in Florida between 2001 and 2006, with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007d). Data from the index nesting beach program in Florida substantiate the dramatic increase in nesting. In 2007, there were 9,455 green turtle nests found just on index nesting beaches, the highest since index beach monitoring began in 1989. The number fell back to 6,385 in 2008, further dropping under 3,000 in 2009, but that consecutive drop was a temporary deviation from the normal biennial nesting cycle for green sea turtles, as 2010 and 2011 saw an increase back to 8,426 and 10,701 nests on the index nesting beaches (FFWCC 2012). The number of green sea turtle nests in 2011 was the highest number recorded during the index nesting beach program since its inception in 1989. Elsewhere in the Atlantic, modeling by Chaloupka *et al*. (2008) using data sets of 25 years or more resulted in an estimated 4.9% annual growth for the Tortuguero, Costa Rica nesting population. The observed increase in nesting of Atlantic green sea turtles suggests that the combined impact to Atlantic green sea turtles from on-going activities as described in the *Status of the Species*, *Environmental Baseline*, *Climate Change*, and *Cumulative Effects* sections are less than what has occurred in the past. The result of which is that more female green sea turtles are maturing and subsequently nesting, and/or are surviving to an older age and producing more nests across their lifetime.

We believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of the green sea turtle in the wild. Although the anticipated mortalities would result in an instantaneous reduction in absolute population numbers, the U.S. populations of green sea turtles would not be appreciably affected. For a population to remain stable, sea turtles must replace themselves through successful reproduction at least once over the course of their reproductive lives, and at least one offspring must survive to reproduce itself. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be exceeded through recruitment of new breeding individuals from successful reproduction of sea turtles that were not seriously injured or killed in the fishery. Since the abundance trend information for green sea turtles is clearly increasing, we believe the lethal interactions attributed to the proposed action will not have any measurable effect on that trend. As described in the *Environmental Baseline*, although the Deepwater Horizon oil spill is expected to have resulted in adverse impacts to green sea turtles, there is no information to indicate, or basis to believe, that a significant population-level impact has occurred that would have changed the species' status to an extent that the expected interactions from the scallop fishery would result in a detectable change in the population status of green sea turtles in the Atlantic. Any impacts are not thought to alter the population status to a degree in which the number of mortalities from the proposed action could be seen as reducing the likelihood of survival and recovery of the species.

As also described in the *Environmental Baseline*, regulatory actions have been taken to reduce anthropogenic effects to green sea turtles in the Atlantic. These include measures to reduce the number and severity of green sea turtle interactions in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries, the scallop dredge fishery, and the Virginia pound net fishery—all of which are causes of green sea turtle mortality in the Atlantic. Since most of these regulatory measures have been in place for several years now, it is likely that current nesting trends reflect the benefit of these measures to Atlantic green sea turtles. Therefore, the current nesting trends for green sea turtles in the Atlantic are likely to continue to improve as a result of the regulatory actions taken for these and other fisheries. There are no new known sources of serious injury or mortality for green sea turtles in the Atlantic other than potential impacts from the Deepwater Horizon oil spill, which would likely be localized to individuals foraging in or migrating through that portion of the northern Gulf of Mexico.

The recovery plan for Atlantic green sea turtles (NMFS and USFWS 1991) lists the following recovery objectives which are relevant to the scallop fishery, and must be met over a period of 25 continuous years:

- The level of nesting in Florida has increased to an average of 5,000 nests per year for at least six years;
- A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.

Green sea turtle nest counts in Florida from 2001-2006 were documented as follows: 2001 - 581, 2002 - 9,201, 2003 - 2,622, 2004 - 3,577, 2005 - 9,644, 2006 - 4,970. This averages to 5,039 nests annually over those six years (2001-2006) (NMFS and USFWS 2007d). Nest counts in subsequent years have, on average, been even higher (*e.g.*, 2007 - 9,455, 2008 - 6,385, 2009 - 3,000, 2010 - 8,426, 2011 - 10,701); thus, this recovery criterion continues to be met.

Several actions are being taken to address the second objective; however, there are currently few studies, and no estimates, available that specifically address changes in abundance of individuals on foraging grounds. Ehrhart *et al*. (2007) found a 661% increase in juvenile green sea turtle capture rates in the central region of the Indian River Lagoon (along the east coast of Florida) over a 24-year study period from 1982-2006. Wilcox *et al*. (1998) found a dramatic increase in the number of green sea turtles captured from the intake canal of the St. Lucie nuclear power plant on Hutchinson Island, Florida beginning in 1993. During a 16-year period from 1976-1993, green sea turtle captures averaged 24 per year (Wilcox *et al*. 1998). Green sea turtle catch rates for 1993, 1994, and 1995 were 745%, 804%, and 2,084% above the previous 16-year average annual catch rates (Wilcox *et al*. 1998). In a study of sea turtles incidentally caught in pound net gear fished in inshore waters of Long Island, New York, Morreale *et al*. (2005) documented the capture of more than twice as many green sea turtles in 2003 and 2004 with less pound net gear fished, compared to the number of green sea turtles captured in pound net gear in the area during the 1990s. Yet other studies have found no difference in the abundance (decreasing or increasing) of green sea turtles on foraging grounds in the Atlantic (Bjorndal *et al*. 2005; Epperly *et al*. 2007). Given the clear increases in nesting, however, it is likely that numbers on foraging grounds have increased by at least the same amount. We assume here that in-water abundance has increased at the same rate as Tortuguero nesting.

Based on the information provided above, the loss of up to two green sea turtles annually in the Atlantic as a result of the continued operation of the fisheries will not appreciably reduce the likelihood of survival for green sea turtles in the Atlantic given the increased nesting trend at the Atlantic nesting sites as well as measures that reduce the number of Atlantic green sea turtles that are injured and killed in the Atlantic (which should result in increases to the numbers of green sea turtles in the Atlantic that would otherwise have not occurred in the absence of those regulatory measures). The scallop fishery has no effects on green sea turtles that occur outside of the Atlantic. Therefore, in light of other ongoing actions affecting green sea turtles in the action area (including climate change), the continued operation of the fishery over the next ten years will not appreciably reduce the likelihood of survival for green sea turtles in the Atlantic. As a result, the proposed action will not appreciably reduce the likelihood of survival of the species.

The annual loss of up to two green sea turtles, together with an increase in nesting, is not expected to materially affect the increasing to stable trend in the number of green sea turtles on the foraging grounds in the Atlantic. Therefore, the continued operation of the fishery under the Scallop FMP will not appreciably reduce the likelihood of recovery for green sea turtles in the Atlantic. Since the fishery has no effects on green sea turtles that occur outside of the Atlantic, its continued operation will not appreciably reduce the likelihood of recovery for the species.

Even amidst an ongoing decline in the overall number of green sea turtles in the Mexican Pacific and the threat of climate change on the species as a whole, the potential loss of two green sea turtles annually from the Atlantic over the next ten years (and potentially beyond) is not likely to result in any additional threat of extinction within the foreseeable future throughout all or a significant portion of its range. The potential loss of two green sea turtles annually due to the scallop fishery only represents a decline in the North Atlantic adult female population by 0.007% at the greatest (two out of 29,243). Taking into account the number of Atlantic green sea turtles in other life stages as well as all those occurring throughout the Pacific (regardless of life stage) indicates how minor this level of annual mortality is in regards to the species achieving its recovery objectives.

8.1.5 Atlantic sturgeon

As explained above, the proposed action may result in the capture of up to one Atlantic sturgeon per year and one lethal removal every 20 years. As noted above, this sturgeon is most likely to be a subadult, but could also be an adult. We have considered the best available information to determine from which DPS this individual is likely to originate. Using a mixed stock analysis explained above, we have determined that Atlantic sturgeon in the action area likely originate from the five DPSs at the following frequencies: NYB 46%; SA 29%; CB 16%; GOM 8%; and Carolina 0.5%. In addition, it is possible that a small fraction (<1%) of Atlantic sturgeon in the action area may be of Canadian origin (*i.e.*, from the St. John River). Based on this information, the NYB DPS is likely to be the most prevalent DPS in the action area; however, this does not necessarily mean that the one Atlantic sturgeon potentially captured annually will come from this

DPS. Based on the available mixed stock analysis and genetics data, it is also possible that it could be from the GOM, CB, Carolina, or SA DPS. As a result, to be conservative we must look at the effects of one annual interaction, and one lethal interaction every 20 years, as if it came from any of the five DPSs.

Gulf of Maine DPS

Individuals originating from the GOM DPS are likely to occur in the action area. The GOM DPS has been listed as threatened. While Atlantic sturgeon occur in several rivers in the GOM DPS, recent spawning has only been documented in the Kennebec River. The capture of a larva 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 and 498 subadults in the GOM DPS. Approximately 1/3 of adults are likely to spawn each year. Gulf of Maine 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.

We have estimated that the proposed action will result in the annual capture of up to one Atlantic sturgeon, which may be a GOM DPS Atlantic sturgeon. The following analysis applies to the anticipated effects of one individual capture per year, and one mortality every 20 years, from the GOM DPS. The mortality of up to one Atlantic sturgeon from the GOM DPS every 20 years represents the removal of a very small percentage (0.15%) of the population only a few times a century. While the death of up to one Atlantic sturgeon every 20 years will reduce the number of GOM 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 population on an infrequent basis. As such, the reproductive potential of the GOM DPS is not expected to be significantly affected in any way other than through a reduction in population size by one individual every 20 years. As most sturgeon captured in trawl gear are anticipated to fully recover from capture, one annual interaction likely will not cause a delay or disruption of any essential behavior including spawning, nor will it cause a reduction in individual fitness or any future reduction in numbers of individuals. A reduction in the number of GOM DPS Atlantic sturgeon would have the effect of reducing the amount of potential reproduction as any dead GOM DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in a small reduction in the number of eggs laid or larvae produced in future years and similarly, a very small effect on the strength of subsequent year classes. Even considering the potential future spawners that would be produced by the individual that would be killed as a result of the proposed action, any effect to future year classes is anticipated to be very small and would not change the status of this species. Additionally, as the proposed action will occur outside of the rivers where GOM DPS fish are expected to spawn (*e.g.*, the Kennebec River), 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.

Based on the information provided above, the annual capture of up to one GOM DPS Atlantic sturgeon in the scallop fishery, and the mortality of one every 20 years, 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). The action will not affect GOM DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle, including reproduction, sustenance, and shelter. This is the case because: (1) the death of up to one GOM DPS Atlantic sturgeon every 20 years represents an extremely small percentage of the species as a whole; (2) the death of up to one GOM DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these GOM DPS Atlantic sturgeon is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of these GOM DPS Atlantic sturgeon is likely to have such a small effect on reproductive output that the loss of these individuals will not change the status or trends of the species; (5) the action will have no effect on the distribution of GOM DPS Atlantic sturgeon in the action area or throughout their range; and, (6) the action will have no effect on the ability of GOM DPS Atlantic sturgeon to shelter or forage.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the GOM DPS will survive in the wild. Here, we consider 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.

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 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 the mortality of one Atlantic sturgeon every 20 years; however, as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the GOM DPS of Atlantic sturgeon. As the reduction in numbers and future reproduction is very small, the loss of these individuals will not change the status or trend 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, which is the improvement in the species' status to the point its listing as threatened is no longer warranted, since the action will cause the mortality of only a very small percentage of the species on an infrequent basis and this mortality is not expected to result in the reduction of overall reproductive fitness for the species. Based on the analysis presented herein, the proposed action, resulting in the mortality of one GOM DPS Atlantic sturgeon every 20 years, is not likely to appreciably reduce the survival and recovery of this species.

New York Bight DPS

Individuals originating from the NYB DPS are likely to occur in the action area. The NYB DPS has been listed as endangered. While Atlantic sturgeon occur in several rivers in the NYB DPS, recent spawning has only been documented in the Delaware and Hudson Rivers. 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. We have also estimated, based on fishery-dependent data, that there are approximately 2,850 subadults in the NYB 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.

We have estimated that the proposed action will result in the annual capture of up to one Atlantic sturgeon, and one mortality every 20 years, which may be a NYB DPS Atlantic sturgeon. The majority of individuals are likely to be Hudson River origin, but some may be Delaware River origin. The mortality of up to one Atlantic sturgeon from the NYB DPS population every 20 years represents a very small percentage (0.03%) of the population. While the death of up to one 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 population. As such, the reproductive potential of the NYB DPS is not expected to be significantly affected in any way other than through a reduction in population size by one individual every 20 years. As most sturgeon captured in trawl gear are anticipated to fully recover from capture, one annual interaction likely will not cause a delay or disruption of any essential behavior including spawning, nor will it cause a reduction in individual fitness or any future reduction in numbers of individuals. A reduction in the number of NYB DPS Atlantic sturgeon would have the effect of reducing the amount of potential reproduction as any dead NYB DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in a small reduction in the number of eggs laid or larvae produced in future years and similarly, a very small effect on the strength of subsequent year classes. Even considering the potential future spawners that would be produced by the individual that would be killed as a result of the proposed action, any effect to future year classes is anticipated to be very small and would not change the status of this species. Additionally, as the proposed action will occur outside of the rivers where NYB DPS fish are expected to spawn (*e.g.*, the Hudson and Delaware Rivers), 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 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.

Based on the information provided above, the annual capture of up to one NYB DPS Atlantic sturgeon in the scallop fishery, and the mortality of one every 20 years, 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). The action will not affect NYB DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle, including reproduction, sustenance, and shelter. This is the case because: (1) the death of up to one NYB DPS Atlantic sturgeon every 20 years represents an extremely small percentage of the species as a whole; (2) the death of up to one NYB DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these NYB DPS Atlantic sturgeon is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of these 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 no effect on the distribution of NYB DPS Atlantic sturgeon in the action area or throughout their range; and, (6) the action will have no effect on the ability of NYB DPS Atlantic sturgeon to shelter or forage.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the NYB DPS will survive in the wild. Here, we consider 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 an extremely small reduction in the number of NYB DPS Atlantic sturgeon and since 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 or affect the adequacy of existing regulatory mechanisms to protect this species. The proposed action is not likely to result in the mortality of one Atlantic sturgeon every 20 years; however, as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the NYB DPS of Atlantic sturgeon. As the reduction in numbers and future reproduction is very small, the loss of these individuals will not change the status or trend of the NYB DPS of Atlantic sturgeon. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery, which is the improvement in the species' status to the point its listing as endangered is no longer warranted, since the action will cause the mortality of only a very small percentage of the species on an infrequent basis and this mortality is not expected to result in the reduction of overall reproductive fitness for the species. Based on the analysis presented herein, the proposed action, resulting in the mortality of one NYB DPS Atlantic sturgeon every 20 years, is not likely to appreciably reduce the survival and recovery of this species.

Chesapeake Bay DPS
Individuals originating from the CB DPS are likely to occur in the action area. The CB DPS has been listed as endangered. While Atlantic sturgeon occur in several rivers in the CB DPS, recent spawning has only been documented in the James River. Using fishery-dependent data, we have estimated that there are 329 adults and 987 subadults 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 and subadults in the CB DPS. CB 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.

We have estimated that the proposed action will result in the annual capture of up to one Atlantic sturgeon, which may be a CB DPS Atlantic sturgeon. The following analysis applies to the anticipated effects of one individual capture per year, and one mortality every 20 years, from the CB DPS. The mortality of up to one Atlantic sturgeon from the CB DPS every 20 years represents the removal of a very small percentage (0.08%) of the population only a few times a century. While the death of up to one Atlantic sturgeon every 20 years will reduce the number of CB DPS Atlantic sturgeon compared to the number that would have been present absent the proposed action, it is not likely that this reduction in numbers will change the status of this species as this loss represents a very small percentage of the population on a highly infrequent basis. As such, the reproductive potential of the CB DPS is not expected to be significantly affected in any way other than through a reduction in population size by one individual every 20 years. As most sturgeon captured in trawl gear are anticipated to fully recover from capture, one annual interaction likely will not cause a delay or disruption of any essential behavior including spawning, nor will it cause a reduction in individual fitness or any future reduction in numbers of individuals. A reduction in the number of CB DPS Atlantic sturgeon would have the effect of reducing the amount of potential reproduction as any dead CB DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in a small reduction in the number of eggs laid or larvae produced in future years and similarly, a very small effect on the strength of subsequent year classes. Even considering the potential future spawners that would be produced by the individual that would be killed as a result of the proposed action, any effect to future year classes is anticipated to be very small and would not change the status of this species. Additionally, as the proposed action will occur outside of the rivers where CB DPS fish are expected to spawn (*e.g.*, the James River), 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 and handling of individuals.

Based on the information provided above, the capture of one CB DPS Atlantic sturgeon per year in the scallop fishery, and the mortality of one every 20 years, 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). The action will not affect CB DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle, including reproduction, sustenance, and shelter. This is the case because: (1) the death of up to one CB DPS Atlantic sturgeon every 20 years represents an extremely small percentage of the species as a whole; (2) the death of up to one CB DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these CB DPS Atlantic sturgeon is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of these CB DPS Atlantic sturgeon is likely to have such a small effect on reproductive output that the loss of these individuals will not change the status or trends of the species; (5) the action will have no effect on the distribution of CB DPS Atlantic sturgeon in the action area or throughout their range; and, (6) the action will have no effect on the ability of CB DPS Atlantic sturgeon to shelter or forage.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the CB DPS will survive in the wild. Here, we consider 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 an extremely small 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 likely to result in the mortality of one Atlantic sturgeon every 20 years; however, as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the CB DPS. As the reduction in numbers and future reproduction is very small, the loss of these individuals will not change the status or trend of the CB DPS of Atlantic sturgeon. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery, which is the improvement in the species' status to the point its listing as endangered is no longer warranted, since the action will cause the mortality of only a very small percentage of the species on an infrequent basis and this mortality is not expected to result in the reduction of overall reproductive fitness for the species. Based on the analysis presented herein, the proposed action, resulting in the mortality of one CB DPS Atlantic sturgeon every 20 years, is not likely to appreciably reduce the survival and recovery of this species.

Carolina DPS

Individuals originating from the Carolina DPS are likely to occur in the action area. The Carolina DPS has been listed as endangered. While Atlantic sturgeon occur in several rivers of the Carolina DPS, recent spawning has only been documented in the Roanoke, Tar-Pamlico, Cape Fear, Waccamaw, and Pee Dee Rivers. Spawning is unknown or believed to be extirpated in other rivers within the DPS. There is no published population estimate for the DPS or total estimate for any river within the DPS. There are estimated to be less than 300 spawning adults (total of both sexes) in each of the five spawning rivers in the Carolina DPS; for a total estimate of less than 1,500 adults. Our fishery dependent estimate is 496. However, it is possible that this is an underestimate of the total number of adults in the Carolina DPS because it is based on the estimate for the SA DPS which may actually only be an estimate for the Savannah and Ogeechee Rivers rather than the DPS as a whole. Carolina DPS origin Atlantic sturgeon are affected by numerous sources of human induced mortality and habitat disturbance throughout the riverine and marine portions of their range. There is currently not enough information to establish a trend for any life stage, for any of the spawning populations, or for the DPS as a whole.

We have estimated that the proposed action will result in the annual capture of up to one Atlantic sturgeon, which may be a Carolina DPS Atlantic sturgeon. The following analysis applies to the anticipated effects of one individual capture per year, and one mortality every 20 years, from the Carolina DPS. The mortality of up to one Atlantic sturgeon from the Carolina DPS every 20 years represents the removal of a very small percentage (0.05%) of the population only a few times a century. While the death of up to one Atlantic sturgeon every 20 years will reduce the number of Carolina 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 population on a highly infrequent basis. As such, the reproductive potential of the Carolina DPS is not expected to be significantly affected in any way other than through a reduction in population size by one individual every 20 years. As most sturgeon captured in trawl gear are anticipated to fully recover from capture, one annual interaction likely will not cause a delay or disruption of any essential behavior including spawning, nor will it cause a reduction in individual fitness or any future reduction in numbers of individuals. A reduction in the number of Carolina DPS Atlantic sturgeon would have the effect of reducing the amount of potential reproduction as any dead Carolina DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in a small reduction in the number of eggs laid or larvae produced in future years and similarly, a very small effect on the strength of subsequent year classes. Even considering the potential future spawners that would be produced by the individual that would be killed as a result of the proposed action, any effect to future year classes is anticipated to be very small and would not change the status of this species. Additionally, as the proposed action will occur outside of the rivers where Carolina DPS fish are expected to spawn, 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 Carolina 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 and handling of individuals.

Based on the information provided above, the capture of one Carolina DPS Atlantic sturgeon per year in the scallop fishery, and the mortality of one every 20 years, 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). The action will not affect Carolina DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle, including reproduction, sustenance, and shelter. This is the case because: (1) the death of up to one Carolina DPS Atlantic sturgeon every 20 years represents an extremely small percentage of the species as a whole; (2) the death of up to one Carolina DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these Carolina DPS Atlantic sturgeon is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of these Carolina 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 no effect on the distribution of Carolina DPS Atlantic sturgeon in the action area or throughout their range; and, (6) the action will have no effect on the ability of Carolina DPS Atlantic sturgeon to shelter or forage.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the Carolina DPS will survive in the wild. Here, we consider 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 an extremely small reduction in the number of Carolina DPS Atlantic sturgeon and since it will not affect the overall distribution of Carolina DPS Atlantic sturgeon. The proposed action will not utilize Carolina 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 likely to result in the mortality of one Atlantic sturgeon every 20 years; however, as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the Carolina DPS. As the reduction in numbers and future reproduction is very small, the loss of these individuals will not change the status or trend of the Carolina DPS of Atlantic sturgeon. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery, which is the improvement in the species' status to the point its listing as endangered is no longer warranted, since the action will cause the mortality of only a very small percentage of the species on an infrequent basis and this mortality is not expected to result in the reduction of overall reproductive fitness for the species. Based on the analysis presented herein, the proposed action, resulting in the mortality of one Carolina DPS Atlantic sturgeon every 20 years, is not likely to appreciably reduce the survival and recovery of this species.

South Atlantic DPS
Individuals originating from the SA DPS are likely to occur in the action area. The SA DPS has been listed as endangered. Spawning occurs in multiple rivers in the SA DPS but spawning populations have been extirpated in some rivers 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 SA 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 indicates that only individuals from the Savannah and Ogeechee Rivers 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. We have estimated, based on fishery-dependent data, that there are also approximately 1,794 subadults in the SA DPS. SA DPS origin

Atlantic sturgeon are affected by numerous sources of human induced mortality and habitat disturbance throughout the riverine and marine portions of their range. There is currently not enough information to establish a trend for any life stage, for any spawning population or for the DPS as a whole.

We have estimated that the proposed action will result in the annual capture of up to one Atlantic sturgeon, and one mortality every 20 years, which may be a SA DPS Atlantic sturgeon. The mortality of up to one Atlantic sturgeon from the SA DPS population every 20 years represents a very small percentage (0.04%) of the population. While the death of up to one Atlantic sturgeon will reduce the number of SA DPS Atlantic sturgeon compared to the number that would have been present absent the proposed action, it is not likely that this reduction in numbers will change the status of this species as this loss represents a very small percentage of the population. As such, the reproductive potential of the SA DPS is not expected to be significantly affected in any way other than through a reduction in population size by one individual every 20 years. As most sturgeon captured in trawl gear are anticipated to fully recover from capture, one annual interaction likely will not cause a delay or disruption of any essential behavior including spawning, nor will it cause a reduction in individual fitness or any future reduction in numbers of individuals. A reduction in the number of SA DPS Atlantic sturgeon would have the effect of reducing the amount of potential reproduction as any dead SA DPS Atlantic sturgeon would have no potential for future reproduction. This small reduction in potential future spawners is expected to result in a small reduction in the number of eggs laid or larvae produced in future years and similarly, a very small effect on the strength of subsequent year classes. Even considering the potential future spawners that would be produced by the individual that would be killed as a result of the proposed action, any effect to future year classes is anticipated to be very small and would not change the status of this species. Additionally, as the proposed action will occur outside of the rivers where SA DPS fish are expected to spawn (*e.g.*, the Altamaha River), 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 and handling in the trawl.

Based on the information provided above, the annual capture of one SA DPS Atlantic sturgeon in the scallop fishery, and one mortality every 20 years, 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). The action will not affect SA DPS Atlantic sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, and it will not result in effects to the environment which would prevent Atlantic sturgeon from completing their entire life cycle, including reproduction, sustenance, and shelter. This is the case because: (1) the death of up to one SA DPS Atlantic sturgeon every 20 years represents an extremely small percentage of the species as a whole; (2) the death of up to one SA DPS Atlantic sturgeon will not change the status or trends of the species as a whole; (3) the loss of these SA DPS Atlantic sturgeon is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the loss of these SA DPS Atlantic sturgeon is likely to have such a small effect on reproductive output that the loss of these individuals will not change the status or trends of the species; (5) the action will have no effect on the distribution of SA DPS Atlantic sturgeon in the action area or throughout their range; and, (6) the action will have no effect on the ability of SA DPS Atlantic sturgeon to shelter or forage.

In certain instances, an action that does not appreciably reduce the likelihood of a species' survival might affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, we have determined that the proposed action will not appreciably reduce the likelihood that the SA DPS will survive in the wild. Here, we consider 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 an extremely small reduction in the number of SA DPS Atlantic sturgeon and since it will not affect 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 likely to result in the mortality of one Atlantic sturgeon every 20 years; however, as explained above, the loss of these individuals and what would have been their progeny is not expected to affect the persistence of the SA DPS of Atlantic sturgeon. As the reduction in numbers and future reproduction is very small, the loss of these individuals will not change the status or trend of the SA DPS of Atlantic sturgeon. The effects of the proposed action will not delay the recovery timeline or otherwise decrease the likelihood of recovery, which is the improvement in the species' status to the point its listing as endangered is no longer warranted, since the action will cause the mortality of only a very small percentage of the species on an infrequent basis and this mortality is not expected to result in the reduction of overall reproductive fitness for the species. Based on the analysis presented herein, the proposed action, resulting in the mortality of one SA DPS Atlantic sturgeon every 20 years, is not likely to appreciably reduce the survival and recovery of this species.

9.0 CONCLUSION

After reviewing the current status of the species, the environmental baseline and cumulative effects in the action area, and the effects of the continued operation of the scallop fishery under the Scallop FMP, it is our biological opinion that the proposed action may adversely affect, but is not likely to jeopardize, the continued existence of loggerhead (specifically, the NWA DPS), leatherback, Kemp's ridley, and green sea turtles, or the GOM, NYB, CB, Carolina, and SA DPSs of Atlantic sturgeon. It is also our biological opinion that the proposed action will not affect shortnose sturgeon, the Gulf of Maine DPS of Atlantic salmon, and hawksbill sea turtles, and is not likely to adversely affect North Atlantic right whales, humpback whales, fin whales, sei whales, blue whales, sperm whales, or designated critical habitat for right whales in the Northwest Atlantic.

10.0 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, unless a special exemption has been granted. Take is defined as "to harass, harm, pursue, hunt, shoot, capture, or collect, or to attempt to engage in any such conduct." Incidental take is defined as take that is incidental to, and not the purpose of, the execution of an otherwise lawful activity. Under the terms of sections 7(b)(4) and 7(o)(2), taking that is incidental to and not intended as part of the action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement (ITS).

The prohibitions against incidental take are currently in effect for all four species of sea turtles and all DPSs of Atlantic sturgeon except the threatened GOM DPS. A final section 4(d) rule for the GOM DPS of Atlantic sturgeon, which we anticipate to be published in the *Federal Register* by the end of July 2012 and to take effect 30 days following publication, will prohibit take. The proposed 4(d) rule for the GOM DPS was published on June 10, 2011 (76 FR 34023). The prohibitions on take of GOM DPS Atlantic sturgeon will take effect on the date the final 4(d) rule is effective and so are the exemptions provided by this ITS pertaining to the GOM DPS.

When a proposed NMFS action is found to be consistent with section 7(a)(2) of the ESA, section 7(b)(4) of the ESA requires NMFS to issue a statement specifying the impact of incidental taking, if any. It also states that reasonable and prudent measures necessary to minimize impacts of any incidental take be provided along with implementing terms and conditions. The measures described below are non-discretionary and must therefore be undertaken in order for the exemption in section 7(o)(2) to apply. Failure to implement the terms and conditions through enforceable measures may result in a lapse of the protective coverage section of 7(o)(2).

Anticipated Amount or Extent of Incidental Take

Based on the Murray (2011) and Warden (2011a) reports, incidental take data from observer reports for the scallop and other fisheries using similar gear, and the distribution and abundance of sea turtles and Atlantic sturgeon in the action area, we anticipate that the continued operation of the scallop fishery may result in the incidental take of ESA-listed species as follows[13]:

- for the NWA DPS of loggerhead sea turtles, we anticipate (a) the annual average take of up to 161 individuals in dredge gear, of which up to 129 per year may be lethal in 2012 and up to 46 per year may be lethal in 2013 and beyond, and (b) the annual average take of up to 140 individuals in trawl gear, of which up to 66 per year may be lethal;

- for leatherback sea turtles, we anticipate the annual lethal take of up to two individuals in dredge and trawl gear combined;

- for Kemp's ridley sea turtles, we anticipate the annual take of up to three individuals in dredge and trawl gear combined (for 2012, up to three takes are anticipated to be lethal, while for 2013 and beyond, up to two takes are anticipated to be lethal);

- for green sea turtles, we anticipate the annual lethal take of up to two individuals in dredge and trawl gear combined;

- for Atlantic sturgeon, we anticipate the annual take of up to one individual from either the GOM, NYB, CB, Carolina, or SA DPS in trawl gear; once every 20 years this take is expected to result in mortality.

Anticipated Impact of Incidental Take

NMFS has concluded that the continued operation of the scallop fishery under the Scallop FMP may adversely affect, but is not likely to jeopardize, the NWA DPS of loggerhead sea turtles, leatherback sea turtles, Kemp's ridley sea turtles, green sea turtles, or the five DPSs of Atlantic sturgeon (GOM, NYB, CB, Carolina, and SA). Nevertheless, NMFS must take action to minimize these takes. The following Reasonable and Prudent Measures (RPMs) have been identified as ways to minimize sea turtle and Atlantic sturgeon interactions with the scallop fishery and to monitor incidental take to provide a trigger for reinitiation and a check on analyses and assumptions in this Opinion. These measures are non-discretionary and must be implemented by NMFS. Some of these measures for sea turtles were included as RPMs in the March 14, 2008, Opinion. They are included here because they still meet the criteria for a RPM and reflect work in progress to minimize the taking of sea turtles in scallop fishing gear.

[13] For sea turtles other than loggerheads, the estimated observed take is for combined gear type. Effort within the fishery may shift from year to year between gear types and, therefore, we believe it is most appropriate to have a total estimated observed take number. For loggerheads, the incidental take statement includes separate estimates for dredges and trawls. This is due to the fact that the take estimates for the gear types are calculated somewhat differently. However, we are choosing to use the upper end of the 95% CI for expected takes in both the dredge and trawl fisheries as calculated by Murray (2011) and Warden (2011a) to ensure consistency across gear types.

Reasonable and Prudent Measures
NMFS has determined that the following RPMs are necessary or appropriate to minimize impacts of the incidental take of sea turtles and the five DPSs of Atlantic sturgeon in the scallop fishery:

1. NMFS must annually monitor and assess the distribution of fishing effort in the Mid-Atlantic scallop dredge fishery during the period of known sea turtle overlap (May through November) to ensure that there are no increases in the likelihood of interactions with sea turtles that may result from increased effort.

2. NMFS must continue to investigate and implement, within a reasonable time frame following sound research, modifications to gears used in these fisheries to reduce incidental takes of sea turtles and Atlantic sturgeon and the severity of the interactions that occur.

3. NMFS must continue to review available data to determine whether there are areas or conditions within the action area where sea turtle and Atlantic sturgeon interactions with fishing gear used in the scallop fishery are more likely to occur.

4. NMFS must continue to quantify the extent to which chain mats and TDDs reduce the number of serious injuries/deaths of sea turtles that interact with scallop dredge gear.

5. NMFS must continue to research the extent to which sea turtle interactions with scallop dredge gear occur on the bottom versus within the water column.

6. NMFS must ensure that any sea turtles incidentally taken in scallop dredge or trawl gear and any Atlantic sturgeon incidentally taken in scallop trawl gear are handled in a way as to minimize stress to the animal and increase its survival rate.

7. NMFS must seek to ensure that monitoring and reporting of any sea turtles and Atlantic sturgeon encountered in scallop fishing gear: (1) detects any adverse effects such as injury or mortality; (2) detects whether the anticipated level of take has occurred or been exceeded; and (3) collects data from individual encounters.

8. NMFS must continue to engage in outreach efforts with commercial fishermen regarding the proper installation and use of chain mats on their scallop dredges.

Terms and Conditions
In order to be exempt from the prohibitions of section 9 of the ESA, and regulations issued pursuant to section 4(d), NMFS must comply with the following terms and conditions, which implement the reasonable and prudent measures described above. These terms and conditions are non-discretionary.

1. To comply with RPM #1 above, NMFS must continue to monitor dredge hours in the Mid-Atlantic scallop dredge fishery during the months of May through November when sea turtle interactions are most likely to occur. NMFS must collect and review effort data as stipulated under the monitoring plan below (*i.e.*, two-year running averages) to determine if dredge effort in the Mid-Atlantic is on the rise, and, if needed, re-evaluate the monitoring plan methodology annually in the event more refined methods become available through discussions within the agency or with the NEFMC or scallop industry. The calculation and comparison of two-year running averages should also be performed on an annual basis, with 2007-2008 serving as the baseline effort levels post-chain mats.

2. To comply with RPM #2 above, NMFS must continue to investigate modifications to scallop dredge and trawl gear to further minimize adverse effects on sea turtles due to collisions with and/or entrainment in the gear. Through continued experimental gear trials from or by any source (*e.g.*, through the Scallop RSA program), NMFS and its partners must review all data collected from those trials, determine the next appropriate course of action (*e.g.*, expanded gear testing, further gear modification, rulemaking to require the gear modification), and initiate management action based on the determination. These trials may include further refinements of and improvements to the TDD as well as continued testing and evaluation of modified trawls (*e.g.*, trawls with TEDs, topless trawls).

3. To comply with RPM #3 above, NMFS must continue to review all available data on the incidental take of sea turtles in the scallop fishery (observable plus unobservable, quantifiable) and other suitable information (*e.g.*, data on observed sea turtle interactions with other trawl fisheries, sea turtle distribution information, or fishery surveys in the area where the scallop fishery operates) to assess whether correlations with environmental conditions (*e.g.*, depth, SST, salinity) or other drivers of incidental take (*e.g.*, gear configuration) can be made for some or all portions of the action area. If additional analysis is deemed appropriate, within a reasonable amount of time after completing the review, NMFS must take action, if appropriate, to reduce sea turtle interactions and/or their impacts.

4. To comply with RPM #4 above, NMFS must continue to use available and appropriate technologies to quantify the extent to which chain mats and TDDs reduce the number of serious injuries/deaths of sea turtles that interact with scallop dredge gear. This information is necessary to better determine the extent to which these two gear modifications reduce injuries leading to death for sea turtles and may result in further modifications of the fishery to ensure sea turtle interactions, including those causing serious injuries and mortalities are minimized.

5. To comply with RPM#5 above, NMFS must continue to use available and appropriate technologies to better determine where (on the bottom or in the water column) and how sea turtle interactions with scallop dredge gear are occurring. Such information is necessary to assess whether further gear modifications in the scallop dredge fishery will

actually provide a benefit to sea turtles by either reducing the number of interactions or the number of interactions causing serious injury and mortality.

6. To comply with RPM #6 above, NMFS must ensure that all Federal permit holders in the scallop fishery possess handling and resuscitation guidelines for sea turtles and Atlantic sturgeon. For sea turtles, all Federally-permitted fishing vessels should have the handling and resuscitation requirements listed in 50 CFR 223.206(d)(1) and as reproduced in Appendix C. For Atlantic sturgeon, NMFS must instruct fishermen and observers to resuscitate any individuals that may appear to be dead by providing a running source of water over the gills.

7. To also comply with RPM #6 above, NMFS must continue to develop and distribute training materials for commercial fishermen regarding the use of recommended sea turtle and Atlantic sturgeon release equipment and protocols. Such training materials would be able to be brought onboard fishing vessels and accessed upon incidental capture (*e.g.*, CD that could be used in on-board computer, placard, etc.).

8. To comply with RPM #7 above, NMFS must continue to place observers onboard scallop dredge and trawl vessels to document and estimate incidental bycatch of sea turtles and Atlantic sturgeon. Monthly summaries and an annual report of observed sea turtle takes in gears primarily landing scallops must be provided to the NERO Protected Resources Division. A similar data reporting plan must be developed for Atlantic sturgeon.

9. To also comply with RPM #7 above, NMFS must continue to instruct observers to tag and take tissue samples from incidentally captured sea turtles as stipulated under their ESA section 10 permit. The current NEFOP protocols are to tag any sea turtles caught that are larger than 26 centimeters in notch-to-tip carapace length and to collect tissue samples for genetic analysis from any sea turtles caught that are larger than 25 centimeters in notch-to-tip carapace length. NMFS must continue to instruct observers to send any genetic samples of sea turtles taken to the NEFSC. NMFS must further instruct observers to take fin clips from all incidentally captured Atlantic sturgeon and send them to NMFS for genetic analysis. Fin clips must be taken according to the procedures outlined in Appendix D and prior to preservation of other fish parts or whole bodies.

10. To also comply with RPM #7 above, NMFS must continue to reconvene the Sea Turtle Injury Working Group in order to better assess and evaluate injuries sustained by sea turtles in scallop dredge and trawl gear, and their potential impact on sea turtle populations. New data should be reviewed on an annual basis.

11. To comply with RPM #8 above, NMFS must distribute information to scallop permit holders specifying the chain mat and TDD regulations and be prepared to provide them assistance to resolve issues that may cause chain mats or any components of the TDD to be rigged improperly or malfunction.

Justification for Proposed Reasonable and Prudent Measures and Terms and Conditions
The RPMs, 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 monitors the impacts of the proposed action in a way that allows for the detection, identification, and reporting of all interactions with ESA-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 Term and Condition #1 are necessary and appropriate because they allow NMFS to continually track changes in scallop fishing effort from year to year so that the agency can adapt its management approach in order to minimize both the quantity and severity of sea turtle interactions with the scallop fishery. In the absence of an effort monitoring program, increases in the susceptibility of sea turtles to takes in the scallop fishery due to increased effort in areas where sea turtles are most abundant could largely go unnoticed due to the use of gear modifications that make observing interactions difficult.

RPM #2 and Term and Condition #2 are necessary and appropriate because they allow NMFS to design, research, and implement the most advanced gear modifications believed to have the lowest potential of interactions with sea turtles. If gear modifications are implemented, rulemaking will be completed in a timely manner in which to minimize any increase in costs or any decrease in efficiency of the fisheries, representing only a minor change to the actions.

RPM #3 and Term and Condition #3 are necessary and appropriate because they allow NMFS to ensure avoidable sea turtle and Atlantic sturgeon takes are not occurring due to currently unknown environmental conditions or other parameters present in the action area. If regulations are implemented, rulemaking will be done in a manner in which to minimize any increase in costs or any decrease in efficiency of the fisheries, representing only a minor change to the actions.

RPM #4 and Term and Condition #4 are necessary and appropriate because they allow NMFS to evaluate the success of and troubleshoot recently implemented gear modifications designed to reduce the severity of interactions between sea turtles and scallop dredge gear. Follow-up studies to Milliken *et al*. (2007) and Smolowitz *et al*. (2010) utilizing similar gear configurations and methods could provide more robust estimates of the conservation benefits of chain mats and TEDs when used both together and separately. Repeated field testing of scallop dredges using photographic and video-based analysis has led to the currently required designs and constructs of both gear modifications, which were adapted over time to successfully exclude sea turtles while also retaining sizeable catches of scallops and minimizing bycatch of non-target fish species. If additional regulations are implemented to further modify the chain mat and TDD designs, rulemaking will be done in a manner in which to minimize any increase in costs or any decrease in efficiency of the fisheries, representing only a minor change to the actions.

RPM #5 and Term and Condition #5 are necessary and appropriate to determine the location in the water column where most sea turtle interactions with scallop dredge gear are occurring. Due to their life histories and foraging behaviors, certain sea turtle species (*e.g.*, hard shelled sea turtles) are likely more prone to interactions on the bottom, while others (*e.g.*, leatherbacks) are likely more prone to interactions in the water column. Such information is necessary to assess whether further gear modifications in the scallop dredge fishery will actually provide a benefit to sea turtles by either reducing the number of interactions or the number of interactions causing serious injury and mortality. ROV work was conducted in 2009, 2010, and 2011 and has already provided information on behavior of sea turtles in waters where the scallop fleet operates. Also from 2009-2011, approximately 40 satellite tags were placed on sea turtles which provided information towards addressing vertical distribution. Continuing this research can provide an even larger, more robust data set on sea turtle depth preferences throughout the action area.

RPM #6 and Terms and Conditions #6 and #7 are necessary and appropriate to ensure that any sea turtles or Atlantic sturgeon that survive capture or entanglement in gear are given the maximum probability of remaining alive and not suffering additional injury or subsequent mortality through inappropriate handling. This is only a minor change as following these procedures is not expected to result in an increase in cost or a decrease in the efficiency of the operation of these fisheries.

RPM #7 and Terms and Conditions #8, #9, and #10 are necessary and appropriate to ensure the proper documentation of any interactions with sea turtles and Atlantic sturgeon as well as requiring that these interactions are reported to NMFS in a timely manner with all the necessary information. This is essential for monitoring the level of incidental take associated with the scallop fishery. Compliance with these terms and conditions will allow NMFS to determine if reinitiation of consultation is necessary at the time that take occurs. The data and information collected can be used to refine our current management measures, and is not just a count of dead or injured individuals. This RPM and its Terms and Conditions represent only a minor change as compliance is not expected to result in an increase in cost or a decrease in the efficiency of scallop fishery operations.

The taking of genetic samples (*e.g.*, biopsies, fin clips) allows NMFS to run genetic analysis to determine the DPS of origin or nesting/spawning stock for sea turtles and Atlantic sturgeon. This allows us to determine if the actual level of take has been exceeded. These procedures do not harm sea turtles or Atlantic sturgeon and are common practices in fisheries science. Tissue sampling does not appear to impair an individual’s ability to swim and is not thought to have any long-term adverse impact. This represents only a minor change as following these procedures will have an insignificant impact on the proposed actions.

RPM #8 and Term and Condition #11 are necessary and appropriate because they allow NMFS to ensure that modified gear requirements are followed by the fishing industry so that sea turtle takes can be minimized to the extent possible. Any outreach activities will be done in a manner in which to minimize any increase in costs or any decrease in efficiency of the scallop fishery, representing only a minor change to the action.

Monitoring
NMFS must continue to monitor levels of sea turtle bycatch in the scallop fishery. Observer coverage has been used as the principal means to estimate sea turtle bycatch in the scallop fishery and to monitor incidental take levels. NMFS must continue to use observer coverage to monitor sea turtle bycatch in commercial dredge and trawl gear that is authorized by the Scallop FMP when that gear is used in areas or at times when chain mats and TDDs are not required.

The use of chain mats and TDDs is expected to greatly reduce the likelihood that sea turtles struck by or incidentally swimming into scallop dredge gear would go under the dredge and enter the dredge bag (71 FR 50361, August 25, 2006; NEFMC 2011b). Therefore, given that scallop dredge vessels are required to use these gear modifications throughout much of the Mid-Atlantic during times when sea turtles are most abundant in the action area (TDD requirement effective on May 1, 2013), injuries to sea turtles that occur as a result of the sea turtle being struck by the dredge gear underwater will continue to occur but will not be observed unless the sea turtle is small enough to pass underneath the low-profile dredge frame and between the chains, where it can enter the dredge bag, or is otherwise caught on the dredge frame and carried to the surface. This also means that observer coverage of scallop dredge vessels will be less effective in monitoring takes of sea turtles in the dredge component of the scallop fishery.

As we did during the development of the 2008 Opinion, we have considered the use of underwater video on scallop dredge vessels to monitor sea turtle interactions with the gear. Based on the information currently available as well as the hardships experienced during previous use of this technology in studies of sea turtle interactions with scallop dredge gear, we believe that the use of underwater video monitoring for monitoring the take of sea turtles in scallop dredge gear remains infeasible (Memo from N. Thompson, NEFSC to P. Kurkul, NERO, October 16, 2007). We have also revisited whether chains mats should be removed from scallop dredge gear during some observed trips to assess the number of sea turtle interactions that were occurring when chain mats were on the gear. However, we have again determined that this is not a reasonable method for monitoring sea turtle interactions with the dredge component of the scallop fishery given that the removal of the chains will likely increase the number of serious injuries and mortalities of sea turtles in comparison to the numbers that would have occurred if chains were present.

As described in the 2008 Opinion on the Scallop FMP, we requested guidance from the NEFSC on methods to monitor sea turtle takes (*e.g.*, captures, collisions) in the dredge component of the scallop fishery once the chain mat rule was approved and implemented. The NEFSC provided information on fishery dependent and fishery independent approaches they considered for monitoring interactions between sea turtles and scallop dredge gear and the reasonableness of each approach. The methods and analyses in Murray (2011) and Warden and Murray (2011) are the results of this initiative.

With the release of these two reports and the success of their methods in determining bycatch rates, we now have two options for monitoring the ITS of this Opinion that we will use in combination. First, we will continue to use dredge hours as a surrogate measure of actual takes, and find that the ITS provided with this Opinion has been exceeded when the fishery operates in a manner that, based on the best available information, would reasonably result in greater sea turtle interactions with scallop dredge gear than what is estimated to have occurred in a given time period (for example, a two-year period from 2007-2008, which are the first two years after chain mats were required and the last two years included in the Murray [2011] analysis). This is what was done in the 2008 Opinion, in which 2003-2004 was used as the benchmark. Given that the likelihood of sea turtle interactions with scallop dredge gear is higher in Mid-Atlantic waters as compared to waters further north (*e.g.*, Georges Bank) and given that sea turtle interactions with scallop dredge gear are likely only from May through November each year, we will monitor sea turtle interactions with scallop dredge gear by:

- using "dredge hour" as the measure of scallop fishing effort for the purpose of monitoring sea turtle interactions with scallop dredge gear;
- using the average of the total number of dredge hours for Mid-Atlantic waters during the period of May through November 2007 and May through November 2008 as the benchmark against which the two-year running average of dredge hours for each subsequent May through November period of each scallop fishing year will be compared; and,
- consider the ITS provided with this Opinion to have been exceeded if the two-year running average of dredge hours in Mid-Atlantic waters (inclusive of NMFS statistical areas between 525 and 700, excluding areas 538, 539, 551, 561, and 562) during the period of May through November of any scallop fishing year is greater than the average of the total number of dredge hours for Mid-Atlantic waters (as far south as Cape Hatteras) during the same period of 2007 and 2008.

In addition, the model developed in Murray (2011) provides a tool to monitor sea turtle interactions with chain mat-equipped dredge gear. The NEFSC undertook this analysis in an attempt to develop a bycatch estimate for the scallop dredge fishery that incorporated both observable and unobservable, quantifiable interactions. However, there are a number of caveats associated with this analysis, especially with TDDs planned for implementation in 2013 (which will further decrease the number of takes observed in the dredge fishery). And with each new year of data, hauls without chain mats will only be from the winter (December through April), and therefore will not represent a random sample. Over the whole time series, hauls without chain mats will be clumped in the early years, and will also become disproportionally smaller in the dataset (Murray 2011). Nonetheless, we will continue to investigate how the methods from Murray (2011) can be used to be monitor the ITS for sea turtles in future years. As a result, we propose that monitoring of the ITS will occur via a combination of the dredge hour surrogate described above and the methods to estimate sea turtle/dredge interactions as described in Murray (2011). The use of both methods is reasonable in that it will help to hedge against the limitations of using just one method on its own.

For the purposes of monitoring this ITS for the trawl component of the fishery, we will continue to use observer coverage as the primary means of collecting incidental take information. The loggerhead sea turtle take estimate described in this Opinion was generated using a statistical estimate that is not feasible to conduct on an annual basis. Conducting such statistical estimates are infeasible on an annual basis due to the data needs; length of time to develop, review, and finalize the estimates; and methodology used. As this estimate depends on take rate information over a several year period, re-examination after one year is not likely to produce any noticeable change in the take rate. For these reasons, approximately every five years, we will re-estimate takes in the scallop trawl fishery using appropriate statistical methods. For sea turtle species other than loggerheads and the five Atlantic sturgeon DPSs, we will use all available information (*e.g.*, observed takes, changes in Mid-Atlantic fishing effort identical to those mentioned above, etc.) to determine if the annual incidental take level in this Opinion has been exceeded.

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 adverse effects of a proposed action on ESA-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/Atlantic sturgeon conservation:

1. NMFS should continue to collect and analyze biological samples from sea turtles and Atlantic sturgeon incidentally taken in fishing gear targeting scallops to determine the nesting/DPS origin of these individuals taken in the scallop fishery in order to better assess the effects of the fishery on nesting groups/recovery units/DPSs and address those effects accordingly. NMFS should review its policies/protocols for the processing of genetics samples to determine what can be done to improve the efficiency and speed for obtaining results of genetic samples taken from all incidentally taken sea turtles and Atlantic sturgeon.

2. NMFS should establish a protocol for bringing to shore any sea turtle or Atlantic sturgeon incidentally taken in scallop fishing gear that is fresh dead, that dies on the vessel shortly after the gear is retrieved, or dies following attempts at resuscitation in accordance with the regulations. Such protocol should include the steps to be taken to ensure that the carcass can be safely and properly stored on the vessel, properly transferred to appropriate personnel for examination, as well as identify the purpose for examining the carcass and the samples to be collected. Port samplers and observers should also be trained in the protocols for notification of the appropriate personnel in the event that a vessel comes into port with a sea turtle or Atlantic sturgeon carcass.

3. NMFS should support studies on the seasonal distribution and abundance of sea turtles and Atlantic sturgeon in the action area, behavioral studies to improve our understanding of sea turtle and Atlantic sturgeon species interactions with fishing gear, foraging studies

including prey abundance/distribution studies (which may influence sea turtle and Atlantic sturgeon distribution), as well as studies and analysis necessary to develop population estimates for sea turtles and Atlantic sturgeon.

4. NMFS should work with the states to promote the permitting of activities (*e.g.*, state permitted fisheries, state agency in-water surveys) that are known to incidentally take sea turtles and Atlantic sturgeon.

5. NMFS should continue to cooperate with the Canadian Government to compare research findings and facilitate implementation in both countries of the most promising risk-reduction practices for sea turtles and Atlantic sturgeon.

6. NMFS should continue to work with the NEFMC to assess trends in the fishery in relation to effort distribution by time and area; landings by port, permit, and gear type; scallop biomass and recruitment amongst Mid-Atlantic and Georges Bank access areas; and stock assessment/bycatch reduction priorities. All of these factors likely influence when and where most scallop fishermen fish and in turn how susceptible sea turtles will be when fishermen are out on the water.

12.0 REINITIATING CONSULTATION

This concludes formal consultation on the continued operation of the Atlantic sea scallop fishery under the Scallop FMP. 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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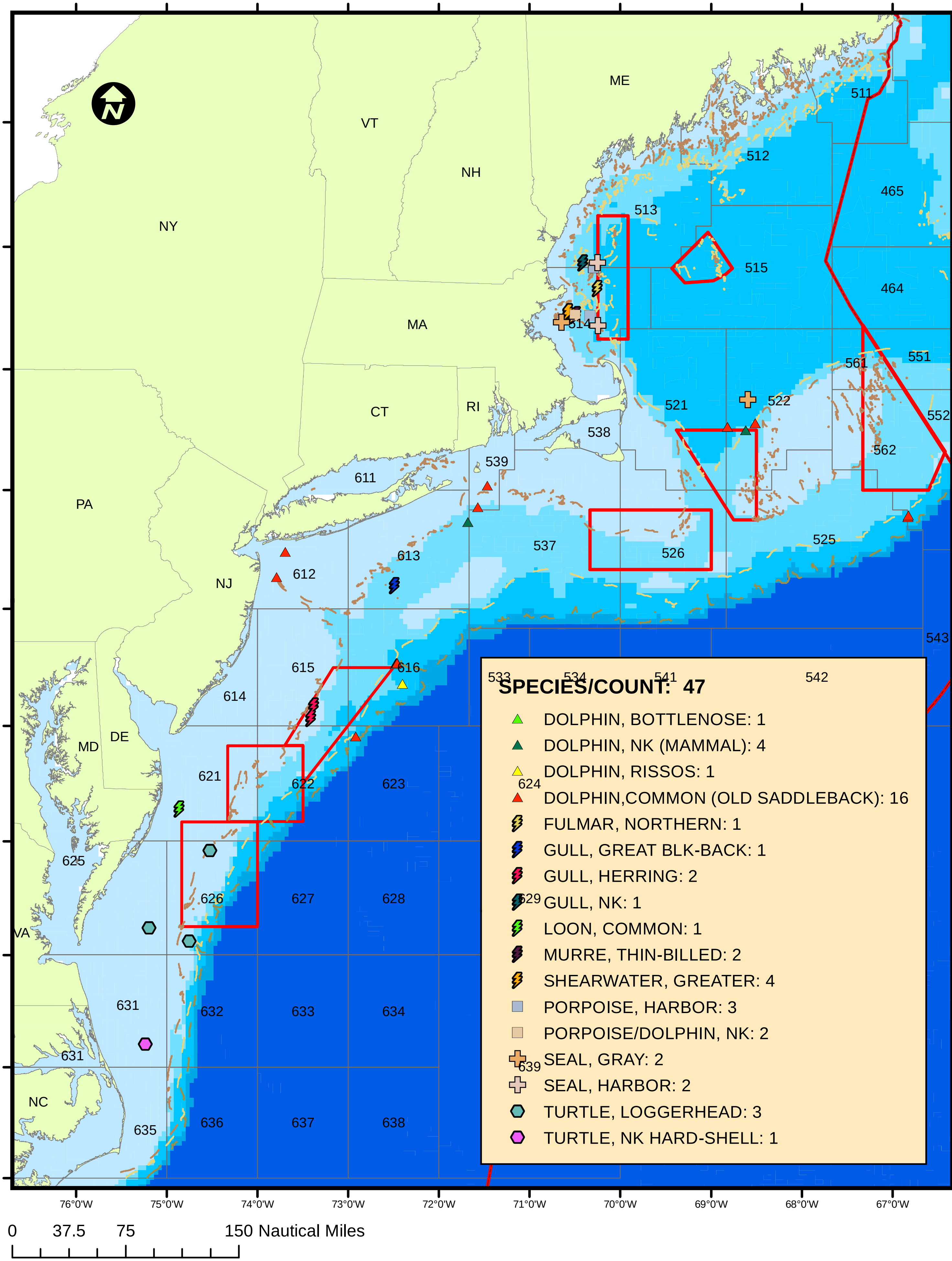
Bottom Otter Trawl, Fish	FLOUNDER, SUMMER	2-Dec-11	TURTLE, LOGGERHEAD	*Caretta caretta*	ALIVE, SEE COMMENTS
Bottom Otter Trawl, Fish	FLOUNDER, WINTER	31-Dec-11	TURTLE, LOGGERHEAD	*Caretta caretta*	ALIVE, SEE COMMENTS
Dredge, Sea Scallop	SCALLOP, SEA	9-Dec-11	TURTLE, LOGGERHEAD	*Caretta caretta*	ALIVE, SEE COMMENTS
Anchored Sink Gill Net	DOGFISH, SPINY	20-Dec-11	LOON, COMMON	*Gavia immer*	DEAD FRESH
Anchored Sink Gill Net	GROUNDFISH, NK	30-Dec-11	SEAL, GRAY	*Halichoerus grypus*	DEAD FRESH
Dredge, Sea Scallop	SCALLOP, SEA	6-Dec-11	GULL, HERRING	*Larus argentatus*	DEAD FRESH
Dredge, Sea Scallop	SCALLOP, SEA	6-Dec-11	GULL, HERRING	*Larus argentatus*	DEAD FRESH
Bottom Otter Trawl, Fish	SQUID, ATL LONG-FIN	3-Dec-11	DOLPHIN,COMMON	*Delphinus delphis*	DEAD FRESH
Bottom Otter Trawl, Fish	SQUID, ATL LONG-FIN	23-Dec-11	DOLPHIN,COMMON	*Delphinus delphis*	DEAD FRESH
Bottom Otter Trawl, Fish	FLOUNDER, SUMMER	5-Dec-11	TURTLE, NK HARD-SHELL	*Cheloniidae spp*	DEAD, SEVERELY DECOMPOSED
Bottom Otter Trawl, Haddock Separator	HADDOCK	4-Dec-11	DOLPHIN,COMMON	*Delphinus delphis*	DEAD FRESH
Bottom Otter Trawl, Haddock Separator	HADDOCK	4-Dec-11	DOLPHIN,COMMON	*Delphinus delphis*	DEAD FRESH
Bottom Otter Trawl, Haddock Separator	HADDOCK	4-Dec-11	DOLPHIN,COMMON	*Delphinus delphis*	DEAD FRESH
Bottom Otter Trawl, Haddock Separator	HADDOCK	4-Dec-11	DOLPHIN,COMMON	*Delphinus delphis*	DEAD FRESH

*100 of the observed incidental takes for December 2011 are represented in this summary

APPROXIMATE LOCATIONS OF INCIDENTAL TAKES
AS REPORTED BY NEFOP OBSERVERS IN
NEFSC DATABASES: DECEMBER 2011



APPENDIX B

NOAA Technical Memorandum NMFS-NE-207

Analysis of Atlantic Sea Scallop (*Placopecten magellanicus*) Fishery Impacts on the North Atlantic Population of Loggerhead Sea Turtles (*Caretta caretta*)

U. S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northeast Fisheries Science Center
Woods Hole, MA
February 2008

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NOAA Technical Memorandum NMFS-NE-207

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Analysis of Atlantic Sea Scallop (*Placopecten magellanicus*) Fishery Impacts on the North Atlantic Population of Loggerhead Sea Turtles (*Caretta caretta*)

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ABSTRACT

An estimated 619 loggerhead turtles of various age and sex classes were taken annually during 1989-2005 in all components of the US Atlantic sea scallop (*Placopecten magellanicus*) fishery. We provide here a quantitative assessment of the potential for these takes to jeopardize the continued existence of the US Atlantic Ocean population of loggerhead sea turtles (*Caretta caretta*). A population viability analysis (PVA) was used to estimate quasi-extinction likelihoods under conditions with and without fishery effects. This PVA used US index nesting beach data for 1989-2005 to estimate the loggerhead population trend μ (mean growth rate) and variance σ^2. The starting population (N_0) for the exercise was the sum of nesting females estimated from the 2005 nest count in the North Carolina to Florida area. The base model (with fishery bycatch) was developed by using estimates of μ (-0.022), σ^2 (0.012), N_0 (34,881) and a quasi-extinction threshold of 250 adult females. Quasi-extinction likelihoods were bootstrapped (1000 iterations) under baseline conditions to derive confidence intervals. The μ for each bootstrap iteration was drawn from a normally distributed random sampling of μ values lying within the 95% confidence interval around the original μ. The model was then rerun with the estimated annual fishery mortality of adult females (102 turtles) added back into the population, thus changing the trend (μ = -0.019, σ^2 = 0.012, and N_0 = 34,881). Results of the two models were similar; the quasi-extinction probabilities were zero at 25, 50, and 75 years, and 0.01 at 100 years for both analyses. Median times to quasi-extinction were 207 years versus 240 years, and the number of bootstrap simulations with extinction probabilities greater than 0.05 in 100 years was 258 and 178, respectively. These results suggest that the annual take of loggerhead sea turtles in the US fisheries for Atlantic sea scallops, though detectable, does not significantly change the calculated risk of extinction of the population of adult female Western North Atlantic loggerheads over the next 100 years.

INTRODUCTION

Loggerhead sea turtles (*Caretta caretta*) are incidentally captured in US dredge and trawl fisheries for Atlantic sea scallops (*Placopecten magellanicus*) in the US Mid-Atlantic region. Increased federal observer coverage of these fisheries allowed the National Marine Fisheries Service (NMFS) to estimate the annual bycatch of loggerhead turtles in the fisheries through 2005 (Murray 2004a, 2004b, 2005, 2007). Recent observer reports document takes through 2007. As loggerhead turtles are a threatened species under the US Endangered Species Act (ESA), NMFS, under Section 7 of the ESA, must ensure that continuation of the sea scallop fisheries is not likely to jeopardize the continued existence of the species.

Impacts of US fisheries (e.g., Atlantic sea scallop, Mid-Atlantic bottom trawl, pelagic longline, and Gulf of Mexico/Southern Atlantic commercial shrimp) on the western North Atlantic loggerhead sea turtle population have been analyzed by Southeast Fisheries Science Center (SEFSC) staff and the loggerhead sea Turtle Expert Working Group (TEWG 1998, 2000; SEFSC 2001; Epperly et al. 2002). However, reduced loggerhead nesting on southeastern US beaches suggests these analyses require updating. The TEWG is currently working on a reanalysis, but the limited data available on current population parameters (e.g., stage specific survival) suggest that the previous demographic models may be difficult to revise.

We provide here an alternative quantitative approach to the assessment of the risk the US Atlantic sea scallop fisheries have of jeopardizing the continued existence of the western North Atlantic Ocean populations of loggerhead sea turtles. This approach is simpler than previously used for western North Atlantic (WNA) loggerheads and is similar to that used by Snover (2005) in her analysis of the impact of the Western Pacific Pelagics Fisheries on several Pacific sea turtle species. We use a population viability analysis (PVA) to estimate quasi-extinction likelihoods under conditions with and without fishery effects. The PVA is count-based (Dennis et al. 1991; Morris et al. 1999; Holmes 2001; Morris and Doak 2002; Snover 2005) which will allow the use of the only relatively complete and available population time series—index nesting beach[1] counts for 1989-2005. As such, the analyses focus on the viability of the adult female portion of the population and should not be considered to model viability of the entire population.

We first present the PVA results under baseline conditions by using the rate of change of the adult female population (which implicitly includes the mortalities from the scallop and other fisheries) and the 2005 count of adult females estimated from all beaches in the Southeast based on an extrapolation from nest counts. We then adjust the rate of change by adding back the fisheries take and rerunning the PVA. The results of these two analyses are then compared by using the probability of quasi-extinction at 100 years to assess the impact of the takes in the Atlantic sea scallop fisheries.

At the outset, we point out three caveats to the interpretation of these analyses. First, the current negative nesting beach trends are at odds with some in-water survey results (e.g., Epperly et al. 2007). Secondly, the current negative trend in adult female abundance has likely been

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[1] Index beaches are a limited series of beaches which are regularly monitored for nesting activity. In Florida, the Index Nesting Beach Survey (INBS) has coordinated a detailed monitoring program since 1989 to measure seasonal productivity, allowing comparisons between beaches and between years. In Florida, 33 beaches (of 190 surveyed beaches) are included in the INBS program. Similar programs exist in states further north.

influenced by mortality events that have occurred over several decades. As such, a model based on current nesting beach trends may overestimate the effect of current takes on the likelihood of extinction for the population. Finally, we stress that our analyses should not be used to assess the likely fate of the population but should only be used to assess the impact of the fisheries for Atlantic sea scallops on the population trajectory of adult female loggerhead sea turtles. A thorough review of loggerhead population trends is provided by Witherington et al. (2006, in review).

METHODS

Data

Population trend data

A time series of population counts (or some index of the population) was needed through 2005 to estimate the population trend for the PVA. The time series needed to be longer than 10 years for the PVA to be more than marginally useful (Morris et al. 1999; Morris and Doak 2002).

Loggerhead nest counts (a proxy for the adult female population) are available for southeastern US index nesting beaches from 1989 to 2005 for the Northern (NC, SC, and GA) and Peninsular Florida subpopulations (NMFS in review, FWRI 2007). These are the subpopulations with the greatest nesting populations. Two other southeastern United States subpopulations have index beach nest counts available from 1996 (Dry Tortugas FL) and 1998 (Northern Gulf [AL, FL]) onwards (NMFS in review). These are the two smallest subpopulations, and since at least 1996 they have constituted a small fraction of the population (e.g., in 2005 they accounted for only 3% of the total number of index beach nests). Because nest counts were available for only a relatively brief period, these two subpopulations were excluded from the trend analysis for 1989-2005. Note that we did include the nest counts for all four subpopulations as part of a supporting analysis for the 1996-2005 period. Finally, these count data were used directly, without any adjustments for remigration[2] or nests per female, to determine the population trend.

Current abundance data

An estimate of adult female abundance in 2005 was necessary for use as the starting point for the PVA. The 2005 estimate of adult female abundance was derived by first summing nest counts from all beaches surveyed in the southeastern United States, including all beaches surveyed in 2005 in NC, SC, GA, FL, and AL (NMFS in review, FWRI 2007, SCDNR 2007). Only index beach nests counts were available for the Dry Tortugas and Northern Gulf subpopulations, so the total nest count is biased low. We then adjusted the sum to estimate adult females:

$$N_{AF} = (\text{Number of nests/Nests per female}) * \text{Remigration interval}$$

[2] Remigration is used here to mean the number of years between visits by adult females to nesting beaches and is not to be confused with the repeat visits within a single year which are included in the nests per female estimate.

Use of a constant value for nests per female and remigration interval is problematic as both parameters vary to some degree. For example, limited food resources can lead to decreased reproductive fitness because of natural and human driven fluctuations in prey availability. Moreover, if the age structure of the population changes, the number of nests per female will change. The available datasets do not characterize this variability, nor is it known whether such variability is random or associated with environmental change. Because of these uncertainties, we generally used conservative parameter values.

Estimates of nests per female vary widely, in part because of observational issues. Estimates adjusted for missed nesting suggest the mean number of nests per female per season in US waters ranges from 2.8 to 4.2 (Frazer and Richardson 1985; Schroeder et al. 2003). We used 4.2 nests per female.

Published estimates for the average remigration intervals of WNA loggerhead sea turtles on US beaches vary from 2.5 to 2.7 years (Richardson et al. 1978; Bjorndal et al. 1983; Schroeder et al. 2003). We used the 2.5 year remigration estimate.

Fishery mortality data

Estimates of loggerhead bycatch in the US Atlantic sea scallop fisheries are available for 2003-2005 for scallop dredge gear and for 2004-2005 for scallop trawl gear (Murray 2004a, 2004b, 2005, 2007). There is a wide range amongst the annual values, and two approaches for deriving an estimate for our model were considered. One approach was based on using the mean annual sea scallop dredge fishery bycatch for 2003-2005 ([749+180+0]/3=310; Murray 2004b, 2007) added to the midpoint of the range of estimated sea scallop trawl fishery bycatch from six bycatch estimates for 2004-2005 (136 turtles; Murray 2007) as the estimate of average annual total loggerhead sea turtles caught in the sea scallop fisheries (446 turtles). An additional 20 loggerheads were estimated to have been caught in groundfish bottom trawl fisheries where sea scallops were the primary catch (Murray 2006). Summing across fisheries suggests that the annual loggerhead bycatch in sea scallop related fisheries in 2004-2005 might be 466 animals.

The second approach used the take estimates in the Atlantic Sea Scallop Fishery Management Plan (FMP) Biological Opinion. This included only the 2003-2004 sea scallop dredge fishery bycatch (biennially 929 loggerhead sea turtles) added to one of the sea scallop trawl fishery bycatch estimates (268 loggerhead sea turtles biennially) and the 20 turtles estimated to be taken annually in groundfish bottom trawls for an average annual bycatch of 619 loggerhead sea turtles in the fishery.

We used the value of 619 loggerhead sea turtles as our estimate of the annual bycatch in the sea scallop fisheries of loggerhead sea turtles of various age and sex classes.

This <u>total</u> loggerhead sea turtle bycatch estimate (N_B=619 turtles) then needed to be adjusted downward to estimate the annual mortality of <u>adult female</u> loggerheads (N_{AF}) associated with the US sea scallop fisheries:

$$N_{AF} = (N_B * F_{US} * F_M * F_{M\text{-}F} * F_L) + (N_B * F_{US} * [1 - F_M] * F_{IM\text{-}F} * F_{IM\text{-}R} * F_L)$$

where:

F_{US} = proportion of the bycatch from the US population

F_M = proportion of bycatch mature

F_{M-F} = proportion of the adult bycatch assumed to be female

F_{IM-F} = proportion of the immature bycatch assumed to be female

F_{IM-R} = relative reproductive value of juvenile neritic turtles

F_L = proportion of the bycatch considered as lethal takes

Again, where there was a range of parameter values, we selected the value that generated the greatest impact by the sea scallop fisheries on the loggerhead population:

1. F_{US} - Genetic samples taken from loggerhead sea turtles captured in the sea scallop fisheries indicated that 88-93% of the animals are from the US nesting population (Haas et al. in review). This is comparable to the ~92% reported by Bass et al. (2004) for the Albemarle-Pamlico Sounds area of NC. We used a value of 93%.
2. F_M - Loggerheads captured in both gear types are expected to be of the same age classes. Loggerhead sea turtles observed bycaught in sea scallop fisheries ranged in size from 62 cm to 107 cm curved carapace length (CCL)(mean = 79.2 cm CCL, SD = 11.6, NE Fishery Observer Program database). The cutoff between sexually immature and mature loggerhead sea turtles appears is in the range of 87 to 100 cm CCL (NMFS in review; SEFSC 2001). CCL data were available for 42 turtles taken in the fishery; 35 (83.3%) were less than 87 cm CCL. As such, we used 0.833 as the proportion of immatures taken in the fisheries.
3. F_{M-F} and F_{IM-F} – There are few data available on the sex classes of loggerheads bycaught in the sea scallop fisheries. We, therefore, used data available from loggerhead captures and strandings. These data suggest that the mature and immature sex ratio in Northeast waters is approximately two females per male (TEWG 2000).
4. F_{IM-R} – Estimated bycatch of immature loggerheads was adjusted to account for the natural mortality expected prior to their recruitment as breeding adults. Wallace et al. (in press) present estimates in the range of 0.28 to 0.32 for the relative reproductive value of the neritic juvenile stage of loggerhead sea turtles found stranded along the US Atlantic coast (mean CCL = 78.5, SD = 16.6). Given the similarity in size of these loggerheads to those taken in the sea scallop fishery (mean CCL = 79.2, SD = 11.6), it appears reasonable to use this estimation of reproductive value for immature juvenile turtles taken in the sea scallop fishery. We, therefore, used 0.32 as the estimate for juvenile reproductive value.
5. F_L - Observer reports from the 2003-2005 fisheries suggest that the percentage of loggerhead sea turtles released alive and uninjured was 22.7-25% for scallop dredge gear and 100% for trawl gear (Murray 2004a, 2004b, 2005, 2007). This compares to the 36% and 88.5% used in the Atlantic Sea Scallop FMP Biological Opinion. We, therefore, used 0.227 and 0.885 for dredge and trawl gear, respectively.

Because of the differences in loggerhead captures in the trawl and dredge fisheries, the number of adult female mortalities was estimated separately for each fishery and then combined.

Together this series of adjustments provides an estimate of the annual mortality (in numbers) of US adult female loggerheads caused by the bycatch in the US Atlantic sea scallop fisheries.

Model

The Dennis Model is a density-independent model of population growth, which uses a diffusion approximation to compute the probability of quasi-extinction (i.e., reaching a low threshold population size) in a randomly varying environment:

$$N_{t+1} = N_t \lambda_t$$

Application of the model requires that two key parameter values be estimated to make inferences regarding population growth rates and quasi-extinction risks:

μ – the arithmetic mean of the log population growth rate
σ^2 – variance of the log population growth rate

Holmes (2001) suggests the use of running sums as a means of reducing bias associated with sampling error and stage-specific counts. We calculated running sums as:

$$R_j = N_i + N_{i+1}$$

where j=1,2,3 … (q-1), q is the number of censuses in dataset, N represents the population size, and R_j represents the population size at time j from the running sums. Without using the running sums approach (1 yr intervals), the trend was -0.0063 and the variance was 0.038. We evaluated running sums of 2 yr, 3 yr, and 4 yr to calculate the annual estimate of R_j and found that the 3 and 4 yr running sums produced the same rate of change (-0.0216), which was slightly different from the 2 yr interval (-0.0220). With the smaller variance in the trend for the 3 and 4 yr running sums (0.006 and 0.003, respectively), the result would be that a 3 or 4 yr interval would lead to reduced probabilities of quasi-extinction in 100 yrs. Following our rule of using conservative parameter values, we decided to use a 2 yr interval for the final analysis.

Then μ was calculated as:

$$\mu = (\sum log(R_{j+1}/R_j\ )/t$$

Similarly, σ^2 is calculated as the variance over the series of log (R_{i+1}/R_i) values. The μ and σ^2 are then used to estimate r (the instantaneous rate of change) and λ (Dennis et al. 1991):

$$r = \mu + \sigma^2/2$$
$$\lambda = e^{(r)}$$

Estimation of the extinction risk requires a population size at extinction (N_{ext}). The population size at extinction can assume several values, with 0 equal to the true extinction. Rather then focusing entirely on total extinction ($N_{ext} = 0$), the concept of quasi-extinction risk has been developed (Ginzburg et al. 1982), where quasi-extinction risk is the probability that a population will fall below a given threshold (N_{ext} > 0). There is no generally agreed upon level for quasi-extinction, though it is commonly considered to be a threshold population size below which the population would be critically endangered or effectively extinct. For large vertebrates, a variety of numerical values have been considered for this threshold (e.g., from 20 to 500). We considered using either 50 or 250 adult females as our estimate of quasi-extinction. Our reasons for considering fifty animals were: (1) there is general consensus in the conservation genetics community that large vertebrate populations cannot fall below 50 breeding animals and still maintain genetic integrity (Shaffer 1981; Franklin 1980), (2) the International Union for Conservation of Nature (IUCN)(2008) considers this to be one of the two threshold numerical values for a "critically endangered" population category, and (3) to provide comparability with the value used in the 2004 Pacific sea turtle bycatch PVA prepared by Snover (2005). IUCN uses 250 mature animals as an alternative threshold value for "critically endangered" populations when there is evidence of a population decline. Given the apparent decline in nesting in the southeastern United States, it appears reasonable to use 250 as our threshold value for quasi-extinction. The IUCN includes all mature animals in this value and not just adult females, so using 250 adult females as the threshold provides a doubly conservative threshold.

Morris and Doak (2002) describe the probability of reaching a quasi-extinction threshold (N_{ext}) by using the following function:

$$g(t|\,\mu,\sigma^2,d) = \frac{d}{\sqrt{2\pi\sigma^2 t^3}}\exp\left[\frac{-(d+\mu t)^2}{2\sigma^2 t}\right]$$

with d = log(N_0/N_{ext}), and N_0 is the population size at the beginning of the analysis period. To calculate the total probability of reaching N_{ext} at some future time T, the cumulative distribution function (which is the preceding function integrated from t = 0 to T) is applied:

$$G(T|\,\mu,\sigma^2,d) = \exp\left[\frac{-2\hat{\mu}d}{\hat{\sigma}^2}\right]\Phi\left[\frac{-d+\hat{\mu}T}{\sqrt{\hat{\sigma}^2 T}}\right]+\Phi\left[\frac{-d-\hat{\mu}T}{\sqrt{\hat{\sigma}^2 T}}\right]$$

where Φ(z) is the standard normal cumulative distribution function (Morris and Doak 2002).

Morris and Doak (2002) outlined an approach for deriving the quasi-extinction time cumulative distribution function confidence intervals by using bootstrap estimation procedures. We used a similar approach, sampling from a random distribution drawn from within the 95% confidence interval for μ and σ^2 and replicated 1000 times to estimate the confidence intervals around the cumulative probability of reaching N_{ext} at some future time T.

Modeling Steps

The base model (with fisheries bycatch) was run over a 1,000 yr period with the estimates of μ, σ^2, N_0 beginning in 2005 and quasi-extinction threshold of 250 adult female loggerheads (Dennis et al. 1991; Holmes 2001; Morris and Doak 2002; Snover 2005). The 1,000 year time horizon was necessary so that we could determine the median time to extinction. Quasi-extinction likelihoods were then bootstrapped under baseline conditions to derive confidence intervals. The μ for each bootstrap iteration was drawn from a normally distributed random sampling of μ values lying within the 95% confidence interval around the original μ.

The model was modified to add back in the annual loggerhead bycatch in the Atlantic sea scallop fisheries. First, we adjusted the annual estimated bycatch in the fisheries (dredge and trawl) of loggerhead sea turtles for all age and sex classes to derive an estimate of total adult females removed from the population. We then calculated the rate of adult female removals for 2005 by dividing the bycatch by the total adult female population in 2005. This rate was then added into the population instantaneous growth rate (r) for each year from 1989 to 2005, and a revised μ and σ^2 was calculated. The model (without fishery bycatch) was then run with the revised estimates of μ, σ^2, and N_0. We bootstrapped quasi-extinction likelihoods under the new model's conditions to derive confidence intervals.

Evaluation of Results

The primary metric we used to compare the results of the two PVAs (with and without the fishery mortalities) was the cumulative probability of quasi-extinction at 100 years (based on recommendations on acceptable risk of extinction in DeMaster et al. 2004). Secondary metrics included the number of bootstrap replicates with a probability of extinction > 0.05 in 100 years and the median times to extinction[3]. We analyzed the sensitivity of the 1989-2005 model to changes in the population trend by comparison with the trend from 1996-2005. We also compared extinction probabilities at take levels that were two and ten times the documented levels of takes in the sea scallop fisheries.

RESULTS

Population Trends to Present

Loggerhead nest counts from the Northern and Peninsular subpopulations were summed (Fig. 1) and analyzed to develop the annual rates (λ) of population change for 1989-2005 (Table 1). The trend (μ = -0.022, σ^2 = 0.012, Table 2) for 1989-2005 for the US Atlantic Ocean loggerhead adult female population suggests the adult female population is declining.

We used an estimate of 58,602[4] nests in 2005 in the southeastern United States (North Carolina to Alabama). This produced an estimate of 34,881 adult females when adjusted for nests per female (4.2 nests per female) and remigration interval (2.5 years).

The annual sea scallop fisheries bycatch mortality of adult female loggerheads was estimated to be 102 turtles (97 in the dredge fishery and 5 in the trawl fisheries). This estimate was derived from the total annual take of 619 loggerheads prorated for area of origin (0.930 from United States), maturity (0.833 immature), female proportion (0.67), reproductive value of juveniles (0.32), and fishery specific mortality (dredge = 0.773 and trawl = 0.115).

Given the 2005 population estimate of 34,881 adult females and a fishery-induced mortality of 102 adult females per year, the rate of adult female removals in the sea scallop

[3] The time when the quasi-extinction probability is 0.50

[4] This includes 2005 counts for all beaches in the Northern (NC = 560, SC = 4,233, GA = 1,145 nests) and Peninsular Florida (51,636 nests) subpopulations and index beaches in the Dry Tortugas (159 nests) and Northern Gulf (869 nests) subpopulations (NMFS in review; FWRI 2007; SCDNR 2007).

fishery was 0.0029 in 2005. These mortalities were added back into the population to produce a revised 1989-2005 μ of -0.019 (σ^2 = 0.012, Table 2).

Viability Analyses

Using the 1989-2005 model, the risk of quasi-extinction (N_{ext} = 250 adult females) at 100 years was 0.01 (Table 2, Fig. 2) with a median time to extinction of 207 years (Table 2). Over 1000 iterations of the model, 258 produced a probability of extinction at 100 years greater than 0.05.

Adding the Atlantic sea scallop fisheries-related loggerhead mortalities back into the population had only a small effect on population trajectory and extinction probabilities. The μ was -0.022 and -0.019 for the analyses with and without the fishery takes. The risk of quasi-extinction at 100 years remained 0.01 (Table 2, Fig. 3). The median time to extinction grew to 240 years (Table 2). Over 1000 iterations of the model, 178 produced a probability of extinction at 100 years greater than 0.05.

Results of the two analyses were similar (Table 2, Fig. 4). Both had quasi-extinction probabilities of zero (0) at 25, 50, and 75 and a probability of 0.01 at 100 years. Median times to quasi-extinction were similar (207 years versus 240 years). The number of simulations with extinction probabilities at 100 years greater than 0.05 was 258 and 178, respectively.

Model Sensitivity

An incorrect estimate of the population trend would significantly affect the model results. Therefore, we repeated this analysis with just the 1996-2005 time series. While this would generally be considered to be too short a time series for analysis, it does provide some insight into the capability of the model to detect risk of extinctions.

Loggerhead nest counts from all four subpopulations were summed (Table 3) and analyzed to develop the annual rates (λ) of population change for 1996-2005 (Table 4). The trend (μ = -0.049, σ^2 = 0.011, Table 2) for 1996-2005 for the US Atlantic Ocean loggerhead adult female population suggests even more strongly than the 1989-2005 analysis that the adult female population is declining. Again with the 2005 population estimate of 34,881 adult females and a fishery-induced mortality of 102 adult females per year, the rate of adult female removals in the sea scallop fishery was 0.0029 in 2005. These mortalities were added back into the population to produce a revised 1996-2005 μ of -0.046 (σ^2 = 0.011, Table 4).

There was little difference between the 1996-2005 analyses with and without the sea scallop fisheries mortalities (Tables 4, Fig. 5). The population trend remains similar; μ equals 0.049 and 0.046 for the two analyses. Cumulative probabilities of extinction are identical up until approximately the 75th year, and the median times to extinction were very similar for both 1996-2005 models (i.e., 98 versus 102 years). The number of simulations with extinction probabilities at 100 years greater than 0.05 was 940 and 922, respectively.

We also evaluated the model's sensitivity to changes in fishery mortality rates. Given that the 1989-2005 model showed probabilities of extinction at 100 years equal to zero for both the original model and the model with takes added back in, it was necessary to use the 1996-2005 model for this evaluation. We compared the results of adding the loggerhead mortalities caused by the Atlantic sea scallop fisheries (102 adult females) with adding back in mortalities that were two and ten times greater than that observed in the sea scallop fisheries (Fig. 6).

Ultimately, it appears that the probability of extinction at 100 years would be reduced to zero if ten times the number of adult females estimated to be taken by the Atlantic sea scallop fisheries were added back to the population.

DISCUSSION

These results suggest that mortalities of loggerhead sea turtles in the US Atlantic sea scallop dredge and trawl fisheries are detectable but have a relatively small effect on the trajectory of the adult female components of the WNA loggerhead sea turtles over the next 100 years. The 1989-2005 population trends, with and without the mortalities, were not significantly different, and the probability of reaching the quasi-extinction threshold (250 adult females) under both scenarios was 0.01. Median times to extinction for both were greater than 200 years. The only obvious difference was in the number of bootstrap simulations with a probability of extinction > 0.05 in 100 years.

The relatively large population size of adult females (34,881), the relatively small negative trend in the adult female population over 1989-2005 (r = -0.022 per year), and the number of adult female mortalities in the fisheries (102 per year) all contribute to the lack of effect. This lack of impact occurred despite the use, wherever possible, of values which generated the greatest consequence of the sea scallop fisheries takes of loggerheads. If less stringent values had been used, the effect would have been less. Patterson and Murray (2008) provide commentary on the effect that application of the precautionary principle to a PVA may have on "robust inference" and defensible policy.

Even a model as simple as the Dennis model is sensitive to parameter values and data inputs. Values calculated or selected for μ, $N_{ext,}$ and σ^2 were all influential. With respect to μ, we found that relatively small changes in the population trend produced profound changes in the probability of quasi-extinction at 100 years. For example, doubling the rate of decline in the base model (from -0.022 to -0.049) greatly increased the probability of extinction at 100 years from 0.01 to 0.54. In contrast, the level of bycatch mortality value removed from the population would need to be much greater than that observed in the sea scallop fisheries to have a major effect on the population trajectory. The comparison of the effect of different background mortalities (Fig. 6) suggests that up to ten times the level of loggerhead mortality in the sea scallop fisheries needs to be removed to stabilize the population. This small effect is important in that it suggests the relatively steep declining trend for 1996-2005 is being driven by some other, larger source of mortality.

Recognizing the influence of the population trend to the analysis, it is important to point out our assumption that the nesting beach data used in this analysis were representative of trends of the US loggerhead population. This was a practical decision; only the index beaches are counted annually in a systematic fashion. However, there is a risk in this assumption. We noted earlier the problem of juvenile in-water counts being at odds with the nesting trends. There is also some concern about the representativeness of the nest counts. If loggerhead nesting shifts systematically between years (either inside or outside of the index beach areas), then trends in the index nesting beach data may not represent the overall trend. For example, if loggerhead nesting is becoming more aggregated at the index sites (because of issues such as habitat protection), then the estimates may be biased high. Alternatively, if turtles nest outside of the time period (for example, earlier nesting caused by warmer climate conditions), then the index site estimates would be biased low. Work underway by the loggerhead TEWG and Florida's

Fish and Wildlife Research Institute will provide a substantive review of these trends. Our focus here was with evaluating the impact of the bycatch mortality in the Atlantic sea scallop fisheries on the future of the loggerhead population, and the impact of such biases on our analysis are likely immaterial. These biases could, however, significantly influence an analysis of population status and perhaps result in inappropriate management decisions.

The quasi-extinction value selected was also influential, but not as dramatically as the population trend. We evaluated N_{ext} values of 50 and 250 adult females. With the 1989-2005 base model, the probabilities of extinction at 100 years were 0.00 and 0.01 for 50 and 250 animals, respectively. Larger differences were observed in the 1996-2005 base model, where the values were 0.07 and 0.42 respectively. The latter, larger effect is likely due to the increased negative population trend. We also considered using the percent of decline approach suggested by Snover and Heppell (in press). We estimated the probability of reaching 50% of the current population size. Although risks of reaching the threshold were much higher (0.97 and 0.95 in 100 years) than with the 50 or 250 animal threshold, there were no significant differences between the base model and the model with takes added back in. Ultimately, we decided to use an absolute value of N_{ext} = 250 adult females largely because this analysis was designed to evaluate the risk of extinction resulting from mortalities in the scallop fisheries, and 250 animals better represents a threshold extinction value than does 50% of the current population size (N_{ext} = 17,441 adult females).

The model is also sensitive to changes in the variance; as the variance increases, the probability of extinction at any point in time increases, and as the variance decreases, probabilities of extinction decrease. Here it was assumed that the variance in the population trend is largely the same with and without the sea scallop fishery takes. Violations of this assumption would not change the interpretation of the sea scallop fisheries impacts, unless the take estimates were much higher relative to the population size and the variance in the takes was large.

However, the largest issue with variance was not the influence on the outcome but the difficulty of providing meaningful tests of significance with large confidence intervals. Using bootstrap techniques produced much tighter confidence intervals, but trajectories would need to vary considerably to find statistical differences.

Finally, this analysis was undertaken to provide a simple evaluation of the effect that loggerhead bycatch in the Atlantic sea scallop fisheries could have on the future viability of the WNA loggerhead population. It was not designed to and should not be used to evaluate population status. For example, here we implicitly assume that adult female recruitment will not change in the future. This is a particularly troublesome assumption because there are data suggesting that the number of juvenile loggerhead sea turtles is increasing (e.g., Epperly et al. 2007). If the increase in juvenile abundance translates into increased adult female recruitment, then our estimates of extinction probabilities would be overestimated; however, the relationship between the models with and without fishery takes would not be fundamentally changed. A staged matrix model, incorporating age-class survival and fecundity, would provide a much better evaluation tool to assess population status (and fishery impacts).

An example of such an evaluation is provided by the US Fish and Wildlife Service's (USFWS) recent quantitative threats analysis for the Florida manatee (*Trichechus manatus latirostris*; Runge et al. 2007). The basis of this threats assessment is a comparative population viability analysis, which involves forecasting the Florida manatee population under different scenarios regarding the presence of threats, while accounting for process variation (environmental, demographic, and catastrophic stochasticity) and parametric and structural uncertainty. Several steps were required: modifying an existing population model to accommodate the threats analysis framework, updating survival rates, estimating the fractions of mortality from various causes, modeling the threats themselves, and developing metrics to measure the impact of the threats. While the conceptual process followed in our analysis of loggerhead sea turtles and that used by the USFWS are similar, the additional information available from the USFWS exercise results from a stage-based projection model for Florida manatees, incorporating environmental and demographic stochasticity, catastrophes, density-dependence, and long-term change in carrying capacity.

However, recent data to support such an analysis of loggerhead sea turtles are incomplete. A comprehensive program to collect these data should be developed and implemented so that scientific analyses, such as those presented here, can be improved and the best possible scientific advice can be provided to NOAA managers tasked with conserving both turtle populations and fisheries.

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Table 1. Counts of loggerhead sea turtle (*Caretta caretta*) nests at index beaches for 1989-2005 by subpopulation, biannual totals, and rates of change (λ and r) by year (NMFS in review, FWRI 2007).

Year	Northern (NC, SC, GA)	Peninsular Florida	Total ($N_{i)}$	Two-year Running Sum (Rj)	Rate of Change (λ)	Inst. rate of change (r)
1989	1,421	39,091	40,512			
1990	2,466	50,266	52,732	93,244		
1991	2,127	52,802	54,929	107,661	1.1546	0.14377
1992	1,844	47,567	49,411	104,340	0.9692	-0.0313
1993	931	41,808	42,739	92,150	0.8832	-0.1242
1994	2,207	51,168	53,375	96,114	1.0430	0.04212
1995	1,484	57843	59,327	112,702	1.1726	0.15921
1996	1,969	52811	54,780	114,107	1.0125	0.01239
1997	1,100	43156	44,256	99,036	0.8679	-0.1417
1998	1,812	59918	61,730	105,986	1.0702	0.06782
1999	2,173	56471	58,644	120,374	1.1358	0.1273
2000	1,475	56277	57,752	116,396	0.9670	-0.0336
2001	1,242	45941	47,183	104,935	0.9015	-0.1037
2002	1,543	38125	39,668	86,851	0.8277	-0.1891
2003	1,998	40726	42,724	82,392	0.9487	-0.0527
2004	549	29547	30,096	72,820	0.8838	-0.1235
2005	1,766	34872	36,638	66,734	0.9164	-0.0873

Table 2. Model results based on 1989-2005 2-year running sum trend with a starting population size of 34,881 adult female loggerhead sea turtles (*Caretta caretta*) and quasi-extinction threshold equal to 250 adult females for base model and model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes added back into population.

	Base Model	With Fishery Takes Added Back In
Population Trend	-0.022	-0.019
Variance of trend	0.012	0.012
Upper confidence limit	0.039	0.042
Lower confidence limit	-0.084	-0.080
Quasi-extinction risk with 95% confidence interval in parentheses		
@ 25 years	0.00 (0, 0)	0.00 (0, 0)
@ 50 years	0.00 (0, 0)	0.00 (0, 0)
@ 75 years	0.00 (0, 0.09)	0.00 (0, 0.02)
@ 100 years	0.01 (0, 0.46)	0.01 (0, 0.31)
Median time to extinction	207 years	240 years

Table 3. Counts of loggerhead sea turtle (*Caretta caretta*) nests at index beaches for 1996-2005 by subpopulation, biannual totals, and rates of change (λ and r) by year (NMFS in review, FWRI 2007). Number in italics were interpolated from adjacent counts.

Year	Northern (NC, SC, GA)	Peninsular Florida	Dry Tortugas (Florida)	Northern Gulf (FL, AL)	Total (N_{i})	Running sum (Rj)	Rate of change (λ)	Inst. rate of change (r)
1996	1,969	52,811	249	*166*	55,195			
1997	1,100	43,156	258	166	44,680	99,875		
1998	1,812	59,918	249	149	62,128	106,808	1.0694	0.0671
1999	2,173	56,471	292	235	59,171	121,299	1.1357	0.1272
2000	1,475	56,277	242	181	58,175	117,346	0.9674	-0.0331
2001	1,242	45,941	213	143	47,539	105,714	0.9009	-0.1044
2002	1,543	38,125	*210*	149	40,027	87,566	0.8283	-0.1883
2003	1,998	40,726	208	95	43,027	83,054	0.9485	-0.053
2004	549	29,547	159	114	30,369	73,396	0.88371	-0.1236
2005	1,766	34,872	*159*	120	36,917	67,286	0.91675	-0.0869

Table 4. Model results based on 1996-2005 2-year running sum trend with a starting population size of 34,881 adult female loggerhead sea turtles (*Caretta caretta*), and quasi-extinction threshold equal to 250 adult females for base model and model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes added back into population.

	Base Model	With Fishery Takes Added Back In
Population trend	-0.049	-0.046
Variance of trend	0.011	0.011
Upper confidence limit	0.037	0.040
Lower confidence limit	-0.135	-0.1322
Quasi-extinction risk with 95% confidence interval in parentheses		
@ 25 years	0.00 (0, 0)	0.00 (0, 0)
@ 50 years	0.00 (0, 0.03)	0.00 (0, 0.02)
@ 75 years	0.10 (0, 0.67)	0.06 (0, 0.57)
@ 100 years	0.54 (0.02, 0.98)	0.42 (0.01, 0.996)
Median time to extinction	98 years	102 years

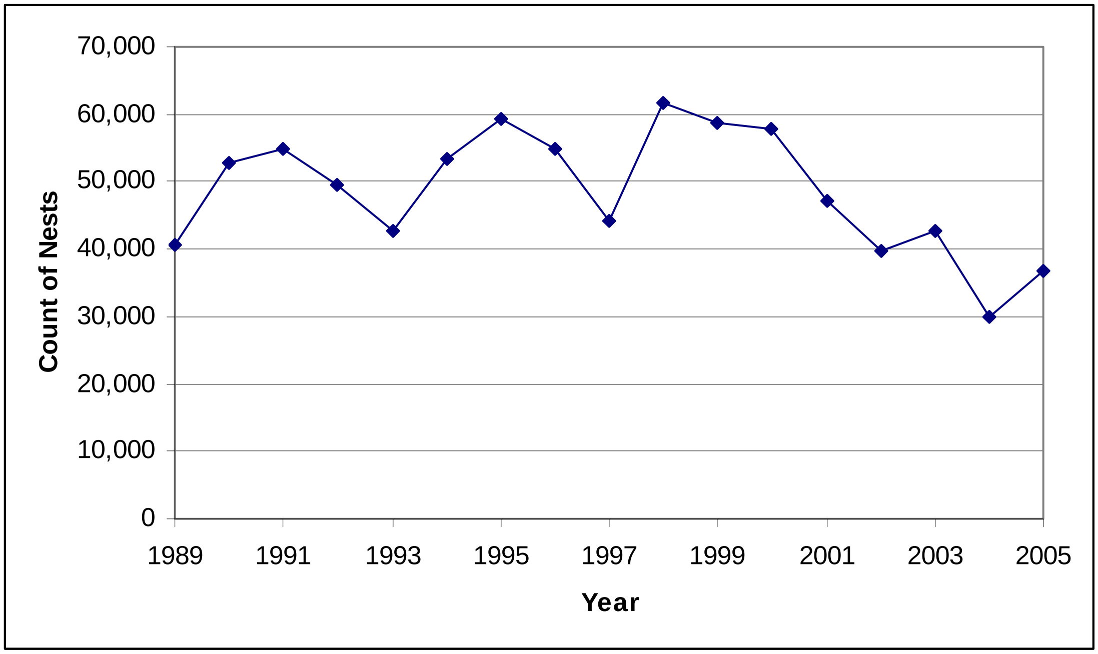

Figure 1. Number of Atlantic Ocean loggerhead sea turtle (*Caretta caretta*) nests recorded at US Northern (NC, SC, GA) and Peninsular Florida index beaches from 1989 to 2005 (NMFS in review, FWRI 2007).

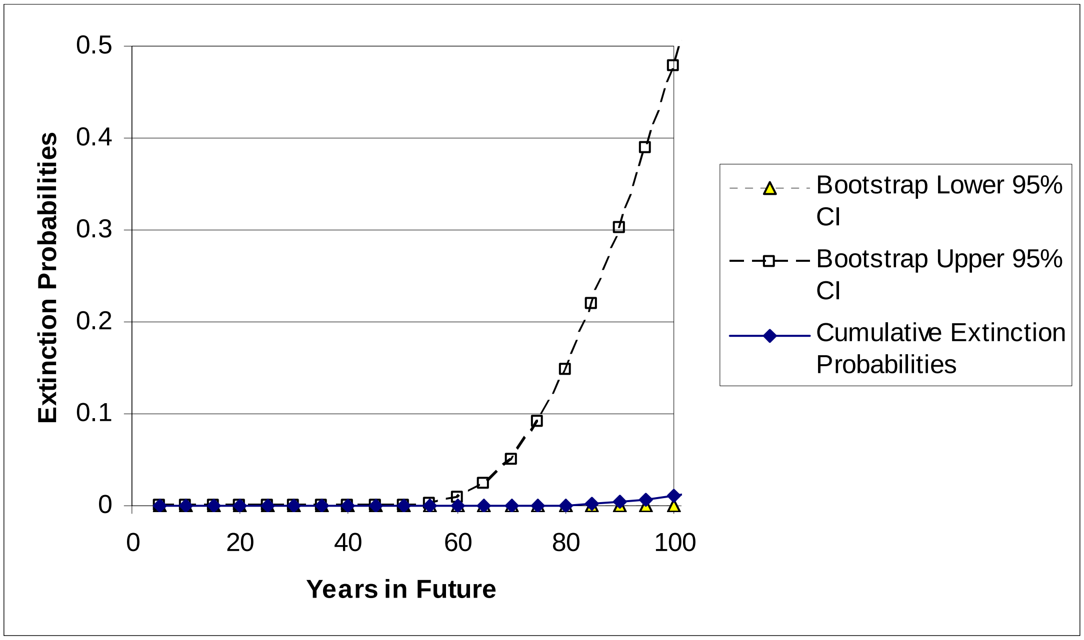

Figure 2. Cumulative quasi-extinction probabilities and confidence intervals (CI) for 1989-2005 base model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes for adult female western North Atlantic loggerhead sea turtles *(Caretta caretta)*. Quasi-extinction is equal to 250 adult female loggerhead sea turtles.

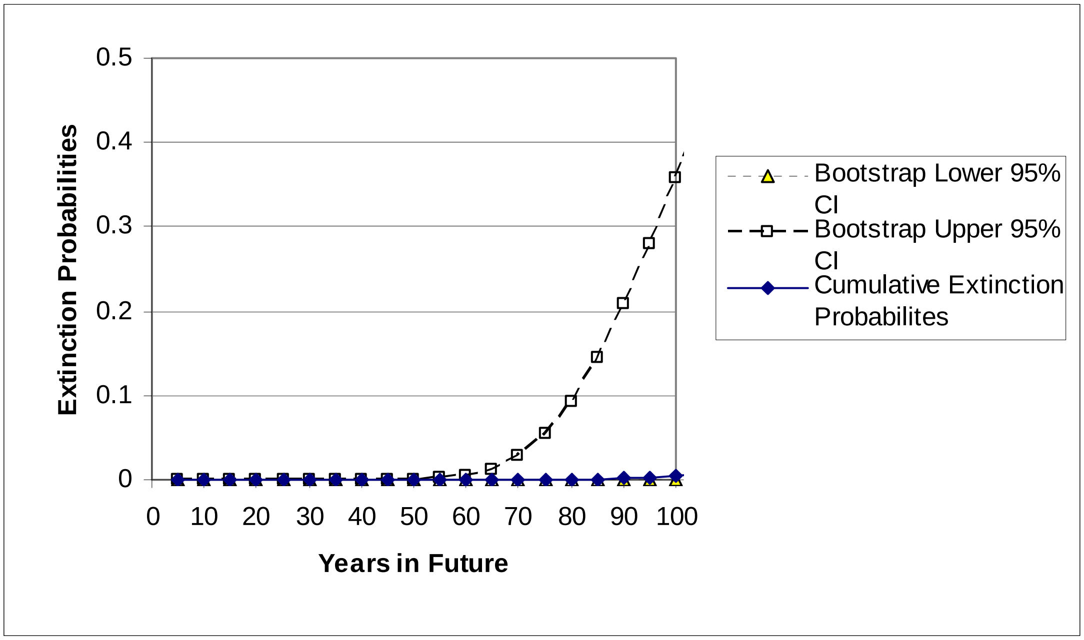

Figure 3. Cumulative quasi-extinction probabilities and confidence intervals (CI) for 1989-2005 model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes for adult female western North Atlantic loggerhead sea turtles *(Caretta caretta)* added back into population. Quasi-extinction is equal to 250 adult female loggerhead sea turtles.

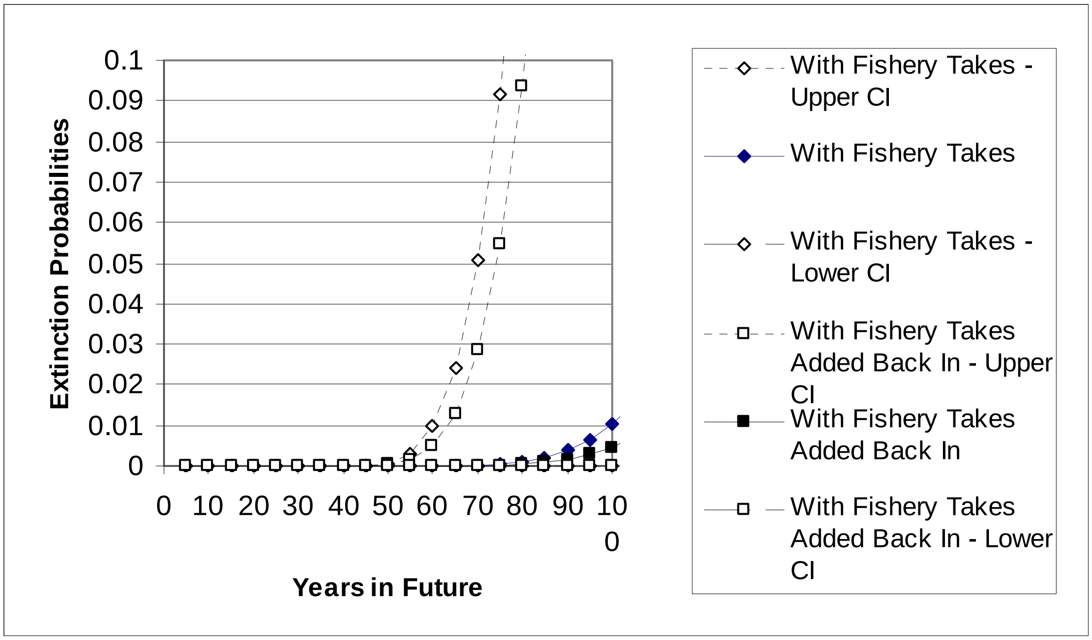

Figure 4. Comparison of cumulative quasi-extinction probabilities and confidence intervals (CI) of 1989-2005 models with and without Atlantic sea scallop (*Placopecten magellanicus*) fishery takes. Quasi-extinction is equal to 250 adult female loggerhead sea turtles *(Caretta caretta).* Note vertical scale runs only through P_{EX} = 0.10.

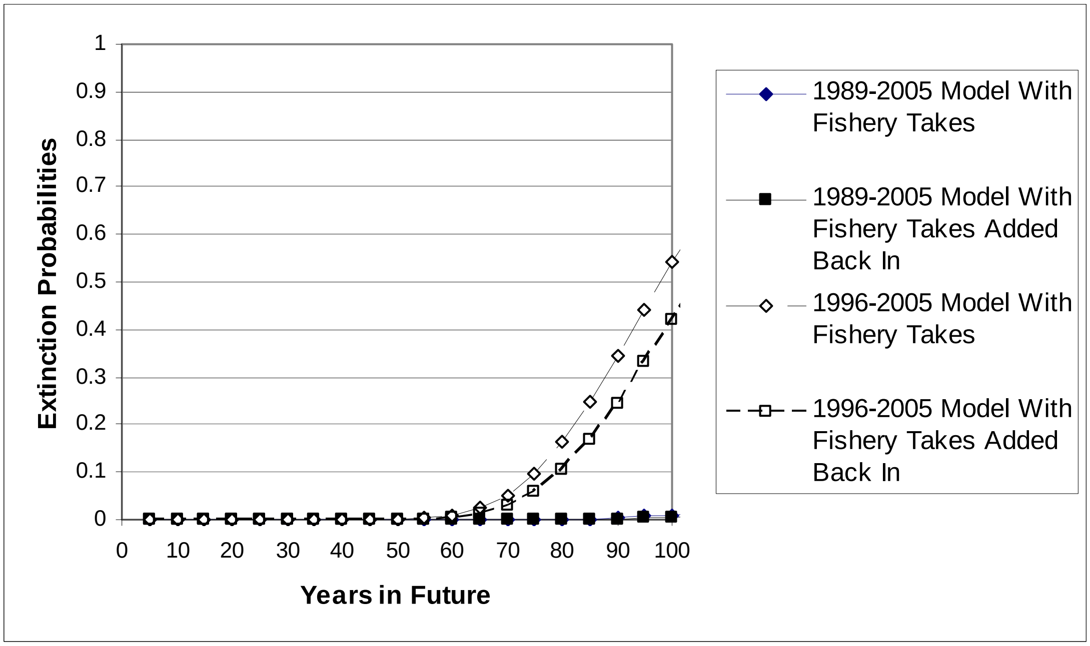

Figure 5. Extinction trajectories for models with and without Atlantic sea scallop (*Placopecten magellanicus*) fishery takes with original 1989-2005 population trajectory compared to 1996-2005 trajectory. Quasi-extinction is equal to 250 adult female loggerhead sea turtles *(Caretta caretta).*

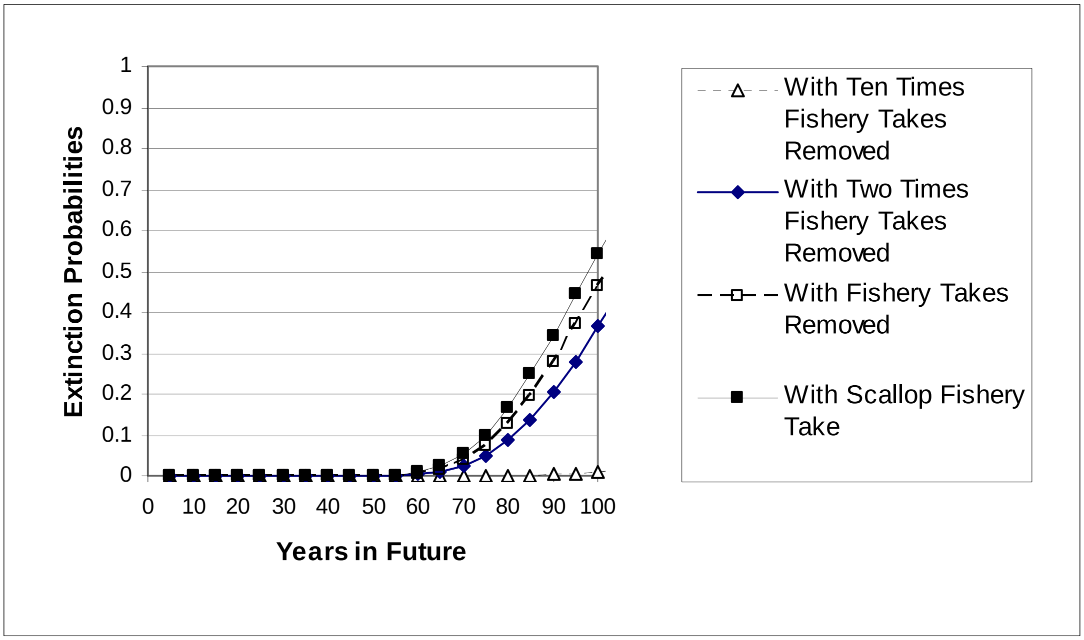

Figure 6. Cumulative quasi-extinction probabilities for 1996-2005 models with various levels of mortality removed from the trend. Fishery takes estimated as one time (the Atlantic sea scallop [*Placopecten magellanicus*] fisheries) versus two and ten times the original sea scallop fishery take level. Quasi-extinction equal to 250 adult females loggerhead sea turtles (*Caretta caretta*).

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APPENDIX C

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.

APPENDIX D

Procedure for obtaining fin clips from Atlantic 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.