ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
BIOLOGICAL OPINION

Action Agency: NMFS, Northeast Regional Office

Activity: Endangered Species Act Section 7 Consultation on the Atlantic Sea Scallop Fishery Management Plan [Consultation No. F/NER/2007/00973] GARFO-2007-00002

Consulting Agency: NMFS, Northeast Region, through its Protected Resources Division

Date Issued: MARCH 14, 2008

Approved by: [signature]
(amended February 5, 2009)

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Section 7(a)(2) of the Endangered Species Act (ESA) (16 U.S.C. 1531 *et seq.*) requires that each Federal agency shall ensure 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 species listed as threatened or endangered, that agency is required to consult with either NOAA Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (FWS), depending upon the species that may be affected. In instances where NMFS or FWS are themselves proposing an action that may affect listed species, the agency must conduct intra-service consultation. Since the action described in this document is authorized 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 that new information on sea turtle takes reveals that the continued authorization of the Atlantic sea scallop fishery (scallop fishery) may affect listed species in a manner or to an extent not previously considered. This document represents NMFS's biological opinion (Opinion) on the continued authorization of the scallop fishery under the Atlantic Sea Scallop FMP (Scallop FMP), and its effects on ESA-listed species under NMFS jurisdiction in accordance with section 7 of the ESA, as amended.

Formal intra-service section 7 consultation on the continued authorization of the scallop fishery was reinitiated on April 3, 2007 [Consultation No. F/NER/2007/00973]. This Opinion is based on the information developed by NMFS NERO and other sources of information, as cited in the Literature Cited section of this document.

1.0 CONSULTATION HISTORY

Cause for Reinitiating

As provided at 50 CFR 402.16, reinitiation of formal consultation is required where discretionary 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 the opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action.

The September 18, 2006, Opinion for the Atlantic sea scallop fishery concluded that continued authorization of the fishery was not likely to jeopardize the continued existence of any ESA-listed species under NMFS jurisdiction. However, sea turtles were expected to interact with scallop dredge and trawl gear used in the fishery such that turtles would come into physical contact with the gear (be struck by or swim into) and would be captured, with the exception that chain-mat equipped dredge gear would prevent most captures of turtles struck by such gear. In accordance with the regulations (50 CFR 402.02), such interactions are considered "incidental takes". An Incidental Take Statement (ITS) was provided with the September 18, 2006, Opinion along with non-discretionary Reasonable and Prudent Measures (RPMs) to minimize the impact of take. As described in the ITS, up to 760 sea turtles (752 in scallop dredge gear and 8 in scallop trawl gear) were anticipated to be taken annually as a result of the continued authorization of the Atlantic sea scallop fishery. Nearly all of the takes (749 of 752 for dredge gear and 5 of 8 for trawl gear) were anticipated to be loggerhead sea turtles.

The anticipated incidental take of loggerhead sea turtles in scallop dredge gear was based on an estimated take of the species in scallop dredge gear during the 2003 scallop fishing year (March 1, 2003 through February 29, 2004)(Murray 2004b; NMFS 2006a), while the anticipated incidental take of loggerhead sea turtles in scallop trawl gear was based on the actual observed take of the species in scallop trawl gear in the 2005 scallop fishing year (NMFS 2006a). The difference in approach reflected differences in the availability of information for loggerhead sea turtle bycatch in scallop dredge and scallop trawl gear at the time of the September 18, 2006, Opinion. In a memorandum dated November 29, 2004, the Regional Administrator of NMFS NERO asked the Northeast Fisheries Science Center (NEFSC) Director whether it was feasible to develop a bycatch estimate for sea turtles in the scallop trawl fishery (memo from P. Kurkul, NER, to J. Boreman, NEFSC, November 29, 2004). In response, the NEFSC Director stated that: (1) Sufficient information to support a scientifically defensible estimate of sea turtle bycatch in the scallop trawl fishery was not available at that time, and (2) a bycatch estimate for sea turtles in the scallop trawl fishery would be provided after the 2005 scallop fishing year, if the proposed level of observer coverage was achieved (memo from J. Boreman, NEFSC to P. Kurkul, NER, December 6, 2004). The 2005 scallop fishing year continued until February 28, 2006 (see 50 CFR 648.53(b)(5) for the definition of a scallop fishing year). When the September 18, 2006, Opinion was completed, it indicated that the scallop trawl estimate was not yet available, but that NMFS would review it once it was available and determine at that time whether it triggered reinitiation of consultation (NMFS 2006a at page 71).

On February 23, 2007, the NEFSC released NEFSC Reference Document 07-04 (Murray 2007). Based on observer data for the scallop trawl fishery for 2004 and 2005, Murray (2007) provided the first estimates of the average annual bycatch of loggerhead sea turtles in scallop trawl gear. As described in Murray 2007, the NEFSC derived six different estimates of the average annual bycatch of loggerheads, using three different methods. The six estimates ranged from 81 to 191 turtles (CVs ranged from 0.32 to 0.50) (Murray 2007). NMFS NERO has reviewed the information presented in the NEFSC reference document and has determined that the method that resulted in an estimated annual take of 134 loggerhead sea turtles (CV=0.45, 95% CI: 37-257) provides the best available information on the number of loggerhead takings in the scallop trawl fishery (Memo to the Record from P. Kurkul dated June 28, 2007). NMFS NERO has also determined that the reference document presents new information regarding the capture of sea turtles in scallop trawl gear that reveals effects of the action that may affect listed sea turtles in a manner or to an extent not previously considered. Therefore, in accordance with the regulations at 50 CFR 402.16, formal consultation was reinitiated on April 3, 2007, to reconsider the effects of the Atlantic sea scallop fishery on ESA-listed sea turtles.

Based on the reinitiation date of April 3, 2007, the 135-day deadline for completion of consultation and preparation of the Opinion was August 15, 2007. However, in July 2007, NMFS entered into an agreement with Oceana and the Fisheries Survival Fund to stay litigation filed regarding the September 18, 2006, Opinion (Oceana v. Gutierrez, D.D.C., Case no. 1: 07-cv-00142-RBW). The conditions of the agreement included that NMFS would consider comments submitted by Oceana and the Fisheries Survival Fund (a defendant intervenor in the litigation) on topics at issue in the September 18, 2006, Opinion. The agreement required that both parties submit comments to NMFS by July 30, 2007, and they have done so. In order for NMFS to fully consider the comments received, NMFS extended the consultation period until December 15, 2007 (Memo from P. Kurkul to The Record, August 7, 2007). Additional time was needed to complete the Opinion, and NMFS indicated that it would issue it on or before March 15, 2008.

Consultation History
The consultation history for the Atlantic sea scallop fishery was reviewed in the previous formal consultation completed September 18, 2006. Briefly, formal consultation on the scallop fishery was initiated December 21, 2001. The Opinion concluded on February 24, 2003, that the continued authorization of the scallop fishery would not jeopardize the continued existence of loggerhead, leatherback, Kemp's ridley, and green sea turtles, or any other ESA-listed species under NMFS jurisdiction (NMFS 2003). An ITS of 97 turtles was provided based on the estimated annual capture of turtles in fishing gear used in the scallop dredge and trawl fisheries. Twenty-nine of the turtles captured were expected to die as a result of capture. RPMs were provided.

Consultation was 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 authorization 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 the listed species or critical habitat not considered in the previous opinion. NMFS subsequently modified the proposed action when it initiated an emergency action for the scallop fishery on January 20, 2004. The consultation was, therefore, revised to consider the effects to ESA-listed species from the modified proposed action. The Opinion concluded (on February 23, 2004) that the continued authorization of the scallop fishery, including implementation of Amendment 10 and emergency measures, would not jeopardize the continued existence of loggerhead, leatherback, Kemp's ridley, and green sea turtles, or any other ESA-listed species under NMFS jurisdiction (NMFS 2004a). An ITS was provided for these four turtle species, along with RPM's.

Subsequently, on September 3, 2004, consultation was reinitiated to consider new information on the effects of the Atlantic sea scallop fishery on sea turtles that was received from the NEFSC. Consultation was completed on December 15, 2004, and concluded that the anticipated capture of 753 turtles (752 loggerheads and 1 leatherback sea turtle) in the scallop fishery, resulting in death of up to 482 loggerheads and 1 leatherback, was not expected to result in jeopardy to loggerhead and leatherback sea turtles (NMFS 2004b). Consultation was reinitiated on November 1, 2005, based on new information on the number of observed turtle takes in the trawl component of the Atlantic sea 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 authorization of the fishery was not likely to result in jeopardy to any ESA-listed species under NMFS jurisdiction (NMFS 2006a). As described above, NMFS anticipated that up to 760 sea turtles (752 in scallop dredge gear and 8 in scallop trawl gear) would be taken annually in the gear as a result of the continued authorization of the Atlantic sea scallop fishery. Of these, up to 490 (482 in scallop dredge gear and 8 in scallop trawl gear) were anticipated to result in death. Nearly all of the takes (749 of 752 for dredge gear and 5 of 8 for trawl gear) were anticipated to be loggerhead sea turtles. The remaining anticipated takes were for leatherback, Kemp's ridley, and green sea turtles.

Other than these formal consultations, Section 7 consultations were conducted and completed informally for other framework adjustments and amendments to the Scallop FMP. Early informal consultations concluded that the action might affect, but was not likely to adversely affect, some ESA-listed species under NMFS jurisdiction listed as threatened or endangered or designated critical habitat, while others concluded that the proposed changes to the fishery did not trigger reinitiation of consultation.

2.0 DESCRIPTION OF THE PROPOSED ACTION

The proposed action is NMFS' continued authorization of the Atlantic sea scallop fishery managed under the Scallop FMP. A summary of the characteristics of the fishery relevant to the analysis of its potential effects on threatened and endangered species is presented below.

2.1 Description of the Current Fishery for Sea Scallops

The current management measures for the commercial sea scallop (*Placopecten magellanicus*) fishery, the history of the fishery, and the general distribution and habitat characteristics of sea scallops are described in the 45th Northeast Regional Stock Assessment Workshop (45th SAW) (NEFSC 2007). Additional information on the fishery can be found in documents prepared in accordance with the National Environmental Policy Act (NEPA) for Amendment 11, Amendment 10, Joint Framework Adjustment 16 to the Scallop FMP and 39 to the Northeast Multispecies FMP (Framework 16/39), and Framework Adjustment 18 (Framework 18) to the Scallop FMP (NEFMC 2003; 2004a; 2005a). Additional information on the distribution and habitat characteristics of sea scallops can be found in the Essential Fish Habitat source documents for sea scallops (Hart and Chute 2004; Packer *et al.* 1999). A summary of the current fishery and its management history based on these sources is provided below.

The fishing year 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 fishery operates year-round in U.S. waters (Hart 2001), although seasonal peaks in sea scallop landings are evident. These peaks may be influenced by management measures, market conditions, weather, and scallop spawning, among other factors.

Sea 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[1] from Virginia to Long Island, on Georges Bank, in the Great South Channel, and in the Gulf of Maine (Hart and Rago 2006). In Georges Bank and the Mid-Atlantic, sea scallops are harvested primarily at depths of 30-100m, while the bulk of landings from the Gulf of Maine are from near-shore, relatively shallow waters (< 40m) (NEFSC 2007). Landings from Georges Bank and the Mid-Atlantic dominate the fishery (NEFSC 2007).

The commercial harvest of Atlantic sea scallops has occurred along the continental shelf from the Gulf of St. Lawrence to Cape Hatteras since the late 1880's (NEFMC 1982). Sea scallop landings in the U.S. increased substantially after the mid-1940s, with peaks occurring around 1960, 1978, 1990, and 2004 (NEFSC 2007). Scallop fishing effort reached its maximum in 1991 at about 52,000 days absent ("day absent" refers to a day that the fishing vessel is absent from port), and then declined during the 1990s so that effort in 1999 was less than half that in 1991 (NEFSC 2007). 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 2007). LPUE more than quadrupled between 1998 and 2001, and remained high during 2001-2006 (NEFSC 2007).

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 2007). Landings in Georges Bank during 1999-2004 were fairly steady, averaging almost 5,000 metric tons (mt) annually, then increased in 2005-2006, primarily due to reopening of portions of the groundfish closed areas to scallop fishing (NEFSC 2007). 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 (NEFSC 2007). Limited openings to allow scallop fishing in closed areas contributed more than half of Georges Bank landings during 1999-2000 and 2004-2006 (NEFSC 2007).

Mid-Atlantic landings during 1962-1982 averaged less than 1,800 mt annually (NEFSC 2007). An upward trend in both recruitment and landings has been evident in the Mid-Atlantic since the mid-eighties (NEFSC 2007). Landings peaked in 2004 at 24,494 mt before declining during 2005-2006 (NEFSC 2007). There have been four rotational scallop closures in the Mid-Atlantic since 1998 (NEFSC 2007). The areas referred to as Hudson Canyon South and Virginia Beach were closed in 1998 and reopened in 2001. Scallop biomass built up in the Hudson Canyon Closed Area during the closure, and substantial landings were obtained from the area during 2001-2005 (NEFSC 2007). A third rotational closed area, named the Elephant Trunk Closed Area, was closed in 2004 and reopened during March 2007 (NEFSC 2007). Preliminary reports indicate very high catch levels consistent with recent survey data of scallop size distribution and abundance in the area (NEFSC 2007). A fourth area in the Mid-Atlantic, named the Delmarva Closed Area, was closed in 2007 (NEFSC 2007).

The Scallop FMP was implemented on May 15, 1982 (NEFSC 2007). From 1982 to 1994, the primary management control was a minimum average meat weight requirement for landings

[1] "Mid-Atlantic" as used here refers to the Mid-Atlantic Bight which is defined as the area between Cape Hatteras, NC, and Long Island, NY.

(NEFSC 2007). Amendment 4 to the FMP, implemented in 1994, changed the management strategy from meat count regulation to effort control for the entire U.S. Exclusive Economic Zone (EEZ) (NEFSC 2007). Effort controls were included that incrementally restricted days-at-sea (DAS), established minimum ring size, and crew limits (NEFSC 2007). Subsequent amendments and framework adjustments to the FMP during the 1990's 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 (NEFSC 2007). During the same time period, three areas of Georges Bank were closed to scallop fishing under the Northeast Multispecies FMP in order to protect regulated groundfish stocks (NEFSC 2007).

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 (permit categories 2, 3, and 4), with a small dredge (categories 5 and 6), or with trawl nets (categories 7, 8, and 9). Open area DAS and sea scallop access area trip allocations to the vessel vary depending on whether the vessel qualified in the full-time, part-time, or occasional permit category. The greatest number of DAS access area trips are allocated to vessels that qualified 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 ft 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.

Of the 345 limited access scallop permits issued for the 2006 scallop fishing year (March 1, 2006 – February 28, 2007), 249 were for full-time dredge vessels, 52 were for full-time small dredge vessels, 3 were for part-time dredge vessels, and 1 was for a dredge vessel in the Occasional permit category (Memo from R. Silva, NER, SFD to E. Keane, NER, PRD, September 24, 2007). The remaining 11 permits issued were for full-time trawl vessels (Memo from R. Silva, NER, SFD to E. Keane, NER, PRD, September 24, 2007). The total number of limited access scallop permits issued each fishing year, as well as the number in each permit category, change somewhat from year to year. For example, in the 2005 scallop fishing year, 363 limited access scallop permits were issued, with 324 of these being for full-time dredge vessels (NMFS, NER, FSO, pers. comm.). Changes in the number of limited access permits issued from one year to another are not the result of new qualifications for the permit. Rather they can be the result of the transfer of a scallop permit from a confirmation of permit history to a new vessel. Although all permits are not typically used to land scallops in each year, vessel owners are required to renew their limited access permits prior to the end of each fishing year. Failure to do so results in permanent loss of the permit (50 CFR 648.4(a)(2)(i)). A limited access scallop permit that is voluntarily relinquished cannot be reissued to the same or another vessel based on that vessel's scallop fishing history (50 CFR 648.4(a)(2)(i)).

In contrast to limited access scallop permits, the general category (open access) permits may be issued to any vessel at any time, regardless of history in the fishery. As described above, Amendment 4 to the Scallop FMP established open access provisions for vessels that did not qualify for a scallop limited access permit. The open access provisions allow vessels to possess and land scallops either: (1) Through possession of a general category permit or (2) in accordance with the exemption for vessels that have neither a limited access or general category permit. Scallop possession and landing limits are the primary effort control mechanism for vessels that do not possess a limited access scallop permit. Vessels that possess a general category permit for the fishery are allowed to retain or land up to 400 lb of shucked scallops, or 50 U.S. bushels of in-shell scallops, per trip. Vessels that have neither a limited access or general category permit (except those that participate exclusively in state waters) are allowed to possess and land up to 40 lb of scallop meat or 5 bushels of shell stock per trip. In either case, the possession and landing limits are far less than for limited access vessels fishing under a DAS (*i.e.*, no landing limits for limited access vessels fishing under a DAS in open areas, and landing limits of up to 18,000 lb of shucked scallops per trip for full-time limited access vessels fishing in a Sea Scallop Access Area).

For the 2006 scallop fishing year a total of 3,011 general category scallop permits were issued (Permit count provided by NER, SFD, December 2007). There are two types of general category permits for the scallop fishery – 1A and 1B permits. The 1B permit allows the vessel to harvest up to 400 lbs of shucked scallops and requires the vessel to use VMS. The 1A permit allows the vessels to retain up to 40 lbs of shucked scallops and the vessel is not required to use VMS. Of the 3,011 general category scallop permits issued for the 2006 scallop fishing year, 1,903 were 1A permits and 1,108 were 1B permits (Permit count provided by NER, SFD, December 2007).

While the number of general category scallop permits issued is much greater than the number of limited access scallop permits, the number of vessels that actually fish under a general category scallop permit is a fraction of the number of vessels that possess a general category permit. In 2003 and 2004, for example, fewer than 20% of the general category permits issued were actually used to land scallops (NEFMC 2005a). Nevertheless, the number of general category permits issued, the number of general category permitted vessels landing scallops, and the total landings of scallops by vessels possessing a general category permit did increase considerably over the period of 2003-2005 (NEFMC 2007). From 1994-2002, general category permitted vessels contributed less than 3% to the total annual scallop landings (NEFMC 2007). In 2003, the share of the total scallop landings by the general category permitted vessels increased to 3.30% and rose sharply from there to 5.26% in 2004, and 14.09% in 2005 (NEFMC 2007). The change in these last several years has occurred as a result of the increasing percentage of general category landings landed by vessels with homeports in the Mid-Atlantic region, and as a result of shifts in fishing effort by general category vessels to Mid-Atlantic fishing grounds (NEFMC 2007).

In 2004, substantive changes were made to management of the scallop fishery as a result of implementation of Amendment 10 and Frameworks 16/39. Amendment 10 retained the basic effort control measures of the FMP but changed the overall approach to managing the fishery by formalizing an area based management system (NEFSC 2007). Amendment 10 includes provisions and criteria for new rotational closures, and separate allocations (in DAS or total allowable catch (TAC)) for reopened closed areas and general open areas (NEFSC 2007). Amendment 10 defined three types of areas for the purpose of managing the fishery within the management unit. These are: Rotational Closed Areas; Sea Scallop Access Areas; and Open Areas. Different management measures (*i.e.*, with respect to DAS use, DAS allocations, trip allocations and access) apply to each type of area. Trip and DAS allocations as well as measures to close areas and/or open access areas are made through framework adjustments to the Scallop FMP approximately every 2 years. Frameworks 16/39 established Sea Scallop Access Areas within each of the three areas on Georges Bank that had been previously closed to scallop fishing. Framework 18 provided the management measures to respond to changes in the scallop resource for the 2006 and 2007 fishing years, including trip allocations, DAS allocations, opening date, seasonal closure, and allocation adjustment measures for the reopening of the Elephant Trunk Area (ETA), a new Mid-Atlantic closed area, identified as the Delmarva Area, and continuation of the Hudson Canyon Access Area (HCAA) for vessels that did not use any or all of their 2005 HCAA trips.

NMFS is in the process of implementing two regulatory actions – Amendment 11 and Framework Adjustment 19 (Framework 19) – that will further modify operation of the scallop fishery. Amendment 11 will create a limited access program for the general category sector in order to constrain effort in this sector of the fishery (NEFMC 2006a; 2007). The Amendment 11 measures would require vessels to qualify for a general category permit based on their fishing/landings history from a specified time period. The qualifying criteria apply to vessels that have previously held a general category scallop permit as well as vessels that possess a limited access scallop permit but have landed scallops under the general category provisions when not fishing under a Scallop Day-At-Sea (DAS). The Amendment 11 measures also include a method for allocating a portion of the scallop TAC to vessels that qualify for a limited entry general category scallop permit. This is expected to also help to control effort compared to the current management measures since there will be an upward bound of how much all of the qualifying general category vessels are expected to land. Several measures to support the overall goal of implementing a limited entry program for the general category scallop fishery are also included in Amendment 11. These include: (a) interim measures for the transition period to limited entry, (b) the change in issuance date of general category permits, (c) provisions to address permitting issues, and (d) the requirement for use of a Vessel Monitoring System (VMS). The mechanism to allow the formation of sectors in the general category fishery was included in Amendment 11 to allow general category scallop vessels the opportunity to form a sector, and establishes 20% as the maximum allocation to a sector. Sectors, if any are formed, would still require approval and would be assessed at that time with respect to its effects on ESA-listed species. NMFS published a proposed rule for Amendment 11 in the *Federal Register* (72 FR

71315, December 17, 2007). On February 27, 2008, NMFS approved Amendment 11. A final rule to implement its regulatory provisions is expected in 2008.

Framework 19 is being implemented to establish the specifications for the 2008 and 2009 scallop fishing years. Based on the Framework 19 measures as submitted by the Council, NMFS proposes to allocate open area DAS to limited access scallop vessels in the 2008 and 2009 scallop fishing years as follows: (a) 35 DAS and 42 DAS, respectively, for full-time limited access permit holders, (b) 14 DAS and 17 DAS, respectively, for part-time limited access permit holders, and (c) 3 DAS in both fishing years for occasional limited access permit holders. These open area DAS allocations are lower than those proposed for Framework 18 for the 2006 and 2007 scallop fishing years, and which were considered with respect to the effects of the fishery on ESA-listed species for the September 18, 2006 Opinion. For example, open area DAS for limited access vessels in 2007 were 51 DAS for full-time limited access permit holders, 20 for part-time limited access permit holders, and 4 for occasional limited access permit holders.

The possession limits proposed for scallop limited access vessels fishing an access area trip in 2008 and 2009 are the same as those considered in the September 18, 2006, Opinion. In addition, the total number of trips proposed to be allocated for the part-time and occasional limited access vessels are the same as those that were considered as part of the overall operation of the fishery for the September 18, 2006, Opinion. The number of trips proposed to be allocated to full-time limited access vessels for 2008 and 2009 are less than the number of access area trips that were considered with respect to the overall operation of the fishery for the September 18, 2006, Opinion[2]. The distribution of access area trips is slightly different for 2008 and 2009 in comparison to those for 2006 and 2007 which were considered as part of the overall operation of the fishery for the September 18, 2006, Opinion. As proposed, occasional limited access permit holders would be able to take their single access area trip in either the Georges Bank Access Areas (Nantucket Lightship (NLCA) or Closed Area II (CAII)) or the ETA. This is similar to the 2007 fishing year in which occasional limited access permit holders could take a single access area trip in either the Georges Bank access areas (NLCA or Closed Area I (CAI)) or the ETA. However, for the part-time and full-time limited access permit holders, Framework 19, as proposed, would allocate 1 less trip for Georges Bank access areas as compared to allocations for the 2007 scallop fishing year.

The Framework 19 measures as submitted by the Council to NMFS also included a measure that would have discontinued the two-month (September-October) seasonal closure for the ETA. This closure was established under Framework 18 to, in part, reduce the likelihood of sea turtle interactions with scallop dredge gear fished in the area. NMFS is proposing to disapprove the measure in Framework 19 that would have discontinued the seasonal closure. Therefore, if Framework 19 is implemented as NMFS proposes, the 2-month seasonal closure for the ETA would continue as established under Framework 18 to the Scallop FMP. Other changes to scallop rotational management areas as a result of Framework 19 are: (a) Closing the existing

[2] The number of total trips allocated to full-time limited access permit holders for the 2007 fishing year was later reduced from 7 to 5 as a result of adjustment of the trip allocation for the ETAA. Framework 19 would allocate 5 access area trips to full-time limited access permit holders in 2008 and 6 trips in 2009, unless the 2009 trip allocation was subsequently reduced based on new scallop survey data.

HCAA to all scallop fishing as well as expire all un-used HCAA trips as of February 29, 2008, and (b) re-opening the Delmarva access area in 2009, one year ahead of the current schedule.

The Framework 19 measures also provide details for the limited entry program proposed in Amendment 11 to the Scallop FMP. These measures include specifying the percentage of the scallop TACs that are allocated to vessels fishing under the general category provisions, and measures for the transition period to an Individual Fishing Quota (IFQ) program. Similarly, Framework 19 specifies the hard-TAC for the Northern Gulf of Maine (NGOM) limited entry program. The NGOM limited entry program was proposed as part of Amendment 11. As proposed, a vessel issued a NGOM scallop permit may not fish for scallops south of 42° 20′ N, and may not possess or land more than 200 lb of shucked or 25 bushels of in-shell scallops at any time, except the vessel may possess up to 50 bushels of in-shell scallops seaward of the VMS demarcation line. Framework 19 would specify the quarterly hard-TAC for the general category sector during the transition period to the IFQ program. Open area, access area, and NGOM scallop landings by directed general category trips would count against the quarterly TAC. Consequently, if a quarterly TAC is caught, all directed general category scallop fishing would cease for the remainder of the quarter; including access area, and open areas, but excluding the NGOM. Other measures included in Framework 19 that would constrain scallop fishing effort by the general category sector are the prohibitions on scallop fishing by general category vessels in: (a) The Georges Bank access areas if such areas are closed due to the yellowtail bycatch TAC having been met, and (b) the Hudson Canyon rotational closed area proposed to be created under Framework 19.

While the Framework 19 measures as currently proposed would increase the number of access area trips allocated for use in the Mid-Atlantic as compared to Georges Bank, the likelihood of sea turtle interactions with scallop fishing gear in the fishery, overall, is not expected to increase compared to that considered for the September 18, 2006, Opinion given that: (a) Effort in the general category sector of the fishery is expected to be reduced or constrained as a result of the Framework 19 measures in combination with the Amendment 11 measures, (b) all un-used HCAA trips will expire at the end of the 2007 scallop fishing year (February 29, 2008); (b) the HCAA will be closed to all scallop fishing rather than re-opened to all scallop fishing; (c) open area DAS allocations are less for 2008 and 2009 than they were for 2007; (d) the ETA will be closed to scallop fishing in September and October – months when sea turtle interactions have been observed in the area at rates higher than those observed elsewhere in the fishery.

NMFS concluded that neither the implementation of Amendment 11 or of Framework 19 would trigger reinitiation of consultation for the scallop fishery (Memo to the Record from P. Kurkul, September 24, 2007; Memo to the Record from P. Kurkul, February 4, 2008). However, since each affects how the fishery operates, the Amendment 11 and Framework 19 measures as proposed by NMFS are described here because this Opinion considers all effects of the fishery as it currently operates and is likely to operate including if Amendment 11 and Framework 19 are implemented as proposed.

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 (hereafter 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). The proposed action adds a transiting provision, and clarifies 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 (35.9 cm). The comment period has closed and NMFS is preparing a final rule (see Issues Advisory, March 12, 2008).

The gear modification is expected to reduce the severity (*e.g.*, mortality and serious injury) of some sea turtle interactions with scallop dredge gear. However, the gear modification is not expected to reduce the number of sea turtle interactions with scallop dredge gear. Based on the condition of 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 turtles out of the dredge bag thus preventing injuries that occur to turtles 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 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. Additional information on the use of chain mats in the fishery is presented in section 6.2, in which the effects of the continued authorization of the fishery, including dredges with chain mats, are analyzed.

2.1.1 Summary of the Fishery

In Georges Bank and the Mid-Atlantic, sea scallops are harvested primarily at depths of 30-100m, while the bulk of landings from the Gulf of Maine are from near-shore relatively shallow waters (< 40m) (Murray 2004b; 2005; NEFSC 2007). Landings from Georges Bank and the Mid-Atlantic dominate the fishery (NEFSC 2007). Scallop biomass increased considerably in both the Georges Bank and Mid-Atlantic areas since the mid-1990's (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 2007). 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 2007). 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 2007). Biomass was relatively stable during 2003-2006 (NEFSC 2007). LPUE in the fishery more than quadrupled between 1998 and 2001, and remained high during 2001-2006 (NEFSC 2007). 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 2007). The number of hours fished during trips to formerly closed areas is considerably less (NEFSC 2007). Overfishing is not occurring in the scallop fishery, and the stock is not overfished (NEFSC 2007).

The scallop fishery is a limited access fishery but vessels that did not qualify for a limited access permit can obtain a (open access) general category scallop permit (type 1A or 1B). An increase in active general category permits and the increase in landings by general category permitted vessels prompted the initiation of Amendment 11 to the Scallop FMP. In particular, it was noted that in these last several years there has been an increasing percentage of general category landings landed by vessels with homeports in the Mid-Atlantic region, and shifts in fishing effort by general category vessels to Mid-Atlantic fishing grounds (NEFMC 2007). Amendment 11 is expected to contribute to the management objectives of the fishery by reducing or constraining effort in the general category sector. The proposed rule for implementing Amendment 11 has been published in the *Federal Register* and the comment period is now closed. NMFS has approved the amendment and is reviewing comments submitted on the proposed rule prior to completion of the final rule in 2008.

Framework 19 is expected to establish the specifications for the scallop fishery for the 2008 and 2009 scallop fishing years. Scallop fishing effort is not expected to increase as a result of Framework 19.

2.2 Action Area

The management unit for the Scallop FMP is defined in the FMP as the range of the sea 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 (north of 35° N latitude, see Appendix 1) and the adjoining state waters that are affected through the regulation of activities of Federal scallop permit holders fishing in those waters.

3.0 STATUS OF THE SPECIES

NMFS has determined that the action being considered in the Opinion may affect the following ESA-listed sea turtle species in a manner that will likely result in adverse effects:

Loggerhead sea turtle	(*Carretta carretta*)	Threatened
Leatherback sea turtle	(*Dermochelys coriacea*)	Endangered
Kemp's ridley sea turtle	(*Lepidochelys kempii*)	Endangered
Green sea turtle	(*Chelonia mydas*)	Endangered[3]

NMFS has determined that the action being considered in the Opinion is not likely to adversely affect shortnose sturgeon (*Acipenser brevirostrum*), the Gulf of Maine distinct population segment (DPS) of Atlantic salmon (*Salmo salar*), hawksbill sea turtles (*Eretmochelys imbricata*), 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 species under the ESA. Thus, these species will not be considered further in this Opinion. The following discussion is NMFS' rationale for these determinations.

Shortnose sturgeon are benthic fish that mainly occupy the deep channel sections of large rivers. They can be found in rivers along the western Atlantic coast from St. Johns River, Florida (possibly extirpated from this system), to 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). Since the scallop fishery does not operate in or near the rivers where concentrations of shortnose sturgeon are most likely found, it is highly unlikely that the scallop fishery will affect shortnose sturgeon.

The wild populations of Atlantic salmon found in rivers and streams from the lower Kennebec River north to the U.S.- Canada border are listed as endangered under the ESA. These populations include those in the Dennys, East Machias, Machias, Pleasant, Narraguagus, Ducktrap, and Sheepscot Rivers and 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. Results from a 2001 post-smolt trawl survey in Penobscot Bay and 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. Therefore, commercial fisheries deploying small mesh active gear (pelagic trawls and purse seines within 10-m of the surface) in nearshore waters of the Gulf of Maine may have the potential to incidentally take smolts. However, 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

3 Green 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 turtles are considered endangered wherever they occur in U.S. waters.

where concentrations of Atlantic salmon are likely to be found and scallop gear operates in the ocean at or near the bottom rather than near the surface. Thus, this species will not be considered further in this Opinion.

The hawksbill turtle is uncommon in the waters of the continental United States. Hawksbills prefer coral reefs, such as those found in the Caribbean and Central America. Hawksbills feed primarily on a wide variety of sponges but also consume bryozoans, coelenterates, and mollusks. The Culebra Archipelago of Puerto Rico contains especially important foraging habitat for hawksbills. Nesting areas in the western North Atlantic include Puerto Rico and the Virgin Islands. There are accounts of hawksbills in south Florida and a number are encountered in Texas. Since operation of the scallop fishery does not occur in waters that are typically used by hawksbill sea turtles, it is highly unlikely that the scallop fishery will affect this turtle species.

During the CeTAP study, sperm whales were observed along the shelf edge, centered around the 1000 meter depth contour but extending seaward out to the 2000 meter depth contour (CeTAP 1982). Although blue whales are occasionally seen in U.S. waters, they are more commonly found in Canadian waters (Waring *et al.* 2000). Given the range and distribution of these species, both are highly unlikely to occur where the scallop fishery operates.

Right whales, humpback whales, 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 1998b). All four 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). Therefore, operation of the scallop fishery may overlap with the distribution of these cetacean 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 cetacean 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, NMFS has determined that this was a unique and very rare event that is extremely unlikely to reoccur given that these large cetaceans have the speed and maneuverability to get out of the way of oncoming scallop fishing gear. Observer coverage of many fishing trips using mobile gear (*e.g.*, dredge, trawl gear) have shown that these gear types do not pose a reasonable risk of entanglement or capture for large cetaceans.

NMFS also determines that the continued authorization of the scallop fishery will not have any adverse effects on cetacean prey. Right whales and sei whales feed on copepods (Horwood 2002; Kenney 2002). The scallop fishery will not affect the availability of copepods for foraging right and sei whales because copepods are very small organisms that will pass through scallop fishing gear rather than being captured in it. Blue whales feed on euphausiids (krill) (Sears 2002) which, likewise, are too small to be captured in scallop fishing gear. Humpback whales and fin whales also 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 authorization of the scallop fishery will not affect the availability of prey for foraging humpback or fin whales. Sperm whales feed on larger organisms that inhabit the deeper ocean regions (Whitehead 2002). The scallop fishery does not operate in these deep water areas. Therefore, the continued authorization of the scallop fishery will not affect the availability of prey for foraging sperm whales.

The Atlantic sea scallop fishery does not operate in low latitude waters where calving and nursing occurs for these large cetacean species (Aguilar 2002; Clapham 2002; Horwood 2002; Kenney 2002; Sears 2002; Whitehead 2002). Therefore, the continued authorization of the scallop fishery will not affect the oceanographic conditions that are conducive for calving and nursing.

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 (non-fishery) dredging operations, for example, affect sea turtles in the neritic zone (defined as the marine environment extending from mean low water down to 200m (660 foot) depths, generally corresponding to the continental shelf (Lalli and Parsons 1997; Encyclopedia Britannica 2008)). 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 200m (Lalli and Parsons 1997))[4]. As a result, sea turtles still face many of the original threats that were the cause of their listing under the ESA.

Sea turtles are listed under the ESA at the species level rather than as subspecies or distinct population segments (DPS). Therefore, information on the range-wide status of each species is included to provide the reader with information on the status of each species, overall. 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; USFWS 1997; Marine Turtle Expert Working Group (TEWG) 1998; TEWG 2000; NMFS and USFWS 2007a; 2007b; 2007c; 2007d; Leatherback TEWG 2007), and recovery plans for the loggerhead sea turtle (NMFS and USFWS 1991a), leatherback sea turtle (NMFS and USFWS 1992; NMFS and USFWS 1998a;), Kemp's ridley sea turtle (USFWS and NMFS 1992), and green sea turtle (NMFS and USFWS 1991b; NMFS and USFWS 1998b).

3.1.1 Loggerhead sea turtle

Loggerhead sea turtles are a cosmopolitan species, found in temperate and subtropical waters. Loggerhead sea turtles are the most abundant species of sea turtle in U.S. waters.

[4] As described in Bolten (2003), oceanographic terms have frequently been used incorrectly to describe sea turtle life stages. In both turtle literature and past Opinions for the continued authorization of the scallop fishery, 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. Turtles can be "benthic" or pelagic" in either the neritic or oceanic zones.

Pacific Ocean. In the Pacific Ocean, major loggerhead nesting grounds are generally located in temperate and subtropical regions with scattered nesting in the tropics. The abundance of loggerhead turtles on nesting colonies throughout the Pacific basin have declined dramatically over the past 10-20 years. Loggerhead sea turtles in the Pacific are represented by a northwestern Pacific nesting group (located in Japan) and a smaller southwestern nesting group that occurs in Australia (Great Barrier Reef and Queensland), New Caledonia, New Zealand, Indonesia, and Papua New Guinea. Data from 1995 estimated the Japanese nesting group at 1,000 female loggerhead turtles (Bolten *et al.* 1996). More recent information suggests that nest numbers have increased somewhat over the period 1998-2004 (NMFS and USFWS 2007a). However, this time period is too short to make a determination of the overall trend in nesting (NMFS and USFWS 2007a). Genetic analyses of loggerhead females nesting in Japan indicates the presence of genetically distinct nesting colonies (Hatase *et al.* 2002).

In Australia, long-term census data has been collected at some rookeries since the late 1960's and early 1970's, and nearly all the data show marked declines in nesting since the mid-1980's (Limpus and Limpus 2003). The nesting group in Queensland, Australia, was as low as 300 females in 1997.

Pacific loggerhead turtles are captured, injured, or killed in numerous Pacific fisheries including gillnet, longline, and trawl fisheries in the western and/or eastern Pacific Ocean (NMFS and USFWS 2007a). In Australia, where turtles are taken in bottom trawl and longline fisheries, efforts have been made to reduce fishery bycatch (NMFS and USFWS 2007a).

In July 2007, NMFS received a petition requesting that loggerhead sea turtles in the North Pacific be classified as a DPS with endangered status and critical habitat designated. The petition also requested that, if the North Pacific loggerhead is not determined to meet the DPS criteria, that loggerheads throughout the Pacific Ocean be designated as a DPS and listed as endangered.

Indian Ocean. Loggerhead sea turtles are distributed throughout the Indian Ocean, along most mainland coasts and island groups (Baldwin *et al.* 2003). Throughout the Indian Ocean, loggerhead sea turtles face many of the same threats as in other parts of the world including loss of nesting beach habitat, fishery interactions, and turtle meat and/or egg harvesting.

In the southwestern Indian Ocean, loggerhead nesting has shown signs of recovery in South Africa where protection measures have been in place for decades. However, in other southwestern areas (*e.g.*, Madagascar and Mozambique) loggerhead nesting groups are still affected by subsistence hunting of adults and eggs (Baldwin *et al.* 2003). The largest known nesting group of loggerheads in the world occurs in Oman in the northern Indian Ocean. An estimated 20,000-40,000 females nest at Masirah, the largest nesting site within Oman, each year (Baldwin *et al.* 2003). In the eastern Indian ocean, all known nesting sites are found in Western Australia (Dodd 1988). As has been found in other areas, nesting numbers are disproportionate within the area with the majority of nesting occurring at a single location. This may, however, be the result of fox predation on eggs at other Western Australia nesting sites (Baldwin *et al.* 2003).

Mediterranean Sea. Nesting in the Mediterranean is confined almost exclusively to the eastern basin (Margaritoulis *et al.* 2003). The greatest number of nests in the Mediterranean are found in Greece with an average of 3,050 nests per year (Margaritoulis *et al.* 2003; NMFS and USFWS 2007a). Turkey has the second largest number of nests with 2,000 nest per year(NMFS and USFWS 2007a). There is a long history of exploitation for loggerheads in the Mediterranean (Margaritoulis *et al.* 2003). Although much of this is now prohibited, some directed take still occurs (Margaritoulis *et al.* 2003). Loggerheads in the Mediterranean also face the threat of habitat degradation, incidental fishery interactions, vessel strikes, and marine pollution (Margaritoulis *et al.* 2003). Longline fisheries, in particular, are believed to catch thousands of juvenile loggerheads each year (NMFS and USFWS 2007a), although genetic analyses indicate that only a portion of the loggerheads captured originate from loggerhead nesting groups in the Mediterranean (Laurent *et al.* 1998).

Atlantic Ocean. Ehrhart *et al.* (2003) provided a summary of the literature identifying known nesting habitat of Atlantic loggerheads as well as known foraging areas within the Atlantic. Information is also provided in the 5-year status review (NMFS and USFWS 2007a). Briefly, nesting occurs on island and mainland beaches on both sides of the Atlantic and both north and south of the Equator (Ehrhart *et al.* 2003). By far, the majority of nesting occurs on beaches of the southeastern U.S. (NMFS and USFWS 2007a). Annual nest counts for loggerhead sea turtles on beaches from other countries are in the hundreds with the exception of Brazil where a total of 4,837 nests were reported for the 2003/2004 nesting season (Marcovaldi and Chaloupka 2007; NMFS and USFWS 2007a). In both the eastern and western Atlantic, waters as far north as 41°-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). Of these, loggerheads that nest and/or forage in U.S. waters of the western Atlantic have been most extensively studied.

Loggerheads commonly occur throughout the inner continental shelf from Florida through Cape Cod, Massachusetts although their presence varies with the seasons due to changes in water temperature (Shoop and Kenney 1992; Epperly *et al.* 1995a; Epperly *et al.* 1995b; Braun and Epperly 1996; Epperly and Braun-McNeill 2002; Mitchell *et al.* 2003). Aerial surveys of continental shelf waters north of Cape Hatteras indicate that loggerhead sea turtles are most commonly sighted in waters with bottom depths ranging from 22 to 49 meters deep (Shoop and Kenney 1992). However, 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; Blumenthal *et al.* 2006; Hawkes *et al.* 2006; McClellan and Read 2007). The presence of loggerhead turtles in an area is also influenced by water temperature. Loggerheads have been observed in waters with surface temperatures of 7-30°C but water temperatures of ≥11°C are favorable to sea turtles (Shoop and Kenney 1992; Epperly *et al.* 1995b).

Within the action area of this consultation, loggerhead sea turtles occur year round in offshore waters off of North Carolina where water temperature is influenced by the Gulf Stream. As coastal water temperatures warm in the spring, loggerheads begin to migrate to North Carolina inshore waters (*e.g.*, Pamlico and Core Sounds) and also move up the coast (Epperly *et al.* 1995a; Epperly *et al.* 1995b; Epperly *et al.* 1995c; Braun-McNeill and Epperly 2004), occurring in Virginia foraging areas as early as April 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 may remain in Mid-Atlantic and Northeast areas until late Fall. By December loggerheads have migrated from inshore North Carolina waters 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; Epperly and Braun-McNeill 2002).

Loggerheads mate in late March-early June, and eggs are laid throughout the summer, with a mean clutch size of 100-126 eggs in the southeastern United States. Individual females nest multiple times during a nesting season, with a mean of 4.1 nests/individual (Murphy and Hopkins 1984). Nesting migrations for an individual female loggerhead are usually on an interval of 2-3 years, but can vary from 1-7 years (Dodd 1988).

The scientific literature for loggerhead sea turtles recognizes five nesting groups in the western North 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 (Márquez 1990; TEWG 2000); and (5) a Dry Tortugas group that nest on beaches of the islands of the Dry Tortugas, near Key West, Florida (NMFS SEFSC 2001). Genetic analyses of mitochondrial DNA, which a turtle inherits from its mother, indicate that there are genetic differences between turtles that nest at and originate from the beaches used by each of the five identified nesting groups of females (TEWG 2000). However, analyses of microsatellite loci from nuclear DNA, which represents the genetic contribution from both parents, indicates little to no genetic differences between turtles originating from nesting beaches of the five western North Atlantic loggerhead 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).

In the western Atlantic, most loggerhead sea turtles nest from North Carolina to Florida and along the Gulf coast of Florida. In 1989, a statewide sea turtle Index Nesting Beach Survey (INBS) program was developed and implemented in Florida. There are currently 33 nesting beaches in the INBS program (letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006). As of 2006, 27 of the 33 beaches had reached the mandatory minimum of 10-years participation for their nest count data to be included in trend evaluations (letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006). Nesting recorded by the INBS program on the 27 beaches represented an average of 65% of all annual nesting by loggerheads in the state for the period 2001-2005 (letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation

Commission, October 25, 2006). Standardized nesting beach survey programs have been implemented in Georgia, South Carolina, and North Carolina as well (Dodd 2003; USFWS and NMFS 2003). A near complete census of the Dry Tortugas nesting beaches were conducted from 1995 – 2004 (excluding 2002). However, no trend in the number of nests laid was detected for the time period and no surveys have been conducted since 2004 (NMFS and USFWS 2007a). Survey effort to counts nests for loggerhead nesting beaches of the Yucatán, Mexico, was consistent from 1987-2001 for seven beaches in Quintana Roo, Mexico (NMFS and USFWS 2007a). However, nesting survey effort overall has been inconsistent among the Yucatán nesting beaches (Zurita *et al.* 2003).

Sea turtle nesting survey data is important in that it provides 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 5-year review for loggerhead sea turtles (NMFS and USFWS 2007a) compiled the most recent information on mean number of loggerhead nests per year, and, where available, the approximated counts of nesting females for each of the five identified western north Atlantic loggerhead nesting groups. These are: (1) For the south Florida nesting group, a mean of 65,460 loggerhead nests per year with approximately 15,966 females nesting per year; (2) for the northern nesting group, a mean of 5,151 nests per year (no estimate of number of females nesting per year provided); (3) for the Florida panhandle nesting group, a mean of 910 nests per year with approximately 222 females nesting per year; (4) for the Dry Tortugas nesting group, a mean of 246 nests per year with approximately 60 females nesting per year; and (5) for the Yucatán nesting group, a range of 903-2,231 nests per year from 1987-2001 (no estimate of number of nesting females provided) (NMFS and USFWS 2007a). As is evident from this information, nests for the south Florida nesting group make up the majority of all loggerhead nests counted along the U.S. Atlantic and Gulf coasts and represents the largest known loggerhead nesting group (in terms of number of nesting females) in the Atlantic (USFWS and NMFS 2003; NMFS and USFWS 2007a). The northern nesting group is the second largest for loggerheads within the United States but smaller than the south Florida nesting group. The remaining three nesting groups (the Dry Tortugas, Florida Panhandle, and Yucatán) are, again, much smaller in terms of the number of nests laid and the estimated number of females laying nests.

In 2006, information was presented at an international sea turtle symposium (Meylan *et al.* 2006) and in a letter to NMFS (letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006) that a trend analysis of the nesting data collected for Florida's INBS program showed a decrease in nesting of 22.3% in the annual nest density of surveyed shoreline over the 17-year period and a 39.5% decline since 1998 (letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006). Data collected in Florida for the 2007 loggerhead nesting season reveals that the decline in nest numbers has continued, with even fewer nests counted in 2007 in comparison to any previous year of the period, 1989-2007 (Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission web posting November 2007). Declines in nesting have been noted for some of the other western Atlantic loggerhead nesting groups as well. Standardized ground surveys of 11 North Carolina, South Carolina, and Georgia nesting beaches showed a significant declining trend of 1.9%

annually in loggerhead nesting from 1983-2005 (NMFS and USFWS 2007a). Aerial surveys conducted by the South Carolina Department of Natural Resources showed a 3.1% annual decline in nesting since 1980 (Dodd 2003; NMFS and USFWS 2007a). The South Carolina data represents approximately 59% of nesting by the northern nesting group (Dodd 2003). A significant declining trend (P=0.04) in loggerhead nesting of 6.8% annually from 1995-2005 has also been detected for the Florida Panhandle nesting group (NMFS and USFWS 2007a). Nesting for the Yucatán nesting group is characterized as having declined since 2001 (NMFS and USFWS 2007a) while no trend is detectable for the Dry Tortugas nesting group (NMFS and USFWS 2007a).

Unlike nesting beach data, 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 western 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.* 2004; Mansfield 2006; Ehrhart *et al.* 2007; Epperly *et al.* 2007). Maier *et al.* (2004) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the southeast coast of the United States (Winyah Bay, South Carolina to St. Augustine, FL) during the period 2000 – 2003. A comparison of loggerhead catch data from this study with historical values suggested that in-water populations of loggerhead sea turtles along the southeastern United States appear to be larger, possibly an order of magnitude higher than they were 25 years ago (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 similarly found a significant increase in catch rates for loggerhead sea turtles for the latter period (Epperly *et al.* 2007). A long-term, on-going, study of loggerhead abundance in the Indian River Lagoon System of Florida found a significant increase in the relative abundance of loggerheads over the last 4 years of the study (Ehrhart *et al.* 2007). However, there was no discernible trend in loggerhead abundance during the 24-year time period of the study (1982-2006) (Ehrhart *et al.* 2007). In contrast to these studies, Morreale *et al.* (2004) observed a decline in the incidental catch of loggerhead sea turtles in pound net gear fished around Long Island, NY, during the period 2002-2004 in comparison to the period 1987-1992, with only two loggerhead sea turtles observed captured in pound net gear during the period 2002-2004. No additional loggerheads were reported captured in pound net gear through 2007, although 2 loggerhead sea turtles were found cold-stunned on Long Island bay beaches in the fall of 2007 (Memo to the File, L. Lankshear, December 2007). 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 1980's. Significantly fewer turtles ($p<0.05$) were observed in both the spring (May-June) and the summer (July-August) of 2001-2004 compared to aerial surveys in the 1980's (Mansfield 2006). A comparison of median densities from the 1980's to the 2000's 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 diversity of a sea turtle's life history leaves them susceptible to many natural and human impacts, including impacts while they are on land, in the benthic environment, and in the pelagic environment. Recent studies have established that the loggerheads 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). One of the studies tracked the movements of adult females post-nesting and found a difference in habitat use was related to body size with larger turtles staying in coastal waters and smaller turtles 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 while others moved off into oceanic waters (McClellan and Read 2007). However, unlike the Hawkes *et al.* study (2006), there was no significant difference in the body size of turtles that remained in neritic waters versus oceanic waters (McClellan and Read 2007). In either case, the research not only supports the need to revise the life history model for loggerheads but also demonstrates that threats to loggerheads in both the neritic and oceanic environments are likely impacting multiple life stages of this species.

The 5-year status review of loggerhead sea turtles recently completed by NMFS and the USFWS provides a summary of natural as well as anthropogenic threats to loggerhead sea turtles (NMFS and USFWS 2007a). Amongst those of natural origin, hurricanes are known to be destructive to sea turtle nests. Sand accretion and rainfall that result from these storms as well as wave action can appreciably reduce hatchling success. Other sources of natural mortality include cold stunning and biotoxin exposure.

Anthropogenic factors that impact hatchlings and adult female turtles on land, or the success of nesting and hatching include: beach erosion, beach armoring and nourishment; artificial lighting; beach cleaning; increased human presence; recreational beach equipment; beach driving; coastal construction and fishing piers; exotic dune and beach 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 and feed on turtle eggs (NMFS and USFWS 2007a). 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.

Sea turtles, including loggerhead sea turtles, 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 report concluded that for juvenile, subadults, and breeders in coastal waters, the most important source of human caused mortality in U.S. Atlantic waters was fishery interactions. Of these, the U.S. south Atlantic and Gulf of Mexico shrimp fisheries were considered to pose the greatest cause of mortality to neritic juvenile and adult age classes of loggerheads accounting for an estimated 5,000 – 50,000 loggerheads deaths each year (NRC 1990). Significant changes to the south Atlantic and Gulf of Mexico shrimp fisheries have occurred since 1990, and the effects of these shrimp fisheries on ESA-listed species, including loggerhead sea turtles, have been assessed several times through section 7 consultation. There is also a lengthy regulatory history with regard to the use of Turtle Excluder Devices (TEDs) in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries (Epperly and Teas 2002; NMFS 2002; Lewison *et al.* 2003). Section 7 consultation was reinitiated in 2002 to, in part, consider the effect of a new rulemaking that would require increasing the size of TED escape openings to allow larger loggerheads (and green sea turtles) to escape from shrimp trawl gear. The resulting Opinion was completed in December 2002 and concluded that, as a result of the new rule, annual loggerhead mortality from capture in shrimp trawls would decline from an estimated 62,294 to 3,947 turtles assuming that all TEDs were installed properly and that compliance was 100% (Epperly *et al.* 2002; NMFS 2002). The total level of take for loggerhead sea turtles as a result of the U.S. south Atlantic and Gulf of Mexico shrimp fisheries was estimated to be 163,160 loggerheads per year (NMFS 2002). On February 21, 2003, NMFS issued the final rule to require the use of the larger opening TED (68 FR 8456). The rule also provided the measures to disallow several previously approved TED designs that did not function properly under normal fishing conditions, and to require modifications to the trynet and bait shrimp exemptions to the TED requirements to decrease mortality of sea turtles.

The NRC report (1990) also stated that other U.S. Atlantic fisheries collectively accounted for 500-5,000 loggerhead deaths each year, but recognized that there was considerable uncertainty in the estimate. Subsequent studies suggest that these numbers were underestimated. For example, the first estimate of loggerhead sea turtle bycatch in U.S. Mid-Atlantic bottom otter trawl gear was completed in September 2006 (Murray 2006). Observers reported 66 loggerhead turtle interactions with bottom otter trawl gear during the period of which 38 were reported as alive and uninjured and 28 were reported as dead, injured, resuscitated, or of unknown condition (Murray 2006). Seventy-seven percent of observed turtle interactions occurred on vessels fishing for summer flounder (50%) and croaker (27%). The remaining 23% of observed takes occurred on vessels targeting weakfish (11%), long-finned squid (8%), groundfish (3%) and short-finned squid (1%) (Murray 2006). Based on observed interactions and fishing effort as reported on VTRs, the average annual loggerhead bycatch in these bottom-otter trawl fisheries combined was estimated to be 616 sea turtles for each year of the period 1996-2004 (Murray 2006).

The U.S. tuna and swordfish longline fisheries that are managed under the Highly Migratory Species Fishery Management Plan (HMS FMP), were estimated to capture 1,905 loggerheads (no more than 339 mortalities) for each 3-year period (NMFS 2004c). NMFS has mandated gear changes for the HMS fishery to reduce turtle bycatch and the likelihood of death from those takes that would still occur (Fairfield-Walsh and Garrison 2007). In 2006, there were 46 observed interactions between loggerhead sea turtles and longline gear used in the HMS fishery. Nearly all of the loggerheads (42 of 46) were released alive but with injuries (Fairfield-Walsh and Garrison 2007). The majority of the injured had been hooked internally (Fairfield-Walsh and Garrison 2007). Based on the observed take, an estimated 561 (range = 318-981) loggerhead sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP in 2006 (Fairfield-Walsh and Garrison 2007). This number is an increase from 2005 when 274 loggerheads were estimated to have been taken in the fisheries but is still lower than some previous years in the period of 1992-2006 (Fairfield-Walsh and Garrison 2007). This fishery represents just one of several longline fisheries operating in the Atlantic. Lewison *et al.* (2004) estimated that 150,000-200,000 loggerheads were taken in the Atlantic longline fisheries in 2000 (includes the U.S. Atlantic tuna and swordfish longline fisheries as well as others).

Summary of Status for Loggerhead Sea Turtles

Loggerheads are a long-lived species and reach sexual maturity relatively late; 20-38 years (NMFS SEFSC 2001). Loggerhead sea turtles are injured and killed by numerous human activities (NRC 1990; NMFS and USFWS 2007a). There are no population estimates for loggerhead sea turtles in any of the ocean basins in which they occur.

Genetic differences exist between 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). 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). Based on the most recent information, a decline in the annual nest counts has been measured or suggested for four of five western Atlantic loggerhead nesting groups. These include the south Florida nesting group which is the largest (in terms of number of nests laid) in the Atlantic.

Based on its 5-year status review of the species, NMFS and the USFWS (2007a) determined that threatened 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 DPS should be identified for the loggerhead turtle, and what the status of any DPSs should be (NMFS and USFWS 2007a). As described above, in July 2007, NMFS received a petition requesting that loggerhead sea turtles in the North Pacific be classified as a DPS with endangered status and critical habitat designated or, alternatively, that loggerheads throughout the Pacific be designated as a DPS and listed as endangered. NMFS received a similar petition on November 15, 2007, that requested loggerhead sea turtles in the western North Atlantic be classified as a DPS with endangered status and critical habitat designated or, alternatively, that loggerhead sea turtles in the Atlantic be classified as a DPS and designated as endangered. NMFS has published a 90-day finding for each of the petitions in the *Federal Register* (72 FR 64585, November 16, 2007, and 73 FR 11849, March 5, 2008, respectively), and concluded that the petitioners presented substantial scientific and commercial information indicating that a reclassification of the loggerhead in the North Pacific and North Atlantic Oceans as DPSs and listing of each of those DPSs with endangered status may be warranted. As described in each of the petition findings, the ESA defines a "species" as "...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." NMFS and the FWS published a joint policy defining the phrase "distinct population segment" on February 7, 1996 (61 FR 4722). Two elements are considered in a decision on whether a population segment qualifies as a DPS under the ESA: discreteness of the population segment in relation to the remainder of the species and significance of the population segment to the species. If a population segment qualifies as a DPS, the conservation status of that DPS is evaluated to determine whether it is threatened or endangered. NMFS is initiating a review of the status of the species to determine whether each petitioned action is warranted, and to determine whether any additional changes to the current listing of the loggerhead sea turtle are warranted.

NMFS has also convened a new loggerhead TEWG to review all available information on Atlantic loggerheads in order to determine what can be said about the status of this species in the Atlantic. A final report from the Loggerhead TEWG is not yet available. An interim update was provided by the Loggerhead TEWG to NMFS in December 2007. In summary, the memo stated that nest counts, fishery dependent data, and stranding data do not provide the necessary insight into loggerhead turtle population dynamics to properly assess species status (Loggerhead TEWG 2007). As has been stated in the literature (Meylan 1982; Ross 1996; Zurita *et al.* 2003; Hawkes *et al.* 2005), the TEWG remarked that nest counts alone provide no insight into the trend/abundance of sexually mature males or of other age classes of either sex (Loggerhead TEWG 2007). In addition, the TEWG stated that interpreting the meaning of a decline in nest counts in terms of the status/trend of the number of nesting females in the population is difficult since converting nest counts to the number of nesting females is confounded by several issues such as variability in the number of nests per female per year; variability in remigration interval; and, as the ability to nest is resource dependent, the effect of habitat changes and the availability of food resources (Loggerhead TEWG 2007). The TEWG is continuing to explore several hypotheses for why nest counts have been declining. These hypotheses will be more fully discussed in the final report (Loggerhead TEWG 2007).

3.1.2 Leatherback sea turtle

Leatherback sea turtles are widely distributed throughout the oceans of the world, and are found in waters of the Atlantic and Pacific Oceans, the Caribbean Sea, and the Gulf of Mexico (Ernst and Barbour 1972). Leatherback sea turtles are the largest living turtles and range farther than any other sea turtles species; their large size and tolerance of relatively low temperatures allows them to occur in northern waters such as 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). However, the most recent population size estimate for the North Atlantic alone is a range of 34,000-94,000 adult leatherbacks (Leatherback TEWG 2007). Thus, there is 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; NMFS and USFWS 1998a; Sarti *et al.* 2000; Spotila *et al.* 2000). Leatherback 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). For example, the nesting group on Terengganu (Malaysia) - which was one of the most significant nesting sites in the western Pacific Ocean - has declined severely from an estimated 3,103 females in 1968 to 2 nesting females in 1994 (Chan and Liew 1996). Nesting groups of leatherback turtles along the coasts of the Solomon Islands, which historically supported important nesting groups, are also reported to be declining (D. Broderick, pers. comm., *in* Dutton *et al.* 1999). In Fiji, Thailand, Australia, and Papua-New Guinea (East Papua), leatherback turtles have only been known to nest in low densities and scattered colonies.

Only an Indonesian nesting group has remained relatively abundant in the Pacific basin. The largest, extant leatherback nesting group in the Indo-Pacific lies on the north Vogelkop coast of Irian Jaya (West Papua), Indonesia, with over 1,000 nesting females during the 1996 season (Suarez *et al.* 2000). During the early-to-mid 1980s, the number of female leatherback turtles nesting on the two primary beaches of Irian Jaya appeared to be stable. However, in 1999, for example, local Indonesian villagers started reporting dramatic declines in sea turtles near their villages (Suarez 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.*, Suarez 1999).

In the western Pacific Ocean and South China Seas, leatherback turtles are captured, injured, or killed in numerous fisheries including Japanese longline fisheries. Leatherback turtles in the western Pacific are also 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, leatherback nesting is declining along the Pacific coast of Mexico and Costa Rica. According to reports from the late 1970s and early 1980s, three beaches located on the Pacific coast of Mexico support as many as half of all leatherback turtle nests. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherback turtles has declined to slightly more than 200 during 1998-99 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. Between 1988 and 1999, the nesting group declined from 1,367 to 117 female leatherback turtles. Based on their models, Spotila *et al.* (2000) estimated that the group could fall to less than 50 females by 2003-2004. An analysis of the Costa Rican nesting beaches indicates a decline in nesting during the past 15 years of monitoring (1989-2004) with approximately 1,504 females nesting in 1988-89 to an average of 188 females nesting in 2000-2001 and 2003-2004 (NMFS and USFWS 2007b). A similar dramatic decline has been seen on nesting beaches in Pacific Mexico, where tens of thousands of leatherback nests were laid on the beaches in the 1980s but where a total of only 120 nests on the four primary index beaches (combined) were counted in the 2003-2004 season.

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 leatherback turtles in the eastern Pacific Ocean. Given the declines in leatherback nesting in the Pacific, some researchers have concluded that the leatherback is on the verge of extinction in the Pacific Ocean (*e.g.,* Spotila *et al.* 1996; Spotila *et al.* 2000).

Indian Ocean. Leatherbacks nest in several areas around the Indian Ocean. These sites include Tongaland, South Africa (Pritchard 2002), and the Andaman and Nicobar Islands (Andrews *et al.* 2002). Intensive survey and tagging work in 2001 provided new information on the level of nesting in the Andaman and Nicobar Islands (Andrews *et al.* 2002). Based on the survey and tagging work, it was estimated that 400-500 female leatherbacks nest annually on Great Nicobar Island (Andrews *et al.* 2002). The number of nesting females using the Andaman and Nicobar Islands combined was estimated around 1000 (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).

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

A 1979 aerial survey of the outer Continental Shelf from Cape Hatteras, North Carolina to Cape Sable, Nova Scotia showed leatherbacks to be present throughout the area with the most numerous sightings made from the Gulf of Maine south to Long Island. Leatherbacks were sighted in water depths ranging from 1-4151m but 84.4% of sightings were in waters less than 180 m (Shoop and Kenney 1992). Leatherbacks were sighted in waters within a sea surface temperature range similar to that observed for loggerheads; from 7-27.2°C (Shoop and Kenney 1992). However, leatherbacks appear to have a greater tolerance for colder waters in comparison to loggerhead sea turtles since more leatherbacks were found at the lower temperatures as compared to loggerheads (Shoop and Kenney 1992). This aerial survey estimated the leatherback population for the northeastern U.S. at approximately 300-600 animals (from near Nova Scotia, Canada to Cape Hatteras, North Carolina). 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 underestimates the leatherback population for the northeastern U.S. Estimates of leatherback abundance of 1,052 turtles (C.V.= 0.38) and 1,174 turtles (C.V.= 0.52) were obtained from surveys conducted from Virginia to the Gulf of St. Lawrence in 1995 and 1998, respectively (Palka 2000). However, since these estimates were also based on sightings of leatherbacks at the surface, the author considered the estimates to be negatively biased and the true abundance of leatherbacks may be 4.27 times the estimates (Palka 2000). Studies of satellite tagged leatherbacks suggest that they spend a 10% - 41% of their time at the surface, depending on the phase of their migratory cycle (James *et al.* 2005a). 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.* 2005a).

Leatherbacks are a long lived species (> 30 years). They mature at a younger age than loggerhead turtles, with an estimated age at sexual maturity of about 13-14 years for females with 9 years reported as a likely minimum (Zug and Parham 1996) and 19 years as a likely maximum (NMFS SEFSC 2001). In the U.S. and Caribbean, female leatherbacks nest from March through July. They nest frequently (up to 7 nests per year) during a nesting season and nest about every 2-3 years. During each nesting, they produce 100 eggs or more in each clutch and can produce 700 eggs or more per nesting season (Schultz 1975). However, a significant portion (up to approximately 30%) of the eggs can be infertile. Therefore, the actual proportion of eggs that can result in hatchlings is less than this seasonal estimate. As is the case with other sea turtle species, leatherback hatchlings enter the water soon after hatching. Based on a review of all sightings of leatherback sea turtles of <145 cm (56.55 in) curved carapace length (CCL), Eckert (1999) found that leatherback juveniles remain in waters warmer than 26° C until they exceed 100 cm (39 in) CCL.

As described in Section 3.1.1, sea turtle nesting survey data is important in that it provides information on the relative abundance of nesting, and the contribution of each population/subpopulation to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually, and as an indicator of the trend in the number of nesting females in the nesting group. The 5-year review for leatherback sea turtles (NMFS and USFWS 2007b) compiled the most recent information on mean number of leatherback nests per year for each of the seven leatherback populations or groups of populations that were identified by the Leatherback TEWG as occurring within the Atlantic. These are: Florida, North Caribbean, Western Caribbean, Southern Caribbean, West Africa, South Africa, and Brazil. 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). An analysis of Florida's Index Nesting Beach Survey 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 (Leatherback TEWG 2007). The TEWG reports an increasing or stable trend for all of the seven populations or groups of populations with the exception of the Western Caribbean and West Africa. However, caution is also warranted even for those that were identified as stable or increasing. In St. Croix, for example, researchers have noted a declining presence of neophytes (first-time nesters) since 2002 (Garner *et al.* 2007). In addition, 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 (Leatherback TEWG 2007), and represents more than half of total nesting by leatherback sea turtles world-wide (Hilterman and Goverse 2004). Nest numbers in Suriname have shown an increase and the long-term trend for the Suriname and French Guiana nesting group seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests for Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). The most recent Leatherback TEWG report (2007) indicates that using nest numbers from 1967-2005, a positive population growth rate was found over the 39-year period for French Guinea and Suriname, with a 95% probability that the population was growing. Nevertheless, given the magnitude of leatherback nesting in this area compared to other nest sites, impacts to this area that negatively impact leatherback sea turtles could have profound impacts on the species, overall.

Tag return data demonstrate that leatherbacks that nest in South America also use U.S. waters. A nesting female tagged May 29, 1990, in French Guiana was later recovered and released alive from the York River, VA. Another nester tagged in French Guiana was later found dead in Palm Beach, Florida (STSSN database). Many other examples also exist. For example, leatherbacks tagged at nesting beaches in Costa Rica have been found in Texas, Florida, South Carolina, Delaware, and New York (STSSN database). Leatherback 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).

Of the Atlantic turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), and their 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. They are also susceptible to entanglement in gillnets (used in various fisheries) and capture in trawl gear (*e.g.*, shrimp trawls, bottom otter trawls). Sea turtles 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.

Leatherbacks are exposed to pelagic longline fisheries in many areas of their range. According to observer records, an estimated 6,363 leatherback sea turtles were caught by the U.S. Atlantic tuna and swordfish longline fisheries between 1992-1999, of which 88 were released dead (NMFS SEFSC 2001). Since the U.S. fleet accounts for only 5-8% of the hooks fished in the Atlantic Ocean, adding up the under-represented observed takes of the other 23 countries actively fishing in the area would likely result in annual take estimates of thousands of leatherbacks over different life stages (NMFS SEFSC 2001).

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). 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). Fixed gear fisheries in the Mid-Atlantic have also contributed to leatherback entanglements. For example, in North Carolina, two leatherback sea turtles were reported entangled in a crab pot buoy inside Hatteras Inlet (NMFS SEFSC 2001). A third leatherback was reported entangled in a crab pot buoy in Pamlico Sound off of Ocracoke. This turtle was disentangled and released alive; however, lacerations on the front flippers from the lines were evident (NMFS SEFSC 2001). In the Southeast, leatherbacks are vulnerable to entanglement in Florida's lobster pot and stone crab fisheries as documented on stranding forms. In the U.S. Virgin Islands, where one of five leatherback strandings from 1982 to 1997 were due to entanglement (Boulon 2000), leatherbacks have been observed with their flippers wrapped in the line of West Indian fish traps (R. Boulon, pers. comm. to Joanne Braun-McNeill, NMFS SEFSC 2001).

Leatherback interactions with the U.S. south Atlantic and Gulf of Mexico shrimp fisheries, are also known to occur (NMFS 2002). Leatherbacks are likely to encounter shrimp trawls working in the coastal waters off the 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, on February 21, 2003, NMFS issued a final rule to amend the TED regulations. 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 turtles (see section 3.1.1 above for further information on the shrimp trawl fishery).

Other trawl fisheries are also known to interact with leatherback sea turtles although on a much smaller scale. In October 2001, for example, a fisheries observer documented the take of a leatherback in a bottom otter trawl fishing for *Loligo* squid off of Delaware. TEDs are not 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 nearshore waters of the Mid-Atlantic states are also known to capture, injure and/or kill leatherbacks when these fisheries and leatherbacks co-occur. Data collected by the NEFSC Fisheries Observer Program from 1994 through 1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54% to 92%. In North Carolina, a leatherback was reported captured in a gillnet set in Pamlico Sound in the spring of 1990 (D. Fletcher, pers.comm. to Sheryan Epperly, NMFS SEFSC 2001). Five other leatherbacks were released alive from nets set in North Carolina during the spring months: one was from a net (unknown gear) set in the nearshore waters near the North Carolina/Virginia border (1985); two others had been caught in gillnets set off of Beaufort Inlet (1990); a fourth was caught in a gillnet set off of Hatteras Island (1993), and a fifth was caught in a sink net set in New River Inlet (1993). In addition to these, in September 1995, two dead leatherbacks were removed from a 11-inch (28.2 cm) monofilament shark gillnet set in the nearshore waters off of Cape Hatteras, North Carolina (STSSN unpublished data reported in NMFS SEFSC 2001).

Fishing gear interactions and poaching are problems for leatherbacks throughout their range. Entanglements are common in Canadian waters where Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland/Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line and crab pot line. Leatherbacks are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo *et al.* 1994; Graff 1995). Gillnets are one of the suspected causes for the decline in the leatherback sea turtle population in French Guiana (Chevalier *et al.* 1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch leatherback 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 2000). An estimated 1,000 mature female leatherback sea turtles are caught annually in fishing nets off of Trinidad and Tobago with mortality estimated to be between 50-95% (Eckert and Lien 1999). However, many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS SEFSC 2001).

Leatherback sea turtles may be more susceptible to marine debris ingestion than other species due to the tendency of floating debris to concentrate in convergence zones that adults and juveniles use for feeding areas (Shoop and Kenney 1992; Lutcavage *et al.* 1997). Investigations of the stomach contents of leatherback sea turtles revealed that a substantial percentage (44% of the 16 cases examined) contained plastic (Mrosovsky 1981). 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 and plastic debris (Mrosovsky 1981). Balazs (1985) speculated that the object may resemble a food item by its shape, color, size or even movement as it drifts about, and induce a feeding response in leatherbacks.

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

Nest counts in many areas of the Atlantic show increasing trends, including for beaches in Suriname and French Guiana which support the majority of leatherback nesting (NMFS and USFWS 2007b). The species as a whole continues to face numerous threats at nesting and marine habitats. The long term recovery potential of this species may be further threatened by observed low genetic diversity, even in the largest nesting groups like French Guiana and Suriname (NMFS and USFWS 2007b).

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

3.1.3 Kemp's ridley sea turtle

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 northern half of the Atlantic Ocean (USFWS and NMFS 1992).

The majority of Kemp's ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; USFWS and NMFS 1992; NMFS and USFWS 2007c). There is a limited amount of scattered nesting to the north and south of the primary nesting beach (NMFS and USFWS 2007c). The number of nesting adult females reached an estimated low of 300 in 1985 (USFWS and NMFS 1992; TEWG 2000; NMFS and USFWS 2007c). 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). From 1985 to 1999, the number of nests observed at Rancho Nuevo, and nearby beaches increased at a mean rate of 11.3% (95% C.I. slope = 0.096-0.130) per year (TEWG 2000). An estimated 5,500 females nested in Tamaulipas over a 3-day period in May 2007 (NMFS and USFWS 2007c).

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 (USFWS and NMFS 1992). Once they leave the beach, neonates presumably enter the Gulf of Mexico where they feed on available sargassum and associated infauna or other epipelagic species (USFWS and NMFS 1992). The presence of juvenile turtles along both the Atlantic and Gulf of Mexico coasts of the U.S., 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 in many areas along the U.S. coast and that these areas may change given resource quality and quantity (TEWG 2000). Foraging areas documented along the Atlantic coast include Pamlico Sound (NC), Chesapeake Bay, Long Island Sound, Charleston Harbor (SC) and Delaware Bay. Developmental habitats are defined by several characteristics, including coastal areas sheltered from high winds and waves such as embayments and estuaries, and nearshore temperate waters shallower than 50m (NMFS and USFWS 2007c). The suitability of these habitats depends on resource availability, with optimal environments providing rich sources of crabs and other invertebrates. A wide variety of substrates have been documented to provide good foraging habitats, including seagrass beds, oyster reefs, sandy and mud bottoms and rock outcroppings (NMFS and USFWS 2007c). Adults are primarily found in near-shore waters of 37m or less that are rich in crabs and have a sandy or muddy bottom (NMFS and USFWS 2007c).

Next to loggerheads, Kemp's ridleys are the second most abundant sea turtle in Virginia and Maryland state waters, arriving in these areas during May and June (Keinath *et al.* 1987; Musick and Limpus 1997). In the Chesapeake Bay, where the seasonal juvenile population of Kemp's ridley sea turtles is estimated to be 211 to 1,083 turtles (Musick and Limpus 1997), ridleys frequently forage in submerged aquatic grass beds for crabs (Musick and Limpus 1997). Kemp's ridleys consume a variety of crab species, including *Callinectes* sp., *Ovalipes* sp., *Libinia* sp., and *Cancer* sp. Mollusks, shrimp, and fish are consumed less frequently (Bjorndal 1997). Upon leaving Chesapeake Bay in autumn, juvenile ridleys migrate down the coast, passing Cape Hatteras in December and January (Musick and Limpus 1997). These larger juveniles are joined there by juveniles of the same size from North Carolina sounds and smaller juveniles from New York and New England to form one of the densest concentrations of Kemp's ridleys outside of the Gulf of Mexico (Epperly *et al.* 1995a; Epperly *et al.* 1995b; Musick and Limpus 1997).

Kemp's ridleys face many of the same natural threats as loggerheads, including destruction of nesting habitat from storm events, natural predators at sea, and oceanic events such as cold-stunning. Although cold-stunning can occur throughout the range of the species, it may be a greater risk for sea turtles that utilize the more northern habitats of Cape Cod Bay and Long Island Sound. For example, as reported in the national STSSN database, in the winter of 1999/2000, there was a major cold-stunning event where 218 Kemp's ridleys, 54 loggerheads, and 5 green turtles were found on Cape Cod beaches. Annual cold stun events do not always occur at this magnitude; the extent of episodic major cold stun events may be associated with numbers of turtles utilizing Northeast waters in a given year, oceanographic conditions and the occurrence of storm events in the late fall. Although many cold-stun turtles can survive if found early enough, cold-stunning events can represent a significant cause of natural mortality.

Like other turtle species, the severe decline in the Kemp's ridley population appears to have been heavily influenced by a combination of exploitation of eggs and impacts from fishery interactions. From the 1940s through the early 1960s, nests from Ranch Nuevo were heavily exploited (USFWS and NMFS 1992), but beach protection in 1966 helped to curtail this activity (USFWS and NMFS 1992). Following World War II, there was a substantial increase in the number of trawl vessels, particularly shrimp trawlers, in the Gulf of Mexico where the adult Kemp's ridley turtles occur. Information from fishers 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 turtle takes in shrimp trawls and other trawl fisheries, including the development and use of TEDs. As described in Section 3.1.1 above, there is lengthy regulatory history with regard to the use of TEDs in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries (Epperly and Teas 2002; NMFS 2002; Lewison *et al.* 2003). The Biological Opinion completed in 2002 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 2002).

Although changes in the use of shrimp trawls and other trawl gear has helped to reduce mortality of Kemp's ridleys, this species is also affected by other sources of anthropogenic impacts similar to those discussed above. For example, in the spring of 2000, a total of five Kemp's ridley carcasses were recovered from the same North Carolina beaches where 275 loggerhead carcasses were found. Cause of death for most of the turtles recovered was unknown, but the mass mortality event was suspected to have been from a large-mesh gillnet fishery operating offshore in the preceding weeks. The five 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.

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; USFWS and NMFS 1992; NMFS and USFWS 2007c). 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 250 nesting females in the entire 1985 nesting season (USFWS and NMFS 1992; TEWG 2000). However, the total annual number of nests at Rancho Nuevo gradually began to increase in the 1990's (NMFS and USFWS 2007c). Based on the number of nests laid in 2006 and the remigration interval for Kemp's ridley sea turtles, there were an estimated 7,000-8,000 adult female Kemps 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 ridleys suggest that the population is female biased (NMFS and USFWS 2007c). Based on its 5-year status review of the species, NMFS and the USFWS (2007c) determined that Kemp's ridley sea turtles should not be reclassified as threatened under the ESA.

3.1.4 Green sea turtle

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

Pacific Ocean. Green turtles occur in the eastern, central, and western Pacific. Foraging areas are also found throughout the Pacific and along the southwestern U.S. coast (NMFS and USFWS 1998b). Nesting is known to occur in the Hawaiian archipelago, American Samoa, Guam, and various other sites in the Pacific but none of these are considered large breeding sites (with 2,000 or more nesting females per year)(NMFS and USFWS 1998b). The main nesting sites for the green sea turtle in the eastern Pacific are located in Michoacan, Mexico, and in the Galapagos Islands, Ecuador (NMFS and USFWS 2007d). The number of nesting females per year exceed 1,000 females at each site (NMFS and USFWS 2007d). However, historically, greater than 20,000 females per year are believed to have nested in Michoacan, alone (Cliffton *et al.* 1982; NMFS and USFWS 2007d). Thus the current number of nesting females is still far below what has historically occurred.

Historically, green turtles were used in many areas of the Pacific for food. They were also commercially exploited and this, coupled with habitat degradation led to their decline in the Pacific (NMFS and USFWS 1998b). Green turtles in the Pacific continue to be affected by poaching, habitat loss or degradation, fishing gear interactions, and fibropappiloma (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 turtle nesting were evident for many of the Indian Ocean Index Sites. While several of these had not demonstrated further declines in the more recent past, only the Comoros Island Index Site in the Western Indian Ocean showed evidence of increased nesting (Seminoff 2004).

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

In the western Atlantic, green sea turtles range from Massachusetts to Argentina, including the Gulf of Mexico and Caribbean (Wynne and Schwartz 1999). Green turtles occur seasonally in Mid-Atlantic and Northeast waters such as Long Island Sound (Musick and Limpus 1997; Morreale and Standora 1998; Morreale *et al.* 2004), presumably for foraging.

Some of the principal feeding pastures in the western Atlantic Ocean include the upper west coast of Florida and the northwestern coast of the Yucatan 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).

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

As is also the case for the other sea turtle species described above, nest count information for green sea turtles provides information on the relative abundance of nesting, and the contribution of each nesting group to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually. The 5-year status review for the species identified eight geographic areas considered to be primary sites for green sea turtle nesting in the Atlantic/Caribbean, and reviewed the trend in nest count data for each (NMFS and USFWS 2007d). These include: (1)Yucatán Peninsula, Mexico, (2) Tortuguero, Costa Rica, (3) Aves Island, Venezuela, (4) Galibi Reserve, Suriname, (5) Isla Trindade, Brazil, (6) Ascension Island, United Kingdom, (7) Bioko Island, Equatorial Guinea, and (8) Bijagos Achipelago (Guinea-Bissau) (NMFS and USFWS 2007d). Nesting at all of these sites was considered to be stable or increasing with the exception of Bioko Island and the Bijagos Archipelago where the lack of sufficient data precluded a meaningful trend assessment for either site (NMFS and USFWS 2007d). Seminoff (2004) likewise reviewed green sea turtle nesting data for eight sites in the western, eastern, and central Atlantic, including all of the above with the exception that nesting in Florida was reviewed in place of Isla Trindade, Brazil. Seminoff (2004) concluded that all sites in the central and western Atlantic showed increased nesting with the exception of nesting at Aves Island, Venezuela, while both sites in the eastern Atlantic demonstrated decreased nesting. These sites are not inclusive of all green sea turtle nesting in the Atlantic. However, other sites are not believed to support nesting levels high enough that would change the overall status of the species in the Atlantic (NMFS and USFWS 2007d).

By far, the most important nesting concentration for green turtles in the western Atlantic is in Tortuguero, Costa Rica (NMFS and USFWS 2007d). Nesting in the area has increased considerably since the 1970's 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). In the U.S., certain Florida nesting beaches have been designated index beaches. Index beaches were established to standardize data collection methods and effort on key nesting beaches. The pattern of green turtle nesting shows biennial peaks in abundance, with a generally positive trend during the ten years of regular monitoring since establishment of the index beaches in 1989, perhaps due to increased protective legislation throughout the Caribbean (Meylan *et al.* 1995).

An average of 5,039 green 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). Occasional nesting has been documented along the Gulf coast of Florida, at southwest Florida beaches, as well as the beaches on the Florida Panhandle (Meylan *et al.* 1995). More recently, green turtle nesting occurred on Bald Head Island, North Carolina just east of the mouth of the Cape Fear River, on Onslow Island, and on Cape Hatteras National Seashore. Increased nesting has also been observed along the Atlantic Coast of Florida, on beaches where only loggerhead nesting was observed in the past (Pritchard 1997).

Green turtles face many of the same natural threats as loggerhead and Kemp's ridley sea turtles. In addition, green turtles appear to be susceptible to fibropapillomatosis, an epizootic disease producing lobe-shaped tumors on the soft portion of a turtle's body. Juveniles are most commonly affected. The occurrence of fibropapilloma tumors may result in impaired foraging, breathing, or swimming ability, leading potentially to death.

As with the other sea turtle species, incidental fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like dredging, pollution, and habitat destruction account for an unknown level of other mortality. Stranding reports indicate that between 200-400 green turtles strand annually along the Eastern U.S. coast from a variety of causes most of which are unknown (STSSN database). Sea sampling coverage in the pelagic driftnet, pelagic longline, southeast shrimp trawl, and summer flounder bottom trawl fisheries has recorded takes of green turtles.

Summary of Status of Green Sea Turtles
A review of 32 Index Sites[5] distributed globally revealed a 48% to 67% decline in the number of mature females nesting annually over the last 3-generations[6] (Seminoff 2004). An evaluation of green sea turtle nesting sites was also conducted as part of the 5-year status review of the species (NMFS and USFWS 2007d). Of the 23 nesting groups assessed in that report, 10 were considered to be increasing, 9 were considered stable, and 4 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). The report also estimates that 108,761 to 150,521 females nest each year among the 46 sites (NMFS and USFWS 2007d). However, given the late age to maturity for green sea turtles, caution is urged regarding the status for any of the nesting groups since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d).

There is cautious optimism that the green sea turtle abundance is increasing in the Atlantic. Seminoff (2004) and NMFS and USFWS (2007d) made comparable conclusions with regard to nesting for four nesting sites in the western Atlantic. Each also concluded that nesting at Tortuguero, Costa Rica represented the most important nesting area for green sea turtles in the

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

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

western Atlantic and that nesting had increased markedly since the 1970's (Seminoff 2004; NMFS and USFWS 2007d). However, the 5-year review also noted that the Tortuguero nesting stock continued to be affected by ongoing directed take at their primary foraging area in Nicaragua (NMFS and USFWS 2007d). As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like dredging, pollution, and habitat destruction account for an unknown level of other mortality.

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 loggerhead, leatherback, Kemp's ridley, and green sea turtles in the action area. The activities generally fall into one of the following three categories: (1) fisheries, (2) other activities that cause death or otherwise impair a turtles ability to function, and (3) recovery activities associated with reducing impacts to ESA-listed sea turtles.

Many of the fisheries and other activities causing death or injury to sea turtles that are identified in this section have occurred for years, even decades. Similarly, while some recovery activities have been in place for years (*e.g.*, nesting beach protection in portions of each species nesting habitat), others have been undertaken more recently following new information on the impact of certain activities on one or more of the ESA-listed sea turtle species considered in this Opinion.

The past impacts of each state, Federal, and private action or other human activity in the action area cannot be particularized in their entirety. However, to the extent they have manifested themselves at the population level, such past impacts are subsumed in the information presented on the status and trends of the species in Sections 3.0 and 6.0, recognizing that the benefits to sea turtles as a result of recovery activities already implemented may not be evident in the status and trend of the population for years given the relatively late age to maturity for sea turtles, and depending on the age class(es) affected.

4.1 Fishery Operations

4.1.1 Federal fisheries

Commercial fisheries in the action area employ gear that has been known to capture, injure, and kill sea turtles. Several federally regulated fisheries that use gillnet, longline, trawl, and trap gear have been documented as unintentionally capturing or entangling sea turtles. In some cases, the entangled turtles are harmed, injured, or killed as a result of the interaction. Available information suggests that sea turtles can be captured in any of these gear types when the operation of the gear overlaps with the distribution of sea turtles.

Formal ESA section 7 consultation has been conducted on the American lobster, Atlantic bluefish, Atlantic herring, Atlantic mackerel/squid/Atlantic butterfish, highly migratory species, monkfish, northeast multispecies, red crab, skate, spiny dogfish, summer flounder/scup/black sea bass, and tilefish fisheries. An Incidental Take Statement (ITS) has been issued for the take of sea turtles in each of the fisheries (Appendix 2). The ITS reflects the impact of the activity on sea turtles and other listed species anticipated from the date of the ITS and forward in time. A summary of each fishery in the action area that has been subject to section 7 consultation is provided below. The information describes times and areas where the fishery presently operates in order to qualitatively assess the likelihood of overlap between operation of the fishery and distribution of sea turtles. Information is also provided, where available, on changes in fishing effort to qualitatively assess whether the likelihood of sea turtle impacts will change in the near future.

As described in Sections 1.0 and 2.1, consultation has also been previously conducted on the Atlantic sea scallop fishery - a fishery with a lengthy fishing history in Mid-Atlantic and Northeast waters. Therefore, the environmental baseline for this action also includes the effects of the past operation of the Atlantic sea scallop fishery.

The *American lobster trap fishery* has been identified as a source of gear causing injuries and mortality of loggerhead and leatherback sea turtles. Previous Opinions for this fishery have concluded that operation of the Federally regulated protion of the lobster trap fishery is likely to jeopardize the continued existence of right whales and may adversely affect loggerhead and leatherback sea turtles. A Reasonable and Prudent Alternative (RPA) to avoid the likelihood that the lobster fishery would jeopardize the continued existence of right whales was implemented. However, these measures were not expected to reduce the number or severity of loggerhead and leatherback sea turtle interactions with the fishery. Consultation on the lobster fishery has been reinitiated following new information on the effectiveness of the RPA for removing the likelihood of jeopardy for right whales.

American lobster occur within U.S. waters from Maine to Virginia. They are most abundant from Maine to New Jersey with abundance declining from north to south (ASMFC 1999). Most lobster trap effort occurs in the Gulf of Maine. Maine and Massachusetts produced 90% of the 2006 total U.S. landings of American lobster, with Maine accounting for 79% of these landings (NMFS 2007a). Lobster landings in the other New England states as well as New York and New Jersey account for most of the remainder of U.S. American lobster landings. However, declines in lobster abundance and landings have occurred from Rhode Island through New Jersey in recent years. The Mid-Atlantic states from Delaware through North Carolina have been granted *de minimus* status under the Lobster ISFMP. Given the distribution of lobster fishing effort, leatherback sea turtles are the most likely sea turtle to be affected since this species occurs regularly in Gulf of Maine waters.

An Interstate Fishery Management Plan (ISFMP) developed through the ASMFC provides management measures for the fishery that are implemented by the states. NMFS has issued regulations for the Federal waters portion of the fishery based on recommendations from the ASMFC. The Federal waters portion of the fishery includes the Federal waters of Lobster Management Areas (LMA) 1, 2, 4, 5 and the Outer Cape as well as all of LMA 3 which occurs entirely within Federal waters (NMFS 1999; 2002). Management measures include a limited access permit system, gear restrictions, and other prohibitions on possession (*e.g.*, of berried or scrubbed lobsters), landing limits for lobsters caught by non-trap gear, a trap tag requirement, and trap limits. These measures include reduction of effort and capping of effort. Management measures have been implemented in most areas in recent years based on "historical participation" to further constrain fishing effort in the lobster fishery. LMA 1, an area of overlap between sea turtles and lobster trap gear, does not currently have a historical participation program.

The *Atlantic bluefish fishery* is known to interact with sea turtles, given the time and locations where the fishery occurs. Two takes of leatherback sea turtles and a take of an unidentified sea turtle have been reported in net gear used in the bluefish fishery. No takes of ESA-listed loggerhead, leatherback, Kemp's ridley or green sea turtles have been reported in bottom otter trawl gear for trips that were targeting bluefish (where > 50% of the catch was bluefish) (NMFS 1999). However, loggerhead and Kemp's ridley sea turtle takes have been observed in bottom otter trawl gear where bluefish was caught but constituted less than 50% of the catch (NMFS 1999). In August 2007, NMFS received an estimate of loggerhead sea turtle takes in bottom otter trawl gear used in the bluefish fishery (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). Using VTR data from 2000-2004 and the average annual bycatch of turtles as described in Murray 2006, the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the bluefish fishery was estimated to be 3 loggerhead sea turtles a year (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). The July 2, 1999, Opinion on the authorization of Amendment 1 to the Bluefish FMP anticipated the annual take of 6 loggerheads annually. Therefore, the information presented by Murray did not represent new information on the effects of the bluefish fishery on loggerhead sea turtles. However, NMFS has received new information on the effects of the fishery on leatherback sea turtles. Therefore, consultation on the continued authorization of the bluefish fishery under the Bluefish FMP was reinitiated on December 18, 2007.

Gillnet and bottom otter trawl gear are the predominant gear types in the commercial bluefish fishery (MAFMC 2007). In 2006, gillnet gear accounted for 32.4% of the total commercial trips targeting bluefish, and landed 72% of the commercial catch for that year (MAFMC 2007). Bottom otter trawls accounted for 44% of the total commercial trips targeting bluefish and landed 20.4% of the catch (MAFMC 2007).

The majority of commercial fishing activity in the North and Mid-Atlantic occurs in the late spring to early fall, when bluefish are most abundant in these areas (NEFSC 2005a). Bluefish migrate south as water temperatures decrease in late fall and winter (NEFSC 2005a). Overall, the majority of bluefish commercial landings are taken in the Mid-Atlantic, with North Carolina reporting the highest landings, followed by New York and New Jersey (NEFSC 2005a).

The ASMFC and the MAFMC jointly manage bluefish under Amendment 5 to the Bluefish FMP (NEFSC 2005a). The bluefish fishery is not a limited access fishery. Bluefish landings are controlled through a coastwide quota, with 83% of the quota allocated to the recreational sector and 17% to the commercial sector (NEFSC 2005a). The commercial bluefish fishery landed 7.1 million pounds in 2006 (NMFS 2007a), down from a peak of 16.1 million pounds in 1981.

Sea turtle interactions with gear used in the *Atlantic herring fishery* have not been reported or observed by NMFS observers. However, NMFS has concluded that sea turtle takes in fishing gear used in the fishery are reasonably likely to occur. An ITS was provided based on the observed capture of sea turtles in other fisheries that use comparable gear. Purse seines, midwater trawls (single), and pair trawls are the three primary gears involved in the Atlantic herring fishery (NEFMC 2006b). However, the gear type accounting for the majority of herring landings changed over the ten-year period from 1995-2005 (NEFMC 2006b). During the 1990's, purse seine and mid-water trawl gear accounted for the majority of annual herring landings. Since 2000, pair trawl gear has accounted for the majority of herring landed each year (NEFMC 2006b).

A FMP for the Atlantic herring fishery was implemented on December 11, 2000. Three management areas, which may have different management measures, were established under the Herring FMP. Management Area 1 includes Gulf of Maine waters and is subdivided into inshore (Area 1A) and offshore (Area 1B) sub-areas. Management Area 2 is referred to as the South Coastal Area and includes state and Federal waters adjacent to the States of Massachusetts, Rhode Island, Connecticut, New York, New Jersey, Delaware, Maryland, Virginia, and North Carolina. Management Area 3 includes waters over Georges Bank (NEFMC 1999). The ASMFC's Atlantic Herring ISFMP provides measures for the management of the herring fishery in state waters that are complementary to the Federal FMP.

Changes to the management of the herring fishery were made in 2007 with the implementation of Amendment 1 to the Herring FMP (72 FR 11252, March 12, 2007). These included making the herring fishery a limited access fishery and measures to issue open access permits to vessels that do not qualify for a limited access permit. Changes were also made to the management area boundaries, as well as other administrative measures (NEFMC 2006b).

The *Atlantic mackerel/squid/butterfish fisheries* are managed under a single FMP. The FMP covers management of four species, given that both short-finned squid (*Illex illecebrosus*) and long-finned squid (*Loligo pealei*) are managed under the FMP. Information for the fisheries was summarized in the Environmental Assessment for the 2008 Atlantic Mackerel, Squid, and Butterfish Specifications (MAFMC 2007). A brief review of each fishery is provided below based on the document prepared by the MAFMC.

Based on NMFS dealer reports, a total of 278 vessels landed 56,641 mt of Atlantic mackerel in 2,424 trips in 2006. Statistical areas 615, 613, 612, and 616 (Mid-Atlantic Bight) accounted for the majority (81%) of commercial Atlantic mackerel landings in 2006 (MAFMC 2007). Statistical areas 622, 537, and 539 collectively accounted for another 18% of landings while all other areas each accounted for less than 1% of the 2006 Atlantic mackerel landings (MAFMC 2007). Although mackerel landings occurred year-round, the primary mackerel fishing season extends from January through April when greater than 95% of the annual landings are taken. The principal gears used to land mackerel in 2006 were mid-water trawls (77%) and bottom otter trawls (19%) (MAFMC 2007).

Based on NMFS dealer reports, a total of 358 vessels landed 15,880 mt of *Loligo* squid in 2006.

Statistical areas 616, 622, 537, and 613 (Mid-Atlantic Bight and southern New England) accounted for the majority (69.5%) of commercial *Loligo* squid landings in 2006 (MAFMC 2007). Although *Loligo* squid landings occurred year-round, the majority of the *Loligo* landings occurred in the fall through winter months (MAFMC 2007). The principal gear used to land *Loligo* squid in 2006 was bottom otter trawls (79.06%) (MAFMC 2007).

Based on NMFS dealer reports, a total of 33 vessels landed 13,837 mt of *Illex* squid on 221 trips in 2006. Statistical areas 622 and 626 (Mid-Atlantic Bight) accounted for the majority (80.54%) of commercial *Illex* squid landings in 2006 (MAFMC 2007). Statisical areas 632, 526, 635, and 537 collectively accounted for another 16.72% of *Illex* landings (MAFMC 2007). All other statistical areas each accounted for less than 1% of the 2006 *Illex* squid landings (MAFMC 2007). The majority (97.17%) of the *Illex* landings occurred from June through October (MAFMC 2007). Most (>79%) of *Illex* landings were taken by otter trawls (MAFMC 2007).

According to NMFS weighout data, a total of 261 vessels landed 554 mt of butterfish in 2006. Statistical areas 611, 537, and 616 accounted for the majority (60.47%) of butterfish landings in 2006 (MAFMC 2007). Butterfish landings occurred throughout the year (MAFMC 2007). Most (>74.68%) were taken by otter trawls (MAFMC 2007).

Takes of sea turtles have been observed in the *Loligo* and *Illex* squid fisheries. In August 2007, NMFS received an estimate of loggerhead sea turtle takes in bottom otter trawl gear used in the mackerel, squid, butterfish fisheries (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). Using VTR data from 2000-2004 and the average annual bycatch of turtles as described in Murray 2006, the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the mackerel, squid, and butterfish fisheries was estimated to be 62 loggerhead sea turtles a year (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). This information represents new information on the capture of loggerhead sea turtles in the mackerel, squid, butterfish fisheries. NMFS has determined that this new information triggers the need to reinitiate section 7 consultation on the Mackerel, Squid, Butterfish FMP.

The *Atlantic sea scallop* fishery has a long history of operation in Mid-Atlantic, as well as New England waters (NEFMC 1982; 2003). The fishery operates in areas and at times that it has traditionally operated and uses traditionally fished gear (NEFMC 1982; 2003). Effort (in terms of days fished) in the Mid-Atlantic is about half of what it was prior to implementation of the Scallop FMP in the 1990's (NEFSC 2007). Additional information on management of the fishery is provided in Section 2.1.

Loggerhead, Kemp's ridley, and green sea turtles have been reported by NMFS-trained observers as being captured in scallop dredge and or trawl gear. The first reported capture of a sea turtle in the scallop fishery occurred in 1996 during an observed trip of a scallop dredge vessel. A single capture in scallop dredge gear was reported for each of 1997 and 1999, as well. In 2001, thirteen sea turtle captures in scallop dredge gear were observed and/or reported by NMFS trained observers. All of these occurred in the re-opened Hudson Canyon and Virginia Beach Access Areas where observer coverage of the scallop fishery was higher in comparison to outside of the Access Areas. Although NMFS was not aware until 2001 that sea turtle interactions with scallop fishing gear were likely to occur, there is no information to suggest that turtle interactions with scallop fishing gear are a new event or are occurring at a greater rate than what has likely occurred in the past. To the contrary, the methods used to detect any sea turtle interactions with scallop fishing gear (dredge or trawl gear) were insufficient prior to increased observer coverage in 2001. In addition, there have been no known changes to the seasonal distribution of loggerhead sea turtles in the Mid-Atlantic north of Cape Hatteras (CeTAP 1982; Lutcavage and Musick 1985; Keinath *et al.* 1987; Shoop and Kenney 1992; Burke *et al.* 1993; Burke *et al.* 1994) with the exception of recent studies (Morreale *et al.* 2004; Mansfield 2006) which suggest a decrease rather than an increase in the use of some Mid-Atlantic loggerhead foraging areas for unknown reasons. Therefore, it is likely that the effect of the scallop fishery on sea turtles, while only quantified and recognized within the last 4 years, has been present for decades. Additional information on the observed capture of sea turtles in the fishery is provided in Section 5.4.4.

Components of the *highly migratory species (HMS)* Atlantic pelagic fishery for swordfish/tuna/shark in the EEZ occur within the action area for this consultation. Use of pelagic longline, pelagic driftnet, bottom longline, hand line (including bait nets), and/or purse seine gear in this fishery has resulted in the take of sea turtles and whales. The Northeast swordfish driftnet portion of the fishery was prohibited during an emergency closure that began in December 1996, and was subsequently extended. A permanent prohibition on the use of driftnet gear in the swordfish fishery was published in 1999. In June 2001, NMFS completed consultation on the HMS pelagic longline fishery and concluded that the Atlantic HMS fisheries, particularly the pelagic longline fisheries that occur outside of the action areas for this Opinion, were likely to jeopardize the continued existence of loggerhead and leatherback sea turtles. An RPA was provided to avoid jeopardy to leatherback and loggerhead sea turtles as a result of operation of the HMS fisheries. Consultation was subsequently reinitiated and a new RPA was developed and implemented following NMFS completion of the Opinion on June 1, 2004. In 2006, the Atlantic HMS pelagic longline fisheries had an estimated 415 interactions with leatherback sea turtles and 561 interactions with loggerhead sea turtles (Fairfield-Walsh and Garrison 2007).

The Federal *monkfish fishery* occurs in all waters under Federal jurisdiction from Maine to the North Carolina/South Carolina border. The current commercial fishery operates primarily in the deeper waters of the Gulf of Maine, Georges Bank, and southern New England, and in the Mid-Atlantic. Monkfish have been found in depths ranging from the tide line to 840 meters, with concentrations between 70 and 100 meters and at 190 meters. The monkfish fishery uses several gear types that may capture ESA-listed species, including gillnet and trawl gear. A consultation conducted on the continued operation of the fishery concluded in 2001 that the fishery may adversely affect sea turtles as a result of entanglement in gear used in the fishery. Although the estimated capture of sea turtles in monkfish gillnet gear is relatively low, there is concern that much higher levels of interaction could occur. Following an event in which over 200 sea turtle carcasses washed ashore in an area where large-mesh gillnetting had been occurring, NMFS published new restrictions for the use of gillnets with larger than 8-inch (20.3 cm) stretched mesh, in the EEZ) off of North Carolina and Virginia (67 FR 71895, December 3, 2002). The rule was subsequently modified on April 26, 2006, by modifying the restrictions to the use of gillnets with ≥ 7-inch (17.9 cm) stretched mesh when fished in Federal waters from the North Carolina/South Carolina border to Chincoteague, VA.

The monkfish fishery is managed in the EEZ through a joint NEFMC and MAFMC Monkfish FMP (NEFSC 2005b). The FMP defines two management areas for monkfish (northern and southern), divided roughly by a line bisecting Georges Bank (NEFSC 2005b). Effort in the fishery is limited through a limited access permit program, as well as DAS and trip allocations that were implemented as initial management measures of the FMP in 1999. Trip allocations differ between the two management areas.

Trawl, scallop dredge, and gillnet gear are the primary gear types that capture monkfish (NEFSC 2005b). The percentage of monkfish landed by gear type has changed somewhat over time. During 1998-2000, trawls accounted for 54% of the total landings, scallop dredges about 17%, and gillnets 29% (NEFSC 2005b). For the period from 2001-2003, trawl, gillnet, and scallop dredge gear accounted for 55%, 36%, and 8% of landings, respectively (NEFSC 2005b). The change in the composition of landings by gear type is likely the result of management measures on the use of these gear types in the fishery. For the 2006 monkfish fishing year (May 1, 2006 – April 30, 2007), trawl gear accounted for 52% of monkfish landings while gillnet gear accounted for 41%, and other gear types (not specified other than not being hook gear) accounted for the remaining 7% of monkfish landings for the timeframe (NMFS FSO 2007).

The *northeast multispecies fishery* operates throughout the year, with peaks in spring and from October through February. Multiple gear types are used in the fishery. Data indicate that sink gillnet gear has seriously injured or killed right whales, humpback whales, fin whales, and loggerhead and leatherback sea turtles. The Northeast multispecies sink gillnet fishery has historically occurred from the periphery of the Gulf of Maine to Rhode Island in waters to 60 fathoms. Participation in this fishery has declined since extensive groundfish conservation measures have been implemented; particularly since implementation of Amendment 13 to the Multispecies FMP. Additional management measures (*i.e.* Framework Adjustment 42) are expected to have further reduced effort in the multispecies fishery.

In August 2007, NMFS received an estimate of loggerhead sea turtle takes in bottom otter trawl gear used in the northeast multispecies fishery (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). Using VTR data from 2000-2004 and the average annual bycatch of turtles as described in Murray 2006, the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the northeast multispecies fishery was estimated to be 43 loggerhead sea turtles a year (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). This information represents new information on the capture of loggerhead sea turtles in the northeast multispecies fishery.

The *red crab fishery* is a pot/trap fishery that occurs in deep waters along the continental slope. The primary fishing zone for red crab, as reported by the fishing industry, is at a depth of 400-800 meters along the continental shelf in the Northeast region, and is limited to waters north of 35° 15.3' N (Cape Hatteras, NC) and south of the Hague Line.

There has been a small, directed fishery for red crab off the coast of New England and the Mid-Atlantic since the 1970s. The fishery was fairly consistent through the 1980's, but landings steadily increased from the mid-1990s (NEFMC 2002). Following concerns that red crab could be overfished, an FMP was developed and became effective on October 21, 2002. The FMP includes management measures to control effort in the fishery (*e.g.*, a limited access permit program, trap limits, a fleet DAS allocation) (NEFMC 2005b). Five vessels are permitted in the limited access red crab fishery; however, one vessel has opted out of the fishery since 2004. In accordance with the Red Crab FMP, the DAS associated with that vessel are reallocated to the fleet. The average catch of red crab by the commercial fleet for the last three fishing years has been approximately 41% of the annual 5.928 million lb quota.

Section 7 consultation was completed on the proposed implementation of the Red Crab FMP, and concluded that the action was not likely to result in jeopardy to any ESA-listed species under NMFS jurisdiction. An ITS was provided that addresses takes of loggerhead and leatherback sea turtles.

The *skate fishery* has typically been composed of both a directed fishery and an indirect fishery. The bait fishery is more historical and is a more directed skate fishery than the wing fishery. Vessels that participate in the bait fishery are primarily from Southern New England and direct primarily on little (90%) and winter skate (10%). The wing fishery is primarily an incidental fishery that takes place throughout the region. For section 7 purposes, NMFS considers the effects to ESA-listed species of the directed skate fishery. Fishing effort that contributes to landings of skate for the indirect fishery is considered during section 7 consultation on the directed fishery in which skate bycatch occurs.

Bottom trawl gear accounted for 94.5% of directed skate landings. Gillnet gear is the next most common gear type, accounting for 3.5% of skate landings. The Northeast skate complex is comprised of seven related skate species. There have been no recorded takes of ESA-listed species in the skate fishery. However, given that sea turtle interactions with trawl and gillnet gear have been observed in other fisheries, sea turtle takes in gear used in the skate fishery may be possible where the gear and sea turtle distributions overlap. Section 7 consultation on the Skate FMP was completed July 24, 2003, and concluded that authorization of the skate fishery may adversely affect ESA-listed sea turtles as a result of interactions with (capture in) gillnet and trawl gear. In August 2007, NMFS received an estimate of loggerhead sea turtle takes in bottom otter trawl gear used in the skate fishery (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). Using VTR data from 2000-2004 and the average annual bycatch of turtles as described in Murray 2006, the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the skate fishery was estimated to be 24 loggerhead sea turtles a year (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). This information represents new information on the capture of loggerhead sea turtles in skate fishery.

The *spiny dogfish fishery* in the EEZ is managed under the Spiny Dogfish FMP for fishing in waters of the EEZ. The NEFMC and MAFMC jointly develop management measures for the fishery that occurs in Federal waters. The ASMFC has also implemented an ISFMP for spiny dogfish in order to coordinate coastwide quotas, and to help enforce state and Federal regulations for the spiny dogfish fishery. In the Northwest Atlantic, spiny dogfish range from Florida to Labrador, but are most abundant from Nova Scotia to Cape Hatteras (NEFSC 2003). They make seasonal inshore-offshore and coastal migrations related to their preferred temperature range (7°-13°C) (NEFSC 2003).

U.S. landings of spiny dogfish increased in the 1990's, reaching 28,000 mt in 1996 before declining to approximately 20,000 mt in both 1997 and 1998, and then to 14,860 mt in 1999 (NEFSC 2003). U.S. landings dropped to about 2,200 mt in 2001 and 2002, and then dropped further to around 1,000 mt in response to the quota restrictions imposed by the Spiny Dogfish FMP and the ASMFC ISFMP (NEFSC 2003; 2006a). Dogfish landings have been reported in all months of the year, but most occur from June through September (NEFSC 2003; 2006a). Massachusetts has been the primary state for landings of spiny dogfish since 1979 (NEFSC 2006a). From 1992 through 1996, North Carolina had the second highest level of landings, accounting for 18.8% - 23.8% of the annual landings in those years (NEFSC 2006a). However, in recent years, these have declined and with the exception of 2004, North Carolina landings in 2001-2007 were negligible (NEFSC 2006a).

The primary gear types for the spiny dogfish fishery have historically been sink gillnets, otter trawls, bottom longline, and driftnet gear (NEFSC 2003). The predominance of any one gear type has varied over time (NEFSC 2003). Landings of spiny dogfish in the 1990's were attributed primarily to sink gillnet gear, followed by otter trawl, longline, and drift gillnet gear (NEFSC 2003). In 2001 and 2002, following implementation of the Spiny Dogfish FMP, longline gear accounted for the majority of landings, followed by sink gillnet and otter trawl gear (NEFSC 2003). Landings for drift gillnet gear were reduced to near zero (NEFSC 2003). In calendar year 2005, 62.1% of landings were taken by sink gillnet gear, followed by 18.4% in otter trawl gear, 2.3% in line gear, and 17.1% in gear defined as Other (excludes drift gillnet gear)(NEFSC 2006b). Sea turtles can be incidentally captured in all gear sectors of the spiny dogfish fishery.

The *summer flounder, scup and black sea bass fisheries* are managed under one FMP. NMFS approved the first FMP for management of the summer flounder fishery in Federal waters in 1988. Management measures for scup and black sea bass were subsequently added to the Federal FMP under Amendments 8 and 9, respectively.

All three species are present in offshore waters throughout the winter and migrate and occupy inshore waters throughout the summer. Otter trawl gear is used in the commercial fisheries for all three species (MAFMC 2007). In addition, floating traps and pots/traps are used in the scup and black sea bass fisheries, respectively (MAFMC 2007).

Commercial landings of summer flounder peaked in 1984 at 37.77 million lbs and then declined to a low of 9.26 million lbs in 1990 (MAFMC 2007). In 2006, the commercial sector of the summer flounder fishery caught 13.97 million lbs of summer flounder of which 12.90 million lbs were landed (MAFMC 2007). The majority of the trips and catch were made by bottom otter and beam trawls (69.8% of trips, 97.0% of catch), followed by gillnets (7.4% of trips, 0.6% of catch), handline "other" (8.8% of trips, 1.0% of catch), scallop dredges (8.9% of trips, 0.8% of catch), and pots and traps (5.1% of trips, 0.6% of catch) (MAFMC 2007).

In general, over 80% of the summer flounder landed in NMFS' Northeast Region from the commercial sector of the fishery have come from statistical areas 537-539 (Southern New England), areas 611-616 (New York Bight), areas 621, 622, 625 and 626 (Delmarva region), and areas 631-632 (Norfolk Canyon area) (NEFSC 2002). In 2006, there were eight statistical areas, which collectively accounted for 80 percent of the summer flounder catch in 2006 (MAFMC 2007). These were statistical areas 616, 537, 622, 612, 626, 613, 621, 611, 539, and 538 (in order of highest to lowest percent contribution to the total catch) (MAFMC 2007).

Commercial landings of scup peaked in 1981 at 21.27 million lbs and then declined to a low of 2.66 million lb in 2000. In 2006, the commercial sector of the scup fishery caught 7.35 million lbs of scup of which 7.23 million lbs were landed (MAFMC 2007). The majority of the trips and catch were made by bottom otter and beam trawls (56.9% of trips and 91.7% of catch) followed by pots and traps (20.5% of trips, 4.8% of catch), hand line "other" (18.0% of trips, 2.8% of catch), gillnets (4.3% of trips, 0.4% of catch), and dredges (0.1% of trips, 0.2% of catch). Five statistical areas collectively accounted for 94 percent of the scup catch (MAFMC 2007). These were statistical areas 616, 613, 539, 611, and 537 (in order of highest to lowest percent contribution to the total catch) (MAFMC 2007).

Commercial landings of black sea bass peaked in 1983 at 4.33 million lbs and then declined to a low of 2.04 million lbs in 1994. In 2006, the commercial sector of the black sea bass fishery caught 2.40 million lbs of black sea bass of which 2.32 million lbs were landed (MAFMC 2007). The majority of the trips and catch were made by bottom otter and beam trawls (48.8% of trips, 38.5 % of catch), followed by pots and traps (33.5% of trips, 56.6% of catch), handline "other" (14.1% of trips, 4.6% of catch), and gillnets (3.4% of trips, 0.3% of catch). Six statistical areas, collectively accounted for greater than 5 percent of the black sea bass catch in 2006 (MAFMC 2007). These were statistical areas 621, 622, 616, 538, 613, and 626 (in order of highest to lowest percent contribution to the total catch) (MAFMC 2007).

An ITS has been provided for the anticipated capture of sea turtles in gear used in the summer flounder, scup, and black sea bass fisheries. In 2006 the NEFSC released an estimate of loggerhead sea turtle takes in bottom otter trawl gear fished in Mid-Atlantic waters during the period 1996-2004 (Murray 2006). Fifty-percent of the observed 66 takes occurred on vessels targeting summer flounder. However, it should also be noted that some of the observed interactions occurred on vessels fishing with TEDs using an allowed (at that time) TED extension with a minimum 5.5" mesh (Murray 2006). Numerous problems were noted by observers with respect to the mesh used in the TED extension including entanglement of sea turtles in the mesh and blocking of the TED by debris (Murray 2006). NMFS addressed these problems in 1999 by requiring that webbing in the TED extension be no more than 3.5" stretched mesh (Murray 2006). Given these changes, the bycatch rates used for the estimate may be higher than current conditions.

In August 2007, NMFS received an estimate of loggerhead sea turtle takes in bottom otter trawl gear used in the summer flounder, scup, black sea bass fisheries (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). Using VTR data from 2000-2004 and the average annual bycatch of turtles as described in Murray 2006, the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the summer flounder, scup, black sea bass fisheries was estimated to be 200 loggerhead sea turtles a year (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). This information represents new information on the capture of loggerhead sea turtles in the summer flounder, scup, black sea bass fisheries

A summary of the current *tilefish fishery* was provided in the 41st Northeast Regional Stock Assessment Report (NEFSC 2005a). The management unit for the *Tilefish* FMP is all golden tilefish under U.S. jurisdiction in the Atlantic Ocean north of the Virginia/North Carolina border (MAFMC 2000). Tilefish have some unique habitat characteristics and are found in a warm water band (9-14° C) along the upper slope of the continental shelf in the southern New England and Mid-Atlantic areas at depths of 80 to 440m (NEFSC 2005a). Because of the restricted habitat and low biomass of tilefish, the tilefish fishery in recent years has occurred in a relatively small area in the Mid-Atlantic Bight, south of New England and east of New Jersey. Over 75% of tilefish landings have come from this area (statistical areas 537 and 616; Appendix 1) since 1991 (NEFSC 2005a).

The directed tilefish fishery is a relatively small fishery in terms of the number of participants. Five vessels accounted for more than 49-93% of the landings during the period 1995-2004 (NEFSC 2005a). Longline gear is the primary gear type used in the tilefish fishery. Since the 1980's, over 85% of the commercial landings of tilefish in the Mid-Atlantic/southern New England region have been taken by longline gear (NEFSC 2005a). The fishery changed from using "J" hooks to circle hooks after 1979 (NEFSC 2005a).

The effects of the tilefish fishery on ESA-listed species were considered during formal consultation on the implementation of the Tilefish FMP, completed in March 2001. Anecdotal information available at that time suggested that loggerhead and leatherback sea turtles have been taken by hook gear in the tilefish bottom longline fishery (MAFMC 2000). Consultation was concluded on March 13, 2001, with the issuance of a biological opinion that includes an ITS for loggerhead and leatherback sea turtles.

4.1.2 Non-federally regulated fisheries

Nearshore and inshore gillnet fisheries occur throughout the Mid-Atlantic in state waters from Connecticut through North Carolina; areas where sea turtles also occur. Captures of sea turtles in these fisheries have been reported (NMFS SEFSC 2001). Two, 10-14 inch (25.6 – 35.9 cm) mesh gillnet fisheries, the black drum and sandbar shark gillnet fisheries, occur in Virginia state waters along the tip of the eastern shore. These fisheries may take sea turtles given the gear type, but no interactions have been observed. Similarly, small mesh gillnet fisheries occurring in Virginia state waters are suspected of taking sea turtles but no interactions have been observed. 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), and no turtle takes were observed. In North Carolina, a large-mesh gillnet fishery for summer flounder in the southern portion of Pamlico Sound was found to contribute to takes of sea turtles in gillnet gear. In 2000, an Incidental Take Permit was issued to the North Carolina Department of Marine Fisheries for the take of sea turtles in the Pamlico Sound large-mesh gillnet fishery. The fishery was closed when the incidental take level for green sea turtles was met (NMFS SEFSC 2001). Long haul seines and channel nets are also known to incidentally capture sea turtles in North Carolina sounds and inshore waters. As described in section 4.4.1 below, NMFS has taken regulatory action to address the potential for sea turtle interactions with gillnet gear with ≥7 inch (17.9 cm) stretched mesh fished in federal waters off of North Carolina and Virginia.

An *Atlantic croaker fishery* using trawl gear also occurs within the action area. Turtle takes have been observed in Atlantic croaker trawl gear (Murray 2006). Between 1994 and 2004, observers documented the capture of 18 loggerhead sea turtles in trawl gear targeting croaker in waters from 41° 30'N/66°W to 35°N/75° 30'W (Murray 2006). Additional observed takes have occurred with 5 takes of loggerhead sea turtles observed in 2006 and 17 takes of loggerhead sea turtles observed in 2007 (NEFSC Fisheries Sampling Branch Website). NMFS is investigating the use of a turtle excluder device for trawl gear used in the Atlantic croaker fishery (72 FR 7382).

The *weakfish fishery* occurs in both state and federal waters but the majority of commercially and recreationally caught weakfish are caught in state waters (ASMFC 2002). The dominant commercial gears include gill nets, pound nets, haul seines and trawls, with the majority of landings occurring in the fall and winter months (ASMFC 2002). Weakfish landings were dominated by the trawl fishery through the mid-1980s after which gill net landings began to account for most weakfish landed (ASMFC 2002). North Carolina has accounted for the majority of the annual landings since 1972 while Virginia ranks second, followed by New Jersey (ASMFC 2002). As described in section 3.1.1, turtle takes in the weakfish fishery have occurred (Murray 2006). Seven of the sixty-six observed loggerhead sea turtle interactions in bottom otter trawl gear fished in Mid-Atlantic waters during the period 1994-2004, were on vessels targeting weakfish. Since observer coverage was low and the fishery uses other gear types known to take turtles, the take of sea turtles in the fishery is likely to have been higher than that which was observed for just the trawl sector.

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

Various *crab fisheries*, such as horseshoe crab and blue crab, also occur in federal and state waters. The crab fisheries may have detrimental impacts on sea turtles beyond entanglement in the fishing gear itself. Loggerheads are known to prey on crab species, including horseshoe and blue crabs. In a study of the diet of loggerhead sea turtles in Virginia waters from 1983-2002, Seney and Musick (2007) found a shift in the diet of loggerheads in the area from horseshoe and blue crabs to fish, particularly menhaden and Atlantic croaker. The authors suggested that a decline in the crab species have resulted in the shift and loggerheads are likely foraging on fish captured in fishing nets or on discarded fishery bycatch (Seney and Musick 2007). The physiological impacts of this shift are uncertain although it was suggested as a possible explanation for the declines in loggerhead abundance noted by Mansfield (2006). Other studies have detected seasonal declines in loggerhead abundance coincident with seasonal declines of horseshoe and blue crabs in the 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.* 2004), commensurate with noted declines in the abundance of horseshoe crab and other crab species raises concerns that crab fisheries may be significantly impacting the forage base for loggerheads in some areas of their range.

The *Virginia pound net fishery* has also been documented as a source of turtle takes. Pound nets with large-mesh leaders set in the Chesapeake Bay have been observed to lethally take turtles as a result of entanglement in the pound net leader. As described in section 4.4.4 below, NMFS has taken regulatory action to address turtle takes in the Virginia pound net fishery.

4.2 Vessel Activity

Past and present adverse effects from federal vessel operations in the action area of this consultation include operations of the U.S. Navy (USN) and the U.S. Coast Guard (USCG), which maintain the largest federal vessel fleets, the Environmental Protection Agency (EPA), the Army Corps of Engineers (ACOE), and NOAA. NMFS has conducted formal consultations with the USCG, the USN and is currently in early phases of consultation with other federal agencies on their vessel operations (*e.g.*, NOAA research vessels). Through the Section 7 process, where applicable, NMFS has and will continue to establish conservation measures for all these agency vessel operations to avoid adverse effects to listed species. At the present time, however, there is the potential for some level of interaction but the level and severity of interactions is not quantified.

4.3 Other Activities

4.3.1 Hopper Dredging

The Sandbridge Shoal is an approved Minerals Management Service borrow site located approximately 3 miles off Virginia Beach. This site has been used in the past for both the Navy's Dam Neck Annex beach renourishment project and the Sandbridge Beach Erosion and Hurricane Protection Project, and is likely to be used in additional beach nourishment projects in the future. The Sandbridge Beach Erosion and Hurricane Protection Project involved hopper dredging of approximately 972,000 cubic yards (cy) of sand during the first year of the project and an anticipated 500,000 cy every two years thereafter. NMFS completed section 7 consultation on this project in April 1993, and anticipated the take of eight loggerhead turtles or one Kemp's ridley or green turtle. Actual dredging did not begin until May 1998, and no sea turtle takes were observed during the 1998 dredge cycle. In June 2001, the ACOE indicated that the next dredge cycle, which was scheduled to begin in the summer of 2002, would require 1.5 million cy of sand initially, with an anticipated 1.1 million cy every two years thereafter. Although the volume of sand had increased from the previous cycle, NMFS reduced the ITS to five loggerheads and one

Kemp's ridley or green turtle due to the lack of observed takes in the previous cycle, along with the levels of anticipated and observed take in hopper dredging projects in nearby locations.

NMFS completed section 7 consultation on the Navy's Dam Neck Annex beach nourishment project in January 1996, which involved the removal of 635,000 cy of material beginning in 1996 and continuing on a 12-year cycle thereafter. NMFS anticipated the take of ten loggerheads and one Kemp's ridley or green sea turtle during each dredge cycle. However, no takes were observed during the 1996 cycle. The Navy reinitiated consultation on June 27, 2003, based on an accelerated dredge cycle (from 12 years to 8 years), an increase in the volume of sand required, and new information on the status of loggerhead sea turtles since the original Opinion was issued in 1996. The consultation was concluded on December 12, 2003, and anticipated the take of four loggerheads and one Kemp's ridley or green sea turtle during each dredge cycle. NMFS concluded that this level of take was not likely to jeopardize the continued existence of any of these species.

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

NMFS has completed section 7 consultation for the issuance of permits to allow for the construction and operation of two Liquid Natural Gas (LNG) terminals within the action area of this consultation. NMFS has concluded that the construction and operation of these facilities will not adversely affect ESA-listed loggerhead, leatherback, Kemp's ridley or green sea turtles.

4.3.3 Pollution

Sources of pollutants in coastal regions of 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 and sewage treatment effluent, and oil spills. Nutrient loading from land-based sources such as coastal community discharges is known to stimulate plankton blooms in closed or semi-closed estuarine systems. The effect to larger embayments is unknown. Contaminants could indirectly degrade habitat if pollution and other factors reduce the food available to marine animals.

4.3.4 Global climate change

There is a large and growing body of literature on past, present, and future impacts of global climate change induced by human activities - frequently referred to in layman's terms as "global warming". Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The Environmental Protection Agency's climate change webpage provides basic background information on these and other measured or anticipated effects (see www. epa.gov/climatechange/index.html). Activities in the action area that may have contributed to global warming include the combustion of fossil fuels by vessels.

The effects of global climate change on sea turtles is typically viewed as being detrimental to the species (NMFS and USFWS 2007a; 2007b; 2007c; 2007d). Changes in water temperature would be expected to affect prey distribution and/or abundance, salinity, and water circulation patterns perhaps even to the extent that the Gulf Stream is disrupted (Gagosian 2003; NMFS and USFWS 2007a; 2007b; 2007c; 2007d). The effects of these on sea turtles cannot, for the most part, be accurately predicted at this time. Several studies have, however, investigated the effects of changes in sea surface temperature and air temperatures on turtle reproductive behavior. 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 (2002).

Air temperatures also play a role in sea turtle reproduction. In marine turtles, sex is determined by temperature in the middle third of incubation with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25-35° C (Ackerman 1997). Based on modeling, a 2° C increase in air temperature is expected to result in a sex ratio of over 80% female offspring for loggerhead nesting beaches in the vicinity of Southport, NC. Farther to the south at Cape Canaveral, Florida, a 2°C increase in air temperature would likely result in production of 100% females while a 3°C increase in air temperature would likely exceed the thermal threshold of turtle clutches resulting in death (Hawkes *et al.* 2007). Thus changes in air temperature as a result of global climate change may alter sex ratios and may reduce hatchling production in the most southern nesting areas of the U.S. Given that the south Florida nesting group is the largest loggerhead nesting group in the Atlantic (in terms of nests laid), a decline in the success of nesting as a result of global climate change could have profound effects on the abundance and distribution of the loggerhead species in the Atlantic.

For green sea turtles, incubation temperatures also appeared to affect hatchling size with smaller turtles produced at higher incubation temperatures (Glen *et al.* 2003). It is unknown whether this effect is species specific and what impact it has on the survival of the offspring.

While the type and extent of effects to sea turtles as a result of global climate change are still speculative, a disruption of the Gulf Stream such as might occur as a result of global climate change (Gagosian 2003), would be expected to have profound effects on every aspect of sea turtle life history from hatching success, oceanic migrations at all life stages, foraging, and nesting.

4.4 Reducing Threats to ESA-listed Sea Turtles

4.4.1 Final Rules for Large-Mesh Gillnets

In March 2002, NMFS published new restrictions for the use of gillnets with larger than 8-inch (20.3 cm) stretched mesh, in Federal waters (3-200 nautical miles) off of North Carolina and Virginia. These restrictions were published in an interim final rule under the authority of the ESA (67 FR 13098) and were implemented to reduce the impact of the monkfish and other large-mesh gillnet fisheries on ESA-listed sea turtles in areas where sea turtles are known to concentrate. Following review of public comments submitted on the interim final rule, NMFS published a final rule on December 3, 2002, that established the restrictions on an annual basis. As a result, gillnets with larger than 8-inch (20.3 cm) stretched mesh were not allowed in Federal waters (3-200 nautical miles) in the areas described as follows: (1) North of the North Carolina/South Carolina border at the coast to Oregon Inlet at all times; (2) north of Oregon Inlet to Currituck Beach Light, NC from March 16 through January 14; (3) north of Currituck Beach Light, NC, to Wachapreague Inlet, VA, from April 1 through January 14; and (4) north of Wachapreague Inlet, VA, to Chincoteague, VA, from April 16 through January 14. On April 26, 2006, NMFS published a final rule (71 FR 24776) that included modifications to the large-mesh gillnet restrictions. The new final rule revised the gillnet restrictions to apply to stretched mesh that is ≥7 inches (17.9 cm). Federal waters north of Chincoteague, VA, 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-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 a rule addressing capture of sea turtles in gillnet gear fished in the southern flounder fishery in Pamlico Sound. NMFS issued a final rule (67 FR 56931), effective September 3, 2002, that closed the waters of Pamlico Sound, NC, to fishing with gillnets with larger than 4 ¼-inch (10.8 cm) stretched mesh from September 1 through December 15 each year to protect migrating sea turtles. The closed area includes all inshore waters of Pamlico Sound south of 35° 46.3' N. lat., north of 35°00' N. lat., and east of 76° 30' W. long.

4.4.2 Revised use of TEDs for the U.S. south Atlantic and Gulf of Mexico shrimp fisheries

On February 21, 2003, NMFS issued a final rule (68 FR 8456) to amend regulations for reducing sea turtle mortality resulting from shrimp trawling in the Atlantic and Gulf Areas of the southeastern United States. TEDs have proven to be effective at excluding sea turtles from shrimp trawls. However, NMFS determined that modifications to the design of TEDs needed to be made to exclude leatherbacks, as well as large, benthic, immature and sexually mature loggerhead and green turtles. In addition, several previously approved TED designs did not function properly under normal fishing conditions. Therefore, NMFS disallowed these TEDs (*e.g.*, weedless TEDs, Jones TEDs, hooped hard TED, and the use of accelerator funnels) as described in the final rule. Finally, the rule also required modifications to the trynet and bait shrimp exemptions to the TED requirements to decrease mortality of sea turtles.

4.4.3 TED requirements for the summer flounder fishery

As mentioned in Section 4.1.1, significant measures have been developed to reduce the take of sea turtles in summer flounder trawls and trawls that meet the definition of a summer flounder trawl (which would include fisheries for other species like scup and black sea bass) by requiring TEDs in trawl nets fished in trawls used in the area of greatest turtle bycatch off the North Carolina and part of the Virginia coast from North Carolina/South Carolina border to Cape Charles, VA. The TED requirements for the summer flounder trawl fishery do not, however, require the use of larger TEDs that are required to be used in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries.

4.4.4 Modification of Gear for Virginia Pound Nets

Existing information indicates that pound nets with traditional large mesh and stringer leaders as used in the Chesapeake Bay, incidentally take sea turtles. NMFS published a temporary rule in June 2001 (66 FR 33489) that prohibited fishing with pound net leaders with a mesh size measuring 8-inches (20.3 cm) or greater, and pound net leaders with stringers in mainstream waters of the Chesapeake Bay and its tributaries for a 30-day period beginning June 19, 2001. NMFS subsequently published an interim final rule in 2002 (67 FR 41196, June 17, 2002) that further addressed the take of sea turtles in large-mesh pound net leaders and stringer leaders used in the Chesapeake Bay and its tributaries. Following new observations of sea turtle entanglements in pound net leaders in the spring of 2003, NMFS issued a temporary final rule (68 FR 41942, July 16, 2003) that restricted all pound net leaders throughout Virginia's waters of the Chesapeake Bay and a portion of its tributaries from July 16 - July 30, 2003. A new final rule was published May 5, 2004 (69 FR 24997) to address sea turtle entanglements with pound net gear that might occur in the Chesapeake Bay during the period May 6 - July 15 each year. That rule prohibited the use of all pound net leaders, set with the inland end of the leader greater than 10 horizontal feet (3 meters) from the mean low water line, from May 6 - July 15 each year in the Virginia waters of the mainstream Chesapeake Bay, south of 37° 19' N and west of 76° 13' W, and all waters south of 37° 13' N to the Chesapeake Bay Bridge Tunnel at the mouth of the Chesapeake Bay, and the James and York Rivers downstream of the first bridge in each tributary. Outside of this area, the prohibition of leaders with greater than or equal to 12 inches (30.5 cm) stretched mesh and leaders with stringers, as established by the June 17, 2002, interim final rule, applied from May 6 - July 15 each year. In response to new information acquired through gear research, on April 17, 2006, NMFS published a proposed rule in the *Federal Register* that would allow the use of offshore pound net leaders meeting the definition of a *modified pound net leader* in a portion of the Chesapeake Bay during the period May 6 to July 15 each year. Modifications to the pound net leader address: (1) The maximum allowed mesh size; (2) placement of the leader in relation to the sea floor; (3) the height of the mesh from the sea floor in relation to the depth at mean lower low water; and (4) the use of vertical lines to hold the mesh in place. Following review of public comments received on the proposed rule, NMFS published a final rule implementing the action on June 23, 2006 (71 FR 36024).

4.4.5 HMS Sea Turtle Protection Measures

As described in *Section 4.1.1* above, NMFS completed the most recent biological opinion on the FMP for the Atlantic HMS fisheries for swordfish, tuna, 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. The RPA is also expected to benefit loggerhead sea turtles by reducing the likelihood of mortality resulting from interactions with the gear. Regulatory components of the RPA have been implemented through rulemaking.

4.4.6 Sea Turtle Handling and Resuscitation Techniques

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

4.4.7 Sea Turtle Entanglements and Rehabilitation

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

4.4.8 Education and Outreach Activities

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

4.4.9 Sea Turtle Stranding and Salvage Network (STSSN)

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

5.0 CUMULATIVE EFFECTS

Cumulative effects include the effects of future State, tribal, local or private actions that are reasonably certain to occur in 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.

Sources of human-induced mortality, injury, and/or harassment of turtles in the action area that are reasonably certain to occur in the future include incidental takes in state-regulated fishing activities, vessel collisions, ingestion of plastic debris, and pollution. While the combination of these activities may affect populations of endangered and threatened sea turtles, preventing or slowing a species' recovery, the magnitude of these effects is currently unknown.

State Water Fisheries - Fishing activities are considered one of the most significant causes of death and serious injury for sea turtles. A 1990 National Research Council report estimated that 550 to 5,500 sea turtles (juvenile and adult loggerheads and Kemp's ridleys) die each year from all other fishing activities besides shrimp fishing. Fishing gear in state waters, including bottom trawls, gillnets, trap/pot gear, and pound nets, take sea turtles each year. NMFS is working with state agencies to address the take of sea turtles in state-water fisheries within the action area of this consultation where information exists to show that these fisheries take sea turtles. Action has been taken by some states to reduce or remove the likelihood of sea turtle takes in one or more gear types. However, given that state managed commercial and recreational fisheries along the Atlantic coast are reasonably certain to occur within the action area in the foreseeable future, additional takes of sea turtles in these fisheries are anticipated. There is insufficient information by which to quantify the number of sea turtle takes presently occurring as a result of state water fisheries as well as the number of sea turtles injured or killed as a result of such takes. While actions have been taken to reduce sea turtle takes in some state water fisheries, the overall effect of these actions on reducing the take of sea turtles in state water fisheries is unknown, and the future effects of state water fisheries on sea turtles cannot be quantified. Further information on past effects of state water fisheries on sea turtles is available in Section 4.1.2.

Vessel Interactions – NMFS' STSSN data indicate that vessel interactions are responsible for a large number of sea turtles strandings within the action area each year. Such collisions are reasonably certain to continue into the future. Collisions with boats can stun or easily kill sea turtles, and many stranded turtles have obvious propeller or collision marks (Dwyer *et al.* 2003). However, it is not always clear whether the collision occurred pre- or post-mortem. As a result an estimate of the number of sea turtles that will likely be killed by vessels is not possible.

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. However, the level of impacts cannot be projected. Marine debris (*e.g.*, discarded fishing line or lines from boats) can entangle turtles in the water and drown them. Turtles commonly ingest plastic or mistake debris for food. Chemical contaminants may also have an effect on sea turtle reproduction and survival. Excessive turbidity due to coastal development and/or construction sites could influence sea turtle foraging ability. As mentioned previously, turtles are not very easily affected by changes in water quality or increased suspended sediments, but if these alterations make habitat less suitable for turtles and hinder their capability to forage, eventually they would tend to leave or avoid these less desirable areas (Ruben and Morreale 1999). Noise pollution has been raised, primarily, as a concern for marine mammals but may be a concern for other marine organisms, including sea turtles. As described above, global warming is likely to negatively affect sea turtles – affecting when females lay their eggs, the survival of the eggs, sex ratios of offspring, and the stability of the Gulf Stream. To the extent that air pollution, for example from the combustion of fossil fuels by vessels, contributes to global warming, then it is also expected to negatively affect sea turtles.

5.1 Summary and synthesis of the Status of Species, Environmental Baseline, and Cumulative Effects sections

The Status of the Species, Environmental Baseline, and Cumulative Effects Sections, taken together, establish a "baseline" against which the effects of the continued authorization of the Atlantic sea scallop fishery are analyzed to determine whether the action—the continued authorization of the Atlantic sea scallop fishery-- is likely to jeopardize the continued existence of the species. Past effects of the scallop fishery are included in this "baseline." To the extent available information allows, this "baseline" (which does not include the future effects of the scallop fishery) would be compared to the backdrop plus the effects of the continued authorization of the fishery from now into the future. The difference in the two trajectories would be reviewed to determine whether the continued authorization of the fishery is likely to jeopardize the continued existence of the species. This section synthesizes the Status of the Species, the Environmental Baseline, and Cumulative Effects sections as best as possible given that some information on sea turtles is quantified, yet much remains qualitative or unknown. Leatherback and Kemp's ridley sea turtles are endangered species, meaning that they are in danger of extinction throughout all or a significant portion of their ranges. The loggerhead sea turtle is a threatened species, meaning that it is likely to become an endangered species in the foreseeable future throughout all or a significant portion of its range. Green sea turtles in U.S. waters are listed as threatened except for the Florida breeding population which is listed as endangered. For purposes of this Opinion, NMFS considers the trend of the sea turtle species considered in this Opinion to be declining for loggerhead, leatherback, and green sea turtles, and stable for Kemp's ridley sea turtles. These trends are the result of past, present, and likely future human activities and natural events, some effects of which are positive, some negative, and some unknown, as discussed previously in the Status of the Species, Environmental Baseline, and Cumulative Effects Sections taken together. Additional information is provided below.

Loggerhead Sea Turtles. Loggerhead sea turtles are listed as a single species classified as "threatened" under the ESA. Loggerhead nesting occurs on beaches of the Pacific, Indian, and Atlantic oceans, and Mediterranean Sea. Genetic analyses of maternally inherited mitochondrial DNA demonstrate the existence of separate, genetically distinct nesting groups between as well as within the ocean basins (TEWG 2000; Bowen and Karl 2007). In response to each of two petitions received in July and November 2007, NMFS has published a 90-day finding in the *Federal Register* (72 FR 64585, November 16, 2007, and 73 FR 11849, March 5, 2008, respectively), and concluded that the petitioners presented substantial scientific and commercial information indicating that a reclassification of the loggerhead in the North Pacific and North Atlantic Oceans as DPSs and listing of each of those DPSs with endangered status may be warranted. As further stated in each finding, NMFS will initiate a review of the status of the species to determine whether the petitioned action is warranted, and to determine whether any additional changes to the current listing of the loggerhead sea turtle is warranted.

It takes decades for loggerhead sea turtles to reach maturity. Once they have reached maturity, females typically lay multiple clutches of eggs within a season, but do not typically lay eggs every season (NMFS and USFWS 1991a). There are many natural and anthropogenic factors affecting survival of turtles prior to their reaching maturity as well as for those adults who have reached maturity. As described in sections 3.1 and 4.0, negative impacts causing death of various age classes occur both on land and in the water. In addition, given the distances traveled by loggerheads in the course of their development, actions to address the negative impacts require the work of multiple countries at both the national and international level (NMFS and USFWS 2007a). Many actions have been taken to address known negative impacts to loggerhead sea turtles. However, many remain unaddressed, have not been sufficiently addressed, or have been addressed in some manner but whose success cannot be quantified.

There are no population estimates for loggerhead sea turtles. Sea turtle nesting data, in terms of the number of nests laid each year, is collected for loggerhead sea turtles for at least some nesting beaches within each of the ocean basins and the Mediterranean Sea. From this, the number of reproductively mature females utilizing those nesting beaches can be estimated based on the presumed remigration interval and the average number of nests laid by a female loggerhead sea turtle per season. These estimates provide a minimum count of the number of loggerhead sea turtles in any particular nesting group. The estimates do not account for adult females who nest on beaches with no or little survey coverage, and do not account for adult males or juveniles of either sex. The proportion of adult males to females from each nesting group, and the age structure of each loggerhead nesting group is currently unknown. For these reasons, nest counts cannot be used to estimate the total population size of a nesting group and, similarly, trends in the number of nests laid cannot be used as an indicator of the population trend (whether decreasing, increasing or stable) (Meylan 1982; Ross 1996; Zurita *et al.* 2003; Hawkes *et al.* 2005; Loggerhead TEWG 2007).

Nevertheless, nest count data are a valuable source of information for each loggerhead nesting group and for loggerheads as a species since the number of nests laid reflect the reproductive output of the nesting group each year, and also provide insight on the contribution of each nesting group to the species. Based on a comparison of the available nesting data, the world's largest known loggerhead nesting group (in terms of estimated number of nesting females) occurs in Oman in the northern Indian Ocean where an estimated 20,000-40,000 females nest each year (Baldwin *et al.* 2003). The world's second largest known loggerhead nesting group occurs along the east coast of the United States where approximately 15,966 females nest per year on south Florida beaches (based on a mean of 65,460 nests laid per year from 1989-2006; NMFS and USFWS 2007a). The world's third largest loggerhead nesting group also occurs in the United States, from approximately northern Florida through North Carolina. However, the mean nest count for this nesting group, the third largest loggerhead nesting group in the world, is 5,151 nests laid per year (NMFS and USFWS 2007a) – less than 1/10th the mean number of nests laid by the south Florida nesting group. Thus, while loggerhead nesting occurs at multiple sites within multiple ocean basins and the Mediterranean Sea, the extent of nesting is disproportionate amongst the various sites and only two geographic areas, Oman and south Florida, U.S., account for the majority of nesting for the species, worldwide.

Declines in loggerhead nesting have been noted at nesting beaches throughout the range of the species. These include nesting for the south Florida nesting group – the second largest loggerhead nesting group in the world and the largest of all of the loggerhead nesting groups in the Atlantic (Dodd 2003; Meylan *et al.* 2006; Letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006; Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission web posting November 2007; NMFS and USFWS 2007a).

In light of the above, for purposes of this Opinion, NMFS considers the trend for loggerheads as a species to be declining. NMFS recognizes that the available nest count data only provides information on the number of females currently nesting, and is not necessarily a reflection of the number of mature females available to nest or the number of immature females that will reach maturity and nest in the future. Also, the trend in the number of nests laid is not a reflection of the overall trend in any nesting group given that the proportion of adult males to females, and the age structure of each loggerhead nesting group is currently unknown. This determination that the trend for loggerheads as a species is declining provides benefit of the doubt to the species given its threatened classification under the ESA, the many on-going negative impacts to the species across all areas of its range and to all age classes, and information to suggest that fewer nests are being laid (potentially reducing the number of offspring that will mature and contribute to the species' continued existence).

Leatherback turtles. Leatherback sea turtles are listed as a single species classified as "endangered" under the ESA. Leatherbacks are widely distributed throughout the oceans of the world, and are found in waters of the Atlantic, Pacific, and Indian Oceans, the Caribbean Sea, Mediterranean Sea, and the Gulf of Mexico (Ernst and Barbour 1972). Leatherback nesting occurs on beaches of the Atlantic, Pacific and Indian Oceans as well as in the Caribbean (NMFS and USFWS 2007b).

Like loggerheads, sexually mature female leatherbacks typically nest in non-successive years and lay multiple clutches in each of the years that nesting occurs. Leatherbacks face a multitude of threats that can cause death prior to and after reaching maturity. Some activities resulting in leatherback mortality have been addressed. However, many others remain to be addressed. Given their range and distribution, international efforts are needed to address all known threats to leatherback sea turtle survival (NMFS and USFWS 2007b).

There are some population estimates for leatherback sea turtles although there appears to be considerable uncertainty in the numbers. In 1980, the global population of adult leatherback females was estimated to be approximately 115,000 (Pritchard 1982). By 1995, this global population of adult females was estimated to be 34,500 (Spotila *et al.* 1996). However, the most recent population size estimate for the North Atlantic alone is 34,000-94,000 adult leatherbacks (Leatherback TEWG 2007; NMFS and USFWS 2007b).

Leatherback nesting in the eastern Atlantic (*i.e.*, off Africa) and in the Caribbean appears to be stable, but there is conflicting information for some sites and it is certain that some nesting groups (*e.g.*, St. John and St. Thomas, U.S. Virgin Islands) have been extirpated (NMFS and USFWS 1995). Data collected for some nesting beaches in the western Atlantic, including leatherback nesting beaches in the U.S. clearly indicate increasing numbers of nests (NMFS SEFSC 2001; NMFS and USFWS 2007b). However, declines in nesting have been noted for beaches in the western Caribbean (NMFS and USFWS 2007b). The largest leatherback rookery in the western Atlantic remains along the northern coast of South America in French Guiana and Suriname. More than half the present world leatherback population is estimated to be nesting on the beaches in and close to the Marowijne River Estuary in Suriname and French Guiana (Hilterman and Goverse 2004). The long-term trend for the Suriname and French Guiana nesting group seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests for Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). Studies by Girondot *et al.* (2007) also suggest that the trend for the Suriname - French Guiana nesting population over the last 36 years is stable or slightly increasing.

Increased nesting by leatherbacks in the Atlantic is not expected to affect leatherback abundance in the Pacific where the abundance of leatherback turtles on nesting beaches has declined dramatically over the past 10 to 20 years (NMFS and USFWS 2007b). Although genetic analyses suggest little difference between Atlantic and Pacific leatherbacks (Bowen and Karl 2007), it is generally recognized that there is little to no genetic exchange between these turtles. In addition, Atlantic and Pacific leatherbacks are impacted by different activities (NMFS and USFWS 1992; 1998a). However, the ESA-listing of leatherbacks as a species means that the effects of a proposed action must, ultimately, be considered at the species level for section 7 consultations. In light of the above, for purposes of this Opinion, NMFS considers the trend for leatherbacks, as a species, to be declining. NMFS recognizes that the nest count data available for leatherbacks in the Atlantic clearly indicates increased nesting at many sites, and that the activities affecting declines in nesting by leatherbacks in the Pacific are not the same as those activities affecting leatherbacks in the Atlantic. However, NMFS also recognizes that the nest count data, including data for leatherbacks in the Atlantic, only provides information on the number of females currently nesting, and is not necessarily a reflection of the number of mature females in the Atlantic that are available to nest or the number of immature females that will reach maturity and nest in the future. Also, the trend in the number of nests laid is not a reflection of the overall trend in any leatherback population given that the proportion of adult males to females, and the age structure of the population(s) is unknown. This determination that the trend for leatherbacks as a species is declining provides benefit of the doubt to the species given its endangered classification under the ESA, the many on-going negative impacts to the species across all areas of its range and to all age classes, the uncertainty in the population estimates, the dramatic decline in leatherback nesting in the Pacific, and the disproportionate nesting of leatherbacks with more than half of the species nesting occurring in one area of the world (thus negative impacts to this area could have very large impacts on reproductive success of the species).

Kemp's Ridley Sea Turtles. Kemp's ridley sea turtles are listed as a single species classified as "endangered" under the ESA. Kemp's ridleys occur in the Atlantic Ocean and Gulf of Mexico. The only major nesting site for ridleys is a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; USFWS and NMFS 1992; NMFS and USFWS 2007c). Approximately 60% of its nesting occurs here with a limited amount of scattered nesting to the north and south of the primary nesting beach (NMFS and USFWS 2007c).

Age to maturity for Kemp's ridley sea turtles occurs earlier than for either loggerhead or leatherback sea turtles. However, maturation may still take 10-17 years (NMFS and USFWS 2007c). As is the case with the other turtle species, adult, female Kemp's ridleys typically lay multiple nests in a nesting season but do not typically nest every nesting season (TEWG 2000; NMFS and USFWS 2007c). Although actions have been taken to protect the nesting beach habitat, and to address activities known to be negatively impacting Kemp's ridley sea turtles, Kemp's ridleys continue to be impacted by anthropogenic activities (see sections 3.1.3 and 4.1).

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

The most recent review of the Kemp's ridley as a species suggests that it is in the early stages of recovery (NMFS and USFWS 2007c). The nest count data indicates increased nesting and an increased number of nesting females in the population. In light of this information, for purposes of this Opinion, NMFS considers the trend for Kemp's ridley sea turtles to be stable. This determination that the trend for Kemp's ridleys as a species is stable provides benefit of the doubt to the species given the species classification of "endangered" under the ESA, the caveats associated with using nesting data as indicators of population size and population trends, that the estimated number of nesting females in the current population is still far below historical numbers (Stephens and Alvarado-Bremer 2003; NMFS and USFWS 2007c), the many on-going negative impacts to the species, and given that the majority of nesting for the species occurs in one area.

Green Sea Turtles. Green sea turtles are listed as both threatened and endangered under the ESA. Breeding colony populations in Florida and on the Pacific cost of Mexico are considered endangered while all others are considered threatened. Due to the inability to distinguish between these populations away from the nesting beach, for this Opinion, green turtles are considered endangered wherever they occur in U.S. waters. Green turtles are distributed circumglobally, and can be found in the Pacific, Indian and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 1991; Seminoff 2004; NMFS and USFWS 2007d).

Green sea turtles appear to have the latest age to maturity of all of the sea turtles with age at maturity occurring after 2-5 decades (NMFS and USFWS 2007d). As is the case with all of the other turtle species mentioned here, mature green sea turtles typically nest more than once in a nesting season but do not nest every nesting season. As is also the case with the other turtle species, green sea turtles face numerous threats on land and in the water that affect the survival of all age classes.

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

The results of genetic analyses show that green sea turtles in the Atlantic do not contribute to green sea turtle nesting elsewhere in the species range (Bowen and Karl 2007). Therefore, increased nesting by green sea turtles in the Atlantic is not expected to affect green sea turtle abundance in other ocean basins in which the species occurs. However, the ESA-listing of green sea turtles as a species across ocean basins means that the effects of a proposed action must, ultimately, be considered at the species level for section 7 consultations. In light of the above, for purposes of this Opinion, NMFS considers the trend for green sea turtles, as a species, to be declining. NMFS recognizes that the nest count data available for green sea turtles in the Atlantic clearly indicates increased nesting at many sites. However, NMFS also recognizes that the nest count data, including data for green sea turtles in the Atlantic, only provides information on the number of females currently nesting, and is not necessarily a reflection of the number of mature females available to nest or the number of immature females that will reach maturity and nest in the future. Also, the trend in the number of green sea turtle nests laid is not an indication of the overall population trend given that the proportion of adult males to females, and the age structure of the population(s) is unknown. Finally, given the late age to maturity for green sea turtles (20 to 50 years; Balazs 1982, Frazer and Ehrhart 1985; Seminoff 2004), caution is urged regarding the trend for any of the nesting groups since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d). This determination that the trend for green sea turtles as a species is declining provides benefit of the doubt to the species given its endangered and threatened classification under the ESA, the many on-going negative impacts to the species across all areas of its range and to all age classes, the declining or uncertain trend in nesting for the majority of the world's nesting sites for green sea turtles, and the lack of up-to-date nesting information for the largest green sea turtle nesting site in the Indian Ocean and possibly the world.

6.0 EFFECTS OF THE PROPOSED ACTION ON ESA-LISTED SEA TURTLES

As described in Section 1.0, NMFS has determined that ESA-listed loggerhead, leatherback, Kemp's ridley, and green sea turtles will continue to be affected by the continued authorization of the scallop fishery as a result of: (a) capture in scallop dredge and trawl gear, and (b) physical contact with chain-mat equipped scallop dredge gear that may or may not result in subsequent capture of the sea turtle in the dredge bag or retention of the turtle against the outside of the dredge bag that is visible upon hauling of the gear. NMFS's assessment of the effects of the scallop gear-sea turtle interactions on loggerhead, leatherback, Kemp's ridley, and green sea turtles is provided below in order for NMFS to make a final determination as to whether the proposed action is likely to jeopardize the continued existence of any of these species.

Sea turtles are known to be killed and injured as a result of being struck by vessels on the water. Fishing vessels operating as a result of the continued authorization of the scallop fishery under the Scallop FMP are unlikely to strike loggerhead, leatherback, Kemp's ridley, or green sea turtles in the action area given that: (a) scallop fishing vessels operate at a relatively slow operating speed, (b) a portion of the fishing occurs in areas in which sea turtles are less likely (*e.g.*, Georges Bank) or not likely (*e.g.*, northern Gulf of Maine) to be present in comparison to Mid-Atlantic waters, (c) a portion of the fishing occurs at times when sea turtles are not likely to be present (the winter period in Mid-Atlantic waters and the late-fall though mid-spring in New England waters) (NMFS 2003; 2004a; 2004b), (d) sea turtles spend part of their time at depths out of range of a vessel collision with boats used in the scallop fishery, (e) the proposed action is not expected to increase the amount of vessel traffic in areas where sea turtles occur, and (f) the fishery will continue as a limited access fishery, and the number of participants are expected to be further constrained by Amendment 11 to the Scallop FMP.

The continued authorization of the scallop fishery will not reduce the availability of prey for loggerhead, leatherback, Kemp's ridley or green sea turtles. Scallop dredge and trawl gear catch horseshoe crabs, other crabs species, whelks and fish as bycatch along with the targeted catch of scallops (NEFMC 2003; NMFS 2007b). None of these are typical prey species of leatherback sea turtles or of neritic juvenile or adult green sea turtles (the age classes anticipated to be captured in the scallop fishery) (Rebel 1974; Mortimer 1982; Bjorndal 1985; USFWS and NMFS 1992; Bjorndal 1997). Therefore, continued authorization of the scallop fishery will not affect the availability of prey for leatherback and green sea turtles in the action area.

Neritic juveniles and adults of both loggerhead and Kemp's ridley sea turtles are known to feed on species that are caught as bycatch in the scallop fishery (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). Some of the bycatch is expected to be returned to the water alive, while the remainder will be returned to the water dead or injured to the extent that the organisms will shortly die. Nevertheless, the continued authorization of the scallop fishery is not expected to affect the availability of prey for loggerhead or Kemp's ridley sea turtles in the action area given that: (a) the turtle food items caught as bycatch will be returned to the water where they could still be preyed upon by sea turtles, particularly loggerheads which are known to eat a variety of live prey as well as scavenge dead organisms, and (b) nesting by Kemp's ridley sea turtles has increased for the last several years strongly suggesting that the species is not food limited.

6.1 Information Available for the Assessment

With only two exceptions, all known turtle interactions with scallop fishing gear have occurred below the water surface. The only visible evidence of these underwater interactions are observations of turtles captured within the scallop dredge or trawl bag or observed on the outside of the gear (*e.g.*, wedged between parts of the dredge frame) upon hauling of the gear. Video cameras have been attached to scallop dredge gear in an attempt to record turtle-scallop dredge interactions below the water surface. However, no interactions occurred in the 80 hours of video recorded by the cameras (Memo to the File from E. Keane, February 24, 2006).

Sea turtles incidentally captured in fishing gear must be reported to NMFS on Vessel Trip Reports that are required for the Federal scallop fishery and other federal fisheries. However, to date, there have been no reports of turtle interactions on VTR forms submitted by scallop fishing vessels. The absence of reports does not mean that interactions were not or are not occurring. Compliance with the Federal requirement for federally permitted fisherman to report sea turtle interactions on their VTRs is very low for all fisheries where VTRs are required. As described further below, NMFS trained observers reported more than 90 interactions between sea turtles and scallop fishing gear (dredge and scallop trawl gear combined) during the period of 1996-2007. There were no VTR reported interactions of sea turtles with scallop fishing gear during the same time period. Contract reports for work on modifications of scallop dredge gear were submitted to NMFS in 2004 and 2005, and included statements that scallop vessel captains noted in the summer of 2000 that sea turtle interactions with scallop dredge gear were occurring (DuPaul *et al.* 2004; Smolowitz *et al.* 2005). Those interactions were not reported to NMFS on VTRs.

In the absence of VTR reporting, the only means by which NMFS has acquired information on sea turtles captured in or retained upon gear used in the commercial scallop fishery is by reporting from NMFS trained observers assigned to fishing vessels on a trip-by-trip basis. Information on the number, condition, and species of sea turtles captured in or retained upon scallop dredge and trawl gear is collected by NMFS trained observers and submitted to the NEFSC, Fisheries Sampling Branch (FSB). In some cases, the observer reported a sea turtle take in scallop fishing gear that was seen only by the vessel's crew (*e.g.*, takes that occurred when the observer was off-duty). In all cases, the observer collected as much of the information as possible.

In order to assess the effects of the continued authorization of the sea scallop fishery on ESA-listed sea turtles, NMFS is using information collected by observers as well as information on the description and operation of scallop fishing gear, life history information for sea turtles, and the effects of fishing gear entanglements on sea turtles that has been published in a number of documents. These documents include sea turtle status reviews and biological reports (NMFS and USFWS 1995; 2007a; 2007b; 2007c; 2007d; TEWG 1998; 2000), recovery plans (NMFS and USFWS 1991a and b; NMFS and USFWS 1992; USFWS and NMFS 1992), the stock assessment report for loggerhead and leatherback sea turtles (NMFS SEFSC 2001), characterization of sea turtle takes in the scallop fishery (Haas *et al.* in review), estimates of sea turtle takes in the scallop fishery (Murray 2004a; 2004b; 2005; 2007), and numerous other sources of information from the published literature as cited below.

6.1.1 Description of the Gear

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; NEFSC pers. comm.). 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 (aka 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 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). TEDs are not required to be used in scallop trawl nets.

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 4-inches (10.3 cm) 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 5 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-ft dredge frame weighs approximately 4,500 lb (Memo to the File, E. Keane, March 2008). Vessels travel at speeds of 4-5 knots when towing dredge gear (NREFHSC 2002; Murray 2004b; Murray 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 2006b).

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 turtles respectively) in the ring bag. The chain mat is designed to have more consistently sized openings which must be 14 inches or less.

6.1.2 Description of the Sea Turtles Caught in Scallop Dredge Gear

The first NMFS observer report of a sea turtle take in scallop dredge gear occurred in 1996 (Murray 2004a). The most recent observer reports of sea turtle takes in scallop dredge gear occurred in 2007. From 1996-2007, a total of 89 sea turtle takes in scallop dredge gear were reported by NMFS trained observers. Sea turtle takes in the scallop dredge fishery as reported by the observers include: (a) sea turtles that are observed to be captured in the gear (either in the dredge bag or caught on parts of the dredge frame), (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 the turtles are at the water surface.

Nine of the 89 sea turtle takes in scallop dredge gear were moderately to severely decomposed turtle carcasses. Since a turtle cannot be caught, killed, and decompose in the time it takes for a dredge tow to be completed, the decomposed turtles could not have been killed as a result of the dredge tow in which each was observed. In addition, since the cause of death could not be identified, it cannot be assumed that any of the decomposed turtles were killed as a result of a previous interaction with scallop dredge gear. Six of the nine turtles were tangled in gillnet gear although, again, the cause of death could not be determined. Therefore, given this information, NMFS is not counting these 9 sea turtle takes as effects of the scallop fishery on ESA-listed sea turtles.

Not all of the 80 non-decomposed sea turtles reported by observers for the scallop dredge fishery were able to be brought aboard the vessel for examination and data collection. Two of the 80 interactions were witnessed when turtles at the surface were observed to bump into the gear (Haas *et al.* in review). Fourteen turtles either fell out/off of the gear or swam away from the gear before the gear could be brought on board. Two other sea turtles were not brought on board but the reasons for this were not described in the observer report (Haas *et al.* in review). Sixty-two turtles captured in or upon the dredge gear were brought on board. Information on species identification and the condition of the turtle (any apparent injuries) are more easily obtained from turtles that are brought on board the vessel. Turtles brought on board are also measured, tagged (if tags are not already present), and photographed when possible.

The majority of sea turtles observed in the fishery were loggerhead sea turtles. Of the 58 sea turtles identified to species, 55 were loggerheads, 2 were Kemp's ridleys, and 1 was a green sea turtle. Twenty-two were unidentified to species. Additional training of observers since 2001 has greatly reduced the number of 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 turtle to species (*e.g.*, when a turtle drops or swims out of the gear before the dredge can be brought on deck).

Loggerhead sea turtles observed captured in scallop dredge gear ranged in length from 62.2 – 107 cm (24.5 – 42.1 in) from notch to tip (curved carapace length (CCL)) (NEFSC, FSB, Observer Database)[7]. One of the two Kemp's ridley sea turtles observed captured in scallop dredge gear measured 24.3 cm (9.5 in) CCL. The other could not be measured due to extensive damage to the fresh carcass. An estimated length of 70 cm (27.3 in) in length was reported for the single green sea turtle observed captured in scallop dredge gear. An actual measurement was not reported.

Of the 80 non-decomposed turtles reported as taken in the scallop dredge fishery from 1996-2007, only 6 were fresh dead or died on the vessel. However, an additional 34 of the 80 were alive but suffered injuries. In many, but not all, cases injuries were described as one or more

[7] A turtle observed taken in 2004 was estimated by the observer to be 170 cm (66.9 in) in length. Since 170 cm is an unusually large size for a loggerhead sea turtle and since the size of the turtle was estimated based on the observer's view of the turtle in the water (the turtle was never brought aboard the vessel), it is likely that the estimated length is an error. Therefore, it is not included here.

cracks to the carapace and/or plastron with blood or soft tissue visible. Twenty-two of the 80 turtles were reported by observers as alive and uninjured while the remaining 18 were alive but of unknown condition.

NEFSC trained observers observed some portion of the scallop dredge trips taken in every month for the years 2003-2005 and 2007 (NEFSC, FSB website). However, takes of sea turtles in scallop dredge gear have only been observed in the months of June through October.

As has been described in previous opinions (NMFS 2004a; 2004b), tissue samples for genetics analysis have been collected from loggerhead sea turtles captured in the scallop fishery. The genetics analysis examines mitochondrial DNA which is inherited from the mother. The results are then compared to the genetics data (haplotypes) that are associated with each nesting group. Twenty-six samples have been analyzed that were collected from 23 loggerheads captured in scallop dredge gear and 3 loggerheads captured in scallop trawl gear from 2002 - 2004 (Haas *et al.* in review). Two sets of results were generated for the 26 samples collected from loggerhead sea turtles captured in scallop fishing gear.

The results from the 26 samples differ depending on whether the data was analyzed: (1) Assuming that there was an equal probability amongst the nesting groups that a sea turtle originating from any group will be caught in the scallop fishery, or (2) assuming that there was an unequal probability such that the number of turtles caught from each nesting group will be in proportion to the size of the nesting group (where size here refers to the estimated number of nesting females in the nesting group). In other words, a nesting group composed of 600 nesting females is likely to produce fewer offspring than a nesting group of 60,000 nesting females. As a result, there would be an unequal probability amongst the nesting groups that a sea turtle originating from any group will be caught in the scallop fishery since there are more or less of some turtles from each nesting group to be caught. Haas *et al.* (in review) present both the unweighted results (equal probability amongst the nesting groups) and the weighted results (unequal probability amongst the nesting groups based on nesting group size) (Table 1).

Table 1. Genetic results from 26 sampled loggerhead sea turtles captured in the scallop fishery from 2002 -2004, and 295 sampled loggerhead sea turtles captured in pound net gear in the Pamlico-Albemarle Estuarine complex from 1996-1997. Results are presented as the percentage of each nesting group represented in the analyzed samples, and with respect to whether the results are weighted by the proportion of nesting contributed by each nesting group to total loggerhead nesting in the Atlantic.

Reference	Sample size	south FL	Northern	Mexico	FL Panhandle	Dry Tortugas	Greece	Turkey	Brazil
Haas *et al.* in review (unweighted)	26	0.63	0.07	0.06	0.11	0.07	0.04	0.01	0.01
Haas *et al.* in review (weighted)	26	0.89	0.03	0.04	0.01	0	0.03	0	0
Bass *et al.* 2004 (weighted)	295	0.80	0.12	0.06	0.02				

The weighted results are similar to those presented by Bass *et al.* (2004) that were previously used to assess the relative impact of the scallop fishery on each loggerhead nesting group in the Atlantic (NMFS 2004b; 2005). However, as noted by Bass *et al.* (2004), weighting genetic results by size of each nesting group may bias the true results. There is increasing evidence that juvenile loggerheads exhibit homing to foraging grounds in the vicinity of their nesting beaches (Bowen *et al.* 2004; 2005). Thus, marine activities, such as fisheries, that capture, injure or kill loggerhead sea turtles may have a greater impact on those turtles that originate from the nesting beaches closest to the operation area of the activity. For the analysis of genetic results from turtles captured in scallop fishing gear, this means that simply weighting the results with respect to the estimated size of each western North Atlantic loggerhead nesting group may skew the true results since the south Florida is the largest of the nesting groups but is not the closest in proximity to the area where the scallop fishery operates.

In light of the above, it is likely that neither the weighted or unweighted genetic results from the 26 sampled loggerhead sea turtles captured in the scallop fishery in 2002-2004, or from Bass *et al.* 2004 that have been previously used, accurately depict the contribution of each nesting group to the loggerheads affected by the scallop fishery. The weighted results of Haas *et al.* (in review) and Bass *et al.* (2004) likely overestimate the number of turtles in the sample that originated from nesting beaches of the south Florida nesting group, and underestimated the number of turtles in the sample that originated from nesting beaches of the northern nesting group (whose nesting beaches are closest in proximity to where the scallop fishery operates). However, the unweighted results of Haas *et al.* likely overestimate the number of turtles in the sample that originated from nesting beaches of the relatively small Florida Panhandle and Dry Tortugas nesting groups. The unweighted results suggest that more of the sampled turtles originated from nesting beaches of the Florida Panhandle nesting group in comparison to the northern nesting group, and that an equal number of turtles originated from nesting beaches of the Dry Tortugas nesting group in comparison to the northern nesting group. Both of these scenarios would seem unlikely given that nesting beaches of the northern nesting group are closer in proximity to the area where the scallop fishery operates than the nesting beaches of either the Florida Panhandle or Dry Tortugas nesting group, and given that the mean annual nest count of the northern nesting group is greater than either the Florida Panhandle or Dry Tortugas nesting groups (5,151 annual mean nest count versus 910 and 246, respectively; NMFS and USFWS 2007a).

The overall pattern of the genetic results from the 26 sampled sea turtles caught in scallop fishing gear is informative, however, and comparable to other genetic studies that sampled loggerhead sea turtles occurring within the area where the scallop fishery operates (Rankin-Baransky *et al.* 2001; Bass *et al.* 2004). Thus, based on the results presented by Haas *et al.* (in review), it can be concluded that 63-89% of loggerheads that are captured in, retained upon, and/or struck by (without subsequent capture in the case of dredge gear fitted with a chain mat) scallop trawl and scallop dredge gear originate from the south Florida nesting beaches.

6.1.3 Description of the Sea Turtles Caught in Scallop Trawl Gear

The first NMFS observer report of a sea turtle take in scallop trawl gear occurred in 2004 (Murray 2007). The most recent observer reports of sea turtle takes in scallop trawl gear occurred in 2007. From 2004-2007, a total of 16 sea turtle takes in the scallop trawl fishery were reported by NMFS trained observers. All of the sea turtle takes in the scallop trawl fishery as reported by the observers have been of sea turtles captured in the net of the trawl gear. One of the 16 sea turtles captured in scallop trawl gear was decomposed. As is the case for scallop dredge gear, a scallop trawl is not towed sufficiently long enough for a live turtle to be captured, killed, and decompose before the gear is hauled. Therefore, the turtle was not killed as a result of the scallop trawl gear in which it was observed. Since the cause of death could not be identified, it cannot be assumed that the sea turtle was killed as a result of a previous interaction with scallop trawl gear. Therefore, NMFS is not counting this sea turtle take as an effect of the scallop fishery on ESA-listed sea turtles.

All of the 15 non-decomposed sea turtles observed captured in scallop trawl gear were brought aboard the fishing vessel. However, one of these fell overboard as the catch was dumped and could not be further examined. All of the remaining 14 sea turtles were identified as loggerheads. Twelve of the 14 sea turtles were measured for size. Sizes of the turtles ranged from 71.0 - 99.0 cm (27.7 - 38.6 in) CCL. As described in section 6.1.2, 63-89% of the loggerhead sea turtles captured in scallop trawl gear are turtles that originate from nesting beaches of the south Florida nesting group.

All of the 14 sea turtles captured in scallop trawl gear and brought on board the vessel were alive. Twelve of the 14 had no apparent injuries and were returned to the water following data collection. One of the 14 turtles appeared to be comatose and another was described as lethargic. Both were resuscitated in accordance with the regulations (50 CFR 223.206(d)(1)(B)) and subsequently released back into the water alive.

The NEFSC has observed scallop trawl trips in all months of the year. Takes of sea turtles in scallop trawl gear have only been observed in the months of June through September.

6.1.4 Information on Factors Affecting Sea Turtle Capture in Scallop Fishing Gear

As described in sections 3.1.1 – 3.1.4, the occurrence of loggerhead, leatherback, Kemp's ridley, and green sea turtles in New England waters and Mid-Atlantic waters north of Cape Hatteras, NC is temperature dependent (Keinath *et al.* 1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998; Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; James *et al.* 2005b; Morreale and Standora 2005). In general, turtles move up the coast from southern wintering areas as water temperatures warm in the spring (Keinath *et al.* 1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998; Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; James *et al.* 2005b; Morreale and Standora 2005). The trend is reversed in the fall as water temperatures cool. By December, turtles have passed Cape Hatteras, returning to more southern waters for the winter (Keinath *et al.* 1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998; Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; James *et al.* 2005b; Morreale and Standora 2005). Recreational anglers have reported sightings of sea turtles in waters defined as inshore waters (bays, inlets, rivers, or sounds; Braun-McNeill and Epperly 2004) as far north as New York as early as March-April, but in relatively low numbers (Braun-McNeill and Epperly 2004). Greater numbers of loggerheads, Kemp's ridleys, and greens are found in Virginia's inshore, nearshore and offshore waters from May through November and in New York's inshore, nearshore and offshore waters from June through October (Keinath *et al.* 1987; Morreale and Standora 1993; Braun-McNeill and Epperly 2004). The hard-shelled turtles appear to be temperature limited to water no further north than Cape Cod. Leatherback sea turtles have a similar seasonal distribution but have a more extensive range in the Gulf of Maine compared to the hard-shelled species (Shoop and Kenney 1992; Mitchell *et al.* 2003; STSSN database).

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

Since the scallop fishery does not operate in bays, inlets, rivers, or sounds, sea turtle distribution would not be expected to overlap with the distribution of scallop fishing gear until May in nearshore and offshore waters off of North Carolina and Virginia, and until June in nearshore and offshore waters off of New York. 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 waters of southern Georges Bank.

The NEFSC has attempted to identify a variable for predicting sea turtle bycatch in the dredge component of the scallop fishery during times and in areas where sea turtle distribution and operation of the scallop fishery overlap (Murray 2004a; 2004b; 2005). Using a modeling approach, sea surface temperature, depth, time-of-day and tow time were identified as variables affecting estimated bycatch rates of sea turtles with scallop dredge gear (Murray 2004a; 2004b; 2005). However, the variable(s) associated with the highest estimated bycatch rates changed from one year to another (*e.g.*, sea surface temperature, depth) or could not be further analyzed (*e.g.*, time-of-day and tow time) because the information is not collected for the entire fishery (Murray 2004a; 2004b; 2005). In addition, while the modeling approach demonstrated that higher rates of turtle bycatch occurred with certain temperature or depth ranges, sea turtle captures in scallop fishing gear also occurred outside of the temperature and depth ranges. Therefore, a single variable has not yet been identified for forecasting sea turtle bycatch with scallop fishing gear, or for identifying when/where higher levels of sea turtle interactions with scallop fishing gear might regularly occur.

NMFS has also considered other factors that might effect the likelihood that a sea turtle will be captured in or otherwise taken by (meaning physical contact without capture in) scallop fishing gear. These other factors include the behaviour of sea turtles in the presence of fishing gear, as well as the effect of certain oceanographic features and fishery practices on sea turtle distribution and abundance. For example, video footage recorded by NMFS, Southeast Fisheries Science Center (SEFSC), Pascagoula Laboratory indicated 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 2002). Turtles have also been observed to dive to the bottom and hunker down when alarmed by loud noise or gear (Memo to the File, L. Lankshear, December 4, 2007), which could place them in the path of bottom gear such as a scallop dredge or trawl. With respect to oceanographic features, a review of the data associated with the 11 sea turtles captured by the scallop dredge fishery in 2001 concluded that the turtles appeared to have been near the shelf/slope front (D. Mountain, pers. comm.). Intense biological activity is usually associated with oceanographic fronts because they are areas where water masses of different densities converge (www.mbari.org/muse/Participants/Robison-Hamner.html; Robison and Hamner, posted February 18, 2004). Such oceanographic features occurring in the same area as the operation of scallop dredge gear may increase the risk of interactions between scallop dredge gear and sea turtles. Scallop fishing practices may also influence sea turtle distribution and abundance in areas where scallop vessels are operating. Scallop fishing gear stirs up and catches turtle prey species. The bycatch, as well the scallop offal, are discarded. The stirring up of prey items as well as the discarding of turtle prey and scallop offal may attract sea turtles to areas where scallop fishing gear is operating, thus increasing the likelihood of sea turtle interactions with the gear. Nevertheless, while all of the above are reasonable circumstances that might be affecting the likelihood of sea turtle interactions with scallop dredge and trawl gear, there is currently no information to support any of these.

Based on the best, currently available information, sea turtle interactions with scallop gear are likely at times when and in areas where sea turtle distribution overlaps with operation of the fishery. Observer data has provided data by which to estimate loggerhead bycatch in the scallop fishery for the year in which the data was collected. However, no predictive variable or set of variables has yet been identified that would enable NMFS to predict the number of future sea turtle takes in the fishery.

6.2 Anticipated Effects of the Proposed Action

NMFS has identified that the proposed action is likely to adversely affect ESA-listed sea turtles when the turtles come into physical contact with scallop fishing gear. Such contact has occurred in the scallop fishery and resulted in injuries, including very severe injuries causing death, to sea turtles. No other direct effects to sea turtles are expected as a result of the proposed action. No indirect effects to sea turtles are expected as a result of the proposed action. In this section of the Biological Opinion, NMFS will determine, given the currently available information, the anticipated number of sea turtles that will be affected by the continued authorization of the scallop fishery defining such effects by species, the estimated mortality of sea turtles that are caught by species, and the age class of turtles that are struck by scallop dredge and trawl gear.

6.2.1 Anticipated number of sea turtle interactions with scallop dredge gear

As described above, no method has yet been identified for predicting the level of sea turtle bycatch in the scallop fishery. The extent of loggerhead bycatch has been estimated for some years based on data collected by observers. Based on data collected by observers for the reported sea turtle captures in or retention upon scallop dredge gear, the NEFSC estimated loggerhead bycatch in the scallop dredge fishery for 2003, 2004 and 2005 (Murray 2004b; 2005; 2007). Estimates were also calculated for 2001 and 2002 but these were applicable only to portions of the scallop dredge fishery operating in Mid-Atlantic waters in those years (Murray 2004a). For the September 18, 2006, Opinion for the Scallop FMP, NMFS determined that the estimated take of 749 sea turtles in the scallop dredge fishery in the 2003 fishing year represented the best available information on the anticipated annual take of loggerhead sea turtles in the scallop dredge fishery in any given year, given that: (a) There was no new information at that point to suggest that sea turtle interactions with scallop dredge gear had increased (observed interactions had declined despite higher levels of observer coverage in the 2004 and 2005 fishing years), (b) NMFS could not predict sea turtle bycatch in the scallop fishery, and (c) the estimate of 749 sea turtles was based on the operation of the fishery before the mandatory or voluntary use of chain mat equipped scallop dredges in the Mid-Atlantic, the consequence of which is that the number of observable interactions is reduced. While all of these points are still accurate, a further review of the information suggests that this approach likely overestimates the annual number of sea turtles that will be struck by and captured in or retained upon scallop dredge gear (or struck but not captured as a result of use of a chain mat). Because observers noted the presence of chain mats on some scallop dredge gear in 2004 and 2005 (prior to the required use of chain mats), NMFS considered the 2003 estimate of observed takes to be the best available for the September 18, 2006, Opinion on the Scallop FMP. However, annual observed turtle bycatch rates varied considerably during the period of 2003-2005, and the use of chain mats on some observed scallop dredge trips in 2004 and 2005 does not account for the differences. For example, Murray (2005) noted that 4.5% of observed dredge hours in Mid-Atlantic waters from June-November 2004 were for dredges fitted with chain mats. Removing these hours from the total observed dredge hours for 2004 would result in 16,894.7 observed dredge hours for non-modified gear; a value more than 50% greater than the 2003 observed dredge hours, but a time period in which observed turtle interactions were more than 50% less than in 2003. Similarly, Murray (2007) noted that 2.4% of observed dredge hours in Mid-Atlantic waters from June – November 2005 were for dredges fitted with chain mats. Removing these hours from the calculations would result in 7,851.8 observed dredge hours; a reduction of approximately 26% from the 2003 observed dredge hours for the same time period, while observed turtle interactions were 100% less than those observed in 2003. NMFS, therefore, has rejected the approach taken in the September 18, 2006, Opinion and does not consider 749 to be a reasonable estimate of the anticipated annual take of loggerhead sea turtles in the scallop dredge fishery.

Previous estimates of loggerhead sea turtle takes in the scallop dredge fishery still provide the best available information for determining the anticipated take of loggerhead sea turtles in the fishery since no predictive variable or set of variables has been found. For the purposes of this Opinion, NMFS is using the estimates generated for 2003 and 2004 (point estimates of 749 and 180, respectively). NMFS is not using the estimate for 2005, since the bycatch estimate resulted in a zero but 3 sea turtles are known to have been captured in scallop dredge gear when the observer was off-watch (Murray 2007)[8]. Counts of turtles that are captured in scallop fishing gear when an observer is off-watch cannot be used for estimating observed bycatch rates (Murray 2007). Off-watch takes that occurred in 2003 and 2004 were subsumed in the respective bycatch estimates since the estimates were based on the observed bycatch rates and the total scallop dredge fishing effort for the designated time period as reported on the VTR's. However, since the observed bycatch rate for 2005 was zero, any estimate of turtle bycatch in the fishery using the observed bycatch rate also equals zero. Thus the 2005 bycatch estimate underestimates the bycatch of sea turtles in the scallop dredge fishery for 2005. Since it cannot be determined how much the 2005 estimate underestimates the actual take, NMFS is not using the 2005 estimate for defining the anticipated take of loggerhead sea turtles in the scallop dredge fishery. NMFS is also not using data collected for 2006. As described above, 1 loggerhead turtle was observed taken by scallop dredge gear in 2006, but no estimate of take is being calculated. In 2006, there was no observer coverage of scallop dredge gear in Mid-Atlantic waters for June and July. In addition, based on the monthly reports of observer coverage posted on the NEFSC, FSB website, observer coverage of scallop dredge trips for August through November was low in comparison to previous years. Finally, approximately one-third of the observer days for scallop dredge trips fished in the Mid-Atlantic occurred on vessels whose dredge gear had been modified with the addition of a chain mat (NEFSC, FSB Monthly Observer Reports).

Therefore, the anticipated take of loggerhead sea turtles in scallop dredge gear biennially is based on the sum of the point estimates for 2003 and 2004 (749 and 180, respectively) which is 929 sea turtles. An estimate of turtle bycatch for 2003 and 2004 using a simple ratio approach of the same data ((number of observed turtle interactions ÷ number of observed dredge hours) x number of VTR dredge hours) yields similar results. These are 809 sea turtles for 2003, and 187 sea turtles for 2004, for a total of 995 for the 2 years combined.

There are no estimates for the capture of Kemp's ridley sea turtles in scallop dredge gear. Since only two Kemp's ridley sea turtles have been observed in the scallop dredge fishery in the period 1996-2007, it is likely that interactions with this species are relatively rare events on an individual haul basis. Nevertheless, given effort in the fishery as a whole, and the seasonal overlap in distribution of this species with operation of scallop dredge gear, Kemp's ridley sea turtles are likely to be captured in scallop dredge gear. A prediction of one take results from using an average of the number of takes for the 1996-2007 time period since a "part" of a turtle cannot be taken. Similarly, there are no estimates for the capture of green sea turtles in scallop dredge gear. NMFS has recorded one capture of a green sea turtle in scallop dredge gear. Therefore, a prediction of one take of a green sea turtle, likewise results from using an average of the number of green sea turtle takes for the 1996-2007 time period.

There have been no confirmed takes of leatherback sea turtles in scallop dredge gear. Tagging studies have shown that leatherbacks, occurring seasonally for foraging in western North

[8] The bycatch estimate result was 0.00014 turtles with a CV = 0.19. For the purposes of biological opinions NMFS typically considers any part of a turtle taken as equal to 1 turtle. However, Murray (2007) interpreted the bycatch estimate results as a zero in accordance with the mathematical rules for rounding numbers.

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 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 was being towed along the bottom. Based on observations of loggerhead turtles taken in scallop dredge gear, NMFS believes some sea turtle interactions with scallop dredge gear occur within the water column. Given the large size of the dredge bag and the presence of leatherback sea turtles in areas where the scallop dredge fishery occurs, NMFS does believe that leatherback sea turtles can be captured in scallop dredge gear when the gear is in the water column (NMFS 2006b). With respect to other mobile gear operating in the area where the sea scallop dredge fishery operates, there have been only two observed takes (capture in the gear) of leatherback sea turtles over the period 1995-2007. This suggests that capture of leatherback sea turtles in any mobile gear operating within the action area, including scallop dredge gear, would be a rare event. However, given the generally low level of observer coverage in the scallop dredge fishery as well as other mobile gear (trawl) fisheries in the action area, it is likely that some interactions with leatherback sea turtles have occurred but were not observed or reported. Therefore, NMFS believes that the actual observed take of one leatherback sea turtle in mobile gear that was operating in the action area for this Opinion provides the best available information on the anticipated annual take of leatherback sea turtles in scallop dredge gear.

As described in Section 2.1, NMFS has approved Amendment 11 to the Scallop FMP and anticipates implementing it in 2008. The purpose of Amendment 11 is to limit effort in the general category sector of the scallop fishery, and may reduce effort in the scallop fishery compared to current circumstances. Nevertheless, the use of turtle bycatch estimates from 2003 and 2004 for determining the anticipated take of sea turtles in scallop dredge gear in the future is still appropriate given that: (1) The 2003 estimate of sea turtle takes in the scallop dredge fishery is based on effort in the fishery prior to the dramatic increase in general category effort in 2004 and 2005, and (2) the 2004 and 2005 observed and estimated sea turtle interactions with scallop dredge gear were lower than in 2003, despite greater fishing effort by the general category sector.

6.2.2 Age class of sea turtles anticipated to be captured in scallop dredge gear

NMFS SEFSC (2001) reviewed size at stage data for Atlantic loggerheads. Depending on the dataset used, the cutoff between pelagic immature and benthic immature loggerhead sea turtles was 42 cm – 49 cm (16.4 – 19.11 in) straight carapace length (SCL) and the cutoff between benthic immature and sexually mature loggerhead sea turtles was described as 83 cm – 90 cm (32.4 – 35.1 in) SCL. Other authors define the benthic immature stage for loggerheads as 36 cm – 100 cm SCL (Bass *et al.* 2004). Loggerhead sea turtles observed captured in scallop dredge gear ranged in length from 62.2 cm – 107 cm (24.5 – 42.1 in) CCL (NEFSC, FSB, Observer Database). When converted to SCL based on the formula for loggerheads provided in Teas (1993), the size range of loggerhead sea turtles observed captured in the fishery is 57.5 cm – 100 cm (22.6 - 39.4 in) SCL. Based on these datasets and observer measurements of loggerhead sea turtles captured in the scallop dredge fishery, NMFS expects that both benthic immature and sexually mature loggerhead sea turtles will be captured in scallop dredge gear as a result of the continued authorization of the scallop fishery.

The first Kemp's ridley sea turtle observed captured in scallop dredge gear measured 24.3 cm (9.5 in) CCL. Using the formula for Kemp's ridley sea turtles provided in Teas (1993), this is a straight carapace length of 23 cm (9 in). The post-hatchling stage for Kemp's ridley sea turtles was defined by the TEWG as Kemp's ridleys of 5 – 20 cm (2 – 8 in) SCL while turtles 20 – 60 cm (8 – 23 in) SCL were considered to be benthic immature (TEWG 2000). The latter stage is described as turtles that have recruited to coastal benthic habitat. In the case of the turtle observed captured in scallop dredge gear, the observed take occurred on the southeastern edge of Georges Bank, far from the coast. For the purposes of this Opinion, NMFS will consider this turtle simply as an immature Kemp's ridley sea turtle. The second Kemp's ridley captured in scallop dredge gear was not measured since only part of the carcass was recovered. However, veterinarian review of the partial carcass determined that the turtle must have been an immature given the size of the features on the partial carcass. 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 (Musick and Limpus 1997; TEWG 2000; Morreale and Standora 2005). Given the life history of the species and the size of Kemp's ridley turtles captured in scallop dredge gear, NMFS expects that immature Kemp's ridley sea turtles will be captured in scallop dredge gear as a result of the continued authorization of the fishery.

The single green sea turtle observed captured in scallop dredge gear was estimated by the observer to be about 70 cm (27 in) in length. Hirth (1997) defined a juvenile green sea turtle as a post-hatchling up to 40 cm (16 in) SCL. A subadult was defined as green sea turtles from 41 cm (16 in) through the onset of sexual maturity (Hirth 1997). Sexual maturity was defined as green sea turtles greater than 70 – 100 cm (27 – 39 in) SCL (Hirth 1997). It is difficult to determine to which age class the green sea turtle observed taken in scallop dredge gear might have belonged given that its size was estimated rather than measured. Like Kemp's ridley sea turtles, Mid-Atlantic 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, NMFS expects that benthic immature and/or sexually mature green sea turtles will be captured in scallop dredge gear as a result of the continued authorization of the scallop fishery.

NMFS believes that leatherback sea turtles may be captured in scallop dredge gear given the large size of the dredge frame and bag and the presence of leatherback sea turtles in areas where the scallop dredge fishery occurs. Stranding and sighting records suggest that both adult and immature leatherback sea turtles occur within the action area where the scallop dredge fishery operates (NMFS and USFWS 1992; NMFS SEFSC 2001). Tracking of tagged leatherbacks also demonstrate the movement of sexually mature leatherbacks over U.S. continental shelf waters (James *et al.* 2005a; 2005b). Therefore, either immature or sexually mature leatherback sea turtles could be captured in scallop dredge gear since both age classes occur in areas where scallop dredge gear operates.

6.2.3 Estimated mortality of sea turtles captured in scallop dredge gear

As suggested by the information provided above, 90% of the non-decomposed, condition known sea turtles reported by observers for the scallop dredge fishery were returned to the water alive. Nevertheless, capture of sea turtles in or their retention upon scallop dredge gear likely results in a higher level of sea turtle mortality than is evident based on the number of turtles returned to the water alive. Injuries suffered by sea turtles captured in or retained upon scallop dredge 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 characterized by scrapes to soft tissue, cracks to the carapace, cracks to the 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 in the future for sea turtles affected by the continued authorization of the scallop fishery.

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 turtles rising from 0% for the first 50 minutes of capture to 70% after 90 minutes of capture (Henwood and Stuntz 1987). However, metabolic changes that can impair a sea turtle's ability to function can occur within minutes of a forced submergence. While most voluntary dives appear to be aerobic, showing little if any increases in blood lactate and only minor changes in acid-base status, the story is quite different in forcibly submerged 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 National Research Council to reexamine the association between tow times and sea turtle deaths, the data set used by Henwood and Stuntz was updated and re-analyzed (Epperly *et al.* 2002; Sasso and Epperly 2006). Seasonal differences in the likelihood of mortality for sea turtles caught in trawl gear were apparent. For example, the observed mortality exceeded 1% after 10 minutes of towing in the winter (defined in Sasso and Epperly 2006 as the months of December-February), while the observed mortality did not exceed 1% until after 50 minutes in the summer (defined as March – November; Sasso and Epperly 2006). In general, tows of short duration (<10 minutes) in either season have little effect on the likelihood of mortality for sea turtles caught in the trawl gear and would likely achieve a negligible mortality rate (defined by the NRC as <1 percent). 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 percent, as a 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 1 hour or less, which should help to reduce the risk of death from forced submergence for turtles caught in scallop dredge gear, but does not eliminate the risk.

Contact injuries involving damage to the carapace and/or plastron of the turtle have been frequently observed in the scallop dredge fishery. As stated in section 6.1 above, no underwater interactions of sea turtles with scallop dredge gear have been observed or photographed. However, given current knowledge of sea turtle life history, the condition of turtles captured in or upon dredge gear as described by the observers, and an understanding of the gear and how it is fished, there are several ways that a turtle might suffer cracks to the carapace and/or plastron during interactions with scallop 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 (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). The scallop dredge fishery is known to capture crabs, whelks, and other organisms as bycatch (NEFMC 2003; NMFS 2007b). Therefore, if loggerhead and Kemp's ridley sea turtles are foraging in areas where scallop dredging occurs, the turtles are likely to be spending some of their time on or very near the bottom where they would be at risk of being struck by scallop dredge gear. Given that the cutting bar rides just 4-inches (10.3 cm) off the seabed (Smolowitz 1998), and the gear weighs approximately 4,500 lb (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 turtle enters the dredge bag, it may be injured by large rocks that are also caught in the dredge bag. A fishery observer report of a sea turtle taken in 1999 indicated that there were large rocks in the bag along with the sea turtle, which had sustained a cracked carapace suggesting that the boulders may have caused the injury. 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 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.

Some observers have reported 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 4-(10.3 cm) 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 likely that a 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. As a result, the turtle would have no observable severe injuries (*i.e.*, cracks to the carapace and/or plastron) upon hauling of the dredge. For these reasons, NMFS believes that some sea turtles are captured in scallop dredge gear when the dredge is in the water column.

As described in section 2.1, NMFS has published a final rule that requires modification of scallop dredge gear 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. While that chain mat requirement is in place, NMFS has issued a proposed rule (72 FR 63537, November 9, 2007) that re-proposes the chain mat requirements with some modifications. The effects of the proposed action (the continued authorization of the scallop fishery) includes the effects of the fishery using chain mats[9]. Since 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 turtles 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 will not increase or decrease the number of turtles that will come into contact with dredge gear used in the fishery. The chain mat simply prevents a turtle encountering the gear from entering the dredge bag where it would be at further risk of injury. Both prior to and since the requirement for the use of chain mats, some sea turtles have been observed caught on the dredge, such as wedged between the bale bars of the dredge frame, rather than caught in the dredge bag. Such interactions could result in further injury to the sea turtle if the dredge gear is not handled with care. However, since the turtle would reasonably have been further injured if it had passed into the dredge bag, the use of a chain mat is not expected to result in additional injuries to sea turtles as a result of preventing the turtle from entering the dredge bag. 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 1 to 4 miles per hour. Turtles that are struck by the chain mat portion of the dredge during haulback are not expected to sustain serious injury leading to death or the failure to reproduce, given the slow speed of the vessel during haulback (NMFS 2006b) and given that contact is made in the water column (a fluid environment) rather than against the bottom.

[9] Chain mats are included in this analysis of the authorization of the scallop fishery under the Scallop FMP even though chain mats are not a measure in the FMP itself and the requirement is not implemented under the Magnuson-Stevens Act, for the following reasons: (1) The chain mat requirement emanates from a term and condition in a previous Opinion on the fishery's authorization under the FMP, and (2) FMP's must comply with all applicable law.

Although many turtles caught in or retained upon scallop dredge gear have some type of obvious injury when first observed, regulations require that fishermen return all turtles (regardless of the level of injury) to the water as soon as possible unless they require resuscitation. Based on the descriptions provided by fisheries observers, it seems probable that some of the injured turtles observed captured in commercial scallop dredge gear and that were returned to the water alive would have subsequently died as a result of those injuries. As described in the December 15, 2004 Opinion on the continued implementation of the scallop fishery under the Scallop FMP, NMFS developed and defined three categories for making serious injury determinations for sea turtles captured in scallop dredge gear (Memo from Mary Colligan to Patricia A. Kurkul dated 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 for 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%. For the purposes of the Opinion, NMFS assigned a 0% chance of survival to Category I injuries, and a 100% chance of survival for Category III injuries (NMFS 2004a). To date, there have been no studies that have investigated the survivability of sea turtles following release from a scallop dredge, or their ability to function, and reproduce. Therefore, it is possible that some turtles with Category III injuries will be seriously injured. Likewise, it is also possible that some turtles with Category I injuries will not die, otherwise fail to function, or reproduce. Nevertheless, NMFS believes that they are reasonable measures of what to expect for sea turtles captured by scallop dredge gear, including that some sea turtles captured in scallop dredge gear and released back into the water alive will likely die as a result of those injuries. 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, NMFS determined that the mortality rate was 64% and the survival rate was 36% (NMFS 2004b).

The use of chain mat-modified scallop dredge gear is not expected to reduce the number of sea turtles that come into contact with scallop dredge gear. The gear modification is, however, expected to reduce the likelihood that a turtle will be seriously injured or killed as a result of coming into contact with scallop dredge gear given that the use of chain mats on scallop dredge gear will: (1) Reduce the likelihood that turtles who encounter the gear on the bottom will enter the dredge bag and be at further risk of injury and death, and (2) reduce the likelihood that turtles who encounter the gear in the water column will enter the dredge bag and be subsequently injured or killed. For these reasons, the serious injury and mortality rate of sea turtles interacting with scallop dredge gear since implementation of the chain mat requirement should be less than that calculated for the December 15, 2004 Opinion since fewer turtles will be subject to injuries occurring within the dredge bag or as a result of dumping the dredge bag on deck. However, NMFS cannot quantify the reduction in mortality rate. Therefore, the 64% mortality rate observed in the scallop dredge fishery in 2003 (prior to the use of chain mats) remains the best available information for defining the number of sea turtle takes in scallop dredge gear that are likely to result in death. Applying the mortality and survival rates to the biennial bycatch estimate of 929 loggerhead sea turtles, NMFS anticipates that, biennially, up to 595 loggerhead sea turtles will be killed immediately or suffer serious injuries leading to death or an inability to reproduce as a result of interactions with scallop dredge gear. NMFS also anticipates that, biennially, up to 334 loggerhead sea turtles will survive following interactions with scallop dredge gear.

As described in section 6.2.1, NMFS also anticipates the annual capture of one leatherback sea turtle, one Kemp's ridley sea turtle, and one green sea turtle in scallop dredge gear. NMFS anticipates that Kemp's ridley and green sea turtles will interact with scallop dredge gear in the same manner as loggerhead sea turtles (*i.e.*, both on the bottom and in the water column). Use of the chain mat would not necessarily prevent juvenile Kemp's ridley sea turtles from entering the dredge bag given their small size in relation to the spacing of the chains. Therefore, NMFS believes that any Kemp's ridley interacting with scallop dredge gear may be immediately killed or suffer serious injuries regardless of whether the dredge is modified by use of a chain mat or not. Use of the chain mat may prevent some green sea turtles encountered from entering the dredge bag, depending on the size of the turtle encountered. Therefore, interactions between green sea turtles and scallop dredge gear modified by use of a chain mat could result in less severe injuries than if the chain mat were not used. Given that injuries can still occur if turtles are struck by the dredge gear on or near the bottom, a green sea turtle interaction with scallop dredge gear on or near the bottom is expected to result in immediate death or serious injury for the turtle. In contrast to the hard-shelled species, NMFS believes that all interactions between scallop dredge gear and leatherback sea turtles will occur when the scallop dredge gear is in the water column (*i.e.*, during haulback) given their largely pelagic existence (Rebel 1974; CeTAP 1982; NMFS and USFWS 1992) and their diving pattern when foraging in western North Atlantic continental shelf waters (James *et al.* 2005a; 2005b). Since the leatherback would not be encountering the gear on the bottom, it is not expected to suffer injuries as a result of being struck by the gear on the bottom. Given the large size of leatherback sea turtles, use of a chain mat on scallop dredge gear is expected to prevent leatherback sea turtles from being caught in the dredge bag when the gear is in the water column and, thus, would remove the likelihood of immediate death or serious injury as a result of the interaction.

6.2.4 Anticipated interactions between sea turtles and scallop trawl gear

NEFSC trained observers recorded 8 on-watch loggerhead sea turtle takes during 79.97 days fished in the 2004 and 2005 fishing years combined. Based on the information reported by observers, the NEFSC calculated six estimates using three different methods for the bycatch of loggerhead sea turtles in scallop trawl gear (Murray 2007). NERO has reviewed the information provided in Murray 2007 and determined that the estimate of 134 loggerhead sea turtles (CV=0.45, 95% CI: 37-257) provides the best available information for the take of loggerhead sea turtles in scallop trawl gear for 2004 and 2005 (Memo to the Record from P. Kurkul dated June 28, 2007).

In addition to the estimate of 134 loggerhead sea turtles in scallop trawl gear, the NEFSC has also estimated that 20 loggerhead sea turtles were captured annually from 2000-2004 as a result of bottom otter trawl gear fished in Mid-Atlantic waters where at least 50% of the catch was scallops (Memo from K. Murray, NEFSC to L.Lankshear NERO, August 2007). Therefore, based on these, an estimated 154 loggerhead sea turtles were captured in trawl gear as a result of the scallop fishery during the years 2004 and 2005. Since there are no variables by which to predict turtle bycatch in the scallop fishery, the annual estimated take of 154 loggerhead sea turtles in trawl gear operating in the scallop fishery for 2004 and 2005 provides the best available information on the anticipated take of loggerhead sea turtles as a result of the continued authorization of the scallop fishery.

There have been no observed captures of leatherback sea turtles in scallop trawl gear. Two leatherback sea turtles have been observed captured in trawl gear operating in the action area over the period 1995-2007. A leatherback sea turtle was observed captured in bottom otter trawl gear used in the *Loligo* squid fishery in 2001, and a second leatherback sea turtle was observed captured in bottom otter trawl gear used in the summer flounder fishery in 2007. Both of these takes occurred within the action area of this Opinion. The very low number of observed leatherback captures in trawl gear used in multiple trawl fisheries in the action area suggests that capture of leatherback sea turtles in any mobile gear operating within the action area would be a rare event. However, given the generally low level of observer coverage in the scallop trawl fishery as well as other mobile gear (trawl) fisheries in the action area, it is likely that some interactions with leatherback sea turtles have occurred but were not observed or reported. Nevertheless, given effort in the fishery as a whole, and the seasonal overlap in distribution of this species with operation of scallop trawl gear, leatherback sea turtles are likely to be captured in scallop trawl gear. A prediction of one take results from using an average of the number of takes for the 1995-2007 time period since a "part" of a turtle cannot be taken.

Similarly, there have been no observed takes of Kemp's ridley or green sea turtles in the scallop trawl fishery. NMFS believes, however, that takes of Kemp's ridley, and green sea turtles may occur given that the distribution of these species overlaps with operation of the scallop trawl fishery. With respect to other mobile gear operating in the area, specifically dredge gear, there have been only two observed take of a Kemp's ridley sea turtle and one observed take of a green sea turtle during the period 1996-2007. As described above, this suggests that the capture of Kemp's ridley and green sea turtles in any mobile gear operating within the action area, including scallop trawl gear, would be a rare event. However, given the low level of observer coverage in the scallop trawl fishery as well as other mobile gear fisheries in the action area it is likely that some interactions have occurred but were not observed or reported. Therefore, based on the average of the number of the takes per year for each of these species, one take of a Kemp's ridley sea turtle and one take of a green sea turtle in scallop trawl gear is anticipated to occur annually as a result of the continued authorization of the scallop fishery.

6.2.5 Age composition and mortality for sea turtles caught in scallop trawl gear

Turtles captured in scallop trawl gear are expected to be of the same age classes as turtles captured in scallop dredge gear since turtles captured in scallop trawl gear are within the same size range as turtles captured in scallop dredge gear. As described above, NMFS anticipates the capture of benthic immature as well as sexually mature loggerhead, leatherback, and green sea turtles in scallop dredge gear as a result of the continued authorization of the scallop fishery. NMFS anticipates the capture of immature Kemp's ridley sea turtles in scallop dredge gear as a result of the continued authorization of the scallop fishery.

As described above, 100% of the non-decomposed turtles reported by observers in scallop trawl gear were returned to the water alive. Contact injuries such as those observed for turtles captured in or retained upon scallop dredge gear have not been observed in the scallop trawl fishery. Two of the turtles were, however, resuscitated in accordance with the regulations prior to be released into the water. Epperly *et al.* (2002) and Sasso and Epperly (2006) found that, in general, tows of short duration have little effect on the mortality of sea turtles caught in the trawl gear. Intermediate tow times result in a rapid escalation to mortality, and eventually reach a plateau of high mortality, but will not equal 100 percent as a turtle caught within the last hour of a long tow will likely survive (Epperly *et al.* 2002; Sasso and Epperly 2006). Tow times for the scallop trawls observed to capture turtles in 2005 and 2006 were 1-2 hours in duration. 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), turtles may die as a result of capture and forced submergence in trawl gear used in the scallop fishery.

NMFS NER has not developed any serious injury criteria for turtles captured in scallop trawl gear. Based on the criteria developed for turtles captured in scallop dredge gear, a turtle requiring resuscitation but with no other apparent contact injuries was considered to have a 50% chance of survival (Category II). Given that the injury, regardless of the gear type fished, appears to be as a result of forced submergence, NMFS NER considers any turtle captured in scallop trawl gear and requiring resuscitation to also have a 50% likelihood of survival. Thus, the overall mortality rate for sea turtles captured in scallop trawl gear would be 12.5%, based on observer data collected for 2004-2005. Applying this to the estimated take of 154 loggerhead sea turtles per year in trawl gear used in the scallop fishery results in an anticipated lethal take of 19.25 loggerhead sea turtles annually as a result of take in scallop trawl gear. Since a part of a turtle cannot be taken, this number is rounded up to 20.

6.3 Summary of anticipated incidental take of sea turtles in the scallop fishery

NMFS anticipates the annual take of up to 929 loggerhead sea turtles biennially as a result of the continued operation of the scallop dredge fishery. The use of chain mats on scallop dredge gear is expected to reduce the number of lethal takes (including serious injuries) for loggerhead, green, and leatherback sea turtles that interact with scallop dredge gear. However, since the reduction in lethal takes as a result of the use of chain mats cannot be quantified, NMFS is using the mortality rates for loggerhead sea turtles captured in the scallop dredge fishery in 2003 – prior to the use of chain mats. Therefore, NMFS anticipates up to, but most likely less than, 595 of the anticipated 929 loggerhead sea turtles captured biennially in the scallop fishery will suffer injuries to the extent that they will die, cease to function in other respects (eventually leading to death) or fail to reproduce.

Loggerhead turtles captured by scallop dredge and trawl gear are expected to include benthic immature and sexually mature turtles. Loggerhead sea turtles originating from nesting beaches of each of the five recognized nesting groups in the western North Atlantic are captured in gear used in the scallop fishery (Haas *et al.* in review). The majority of the turtles captured originate from nesting beaches of the south Florida nesting group (Haas *et al.* in review).

Continued operation of the scallop fishery is also expected to result in the annual capture of one leatherback, Kemp's ridley, and green sea turtle in scallop dredge gear. Takings of Kemp's ridley and green sea turtles in scallop dredge gear are expected to be either lethal or non-lethal. Takings of leatherback sea turtles in scallop dredge gear are expected to be non-lethal given the use of chain mats and the location of leatherbacks within the water column.

Scallop trawl gear is expected to result in the annual capture of 154 loggerhead sea turtles of which up to 20 are expected to be lethal takes. Scallop trawl gear is also expected to result in the annual take of one leatherback sea turtle, one Kemp's ridley sea turtle, and one green sea turtle annually. These takes may be either lethal or non-lethal.

7.0 INTEGRATION AND SYNTHESIS OF EFFECTS

The *Status of Affected Species*, *Environmental Baseline*, and *Cumulative Effects* sections of this Opinion discuss the natural and human-related phenomena that caused listed species to become threatened or endangered and may continue to place those sea turtle species at high risk of extinction. "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 CFR402.02). The present section of this Opinion applies that definition by examining the effects of the continued authorization of the scallop fishery under the FMP (described in Section 6.2) in the context of information presented in the status of the species, environmental baseline, and cumulative effects sections to determine: (a) If those effects due to the fishery would be expected to reduce the reproduction, numbers, or distribution of threatened or endangered species in the action area, and (b) if any reduction in the reproduction, numbers, or distribution of listed species causes an appreciable reduction in the species' likelihood of surviving and recovering in the wild. The meaning of "reduce appreciably," as it appears in the regulatory definition of "jeopardize the continued existence," has not been defined by the Section 7 consultation regulations. When the Section 7 consultation terms, "jeopardize the continued existence of" and "destruction or adverse modification of critical habitat," were first defined in 1978 (43 FR 870, January 4, 1978), both definitions used the word "appreciably." In 1986, the Services revised the Section 7 regulations, maintaining the word "appreciably" in the definitions of both "jeopardize the continued existence of" and "destruction or adverse modification of critical habitat," yet still not specifying what it means (see 51 FR 19926, June 3, 1986).

While NMFS has not expressly addressed in the consultation regulations the meaning of the jeopardy definition's term "reduce appreciably," its Consultation Handbook addressed the word "appreciably" in the analogous context of the regulatory definition of "destruction or adverse modification of critical habitat."[10] "Destruction or adverse modification" was defined as "a direct or indirect alteration that appreciably diminishes the value of critical habitat for both the survival and recovery of a listed species" (51 FR at 19958). The Consultation Handbook interprets the term "appreciably diminish the value of" to mean "to considerably reduce the capability of designated or proposed critical habitat to satisfy the requirements essential to both the survival and recovery of a listed species" (FWS and NMFS, 1998; emphasis added). A

[10] While the regulatory definition of destruction or adverse modification of critical habitat has been struck down and NMFS no longer uses it, the reasons for its rejection do not pertain to the meaning of "appreciably."

common dictionary definition of "appreciable" indicates it means capable of being perceived or measured. "To measure" something includes not only quantifying dimensions, capacity or amount, but also conducting qualitative evaluations of extent or degree. The latter aspect is consistent with the meaning of "appreciable" as including "capable of being perceived."

Given the lack of a regulatory definition of, or agency statement on, the meaning of "appreciably reduce" within the definition of "jeopardize the continued existence," for purposes of this Opinion, NMFS is taking the following approach to applying the term "appreciably reduce" the likelihood of survival and recovery of a listed species in the wild. It attempts to meld together the different clues as to the meaning of "appreciably reduce" in light of the quantitative or qualitative nature of the jeopardy analyses for the different species affected. For the analysis provided below on loggerhead sea turtles, NMFS is able to use a quantitative approach for determining whether the proposed action is likely to jeopardize the continued existence of loggerhead sea turtles. As described in more detail below, for three measurements of the likelihood of survival, it reviews whether there is a difference in those measures with the effects of the fishery factored in compared to the measures without the effects of the fishery factored in. If there is a difference in any of the measures, NMFS will determine whether that difference translates into a considerable or material reduction in the likelihood of survival and recovery. The term "appreciably reduce the likelihood of survival and recovery" is also interpreted to accommodate the use of tests of statistical significance, which are designed to deal with uncertainty as to whether an observed difference in a measure is a real reduction in the likelihood of survival and recovery. This interpretation is consistent with ESA's requirement that NMFS use the best available scientific information, which in this case includes the results of a population viability analysis relying on tests of statistical significance commonly used in scientific analyses.

For leatherback, Kemp's ridley, and green sea turtles, quantitative analyses of the effects of the scallop fishery on the likelihood of survival and recovery were not performed due to data deficiencies and the likely inability to detect effects on the likelihood of survival and recovery of the loss of one or two animals from each species. For these species, NMFS is not able to measure in a quantitative manner the effects on the likelihood of survival and recovery, but is only able to measure them qualitatively by evaluating the effects of the fishery on the likelihood of survival and recovery using information on trends and the number of each sea turtle species killed by the scallop fishery. Through that process, NMFS judged whether the loss of a specific number of individuals is likely to have a considerable or material effect on the likelihood of survival and recovery of each species.

7.1 Integration and Synthesis of Effects on Sea Turtles

This Opinion has identified in Section 6 (*Effects of the Proposed Action*) that the proposed action-- continued authorization of the fishery under the Scallop FMP will directly affect loggerhead, leatherback, Kemp's ridley, and green sea turtles as a result of interactions (including capture) with scallop dredge and scallop trawl gear. No other direct or indirect effects to ESA-listed species are expected as a result of the activity. The following discussion in Sections 7.1.1 through 7.1.4 below provide NMFS' determinations of whether there is a reasonable expectation that loggerhead, leatherback, Kemp's ridley, and green sea turtles 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.

7.1.1 Loggerhead Sea Turtle

Based on information provided in this Opinion, NMFS anticipates the taking of up to 929 loggerhead sea turtles biennially in scallop dredge gear as a result of the continued authorization of the scallop fishery. The use of chain mats on scallop dredge gear is expected to reduce the number of lethal takes (including serious injuries) for loggerhead sea turtles that interact with scallop dredge gear. However, since the reduction in lethal takes as a result of the use of chain mats cannot be quantified, NMFS is using the mortality rates for loggerhead sea turtles captured in the scallop dredge fishery in 2003 – prior to the use of chain mats. Therefore, NMFS anticipates up to, but most likely less than, 595 of the anticipated 929 loggerhead sea turtles will suffer injuries to the extent that they will die, cease to function in other respects (eventually leading to death) or fail to reproduce. The remaining 334 loggerhead turtles that are taken by scallop dredge gear biennially and released alive are not expected to suffer any ill effects as a result of capture and there should be no negative impact to the species from the capture of these 334 turtles. Loggerhead sea turtles "taken" in scallop dredge gear are those that are captured in or retained upon scallop dredge gear as well as those that have physical contact with scallop dredge gear fitted with a chain mat, with or without subsequent capture in or retention upon the dredge.

NMFS anticipates the take of up to 154 loggerhead sea turtles annually in scallop trawl gear as a result of the continued authorization of the scallop fishery. Up to 20 of the 154 turtles are expected to be immediately killed or seriously injured to the extent that they will die, cease to function on other respects (eventually leading to death) or fail to reproduce as a result. Loggerhead sea turtles "taken" in scallop trawl gear (includes gear that is identified as bottom otter trawl and fished in the scallop fishery) are those that are captured in the gear.

In total, therefore, the continued authorization of the scallop fishery is expected to result in the death and serious injury of 635 loggerhead sea turtles from the Atlantic every two years (595 biennially in dredge gear and 20 annually in trawl gear). The loss of these loggerhead sea turtles from the Atlantic will reduce the number of loggerhead sea turtles as compared to the number of loggerhead sea turtles that would have been present in the absence of the proposed action assuming all other variables remained the same. Similarly, the loss of female loggerhead sea turtles as a result of the proposed action are expected to reduce the reproduction of loggerheads in the Atlantic compared to the reproductive output of Atlantic loggerheads in the absence of the proposed action. As described in section 5.1, NMFS considers the trend for loggerheads, as a species, to be declining. Nest counts in many areas of the species range reflect decreased nesting. These include nest sites for loggerhead sea turtles in the western North Atlantic. A trend analysis of the nesting data collected for Florida's INBS program showed a decrease in nesting of 22.3% in the annual nest density of surveyed shoreline over the 17-year period and a 39.5% decline since 1998 (letter to NMFS from the Director, Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission, October 25, 2006; Meylan *et al.* 2006). Data collected in Florida for the 2007 loggerhead nesting season reveals that the decline in nest numbers has continued, with even fewer nests counted in 2007 in comparison to any previous year of the period, 1989-2007 (Fish and Wildlife Research Institute, Florida Fish and Wildlife Conservation Commission web posting November 2007). Standardized ground surveys of 11 North Carolina, South Carolina, and Georgia nesting beaches showed a significant declining trend of 1.9% annually in loggerhead nesting from 1983-2005 (NMFS and USFWS 2007a). Aerial surveys conducted by the South Carolina Department of Natural Resources showed a 3.1% annual decline in nesting since 1980 (Dodd 2003; NMFS and USFWS 2007a). Nesting for the Yucatán nesting group is characterized as having declined since 2001 (NMFS and USFWS 2007a) while no trend is detectable for the Dry Tortugas nesting group (NMFS and USFWS 2007a).

As described in section 3.1.1, adult female loggerhead sea turtles exhibit natal homing which has resulted in detectable differences in mitochondrial DNA. Based on these differences, five loggerhead nesting groups have been identified as occurring in the western Atlantic (Márquez 1990; TEWG 2000; SEFSC 2001). Genetic analysis of samples collected from sea turtles captured in the scallop fishery demonstrate that turtles originating from beaches of each of the five western Atlantic nesting groups occur within the action area and are caught in scallop fishing gear. The genetic results also suggest the presence of turtles originating from Greece, Turkey, and Brazil (Haas *et al.* in review). These results are similar to Bass *et al.* (2004) which examined the origin of juvenile loggerheads captured on foraging grounds in the Pamlico-Albemarle Estuarine complex (inshore of the southern limit of the action area for this Opinion). Using the results of Bass *et al.* (2004) and Haas *et al.* (in review), and the anticipated biennial lethal take of loggerhead sea turtles in the scallop fishery (dredge and trawl gear combined), it is found that of the 635 loggerhead sea turtles expected to be removed from the Atlantic every two years: (1) 399-563 would originate from nesting beaches of the south Florida nesting group, (2) 19-76 would originate from nesting beaches of the northern nesting group, (3) 7-69 would originate from nesting beaches of the Florida Panhandle nesting group, (4) 0-45 would originate from nesting beaches of the Dry Tortugas nesting group, (5) 26-38 would originate from nesting beaches of the Mexico nesting group, (6) 13-26 would originate from nesting beaches in Greece, and (7) 0-7 would originate from nesting beaches in Brazil and Turkey. The only estimate of size that we have for each nesting group is the estimated number of nesting females in that group. The total numbers of turtles in the Atlantic that originate from each of the nesting groups (adult males, immature males, immature females) is unknown. Therefore, it is impossible to determine whether the anticipated take of loggerhead sea turtles in the scallop fishery, apportioned by nesting group origin, would reduce the numbers or reproduction of any one group to an extent that it would affect the continued existence of the nesting group and, therefore, distribution of the species. Since there are estimates of the number of nesting females for each of the nesting groups, NMFS considered whether to estimate the number of females anticipated to be taken from each nesting group by further apportioning the takes to account for the expected sex ratio of Atlantic loggerheads, and then using those numbers to assess what effect the continued authorization of the scallop fishery would have on the number of females from each nesting group. NMFS rejected this approach for several reasons. First, while the number of nests serves as a proxy for the size of the adult nesting female population, converting nests counts to females is confounded by several issues such as the variability in number of nests per female per year, variability in remigration interval, and, as the ability to nest is resource-dependent, the effect of habitat changes and the availability of food resources (Loggerhead

TEWG 2007). Even if NMFS were to use the values for number of nests per female per year, and for remigration interval that are commonly reported in the scientific literature for loggerheads (4.1 nests per female per year and 2.5 years remigration interval), we would need to further determine what effect the removal of that number of females would mean to the nesting group, subsequently what the effect to the nesting group would mean for loggerheads in the Atlantic, what the effect to loggerheads in the Atlantic would have on the species and, finally, whether the effect to the species would appreciably reduce the likelihood of survival and recovery of the species. There is currently insufficient information by which to make such determinations.

For the September 18, 2006, Opinion on the continued authorization of the scallop fishery, NMFS used the results of the NMFS Southeast Fisheries Science Center (SEFSC) modeling (NMFS SEFSC 2001) which examined the anticipated effect to the growth rate of the northern nesting group as a result of the change in the TED requirements for the U.S. southeast Atlantic and Gulf of Mexico shrimp fisheries, and increases in loggerhead pelagic immature survival. The modeling results showed that the trend for the northern nesting group would increase as a result of these changes. Since the other western Atlantic nesting groups were believed to have a more positive trend than the northern nesting group, NMFS SEFSC (2001) concluded that the changes to the TED requirements and to pelagic immature survival would improve the trend of all of the nesting groups, and to a greater degree than that predicted by the modeling for the northern nesting group. Based on these results and knowledge of the scallop fishery, including its long term operation in the Mid-Atlantic, NMFS concluded for the September 18, 2006, Opinion that the continued authorization of the scallop fishery would not jeopardize the continued existence of loggerhead sea turtles since the takes from the scallop fishery were part of the starting growth rates used in the Southeast Fisheries Science Center modeling, and takes in the fishery as a result of the continued authorization of the scallop fishery were not expected to greater than what was subsumed by the starting growth rates of NMFS SEFSC 2001. Additional information on nesting trends for the western Atlantic nesting groups has become available since completion of the September 18, 2006, Opinion. As described in section 3.1.1, nest counts for four of the five western Atlantic loggerhead nesting groups show a declining trend. No trend is apparent for the fifth nesting group. The Loggerhead TEWG has examined the nesting data and concluded that the declines in nest counts are real (Loggerhead TEWG 2007). The cause(s) of the decline in nesting, and the impact to the loggerhead nesting groups is unknown (Loggerhead 2007). In light of this information, NMFS has chosen not to use the NMFS SEFSC (2001) results for this Opinion since it is unknown whether it is correct to assume that the northern nesting group has the least favorable status compared to the other western Atlantic nesting groups.

Another option considered by NMFS to determine whether the continued authorization of the scallop fishery would reduce appreciably the likelihood of survival and recovery for the species was to use a population viability analysis (PVA) based on one that had been used to assess the effects of the Hawaii deep-set pelagic longline fishery on ESA-listed sea turtles, including loggerheads, in the Pacific (NMFS 2005; Snover 2005). NMFS chose to use this quantitative approach. The PVA is for adult females, only, as was the case for the PVA used to assess the effect of the continued authorization of the Hawaii deep-set pelagic longline fishery on ESA-listed turtles in the Pacific. A PVA for the whole Atlantic loggerhead population cannot be constructed since there are no estimates of the number of mature males, immature males, and immature females in the population, and the age structure of the population is unknown.

In using the PVA for making the jeopardy determination for this Opinion, NMFS has:

- used 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;
- uses three measures to assess the likelihood of quasi-extinction which are the probability of quasi-extinction (at 25, 50, 75, and 100 years), the median time to quasi-extinction, and the number of simulations with quasi-extinction probabilities at 25, 50, 75, or 100 years greater than 0.05; and,
- uses statistical tests to inform whether any detected differences in the three measures for the comparison of the baseline to the baseline minus effects of the fishery are real.

The PVA was conducted for the adult female portion of loggerheads nesting in the western Atlantic Ocean. NMFS considered running the PVA at the nesting group level for the effects analysis, but did not pursue that option for two major reasons. First, sufficient data are not available to develop a PVA model for each of the nesting groups. Second, it is unclear how PVA outputs at a nesting group can be reconciled to assess the effects of the proposed action on the western Atlantic Ocean stock or the species overall. This is problematic because the jeopardy determination must ultimately be made at the species level.

Sufficient data are available to conduct a PVA of the northern nesting group and the south Florida nesting groups. It is unlikely that the results of a PVA on these two separate nesting groups would differ significantly from the results of the PVA on adult female loggerheads of the western Atlantic Ocean taken as a whole. This is for two reasons. First, the south Florida nesting group already drives the results of the western Atlantic Ocean analysis; index sites there represented 95% of the 2005 nests counted. As such, the viability of the south Florida nesting group will be very similar to that predicted for the overall western Atlantic Ocean stock of loggerheads. Second, the much smaller northern nesting group has shown considerable inter-annual variability in nest counts. Whether this is due to true environmental variability or process error is unknown. This high level of variability blurs our ability to detect real effects of the fishery, because high variance means that only large effects can be statistically significant. While it is likely that a PVA of the northern nesting group would show a differences between the projected extinction risk with and without the takes from the scallop fishery (as is the case with the PVA on adult female loggerheads nesting in the western Atlantic Ocean; see below), it is likely that these two projections would fall within the confidence intervals of each other. Therefore, these differences would not be statistically significant. In other words, given available data, we are more likely to detect a real effect of the fishery on quasi-extinction of adult female loggerhead sea turtles in the Atlantic by conducting the PVA at the stock level (western North Atlantic) than if the PVA was conducted on the much smaller northern nesting group, alone, because conducting the PVA at the stock level reduces the variability thus improving the ability to detect real effects of the fishery.

The PVA does not address loggerheads that nest in Greece, Turkey and Brazil since the PVA was run for adult female loggerheads in the western Atlantic, only. Data to conduct a PVA for adult female loggerheads in the Atlantic as a whole are not available. However, given that the south Florida and northern nesting groups are the first and second largest of the loggerhead nesting groups in the Atlantic, respectively, the result of a PVA for adult female loggerheads in the Atlantic would be expected to be driven by the western Atlantic nesting groups even if data to conduct a PVA for the Atlantic as a whole were available.

The details of the PVA addressing the effects of the continued authorization of the scallop fishery on Atlantic loggerheads are provided in Appendix 3. In short, the PVA established a baseline 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 U.S. based on an extrapolation from nest counts (Merrick and Haas 2008). The rate of change was then adjusted by adding back the fisheries take (converted to adult female equivalents), and rerunning the PVA. The results of these two analyses were then compared. Values for inputs were used throughout such that the PVA would be more, rather than less, likely to show a significant difference in quasi-extinction between the baseline and the baseline adjusted by adding back in the fisheries take.

Using this approach, it was determined that both the baseline and adjusted baseline (adding back the fisheries take) had quasi-extinction probabilities of zero (0) at 25, 50, and 75 years, and a probability of 1% at 100 years. Median times to quasi-extinction were similar (207 years versus 240 years). Over 1,000 iterations of the model, the number of iterations with quasi-extinction probabilities at 100 years greater than 0.05 were higher for the baseline compared to the adjusted baseline (258 and 178, respectively) and were significantly different (Chi square = 18.3, P = 0.00) (Merrick and Haas 2008).

The results suggest that the continued authorization of the scallop fishery, resulting in mortalities of loggerhead sea turtles, will not have an appreciable effect on the number of adult female loggerhead sea turtles in the western Atlantic over the next 100 years. While a statistically significant difference is 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, there is no difference in the probability of quasi-extinction at 100 years is the same (1%) under both baseline conditions, and when the baseline is adjusted by removing takes as a result of the scallop fishery. In addition, while median times to quasi-extinction differed between the baseline and the adjusted baseline, the difference was small and median times for both were greater than 200 years. Therefore, based on the median times to quasi-extinction, loggerhead sea turtles in the western Atlantic will not go extinct within the next 100 years regardless of the continued authorization of the scallop fishery. Based on these results, the continued authorization of the scallop fishery is not expected to appreciably reduce the survival of loggerhead sea turtles in the Atlantic. The scallop fishery has no direct or indirect effects on loggerhead sea turtles that occur outside of the Atlantic. Therefore, since the continued authorization of the scallop fishery is not expected to appreciably reduce the survival of loggerhead sea turtles in the Atlantic, the proposed action will not appreciably reduce the survival of the species.

The 5-year status review for the species reviewed the recovery criteria provided with the 1991 recovery plan for loggerhead sea turtles in the Atlantic, and the progress made in meeting each objective (NMFS and USFWS 2007a). These are that the southeastern United States population of the loggerhead can be considered for delisting if, over a period of 25 years, the following conditions are met: (1) The adult female population in Florida is increasing and in North Carolina, South Carolina, and Georgia it has returned to pre-listing levels; (2) at least 25% (560 km) of all available nesting beaches (2240 km) is in public ownership, is distributed over the entire nesting range and encompasses greater than 50 percent of the nesting activity; (3) all priority one tasks have been successfully implemented (address a multitude of measures in areas of nesting habitat, marine habitat, lighting, sea turtle research to better elucidate life history, and anthropogenic effects from commercial and recreational fisheries). As described above and elsewhere in this Opinion, the continued authorization of the scallop fishery is expected to kill and injure loggerhead sea turtles as a result of physical contact between the turtles and scallop fishing gear that includes capture of the turtles in the gear with the exception of scallop dredge gear fitted with chain mats. No other direct or indirect effects to loggerhead sea turtles are expected as a result of the proposed action. The continued authorization of the scallop fishery will not affect ownership of nesting habitat, the ability to put nesting habitat into public ownership, nor will it affect the protection of nesting beaches and the marine environment or compromise the ability of researchers to conduct scientific studies. Therefore, the continued authorization of the scallop fishery will have no effect on recovery criteria #2 and #3.

The lethal take of up to 635 loggerhead sea turtles from the Atlantic every two years (595 biennially in dredge gear and 20 annually in trawl gear) will reduce the number of loggerhead sea turtles as compared to the number of loggerhead sea turtles that would have been present in the absence of the proposed action (assuming all other variables remained the same). The loss of female loggerhead sea turtles as a result of the proposed action are expected to reduce the reproduction of loggerheads in the Atlantic compared to the reproductive output of Atlantic loggerheads in the absence of the proposed action. These are relevant to recovery criteria #1. Nesting data demonstrate declines in the number of nests laid for nearly all of the western North Atlantic nesting groups. The reasons for the declines are unknown as is whether the declines in nest counts reflect a decline in the number of adult females or a decline in the population or stock as a whole (Loggerhead TEWG 2007). Regardless of the cause(s) of the decline, a reduction in nests means that there are fewer offspring produced and, therefore, potentially fewer turtles that will mature and reproduce in the future. Depending on the extent of the decline, recovery of the species may be delayed or prevented. With respect to the effects of the scallop fishery on the projected baseline for female loggerhead sea turtles, the PVA as described above demonstrated that the continued authorization of the scallop fishery will not appreciably reduce the number of adult females in the western Atlantic compared to the numbers of adult females that would be present in the absence of the proposed action, even though the input values selected for the PVA (*e.g.*, number of nests per female, sex ratio, quasi-extinction level of 250 females) were chosen to maximize the chance that the PVA would show an effect from the fishery. Therefore, the continued authorization of the scallop will not appreciably reduce the likelihood of recovery for loggerheads in the Atlantic. Since Atlantic loggereheads do not contribute to reproduction for loggerheads outside of the Atlantic (Bowen and Karl 2007), the continued authorization of the scallop fishery is not expected to appreciably reduce the likelihood of recovery for the species.

7.1.2 Leatherback Sea Turtle

There have been no documented captures of leatherback sea turtles in scallop trawl or scallop dredge gear. However, reporting of sea turtle takes in the fishery on VTR's is non-existent, and observer coverage of the trawl and dredge components of the fishery was very low prior to 2004 and 2003, respectively. Takes of leatherback sea turtles in the fishery are reasonably likely to occur given: (1) That the distribution of leatherbacks overlaps with operation of scallop trawl and dredge gear, (2) two leatherback sea turtles were observed captured in bottom otter trawl gear – a gear type very similar to scallop trawl gear-- used in the *Loligo* squid fishery and the summer flounder fishery operating in Mid-Atlantic waters where the scallop fishery (trawl and dredge components) also operates, and (3) the mouth/opening of a scallop dredge bag is large enough to incidentally catch a leatherback when the gear is hauled through the water column.

Based on results from the U.S. south Atlantic and Gulf of Mexico shrimp trawl fisheries (Epperly *et al.* 2002; Sasso and Epperly 2006), any capture of a leatherback sea turtle in scallop trawl gear could result in death due to forced submergence, given that there are no regulatory controls on tow-times in the scallop fishery and some trawl and dredge tows that have been observed to capture loggerhead sea turtles have exceeded one hour in duration (NEFSC, FSB database). However, the use of chain mats is expected to prevent the capture of leatherback sea turtles in scallop dredge gear since the chains form a pattern of openings across the mouth of the dredge bag that are too small for a leatherback to pass through. Since the chain mat will prevent leatherback sea turtles from entering the dredge bag, leatherback sea turtles are not expected 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. Given that leatherbacks forage within the water column rather than on the bottom, interactions between leatherback sea turtles and scallop dredge gear are expected to occur when the gear is traveling through the water column versus on the bottom. Since the dredge gear is hauled through the water column at a relatively slow speed and contact between the turtle and the gear would occur in a fluid environment versus on the bottom, leatherbacks occurring within the water column are not expected to be injured or killed as a result of physical contact with a chain-mat equipped scallop dredge.

In summary, based on the observed capture of two leatherback sea turtles in bottom otter trawl gear used in the *Loligo* squid and summer flounder fisheries within Mid-Atlantic waters of the action area, the continued authorization of the scallop fishery (trawl and dredge gear components combined) is anticipated to result in the annual taking of up two leatherback sea turtles. One of these is anticipated to result in lethal take given the risk of forced submergence leading to death as a result of capture in scallop trawl gear. The second is expected to be non-lethal since leatherback sea turtles are expected to encounter scallop dredge gear in the water column, and the use of chain mats will prevent the turtle from entering the dredge bag.

The lethal removal of one leatherback sea turtle annually, whether male or female or immature or mature, would be expected to reduce the number of Atlantic leatherback sea turtles as compared to the number of leatherback sea turtles in the Atlantic that would have been present in the absence of the proposed action assuming all other variables remained the same. The loss of a female leatherback sea turtle, annually, would be expected to reduce the reproduction of Atlantic leatherback sea turtles as compared to the reproductive output of leatherback sea turtles in the Atlantic in the absence of the proposed action. As described in Section 5.1, NMFS considers the trend for leatherbacks, as a species, to be declining. Nevertheless, the lethal removal of one leatherback sea turtle annually from the Atlantic as a result of the continued authorization of the scallop fishery will not appreciably reduce the likelihood of survival for the species for the following reasons. Unlike leatherbacks in the Pacific, the nesting trend (in terms of number of nests laid) for leatherbacks in the Atlantic is stable or increasing for nearly all Atlantic leatherback nesting sites. The 2007 Leatherback TEWG report identified seven leatherback populations or groups of populations in the Atlantic: Florida, North Caribbean, Western Caribbean, Southern Caribbean, West Africa, South Africa, and Brazil. The Leatherback TEWG concluded that there was an increasing or stable trend in nesting for all of these with the exception of the Western Caribbean and West Africa. For example, the Florida Statewide Nesting Beach Survey Program has documented an increase in leatherback nesting numbers in that state from 98 in 1988 to between 800 and 900 nests in the early 2000s (NMFS and USFWS 2007b). In 2001, the number of nests for Suriname and French Guiana, the largest known nesting areas for leatherbacks worldwide, was 60,000 (Hilterman and Goverse 2004). This is one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). 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 *Environmental Baseline* -- which includes the effects of the past operation of the Atlantic sea scallop fishery--, *Cumulative Effects*, and the *Status of the Species* (for those activities that occur outside of the action area of this Opinion). 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.

As described in the *Status of the Species* and *Environmental Baseline*, action has been taken to reduce anthropogenic effects to Atlantic leatherbacks. These include regulatory measures to reduce the number and severity of leatherback interactions with the two leading known causes of leatherback fishing mortality in the Atlantic - the U.S. Atlantic longline fisheries (measures first implemented in 2000 and subsequently revised) , and measures implemented in 2002 for the U.S. south Atlantic and Gulf of Mexico shrimp fisheries. 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 the regulatory measures are relatively recent, it is unlikely that current nesting trends reflect the benefit of these actions to Atlantic leatherbacks. Therefore, the current nesting trends for Atlantic leatherbacks are likely to improve as a result of regulatory action taken for the U.S. Atlantic longline fisheries and the U.S. south Atlantic and Gulf of Mexico shrimp fisheries. There are no new known sources of injury or mortality for leatherback sea turtles in the Atlantic.

Based on the information provided above, the loss of one leatherback sea turtle annually in the Atlantic as a result of the continued authorization of the scallop fishery will not appreciably reduce the likelihood of survival for leatherbacks in the Atlantic given the increased and stable nesting trend at the 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 direct or indirect effects on leatherback sea turtles that occur outside of the Atlantic. Therefore, since the continued authorization of the scallop fishery will not appreciably reduce the likelihood of survival for leatherbacks in the Atlantic, the proposed action will not appreciably reduce the likelihood of survival of the species.

The 5-year status review for the species reviewed the recovery criteria provided with the 1992 recovery plan for leatherbacks in the Atlantic, and the progress made in meeting each objective (NMFS and USFWS 2007b). These are: (1) 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.V.I., and along the east coast of Florida; (2) nesting habitat encompassing at least 75 percent of nesting activity in U.S.V.I., Puerto Rico, and Florida is in public ownership; (3) all priority one tasks have been implemented (address a multitude of measures in areas of nesting habitat protection, scientific studies, marine debris, oil and gas exploration, amongst others) (NMFS and USFWS 1992). As described in this Opinion, the continued authorization of the scallop fishery is expected to kill one Atlantic leatherback sea turtle annually, and non-lethally interact with one Atlantic leatherback sea turtle, annually. No other direct or indirect effects to leatherback sea turtles are expected as a result of the proposed action. The continued authorization of the scallop fishery will not affect ownership of nesting habitat, nor will it affect the protection of nesting beaches and the marine environment or compromise the ability of researchers to conduct scientific studies. The continued authorization of the scallop fishery will not affect ownership of nesting habitat, nor will it affect the protection of nesting beaches and the marine environment or compromise the ability of researchers to conduct scientific studies. Therefore, the continued authorization of the scallop fishery will have no effect on recovery criteria #2 and #3.

The lethal take of one Atlantic leatherback sea turtle, annually, as a result of the proposed action is expected to reduce the number of leatherbacks in the Atlantic compared to the number that would have been present in the absence of the proposed action, and will, similarly, reduce leatherback reproduction in the Atlantic as a result of the capture and killing if the leatherback is a female. These conclusions are relevant to recovery criteria #1 of the 1992 recovery plan for leatherbacks in the Atlantic. As described in the 5-year status review, the number of nests counted in Puerto Rico increased from 9 in 1978 to a minimum of 469-882 nests recorded each year from 2000-2005. Based on the nesting numbers, the annual female population growth rate was positive for the 28 year time period from 1978-2005. In the U.S.V.I., on the island of St. Croix, leatherback nesting increased from a low of 143 in 1990 to a high of 1,008 in 2001. Based on the nesting numbers, the annual female population growth rate was positive for the 19 year time period from 1986-2004. In Florida, nests have increased from 98 nests in 1989 to 800-900 nests per season in the early 2000s (NMFS and USFWS 2007b). Based on the nesting numbers, the annual female population growth rate was positive for the 18 year time period from 1989-2006 (NMFS and USFWS 2007b). The annual loss of one leatherback sea turtle, together with an increase in nesting, is not expected to materially effect the positive growth rate in the female population of leatherback sea turtles nesting in Puerto Rico, St. Croix, and Florida. Therefore, the continued authorization of the scallop fishery will not appreciably reduce the likelihood of recovery for leatherback sea turtles in the Atlantic. Since the scallop fishery has no direct or indirect effects on leatherback sea turtles that occur outside of the Atlantic, the continued authorization of the scallop fishery will not appreciably reduce the likelihood of recovery for the species.

7.1.3 Kemp's Ridley Sea Turtle

There have been no known takes of Kemp's ridley sea turtles in scallop trawl gear. The distribution of Kemp's ridleys overlaps seasonally with the use of scallop trawl gear, and Kemp's ridley sea turtles are captured in other types of trawl gear. Based on observer data, the capture of Kemp's ridley sea turtles in any mobile gear operating within the action area, including scallop trawl gear, would be a rare event. However, given the low level of observer coverage in the scallop trawl fishery as well as other mobile gear fisheries in the action area it is likely that some interactions have occurred but were not observed or reported. Therefore, based on the average of the number of the takes per year for the period 1996-2007, one take of a Kemp's ridley sea turtle in scallop trawl gear is anticipated to occur annually as a result of the continued authorization of the scallop fishery. Based on results from the U.S. south Atlantic and Gulf of Mexico shrimp trawl fisheries (Epperly *et al.* 2002; Sasso and Epperly 2006), any capture of a Kemp's ridley sea turtle in scallop trawl gear could result in death due to forced submergence, given that there are no regulatory controls on tow-times in the scallop trawl fishery and some trawl tows that have been observed to take loggerhead sea turtles have exceeded one hour in duration (NEFSC, Fisheries Sampling Branch database).

There have been two confirmed captures of Kemp's ridley sea turtles in scallop dredge gear. One of these was killed as a result of the interaction. Both of the turtles were likely immatures based on their size. 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 Kemp's ridley sea turtle struck by the gear from entering the dredge bag whether the turtle encountered is an immature or mature. Therefore, Kemp's ridley interactions with scallop dredge gear may result in death and injury 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 injury and/or death to the turtle.

In summary, based on the observed capture of two Kemp's ridley sea turtles in scallop dredge gear and the observed capture of Kemp's ridley sea turtles in other mobile fishing gear in waters of the action area, the continued authorization of the scallop fishery (trawl and dredge gear components combined) is anticipated to result in the annual lethal take of up to two Kemp's ridley sea turtles. It is assumed that there is an equal chance of lethally taking male or female Kemp's ridley sea turtles since available information suggests that both sexes occur in the action area. Kemp's ridley sea turtles taken as a result of scallop dredge or trawl gear are expected to be immatures.

The lethal removal of up to 2 Kemp's ridley sea turtles annually, whether males or females, immature or mature animals, would be expected to reduce the number of Kemp's ridley sea turtles as compared to the number of Kemp's ridleys that would have been present in the absence of the proposed action assuming all other variables remained the same. The loss of up to 2 female Kemp's ridley sea turtles, annually, would be expected to reduce the reproduction of Kemp's ridley sea turtles as compared to the reproductive output of Kemp's ridley sea turtles in the absence of the proposed action. As described in Section 5.1, NMFS considers the trend for Kemp's ridley sea turtles to be stable. Nevertheless, the lethal removal of up to 2 Kemp's ridley sea turtles annually as a result of the continued authorization of the scallop fishery will not appreciably reduce the likelihood of survival for the species for the following reasons. From 1985 to 1999, the number of Kemp's ridley nests observed at Rancho Nuevo and nearby beaches increased at a mean rate of 11.3% per year. An estimated 4,047 females nested in 2006 and an estimated 5,500 females nested in Tamaulipas (the primary but not sole nesting site) over a 3-day period in May 2007 (NMFS and USFWS 2007c). Based on the number of nests laid in 2006 and the remigration interval for Kemp's ridley sea turtles, there were an estimated 7,000-8,000 adult female Kemp's ridleys in 2006 (NMFS and USFWS 2007c). 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 *Environmental Baseline* -- which includes the effects of the past operation of the Atlantic sea scallop fishery--, *Cumulative Effects*, and the *Status of the Species* (for those activities that occur outside of the action area of this Opinion) 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.

As described in the *Status of the Species* and *Environmental Baseline*, action has been taken to reduce anthropogenic effects to Kemp's ridley sea turtles. These include regulatory measures implemented in 2002 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 -- a leading known cause of Kemp's ridley sea turtle mortality. Since these regulatory measures are relatively recent, it is unlikely that current nesting trends reflect the benefit of these measures to Kemp's ridley sea turtles. Therefore, the current nesting trends for Kemp's ridley sea turtles are likely to improve as a result of regulatory action taken for the U.S. south Atlantic and Gulf of Mexico shrimp fisheries. There are no new known sources of injury or mortality for Kemp's ridley sea turtles.

Based on the information provided above, the loss of up to 2 Kemp's ridley sea turtles annually as a result of the continued authorization of the scallop fishery will not appreciably reduce the likelihood of survival for Kemp's ridley sea turtles given the increased nesting trend, and given 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).

Section 4(a)(1) of the ESA requires listing of a species if it is endangered or 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. NMFS is using these factors to assess whether the continued authorization of the scallop fishery will appreciably reduce the likelihood of recovery for the species given that recovery is defined as improvement in the status of the listed species to the point at which listing is no longer appropriate under the criteria seat out in section 4(a)(1) of the ESA (50 CFR 402.02).[11] As described in this Opinion, the continued authorization of the scallop fishery is expected to kill up to 2 Kemp's ridley sea turtles annually. No other direct or indirect effects to Kemp's ridley sea turtles such as on habitat, disease, predation and other natural influences on survival, are expected as a result of the proposed action. The loss of 2 Kemp's ridleys annually is not expected to modify, curtail or destroy their range. The scallop fishery does not utilize Kemp's ridleys for recreational, scientific or commercial purposes, or affect the adequacy of existing regulatory mechanisms to protect Kemp's ridley sea turtles. Therefore, the continued authorization of the scallop fishery will have no effect on #1-#4 of the ESA listing factors.

The lethal taking of up to 2 Kemp's ridley sea turtles annually in the scallop fishery is expected to reduce the number of Kemp's ridley sea turtles compared to the number that would have been present in the absence of the proposed action, and will, similarly, reduce Kemp's ridley reproduction as a result of the capture and killing if the Kemp's ridley sea turtles are females. These conclusions are relevant to listing factor #5 of the ESA. As described in the 5-year status review, Kemp's ridley sea turtles are experiencing considerable increases in nesting (NMFS and USFWS 2007c). From 1985 to 1999, the number of Kemp's ridley nests observed at Rancho Nuevo and nearby beaches increased at a mean rate of 11.3% per year. Nesting has increased from 247 nesting females in the 1985 nesting season to 4,047 nesting females in 2006. In May 2007, an estimated 5,500 females nested in Tamaulipas (the primary but not sole nesting site) over a 3-day period (NMFS and USFWS 2007c). Based on the number of nests laid in 2006 and the remigration interval for Kemp's ridley sea turtles, there were an estimated 7,000-8,000 adult female Kemp's ridleys in 2006 (NMFS and USFWS 2007c). The observed increase in nesting of Kemp's ridley sea turtles suggests that the manmade factors which contributed to its being listed under the ESA as an endangered species have been reduced to the extent that more female Kemp's ridley sea turtles are reaching maturity and nesting and/or mature females are living longer thus producing more nests over their lifetime. The loss of 2 Kemp's ridleys annually is not expected to change the trend in increased nesting especially if the Kemp's ridleys killed in the scallop fishery are males. With an increasing trend, the loss of 2 Kemp's ridleys will not compromise the continued existence of the species, which is the focus of the listing factor #5. Therefore, the continued authorization of the scallop fishery will not appreciably reduce the likelihood of recovery for the species.

7.1.4 Green Sea Turtle

There have been no known takes of green sea turtles in scallop trawl gear. The distribution of green sea turtles overlaps seasonally with the use of scallop trawl gear, and green sea turtles are captured in other types of trawl gear. Based on observer data, the capture of green sea turtles in any mobile gear operating within the action area, including scallop trawl gear, would be a rare event. However, given the low level of observer coverage in the scallop trawl fishery as well as

[11] NMFS is not using the recovery criteria of the 1992 recovery plan for Kemp's ridley sea turtles since the criteria provided address measures that need to be met for the species to be considered for downlisting rather than delisting.

other mobile gear fisheries in the action area it is likely that some interactions have occurred but were not observed or reported. Therefore, based on the average of the number of the takes per year for the period 1996-2007, one take of a green sea turtle in scallop trawl gear is anticipated to occur annually as a result of the continued authorization of the scallop fishery. Based on results from the U.S. south Atlantic and Gulf of Mexico shrimp trawl fisheries (Epperly *et al.* 2002; Sasso and Epperly 2006), any capture of a green sea turtle in scallop trawl gear could result in death due to forced submergence, given that there are no regulatory controls on tow-times in the scallop trawl fishery and some trawl tows that have been observed to take loggerhead sea turtles have exceeded one hour in duration (NEFSC, FSB database).

There has been one confirmed capture of a green sea turtle in scallop dredge gear. It is difficult to determine to which age class the green sea turtle observed taken in scallop dredge gear might have belonged given that its size was estimated rather than measured. Mid-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 authorization 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, such as would likely be the case if the green sea turtle encountered was immature, 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 the turtle was large enough to be prevented from entering the dredge bag by the chain mat, such as would likely be the case if the green sea turtle encountered was mature, 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.

In summary, based on the observed capture of a green sea turtle in scallop dredge gear and the observed capture of green sea turtles in other mobile fishing gear in waters of the action area, the continued authorization of the scallop fishery (trawl and dredge gear components combined) is anticipated to result in the annual lethal take of up to two green sea turtles. It is assumed that there is an equal chance of lethally taking male or female green sea turtle since available information suggests that both sexes occur in the action area. Green sea turtles taken as a result of scallop dredge or trawl gear are expected to be either neritic immatures or adults.

The lethal removal of up to 2 green sea turtles annually from the Atlantic, whether males or females, immature or mature animals, would be expected to reduce the number of green sea turtles in the Atlantic as compared to the number of green sea turtles that would have been present in the absence of the proposed action assuming all other variables remained the same. The loss of up to 2 female green sea turtles, annually, would be expected to reduce the reproduction of green sea turtles in the Atlantic as compared to the reproductive output of green sea turtles in the Atlantic in the absence of the proposed action. As described in Section 5.1, NMFS considers the trend for green sea turtles, as a species, to be declining. Nevertheless, the lethal removal of up to 2 green sea turtles annually from the Atlantic as a result of the continued authorization of the scallop fishery will not appreciably reduce the likelihood of survival for the species for the following reasons. Unlike green sea turtles that occur elsewhere in the species range, green turtle nesting in the Atlantic shows a generally positive trend during the ten years of regular monitoring since establishment of the index beaches in 1989 (Meylan *et al.* 1995). In the continental U.S., an average of 5,039 nests have been laid annually in Florida between 2001-2006 with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007d). Seminoff (2004) reviewed green turtle nesting at five western Atlantic sites. All of these showed increased nesting compared to prior estimates with the exception of nesting at Aves Island, Venezuela (Seminoff 2004). 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 1970's and nest count data from 1990-2003 suggests that 17,402-37,290 adult females nested each year (NMFS and USFWS 2007d). 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 *Environmental Baseline* -- which includes the effects of the past operation of the Atlantic sea scallop fishery--, *Cumulative Effects*, and the *Status of the Species* (for those activities that occur outside of the action area of this Opinion) 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.

As described in the *Status of the Species* and *Environmental Baseline*, action has been taken to reduce anthropogenic effects to green sea turtles in the Atlantic. These include regulatory measures implemented in 2002 to reduce the number and severity of green sea turtle interactions in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries -- a leading known cause of green sea turtle mortality in the Atlantic. Since these regulatory measures are relatively recent, it is unlikely 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 improve as a result of regulatory action taken for the U.S. south Atlantic and Gulf of Mexico shrimp fisheries. There are no new known sources of injury or mortality for green sea turtles in the Atlantic.

Based on the information provided above, the loss of up to 2 green sea turtles annually in the Atlantic as a result of the continued authorization of the scallop fishery 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, and given measures that reduce the number of Atlantic green sea turtles 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 direct or indirect effects on green sea turtles that occur outside of the Atlantic. Therefore, since the continued authorization of the scallop fishery will not appreciably reduce the likelihood of survival of green sea turtles in the Atlantic, the proposed action will not appreciably reduce the likelihood of survival for the species.

The 5-year status review for the species reviewed the recovery criteria provided with the 1991 recovery plan for green sea turtles in the Atlantic, and the progress made in meeting each objective (NMFS and USFWS 2007d). These are that the U.S. population of green turtles can be considered for delisting if, over a period of 25 years, the following conditions are met: (1) the level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years; (2) at least 25% (105 km) of all available nesting beaches (420 km) is in public ownership and encompasses greater than 50% of the nesting activity; (3) a reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds; (4) all priority one tasks have been successfully implemented (these address a multitude of measures in areas of nesting habitat, marine habitat, disease, species protection, data collection and management amongst others; NMFS and USFWS 1991b). As described in this Opinion, the continued authorization of the scallop fishery is expected to kill up to 2 Atlantic green sea turtles annually. No other direct or indirect effects to green sea turtles are expected as a result of the proposed action. The continued authorization of the scallop fishery will not affect ownership of nesting habitat, nor will it affect the protection of nesting beaches and the marine environment or compromise the ability of researchers to conduct scientific studies. Therefore, the continued authorization of the scallop fishery will have no effect on recovery criteria #2 and #4.

The lethal taking of up to 2 green sea turtles annually in the scallop fishery is expected to reduce the number of green sea turtles in the Atlantic compared to the number that would have been present in the absence of the proposed action, and will, similarly, reduce green sea turtle reproduction in the Atlantic as a result of the capture and killing if the green sea turtles are females. These conclusions are relevant to recovery criteria #1 and #3 of the 1991 recovery plan for green sea turtles in the Atlantic. As described in the 5-year status review for the species (NMFS and USFWS 2007d), an average of 5,039 green sea turtle nests have been laid annually over the past 6 years in Florida. Thus, recovery criteria #1 has been met, and the annual loss of 2 green sea turtles which may be male or female, mature or immature, is not expected to materially affect the 6-year average of nests on Florida beaches. With respect to recovery criteria #3, there is evidence of substantial increases in the number of green sea turtles on foraging grounds within the western Atlantic. 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 the 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 the 16-year period from 1976-1993, green sea turtle captures averaged 24 per year (Wilcox *et al.* 1998). The green turtle catch for 1993, 1994, and 1995 was 745%, 804%, and 2084%, respectively, above the previous 16-year average annual catch (Wilcox *et al.* 1998). Such changes are not as dramatic elsewhere. In a study of sea turtles incidentally caught in pound net gear fished in inshore waters of Long Island, NY, Morreale *et al.* (2004) 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 1990's. 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). The annual loss of 2 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 authorization of the scallop fishery will not appreciably reduce the likelihood of recovery for green sea turtles in the Atlantic. Since the scallop fishery has no direct or indirect effects on green sea turtles that occur outside of the Atlantic, the continued authorization of the scallop fishery will not appreciably reduce the likelihood of recovery for the species.

8.0 CONCLUSION

After reviewing the current status of loggerhead, leatherback, Kemp's ridley, and green sea turtles, the environmental baseline and cumulative effects in the action area, the effects of the continued authorization of the Scallop FMP (including the seasonal use of chain mat modified scallop dredge gear in Mid-Atlantic waters), it is NMFS' biological opinion that the proposed activity may adversely affect but is not likely to jeopardize the continued existence of loggerhead, leatherback, Kemp's ridley and green sea turtles.

9.0 INCIDENTAL TAKE STATEMENT (as amended, February 5, 2009)

Section 9 of the Endangered Species Act 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).

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 data from observer reports for the scallop fishery, and the distribution and abundance of turtles in the action area, NMFS anticipates that the continued implementation of the Scallop FMP, may result in the taking of sea turtles as follows:

- for scallop *dredge* gear, NMFS anticipates the **biennial** take of up to 929 loggerheads of which up to 595 will be lethal takes (includes serious injuries), as well as the annual take of 1 leatherback sea turtle (non-lethal), 2 Kemp's ridley sea turtles (lethal or non-lethal), and 2 green sea turtles (lethal or non-lethal); and,

- for scallop *trawl* gear, NMFS anticipates the annual take of up to 154 loggerhead sea turtles of which up to 20 will be lethal takes, as well as 1 leatherback, 1 Kemp's ridley, and 1 green sea turtle, all of which may be lethal or non-lethal takes.

The number of loggerhead sea turtles expected to be killed or suffer serious injuries as a result of interactions with scallop dredge gear is based on data collected in the 2003 fishing year, prior to the use of chain mats. Therefore, while the estimated 595 loggerhead takes, biennially, resulting in immediate death or serious injury is based on the best currently available information, it is also likely a worst case scenario.

Anticipated Impact of Incidental Take

NMFS has concluded that the continued operation of the scallop fishery may adversely affect but is not likely to jeopardize loggerhead, leatherback, Kemp's ridley or green sea turtles. 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 interactions with the scallop fishery now and to generate the information necessary in the future to continue to minimize incidental takes. These measures are non-discretionary and must be implemented by NMFS. Many of these measures were included as RPMs with the September 18, 2006 Opinion. They are repeated here because they still meet the criteria for an RPM and reflect work in progress to minimize the taking of sea turtles in scallop dredge and/or scallop trawl gear.

Reasonable and Prudent Measures

NMFS has determined that the following reasonable and prudent measures are necessary or appropriate to minimize impacts of incidental take of sea turtles:

1. NMFS must limit the amount of allocated scallop fishing effort by "Limited access scallop vessels" as such vessels are defined in the regulations (50 CFR 648.2), that can be used in the area and during the time of year when sea turtle distribution overlaps with scallop fishing activity. **(amended February 5, 2009)**

2. NMFS must continue to investigate and implement, as appropriate, gear modifications for scallop dredge and trawl gear to reduce the capture of sea turtles and/or the severity of the interactions that occur.

3. NMFS must review available data to determine whether there are areas (*i.e.*, "hot spots") within the action area where sea turtle interactions with scallop dredge and/or trawl gear are more likely to occur.

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

5. NMFS must determine (a) the extent to which sea turtle interactions with scallop dredge gear occur on the bottom vs. within the water column and (b) the effect on sea turtles of being struck by the scallop dredge.

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 1 above, no later than the 2010 scallop fishing year, NMFS must limit the amount of allocated limited access scallop fishing effort that can be used in waters south of the northern boundaries of statistical areas 612, 613, 533, 534, 541-543 during the periods in which turtle takes have occurred. Restrictions on fishing effort described

above shall be limited to a level that will not result in more than a minor impact on the fishery. **(amended February 5, 2009)**

2. To comply with 2 above, NMFS must continue to investigate modifications of scallop trawl and dredge gear. Within a reasonable amount of time following completion of an experimental gear trial from or by any source, NMFS must review all data collected from the experimental gear trials, determine the next appropriate course of action (*e.g.*, expanded gear testing, further gear modification, rulemaking to require the gear modification), and initiate action based on the determination. The goal of this RPM is ultimately to require modification of fishing gear used in the scallop fishery operating under the Atlantic Sea Scallop FMP within a reasonable timeframe following sound research that demonstrates that the gear modification is reasonable and feasible and will help to minimize the number and/or severity of sea turtle interactions with scallop fishing gear.

3. To comply with 3 above, NMFS must review all data available on the observed take of sea turtles in the scallop fishery and other suitable information (*i.e.*, data on observed turtle interactions for other fisheries or fishery surveys in the area where the scallop fishery operates) to assess whether there is sufficient information to identify "hot spots" within the action area. Within a reasonable amount of time after completing the review, if NMFS determines that "hot spots" do exist, NMFS must take appropriate action to reduce sea turtle interactions and/or their impacts within any identified hot spot.

4. To comply with 4 above, NMFS must use available and appropriate technologies (*e.*, underwater video as part of an experiment using scallop dredge gear in either the natural or controlled environment, computer modeling, etc.) to quantify the extent to which chain mats 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 chain mats do reduce injuries leading to death for sea turtles and may result in further modifications of the fishery to ensure sea turtle interactions and/or interactions causing death are minimized. Initiate study no later than fiscal year 2009.

5. To comply with 5 above, NMFS must 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 mortal injuries. Initiate study no later than fiscal year 2009.

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 provided in biological opinions for the scallop fishery. NMFS will continue to use observer coverage to monitor sea turtle bycatch in trawl gear and sea turtle bycatch in scallop dredge gear when that gear is used in areas or at times when chain mats are not required.

The use of chain mats on scallop dredge gear is expected to greatly reduce the likelihood that turtles struck by or incidentally swimming into scallop dredge gear would enter the dredge bag (NMFS 2006b). Therefore, given that scallop dredge vessels are required to use chain mats on scallop dredge gear 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 (71 FR 50361, August 25, 2006), injuries to sea turtles that occur as a result of the turtle being struck by the dredge gear underwater will continue to occur but will not be observed unless the turtle is small enough to pass between the chains and 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.

NMFS' NERO has considered the use of underwater video on scallop dredge vessels to monitor sea turtle interactions with the gear. However, based on the information currently available as well as the previous use of this technology in studies of turtle interactions with scallop dredge gear, the use of underwater video monitoring for monitoring the take of sea turtles in scallop dredge gear is infeasible (Memo from N. Thompson, NEFSC to P. Kurkul, NERO, October 16, 2007). NMFS' NERO has also considered whether chains mats should be removed from scallop dredge gear during some observed trips to assess the number of turtle interactions that were occurring when chain mats were on the gear. However, NMFS' NERO has also determined that this is not a feasible method for monitoring the sea turtle interactions with the dredge component of the scallop fishery given that the removal of the chains will likely increase the number of sea turtle deaths in comparison to the number that would have occurred if chains were present, and it is uncertain whether the take estimates generated from non-chain mat modified dredges will correctly estimate takes in chain mat equipped dredges since the dredges may perform differently based on the presence of absence of the chain mat (Memo from N. Thompson, NEFSC to P. Kurkul, NERO, October 16, 2007).

As described in the September 18, 2006 Opinion on the continued authorization of the scallop fishery under the FMP, NMFS' NERO requested guidance from the NEFSC on methods to monitor sea turtle takes (*e.g.*, capture) in the dredge component of the scallop fishery in the event that the chain mat rule was approved and implemented. In response to this request, the NEFSC provided information on fishery dependent and fishery independent approaches considered by the NEFSC for monitoring interactions between sea turtles and scallop dredge gear and the reasonableness of each approach. The NEFSC concluded, however, that none of the approaches could provide a "scientifically accurate and robust" take estimate and, as such, the NEFSC could not support or recommend any one of these approaches. Based on information provided by the NEFSC, NERO concluded that a method does not currently exist for enumerating sea turtles taken by chain mat equipped scallop dredge gear which meets the NEFSC's definition of a scientifically robust and accurate take estimate and the guiding principles for the preparation of biological opinions provided in the Final ESA Section 7 Handbook developed jointly by the FWS and NMFS. In the absence of a method for enumerating most takes to monitor the ITS on the scallop fishery as a whole, NMFS will, therefore, 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 likely result in greater sea turtle interactions with scallop dredge gear than what is estimated to have occurred in 2003 and 2004. 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, NMFS 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 2003 and May through November 2004 as the benchmark against which the 2-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 2-year running average of dredge hours in Mid-Atlantic waters (as far south as Cape Hatteras, NC) 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, NC) during the same period of 2003 and 2004.

10.0 CONSERVATION RECOMMENDATIONS

In addition to section 7(a)(2), which requires agencies to ensure that proposed projects will not jeopardize the continued existence of listed species, section 7(a)(1) of the ESA places a responsibility on all Federal agencies to utilize their authorities in furtherance of the purposes of the Act by carrying out programs for the conservation of endangered species. Conservation Recommendations are discretionary activities designed to minimize or avoid adverse effects of a proposed action on listed species or critical habitat to help implement recovery plans, or to develop information. The following additional measures are recommended regarding incidental take and sea turtle conservation:

1. Given the very low rate of compliance with the reporting on VTRs of sea turtle interactions with federally-permitted fishing gear, NMFS should work to increase compliance with this requirement.

2. NMFS must continue to collect and analyze samples from loggerhead sea turtles captured in scallop dredge and trawl gear to determine the nesting origin of loggerhead sea turtles taken in the scallop fishery in order to better assess the effects of the scallop fishery on loggerhead nesting groups and address those effects accordingly. NMFS should review its policy/protocol(s) 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 captured sea turtles.

3. NMFS should establish a protocol for bringing to shore any sea turtle captured in scallop dredge or trawl 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.

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.

5. NMFS should support (*i.e.*, fund, advocate, promote) in-water abundance estimates of sea turtles in the action area. This information is required to provide more current information on the distribution and abundance of sea turtles than that provided by the CeTAP surveys conducted in the 1980s.

6. NMFS should reestablish a long term in-water index study for sea turtles to monitor recruitment and health in the action area.

7. NMFS, NER should work with NMFS, F/PR2 to evaluate whether the existing sea turtle resuscitation and handling guidelines should accommodate the treatment of seriously

injured turtles (*e.g.* cracked carapaces) that have been recorded in the scallop dredge component.

8. NMFS should provide guidance to permitted scallop fishermen on the sea turtle handling and resuscitation criteria, as well as guidance to scallop dredge fishermen on the dumping of the dredge bag and lowering of the cutting bar to reduce the risk of injury to sea turtles that may be caught in dredge gear.

11.0 REINITIATING CONSULTATION

This concludes formal consultation on the continued authorization of the Atlantic sea scallop fishery as it operates 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 take is exceeded, NMFS, NERO must immediately request reinitiation of formal consultation.

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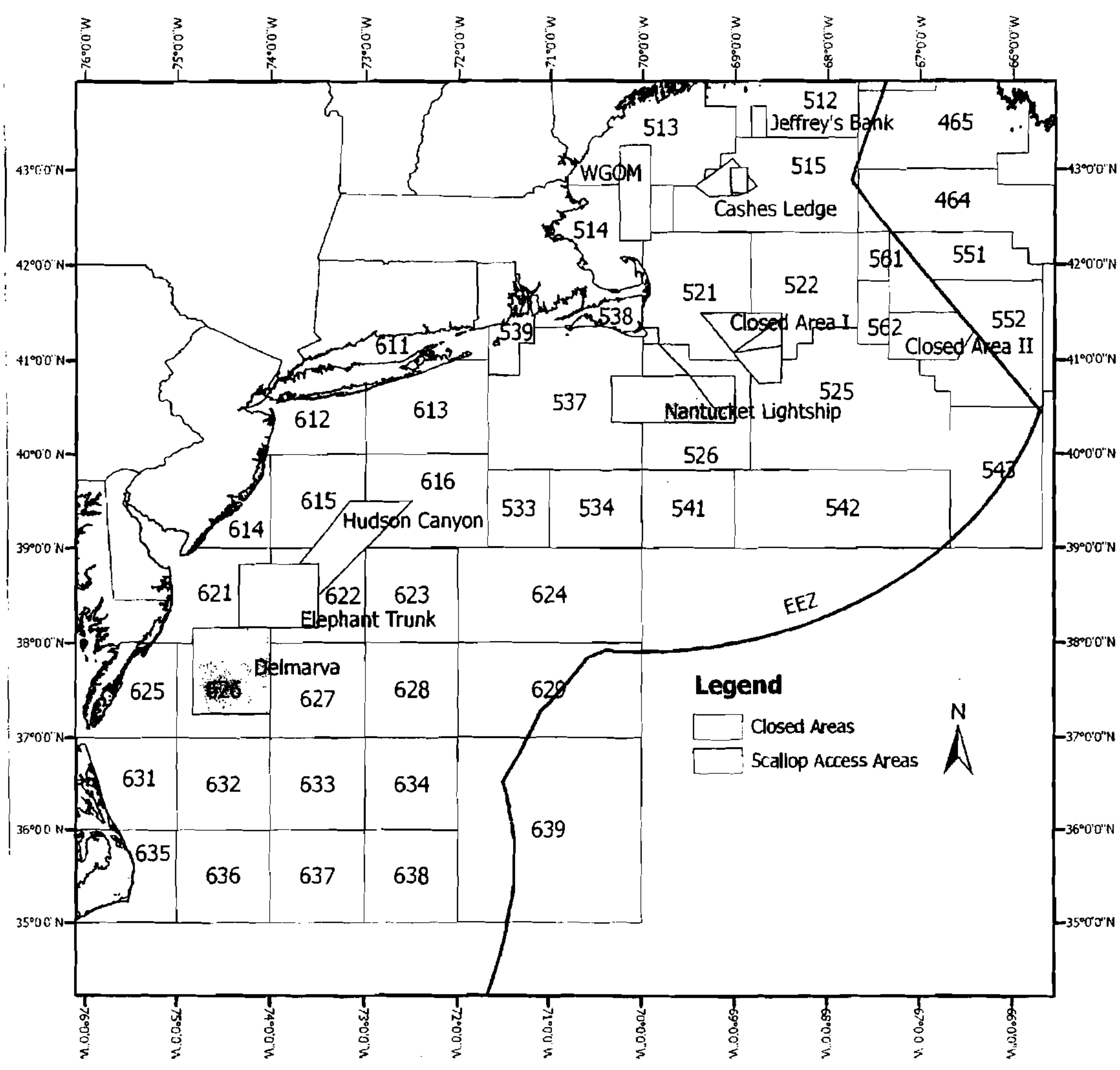
Appendix 1. Northeast Region Statistical Areas



Appendix 2. The anticipated Incidental Take of loggerhead, leatherback, Kemp's ridley and green sea turtles as currently determined in the most recent Biological Opinion's for NOAA Fisheries implementation of the Bluefish, Herring, Multispecies, Mackerel/Squid/Butterfish, Red Crab, Monkfish, Skate, Spiny Dogfish, Summer Flounder/Scup/Black Sea Bass, Tilefish, and Highly Migratory Species fishery management plans as well as for the American Lobster fishery operating in Federal waters, and hopper dredging projects of the ACOE and USN operating off of Virginia. Takes are represented as anticipated annual take unless otherwise noted.

Fishery	**Sea Turtle Species**			
	Loggerhead	**Leatherback**	**Kemp's Ridley**	**Green**
Bluefish	6-no more than 3 lethal	None	6 lethal or non-lethal	None
Herring	6-no more than 3 lethal	1 lethal or non-lethal	1 lethal or non-lethal	1 lethal or non-lethal
HMS	1869 for 2004-2006 and 1905 for each subsequent 3-year period	1981 for 2004-2006 and 1764 for each subsequent 3-year period	105 total for each 3-year period beginning 2004-2006 (Kemp's ridleys, green, olive ridley or hawksbill in combination)	
Lobster	2 lethal or non-lethal	4 lethal or non-lethal	None	None
Mackerel/Squid/ Butterfish	6-no more than 3 lethal	1 lethal or non-lethal	2 lethal or non-lethal	2 lethal or non-lethal
Monkfish (gillnet)	3	1 leatherback, Kemp's ridley or green		
Monkfish (trawl)	1 loggerhead, leatherback, Kemp's ridley or green			
Multispecies	1 lethal or non-lethal	1 lethal or non-lethal	1 lethal or non-lethal	1 lethal or non-lethal
Red Crab	1 lethal or non-lethal	1 lethal or non-lethal	None	None
Skate	1 (either a loggerhead, leatherback, Kemp's ridley or green) - lethal or non-lethal			
Spiny Dogfish	3-no more than 2 lethal	1 lethal or non-lethal	1 lethal or non-lethal	1 lethal or non-lethal
Summer Flounder/Scup/ Black Sea Bass	19-no more than 5 lethal (total - either loggerheads or Kemp's ridley)	None	See loggerhead entry	2 lethal or non-lethal
Tilefish	6-no more than 3 lethal or having ingested the hook	1 lethal or non-lethal take (includes having ingested the hook)	None	None
(ACOE) Sandbridge Protection Project	5	None	1 Kemp's ridley or green	
(USN) Dam Neck Nourishment Project	4 per dredge cycle	None	1 Kemp's ridley or green per dredge cycle	

Appendix 3. Analysis of Atlantic Sea Scallop Fishery Impacts on the North Atlantic Population of Loggerhead Turtles (*Caretta caretta*)

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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February 2008

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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 $\Phi(z)$ 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

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[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 ($\sigma^2 = 0.012$, 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 ($\mu = -0.049$, $\sigma^2 = 0.011$, 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 ($\sigma^2 = 0.011$, 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.

ACKNOWLEDGMENTS

Thanks to Sheryan Epperly, Selina Heppell, Chris Legault, Kimberly Murray, Tim Ragen, Paul Richards, Fred Serchuk, Chris Sasso, and Melissa Snover for helpful comments in the development of this analysis and manuscript. Also, we are grateful to Blair Witherington, the Florida Fish and Wildlife Research Institute, and the South Carolina Department of Natural Resources for use of their data. Finally, Jarita Davis significantly improved the manuscript through her technical and grammatical edits.

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

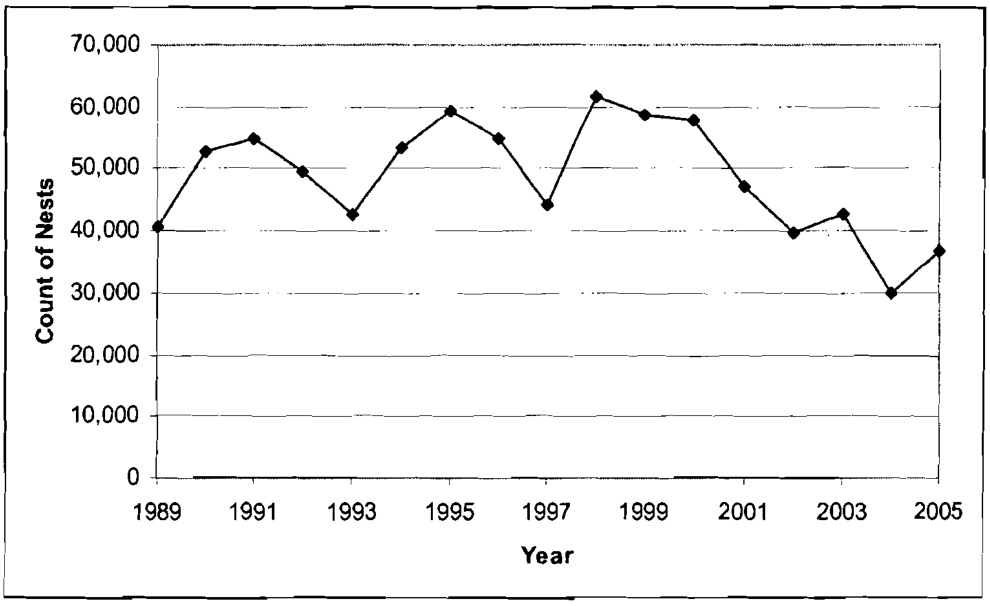

Figure 1. Number of Atlantic Ocean loggerhead sea turtle (*Caretta caretta*) nests recorded at US Northern (NC, SC, GA) and Peninsular Florida index beaches from 1989 to 2005 (NMFS in review, FWRI 2007).

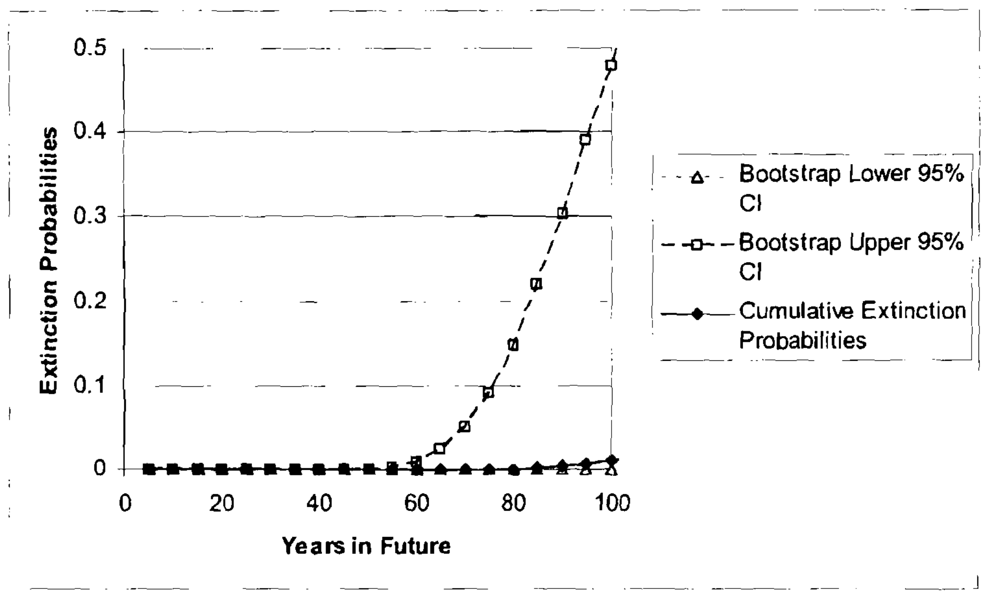

Figure 2. Cumulative quasi-extinction probabilities and confidence intervals (CI) for 1989-2005 base model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes for adult female western North Atlantic loggerhead sea turtles *(Caretta caretta)*. Quasi-extinction is equal to 250 adult female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

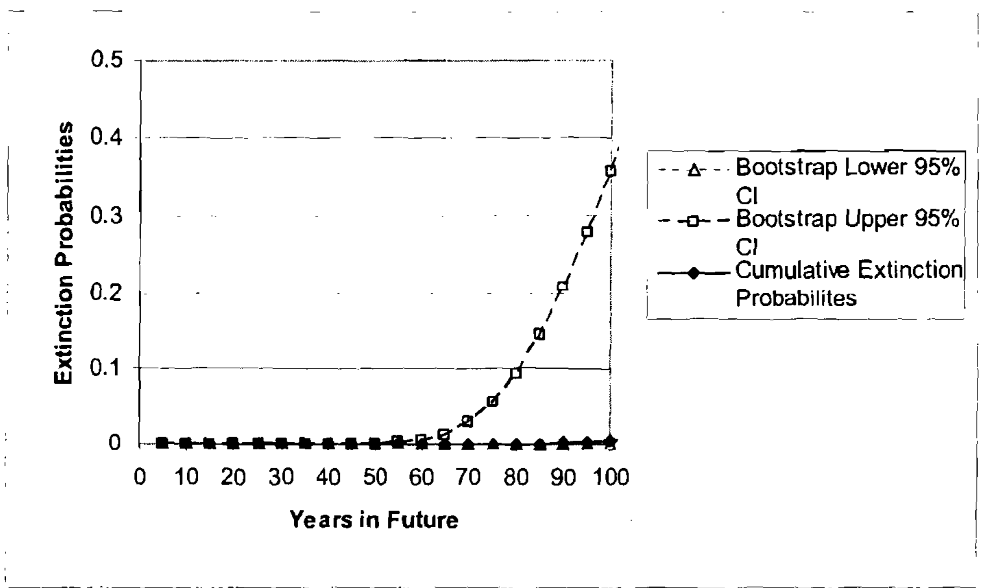

Figure 3. Cumulative quasi-extinction probabilities and confidence intervals (CI) for 1989-2005 model with Atlantic sea scallop (*Placopecten magellanicus*) fishery takes for adult female western North Atlantic loggerhead sea turtles *(Caretta caretta)* added back into population. Quasi-extinction is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

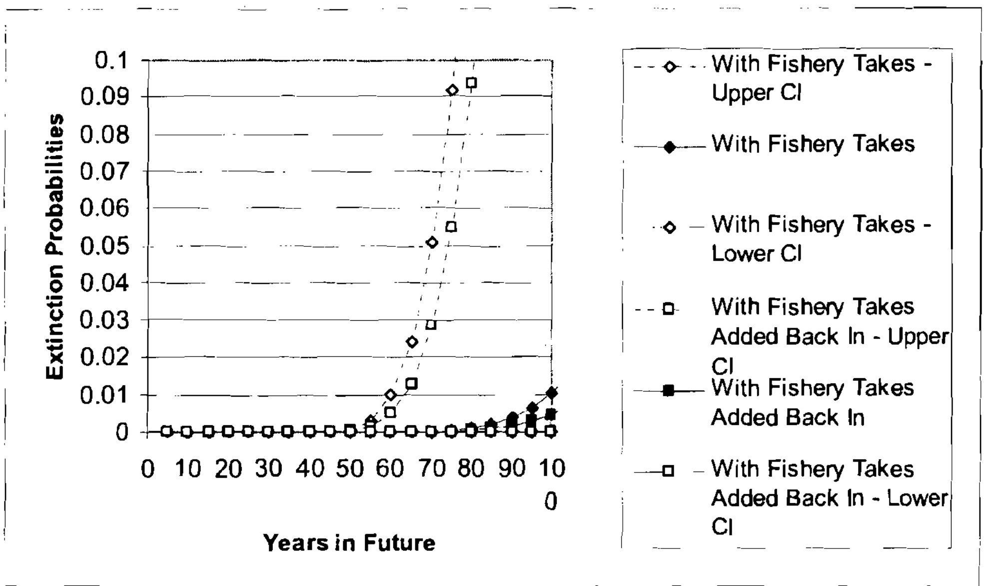

Figure 4. Comparison of cumulative quasi-extinction probabilities and confidence intervals (CI) of 1989-2005 models with and without Atlantic sea scallop (*Placopecten magellanicus*) fishery takes. Quasi-extinction is equal to 250 adult female loggerhead sea turtles *(Caretta caretta)*. Note vertical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

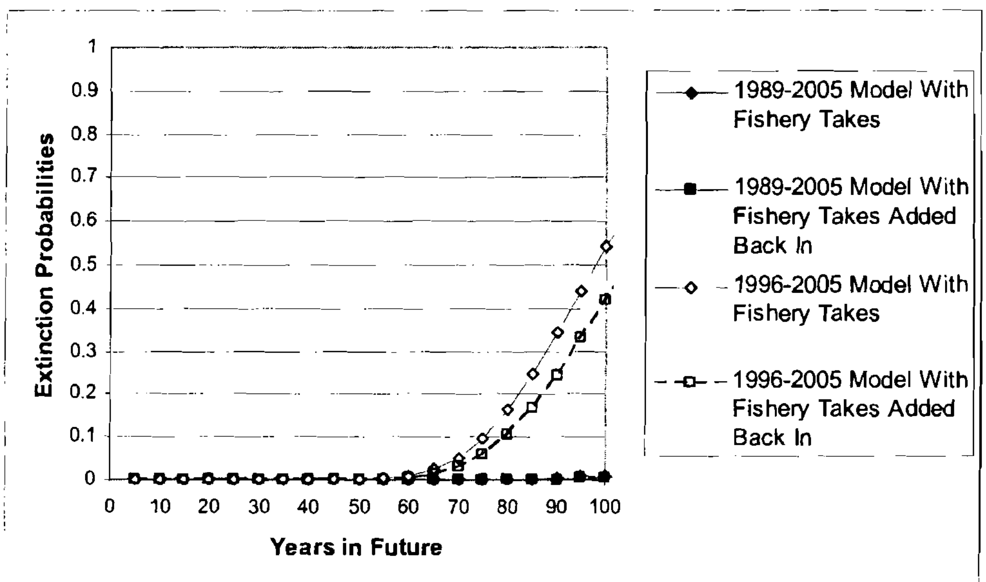

Figure 5. Extinction trajectories for models with and without Atlantic sea scallop (*Placopecten magellanicus*) fishery takes with original 1989-2005 population trajectory compared to 1996-2005 trajectory. Quasi-extinction is equal to 250 adult female loggerhead sea turtles *(Caretta caretta)*.

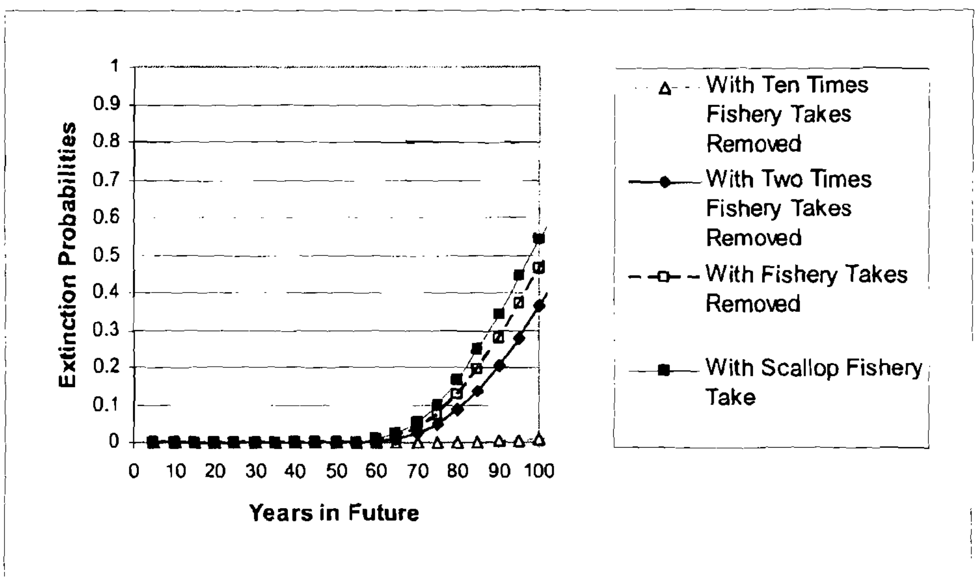

Figure 6. Cumulative quasi-extinction probabilities for 1996-2005 models with various levels of mortality removed from the trend. Fishery takes estimated as one time (the Atlantic sea scallop [*Placopecten magellanicus*] fisheries) versus two and ten times the original sea scallop fishery take level.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

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