ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
BIOLOGICAL OPINION

Action Agency: National Marine Fisheries Service, Northeast Regional Office

Activity: Authorization of fisheries under the Monkfish Fishery Management Plan [Consultation No. F/NER/2008/01754]
GARFO-2008-00001

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

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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 reinitiated formal intra-service consultation on the continued operation of the monkfish fishery, in accordance with section 7(a)(2) of the ESA and 50 CFR 402.16, given new information on sea turtle takes in the fishery and the Atlantic Large Whale Take Reduction Plan (ALWTRP) modifications that may affect the fishery in a manner or to an extent not previously considered. This document represents NMFS' biological opinion (Opinion) on the continued operation of the monkfish fishery under the Monkfish 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 operation of the monkfish fishery was reinitiated on April 2, 2008 [Consultation No. F/NER/2008/01754]. 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

1.1 Consultation Review

The consultation history for the monkfish fishery was reviewed in the previous formal consultation completed April 14, 2003 Opinion [Consultation number F/NER/2002/00196]. In brief, formal consultation on the fishery was first initiated in 1998 and concluded that the operation of the fishery would not result in jeopardy to any ESA protected species under NMFS jurisdiction provided that the gillnet portion of this fishery was modified by the application of the Atlantic Large Whale Take Reduction Plan (ALWTRP). The Opinion also concluded that the gillnet sector might adversely affect sea turtles, and an Incidental Take Statement (ITS) with Reasonable and Prudent Measures (RPMs) to minimize take was provided. Consultation was reinitiated in 2000 after new information indicated a change in the status of right whales, and observer data indicated that the ITS for sea turtles in the monkfish fishery was exceeded during Year 1 (November 8, 1999 - April 30, 2000) of the FMP. The consultation [Consultation number F/NER/2001/00546] was concluded on June 14, 2001, and resulted in a jeopardy finding for North Atlantic right whales. In response to the jeopardy conclusion, the other contained one Reasonable and Prudent Alternative (RPA) with multiple management components that collectively were designed to avoid the likelihood of the federal monkfish fishery jeopardizing the continued existence of the endangered right whale. Incidental take of sea turtles was also anticipated but was not expected to lead to jeopardy for any of the affected sea turtle species. An ITS was provided along with RPMs to minimize the taking of sea turtles in the monkfish fishery.

In 2002, following the NMFS rejection of the proposed Framework Adjustment 1, the agency published an Emergency Interim Final Rule to establish the Year 4 specifications for the monkfish fishery. The Emergency Interim Final Rule included deferral of the Year 4 default that would have reduced DAS in the monkfish fishery to zero, effectively eliminating the directed monkfish fishery. Since the June 14, 2001 Opinion had not considered the effects of monkfish fishing effort on ESA-listed species for year 4 of the FMP, NMFS concluded that deferral of the Year 4 measures for one year may adversely affect ESA-listed species. NMFS, therefore, reinitiated section 7 consultation on the continued implementation of the monkfish fishery and on May 14, 2002, concluded that the fishery was not likely to jeopardize any ESA-listed species under NMFS jurisdiction. A new ITS and RPMs to address the anticipated take of sea turtles in the fishery for Year 4 were provided.

Consultation was reinitiated on February 12, 2003 to consider the effects to protected species from actions proposed under Framework Adjustment 2. This consultation concluded in a Biological Opinion, issued on April 14, 2003, that the implementation of Framework Adjustment 2 to the Monkfish FMP may adversely affect but may not likely to jeopardize the continued existence of ESA-listed species. A new ITS and RPMs to address the anticipated take of sea turtles were provided.

Regulations implementing Amendment 2 to the Monkfish FMP were approved and took effect on May 1, 2005. The regulations included measures to increase fishing opportunities and provide for additional flexibility, while also meeting the conservation objectives of the FMP. Amendment 2 also contains gear modifications and closures to protect Essential Fish Habitat (EFH). Amendment 2 did not change the existing effort control measures that link Northeast (NE) monkfish and Atlantic sea scallop days-at-sea (DAS) to monkfish DAS.

Due to changes in the ALWTRP, which included elimination of the Dynamic Area Management (DAM) program as of April 7, 2008, and elimination of the Seasonal Area Management (SAM) program as of October 6, 2008[1], as well as new information about effects the monkfish fishery has on sea turtle takes, formal consultation has been reinitiated in accordance with the regulations at 50 CFR 402.16, to reconsider the effects of the continued operation of the monkfish fishery on ESA-listed cetaceans and sea turtles.

Other than these formal consultations, Section 7 consultations were conducted and completed informally for other framework adjustments and amendments to the Monkfish FMP. None of these met the triggers for reinitiating formal consultation.

1.2 Causes for Reinitiating

[1] Effective October 5, 2008, NMFS reinstituted the DAM program under the ALWTRP pursuant to a preliminary injunction issued in the case The Humane Society of the United States, *et al.* v. Gutierrez, *et al.* (Civil Action No. 08-cv-1593 (ESH)). The DAM program was effective through 2400 hrs April 4, 2009, and expired at this time when the broad-based sinking groundline requirement for Atlantic trap/pot fisheries became effective on April 5, 2009.

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 June 14, 2001 Opinion for the monkfish fishery concluded that continued operation of the fishery was likely to jeopardize the continued existence of ESA-listed right whales as a result of entanglement in gillnet gear used in the fishery. An RPA was provided to remove the likelihood of jeopardy and was considered part of the action in April 14, 2003 Opinion. The RPA included, in part, implementation of a SAM program and a DAM program to reduce the likelihood of right whale interactions with gillnet gear used in the monkfish fishery. The RPA measures were implemented as part of the ALWTRP. On October 5, 2007, NMFS published a final rule in the *Federal Register* (72 FR 57104) that made many changes to the ALWTRP affecting the use of fixed gillnet gear in the monkfish fishery, amongst others. These changes included elimination of the DAM program as of April 7, 2008, and elimination of the SAM program as of October 6, 2008[2]. The changes to the ALWTRP, therefore, modified the RPA in a manner that causes an effect to listed species not considered in the April 14, 2003 Opinion for the fishery. NMFS reinitiated formal consultation on the monkfish fishery on April 2, 2008, in accordance with the regulations at 50 CFR 402.16 to reconsider the effects of the continued operation of the monkfish fishery on ESA-listed cetaceans and sea turtles.

The April 14, 2003 Opinion for the monkfish fishery concluded that continued operation of the fishery may adversely affect ESA-listed sea turtles. An ITS was provided in the Opinion that described the anticipated annual take (lethal or non-lethal) in gillnet or trawl gear used in the fishery. In 2006, the Northeast Fisheries Science Center (NEFSC) released reference document 06-19 (Murray 2006) that reported on the annual estimated taking of loggerhead sea turtles in bottom-otter trawl gear fished in Mid-Atlantic waters during the period of 1996-2004. As a follow-up, and in response to a request from NERO, the bycatch rate identified in Murray 2006 was used to estimate the take of loggerhead sea turtles in all fisheries (by FMP group) using bottom otter trawl gear fished in Mid-Atlantic waters during the period of 2000-2004 (Murray 2008). Based on the approach as described in Murray 2008, the average annual take of loggerhead sea turtles in bottom otter trawl gear for the period of 2000-2004 was estimated to be 2 for trawl gear used in the monkfish fishery. Therefore, this information reveals effects of the monkfish fishery on sea turtles that were not previously considered. In accordance with the regulations at 50 CFR 402.16, formal consultation was reinitiated to reconsider the effects of the continued operation of the monkfish fishery on ESA-listed sea turtles and cetaceans.

[2] Effective October 5, 2008, NMFS reinstituted the DAM program under the ALWTRP pursuant to a preliminary injunction issued in the case The Humane Society of the United States, *et al.* v. Gutierrez, *et al.* (Civil Action No. 08-cv-1593 (ESH)). The DAM program was effective through 2400 hrs April 4, 2009, and expired at this time when the broad-based sinking groundline requirement for Atlantic trap/pot fisheries became effective on April 5, 2009.

2.0 DESCRIPTION OF THE PROPOSED ACTION

The proposed action is the continued operation of the monkfish fishery managed under the Monkfish FMP. A summary of the characteristics of the fishery relevant to the analysis of its potential effects on ESA-listed species and critical habitat is presented below.

It is important to note that commercial and recreational fishing vessels are often permitted to operate within multiple federal fisheries and species of fish managed under multiple FMPs are commonly landed concurrently, for the purposes of this Opinion, fishing effort under the Monkfish FMP includes actions that result in landings of monkfish by federally permitted vessels operating within the action area described below in Section 2.2. In order to identify and analyze fishery impacts on ESA-listed species, ideally, documented takes of listed species would be linked to FMPs proportionally based on the fish catch composition of the fishing trip. As an example, fishing effort and estimated bycatch of ESA-listed species for a trip that landed 40% spiny dogfish, 35% haddock (a species managed under the Multispecies FMP) , and 25% monkfish would be allocated proportionately to the Spiny Dogfish FMP (40%), Multispecies FMP (35%), and Monkfish FMP (25%). The overall estimated bycatch for each FMP is the sum of the proportionally allocated bycatch estimates.

However, currently, data on take of protected species does not completely align with this ideal definition of the fishery. We have the benefit of scientifically produced estimates of loggerhead sea turtle bycatch in commercial trawl and gillnet fisheries pertaining to the action area considered in this consultation (Murray 2008 and Murray 2009a). The bycatch estimate for trawl fisheries attributes takes to the most abundant (by weight) fish species (which are used as a proxy for associated FMPs) landed per trip. Alternatively, the gillnet loggerhead bycatch estimate is more closely aligned with our ideal definition of the fishery as it proportionally attributes sea turtle takes consistent with the composition of the fish catch for that trip. For leatherback, Kemp's ridley, and green sea turtles observed takes of sea turtles are attributed to the FMP that covers the species which makes up the majority (by weight) of the catch for the trip during which sea turtle(s) were caught. The number of observed non loggerhead sea turtle takes attributable to a specific fishery is a small sample size. Given that we know these are underestimates since they are a tally of observations rather than an overall estimate, we have selected to use the total number of leatherback, Kemp's ridley and green sea turtle takes by species and gear type as the estimated take level. While this may attribute the same take of a turtle to multiple fisheries using the same gear type, and in that way count that individual take more than once, this is offset by the fact that the number of observed takes is less than the number of actual takes occurring in the fishery. For listed large whales, we can only rarely attribute takes to a specific fishery. We, therefore, attribute takes by gear type and assume that any one of the fishery management plans that authorize the use of that gear may be responsible for that take.

Stranding data provide some evidence of interactions between recreational hook and line gear and ESA-listed species, but assigning the gear to a specific fishery is rarely, if ever, possible. Presently, there are no other data sets available to provide estimates of incidental take for recreational fishing activities in an area as extensive as the action area for this consultation. There is an effort to include questions about interactions with ESA-listed species in a survey similar to the Marine Recreational Fisheries Statistics Survey (MRFSS), but the development of the survey has not been completed. Based on perspectives of fishery managers within NMFS a recreational fishery for monkfish seems to be absent, therefore, NMFS will instead focus the majority of the effects analysis on the commercial component of the fishery.

2.1 Description of the Current Monkfish Fishery

Monkfish (also known as goosefish) are harvested for their livers and the tender meat in their tails, but are also landed as whole gutted fish. The species is distributed widely throughout the Northwest Atlantic, from the northern Gulf of St. Lawrence to Cape Hatteras, NC, and is known to inhabit waters from the tide-line to depths as great as 900 meters across a wide range of temperatures. Adults have been found on a variety of substrate types including hard sand, gravel, broken shell, and soft mud. Monkfish rest partially buried on soft substrates while attracting prey using their modified first dorsal fin rays as a lures. Growth is rapid in goosefish, growing about 10 cm per year for both sexes, until the age of 6 years. It is rare for a male to live longer than 7 years, but females are capable to live 12-14 years or older. Spawning primarily occurs from spring to early summer from Cape Hatteras to southern New England, but may occur as early as January and as late as August (Johnson *et al.* 2008).

Although there is no strong evidence (growth, maturity, and genetic information) of separate biological stocks, monkfish are managed as two regional stocks to accommodate differences in fishery practices. These stocks consist of a northern stock from Maine to Cape Cod, MA, and a southern stock from Cape Cod to North Carolina. There is no known recreational fishery for monkfish, but they may rarely be taken by anglers fishing for other bottom dwelling fish. The monkfish fishery is jointly managed by the New England Fishery Management Council (NEFMC) and the Mid-Atlantic Fishery Management Council (MAFMC), with the NEFMC having the administrative lead. During the early 1990s, commercial fishermen and dealers in the monkfish fishery addressed both the New England and Mid-Atlantic Councils with concerns about the increasing amount of small fish being landed commercially, the increasing frequency of gear conflicts between monkfish vessels and those in other fisheries, and the expanding directed monkfish trawl fishery. In response, the Councils developed the joint FMP that was implemented in 1999. For the first eight years under the FMP, the fishery was in a rebuilding plan since the stocks were considered overfished (below the biomass target). The FMP was designed to stop overfishing and rebuild the stocks through a number of measures, including: limiting the number of vessels with access to the fishery and allocating days-at-sea (DAS) to those vessels; setting trip limits for vessels fishing for monkfish; minimum fish size limits; gear restrictions; mandatory time out of the fishery during the spawning season; and a framework adjustment process.

Reported landings of monkfish increased dramatically from the late 1970s until the mid-1990s and have remained high. Burgeoning markets for monkfish tails and livers in the 1980s allowed fishermen to fish profitably for monkfish, landing increasingly smaller monkfish as the stocks became depleted. Since the implementation of the FMP, however, vessels are more commonly landing large, whole monkfish for export to Asian markets. Revenues have generally increased since the mid-1980s and the relative value of monkfish has recently been at its highest point since 1996, despite a temporary drop in value during 2001-2003.

No recreational monkfish fishery exists. The two gears predominantly used in the directed monkfish fishery are bottom trawls and bottom gillnets. Trawl gear accounts for most of the landings in the northern area (75% during 2000-2006), while gillnets account for the majority of the landings in the southern area (54% during 2000-2006). During 2000-2006, 53% of U.S. monkfish landings were taken in otter trawls, 7% in sea scallop dredges, 35% in gillnets, and 5% in other gear (Northeast Data Poor Stocks Working Group, 2007). Dredges, spears, and hook gear are minor components of effort and landings in this fishery. All incidental loggerhead sea turtle takes in sea scallop dredges have been attributed to the sea scallop fishery in the Murray (2007) report which is the best estimate of loggerhead sea turtle bycatch in dredge gear since it represents the best available information and analysis for bycatch of loggerheads in mid and North Atlantic scallop dredge fisheries. Subsequently, the most recent ESA Section 7 Consultation on the Atlantic Sea Scallop Fishery Management Plan (NMFS 2008b) attributed incidental takes of all sea turtles species to the Atlantic sea scallop fishery. Therefore, takes of sea turtles in scallop dredges do not need to be evaluated in this Opinion. Because the vast majority of directed effort and landings for monkfish occurs with bottom trawls and gillnets, this Opinion will focus on potential effects from these gear types.

Most monkfish catch went unreported until the mid-1970s. Annual US commercial landings (live weight) increased to 6,000 mt in 1978 (NEFSC 2005), remained stable at 8,000-10,000 mt during the 1980s, and then increased rapidly in the 1990s, averaging 23,000 mt and peaking at 28,300 mt in 1997. The time period 2000-2005 had an annual average of 22,000 mt landed, and in 2006 declined to 14,500 mt, the lowest level since 1990, due to management regulations. Maximum sizes have also declined, from about 110 cm during the 1960s to 90 cm since the early 1990s in the north, and from about 100 cm in the 1960s to 75 cm since the 1990s in the south (Northeast Data Poor Stocks Working Group, 2007).

Amendment 1 to the Monkfish FMP, enacted April 1999, implemented the essential fish habitat (EFH) provisions of the Magnuson-Stevens Act. Amendment 2, which was implemented in May 2005, included restrictions on otter trawls in certain areas, made the minimum fish size consistent in all areas, closed two offshore canyons to monkfish fishing, created a monkfish research DAS set-aside program, and created new permit categories for fishing in designated areas, among other measures. In 2007, the Councils proposed Framework 4 to set catch targets (TTACs) at 5,000 mt and 5,100 mt for the Northern Fishery Management Area (NMA) and Southern Fishery Management Area (SMA), respectively. The Councils requested the Northeast Data Poor Stocks Working Group (DPWG) to evaluate the impact of applying those TTACs for the 2007-2009 fishing years. The DPWG concluded that under those catch targets, fishing mortality rates would remain below the threshold, and biomass would remain in an upward trend above the biomass target. Upon receiving the DPWG report, NMFS approved Framework 4 which included an automatic extension of the TTACs beyond FY2009 if the Councils did not adopt new targets.

In 2007, the Northeast Data Poor Stocks Working Group (DPWG) completed a monkfish stock assessment and recommended revisions to the biomass reference points. The Councils adopted the new reference points as Framework Adjustment 5 to the Monkfish FMP, implemented in May 2008, for monkfish stocks and both the northern and southern stock are considered "not overfished." Also in 2007, the Magnuson-Stevens Act was reauthorized (MSRA) and revised to include, among other things, the requirement that all FMPs establish Annual Catch Limits (ACLs) and measures to ensure accountability (AMs). For stocks not subject to overfishing, such as monkfish, the MSRA set a deadline of 2011 for the implementation of ACLs and AMs. In 2009, NMFS published revised National Standard 1 Guidelines which the Councils have used to develop ACLs and AMs for all FMPs.

Amendment 5 is currently being drafted. The Councils are addressing two primary purposes with this amendment: (1) to implement the MSRA mandated ACLs and AMs, and (2) to set the specifications of DAS, trip limits and other management measures to replace those adopted in Framework 4. The Councils are also proposing to make modifications to the FMP to improve the Research Set Aside (RSA) Program, to minimize bycatch resulting from trip limit overages, and to enable vessels to land monkfish heads separate from the body.

2.1.1 Summary of the Monkfish Fishery

Commercial fisheries for monkfish occur year round using gillnets, trawls, scallop dredges and other gear. Trawls and gillnets are the two gears used in the directed monkfish fishery. No recreational fishery exists. Peak fishing activity occurs during November through June, and value of the catch is highest in the fall due to the high quality of livers during this season.

U.S. fisheries for monkfish are managed under the Monkfish Fishery Management Plan (FMP) by the New England and Mid-Atlantic Fishery Management Councils. The primary goals of the Monkfish FMP are to end and prevent overfishing and to optimize yield and economic benefits to various fishing sectors involved with the fishery (NEFMC and MAFMC 1998). Current regulatory measures vary with type of permit but include limited access, limitations on days at sea, mesh size restrictions, trip limits, minimum size limits and other measures.

2.2 Action Area

The management unit for the Monkfish FMP is defined in the FMP as the range of the monkfish resource along the U.S. Atlantic Coast. When considering the range of the species covered in total, the overall range of monkfish is from Maine to Cape Hatteras, NC.

The direct and indirect effects of the monkfish fishery managed under the Monkfish 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 monkfish fishery (the action area) is the area in which the monkfish fishery operates, broadly defined as all United States Exclusive Economic Zone (EEZ) waters from Maine through Cape Hatteras, NC, and the adjoining state waters that are affected through the regulation of activities of Federal monkfish 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 species in a manner that will likely result in adverse effects:

North Atlantic right whale	(*Eubalaena glacialis*)	Endangered
Humpback whale	(*Megaptera novaengliae*)	Endangered
Fin whale	(*Balaenoptera physalus*)	Endangered
Sei whale	(*Balaenoptera borealis*)	Endangered
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*), 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 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 monkfish fishery does not operate in or near the rivers where concentrations of shortnose sturgeon are most likely found, it is highly unlikely that the monkfish fishery will affect shortnose sturgeon.

The wild populations of Atlantic salmon whose freshwater range occurs in the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River are listed as endangered under the ESA. Juvenile salmon in New England rivers typically migrate to sea in May after a two to three year period of development in freshwater streams, and remain at sea for two winters before returning to their U.S. natal rivers to spawn (Reddin 2006). The preferred habitat of post-smolt salmon in the open ocean is principally the upper 10 meters of the water column (ICES SGBYSAL, 2005); although there is evidence of forays into deeper water for shorter periods, in contrast adult Atlantic salmon demonstrate a wider depth profile (ICES SGBYSAL, 2005). Results from a 2001-2003 post-smolt trawl survey in the nearshore waters of the Gulf of Maine indicate that Atlantic salmon post-smolts are prevalent in the upper water column throughout this area in mid to late May (Lacroix and Knox 2005). Therefore, fishing close to the bottom, as practiced in the monkfish fishery, reduces the potential for catching Atlantic salmon as either post-smolts or adults. In addition, commercial fisheries deploying small

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.

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 post-smolts, however, the monkfish fishery does not occur in or near the rivers where concentrations of Atlantic salmon are likely to be found and generally use gear with larger mesh sizes that are not likely to catch salmon post-smolts.

In its report on salmon bycatch, the ICES Working Group for North Atlantic Salmon (WGNAS) concluded that bycatch of Atlantic salmon in Northeast Atlantic commercial fisheries was not an obvious concern for Atlantic salmon. The 2006 WGNAS report also discussed potential salmon bycatch implication from these fisheries and believed there is insufficient information to quantify bycatch although, based on the information that was available, there was no evidence of major bycatch of salmon in these Northeast fisheries. NMFS finds it is highly unlikely that the action being considered in this Opinion will harm or harass the Gulf of Maine DPS of Atlantic salmon. Thus, this species will not be considered further in this Opinion.

The hawksbill sea 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 individuals have been sighted along the east coast as far north as Massachusetts; however, sightings north of Florida are rare. Hawksbills have been described stranded as far north as Cape Cod, Massachusetts; however, many of these strandings were observed after hurricanes or offshore storms. Since operation of the monkfish fishery does not occur in waters that are typically used by hawksbill sea turtles, it is highly unlikely that the monkfish fishery will affect this turtle species.

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

Unlike blue whales, sperm whales do regularly occur in waters of the U.S. EEZ. However, the distribution of the sperm whale in the U.S. EEZ occurs on the continental shelf edge, over the continental slope, and into mid-ocean regions (Waring *et al.* 2007). In contrast, the monkfish fishery operates in continental shelf waters. The average depth of sperm whale sightings observed during the CeTAP surveys was 1,792m (CeTAP 1982). Female sperm whales and young males almost always inhabit waters deeper than 1000m and at latitudes less than 40° N (Whitehead 2002). Sperm whales feed on larger organisms that inhabit the deeper ocean regions

(Whitehead 2002). Calving for the species occurs in low latitude waters outside of the area where the monkfish fishery operates. Given that sperm whales are unlikely to occur in areas (based on water depth) where the monkfish fishery operates, and given that the operation of the monkfish fishery will not affect the availability of sperm whale prey or areas where calving and nursing of young occurs, NMFS has determined that the continued operation of the monkfish fishery is not likely to adversely affect sperm whales.

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

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

3.1 Status of Large Whales

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

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

3.1.1 North Atlantic right whales

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

The North Atlantic right whale (*Eubalaena glacialis*) has been listed as endangered under the Endangered Species Act (ESA) since 1973. It was originally listed as the "northern right whale" as endangered under the Endangered Species Conservation Act, the precursor to the ESA in June 1970. The species is also designated as depleted under the Marine Mammal Protection Act (MMPA).

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

The International Whaling Commission (IWC) recognizes two right whale populations in the North Atlantic: a western and eastern population (IWC 1986). It is thought that the eastern population migrated along the coast from northern Europe to Northwest Africa. The current distribution and migration patterns of the eastern North Atlantic right whale population, if extant, are unknown. Sighting surveys from the eastern Atlantic Ocean suggest that right whales present in this region are rare (Best *et al.* 2001) and it is unclear whether a viable population in the eastern North Atlantic still exists (Brown 1986, NMFS 1991a). Photo-identification work has shown that some of the whales observed in the eastern Atlantic were previously identified as western Atlantic right whales (Kenney 2002). This Opinion will focus on the North Atlantic right whale (*Eubalaena glacialis*) which occurs in the action area.

Habitat and Distribution

Western North Atlantic right whales generally occur from the Southeast U.S. to Canada (*e.g.*, Bay of Fundy and Scotian Shelf) (Kenney 2002; Waring *et al.* 2009). Like other right whale species, they follow an annual pattern of migration between low latitude winter calving grounds and high latitude summer foraging grounds (Perry *et al.* 1999; Kenney 2002).

The distribution of right whales seems linked to the distribution of their principal zooplankton prey, calanoid copepods (Winn *et al.* 1986; NMFS 2005a; Baumgartner and Mate 2005; Waring *et al.* 2009). Right whales are most abundant in Cape Cod Bay between February and April (Hamilton and Mayo 1990; Schevill *et al.* 1986; Watkins and Schevill 1982) and in the Great South Channel in May and June (Kenney *et al.* 1986; Payne *et al.* 1990; Kenney *et al.* 1995; Kenney 2001) where they have been observed feeding predominantly on copepods of the genera *Calanus* and *Pseudocalanus* (Baumgartner and Mate 2005; Waring *et al.* 2009). Right whales also frequent Stellwagen Bank and Jeffrey's Ledge, as well as Canadian waters including the Bay of Fundy and Browns and Baccaro Banks in the summer through fall (Mitchell *et al.* 1986; Winn *et al.* 1986; Stone *et al.* 1990). The consistency with which right whales occur in such locations is relatively high, but these studies also highlight the high interannual variability in right whale use of some habitats. Calving is known to occur in the winter months in coastal waters off of Georgia and Florida (Kraus *et al.* 1988). Calves have also been sighted off the coast of North Carolina during winter months suggesting the calving grounds may extend as far north as Cape Fear. In the North Atlantic it appears that not all reproductively active females return to the calving grounds each year (Kraus *et al.*, 1986; Payne, 1986). Patrician *et al.* (2009) analyzed photographs of a right whale calf sighted in the Great South Channel in June of 2007 and determined the calf appeared too young to have been born in the known southern calving area. Although it is possible the female traveled south to New Jersey or Delaware to give birth, evidence suggests that calving in waters of the Northeastern U.S. is possible. The location of some portion of the population during the winter months remains unknown (NMFS 2005a). However, recent aerial surveys conducted under the North Atlantic Right Whale Sighting Survey (NARWSS) program have indicated that some individuals may reside in the northern Gulf of Maine during the winter. In 2008 and 2009, right whales were sighted on Jeffrey's and Cashes Ledge, Stellwagen Bank, and Jordan Basin from December to February (Khan *et al.* 2009, 2010).

While right whales are known to congregate in the aforementioned areas, much is still not understood about their seasonal distribution and movements within and between these areas are extensive (Waring *et al.* 2009). In the winter, only a portion of the known right whale population is seen on the calving grounds. The winter distribution of the remaining right whales remains uncertain (NMFS 2005a, Waring *et al.* 2009). Results from winter surveys and passive acoustic studies suggest that animals may be dispersed in several areas including Cape Cod Bay (Brown *et al.* 2002) and offshore waters of the southeastern U.S. (Waring *et al.* 2009). On multiple days in December 2008, congregations of more than forty individual right whales were observed in the Jordan Basin area of the Gulf of Maine, leading researchers to believe this may be a wintering ground (NOAA 2008). Telemetry data have shown lengthy and somewhat distant excursions into deep water off of the continental shelf (Mate *et al.* 1997) as well as extensive movements over the continental shelf during the summer foraging period (Mate *et al.* 1992; Mate *et al.* 1997; Bowman *et al.* 2003; Baumgartner and Mate 2005). Knowlton *et al.* (1992) reported several long-distance movements as far north as Newfoundland, the Labrador Basin, and southeast of Greenland; in addition, resightings of photographically identified individuals have been made off Iceland, arctic Norway, and in the old Cape Farewell whaling ground east of Greenland. The Norwegian sighting (September 1999) represents one (1) of only two (2) sightings this century of a right whale in Norwegian waters, and the first since 1926. Together, these long-range matches indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not presently well described. Similarly, records from the Gulf of Mexico (Moore and Clark, 1963; Schmidly *et al.*, 1972) represent either geographic anomalies or a more extensive historic range beyond the sole known calving and wintering ground in the waters of the southeastern United States. The frequency with which right whales occur in offshore waters in the southeastern U.S. remains unclear (Waring *et al.* 2009).

Abundance estimates and trends

An estimate of the pre-exploitation population size for the North Atlantic right whale is not available. As is the case with most wild animals, an exact count of North Atlantic right whales cannot be obtained. However, abundance can be reasonably estimated as a result of the extensive study of the western North Atlantic right whale population. IWC participants from a 1999 workshop agreed to a minimum direct-count estimate of 263 right whales alive in 1996 and noted that the true population was unlikely to be greater than this estimate (Best *et al.* 2001). Based on a census of individual whales using photo-identification techniques and an assumption of mortality for those whales not seen in seven years, a total 299 right whales was estimated in 1998 (Kraus *et al.* 2001), and a review of the photo-ID recapture database on October 10, 2008, indicated that 345 individually recognized whales were known to be alive during 2005 (Waring *et al.* 2009). Because this 2008 review was a nearly complete census, it is assumed this estimate represents a minimum population size. The minimum number alive population index for the years 1990-2005 suggests a positive trend in numbers. These data reveal a significant increase in the number of catalogued whales alive during this period, but with significant variation due to apparent losses exceeding gains during 1998-1999. Mean growth rate for the period 1990-2005 was 1.8% (Waring *et al.* 2009).

A total of 235 right whale calves have been born from 1993-2007 (Waring *et al.* 2009). The mean calf production for the 15-year period from 1993-2007 is estimated to be 15.6/year

(Waring *et al.* 2009). Calving numbers have been sporadic, with large differences among years, including a record calving season in 2000/2001 with 31 right whale births (Waring *et al.* 2009). The three calving years (97/98; 98/99; 99/00) prior to this record year provided low recruitment levels with a total of only 11 calves born. The calving seasons from 2000-2007 have been remarkably better with 31, 21, 19, 17, 28, 19, and 23 births, respectively (Waring *et al.* 2009). A calf count for the 2008/2009 season indicates a new record calving season of 39 calves (Zoodsma, pers. comm.). However, the western North Atlantic stock has also continued to experience losses of calves, juveniles and adults.

As is the case with other mammalian species, there is an interest in monitoring the number of females in this western North Atlantic right whale population since their numbers will affect the population trend (whether declining, increasing or stable). Kraus *et al.* (2007) reported that as of 2005, 92 reproductively-active females had been identified and Schick *et al.* (2009) estimated 97 breeding females. From 1983-2005, the number of new mothers recruited to the population (with an estimated age of 10 for the age of first calving), varied from 0-11 each year with no significant increase or decline over the period (Kraus *et al.* 2007). Between 1980 and 2005, 16 right whales had produced at least six (6) calves each, and four (4) cows had at least seven (7) calves. Two (2) of these cows were at an age which indicated a reproductive life span of at least 31 years (Kraus *et al.* 2007). As described above, the 2000/2001 - 2006/2007 calving seasons had relatively high calf production and have included additional first time mothers (*e.g.*, eight new mothers in 2000/2001). However, over the same time period there have been continued losses to the western North Atlantic right whale population including the death of mature females as a result of anthropogenic mortality (like that described in Glass *et al.* 2009, below). Of the 15 serious injuries and mortalities between 2003-2007, at least nine (9)were adult females, three (3) of which were carrying near-term fetuses and four (4) of which were just starting to bear calves (Waring *et al.* 2009). Since the average lifetime calf production is 5.25 calves (Fujiwara and Caswell 2001), depending on how many calves each female previously had, the deaths of these nine females may represent a loss of reproductive potential of as many as 47 animals. However, it is important to note that not all right whale mothers are equal with regards to calf production. Right whale #1158 had only one (1) calf over a 25-year period (Kraus *et al.* 2007). In contrast, one of the largest right whales on record was a female nicknamed "Stumpy," who was killed in February 2004 of an apparent ship strike (NMFS 2006a). She was first sighted in 1975 and known to be a prolific breeder, successfully rearing calves in 1980, 1987, 1990, 1993, and 1996 (Moore *et. al* 2007). At the time of her death, she was estimated to be 30 years of age and carrying her sixth calf; the near-term fetus also died (NMFS 2006a).

Abundance estimates are an important part of assessing the status of the species. However, for Section 7 purposes, the population trend (*i.e.,* whether increasing or declining) provides better information for assessing the effects of a proposed action on the species. As described in previous Opinions, data collected in the 1990s suggested that right whales were experiencing a slow but steady recovery (Knowlton *et al.* 1994). However, Caswell *et al.* (1999) used photo-identification data and modeling to estimate survival and concluded that right whale survival decreased from 1980 to 1994. Modified versions of the Caswell *et al.* (1999) model as well as several other models were reviewed at the 1999 IWC workshop (Best *et al.* 2001). Despite differences in approach, all of the models indicated a decline in right whale survival in the 1990s relative to the 1980s with female survival, in particular, apparently affected (Best *et al.* 2001, Waring *et al.* 2009). In 2002, NMFS' NEFSC hosted a workshop to review right whale population models to examine: (1) potential bias in the models and (2) changes in the subpopulation trend based on new information collected in the late 1990s (Clapham *et al.* 2002). Three different models were used to explore right whale survivability and to address potential sources of bias. Although biases were identified that could negatively affect the results, all three modeling techniques resulted in the same conclusion; survival had continued to decline in the 1990s and seemed to greatly affect females (Clapham *et al.* 2002). Mortalities, including those in the first half of 2005, suggest an increase in the annual mortality rate (Kraus *et. al* 2005). Calculations indicate that this increased mortality rate would reduce population growth by approximately 10% per year (Kraus *et. al* 2005). Despite the preceding, examination of the minimum number alive population index calculated from the individual sightings database, as it existed on 10 October 2008, for the years 1990-2005 suggest a positive trend in numbers (Waring *et al.* 2009). These data reveal a significant increase in the number of catalogued whales alive during this period, but with significant variation due to apparent losses exceeding gains during 1998-1999 (Waring *et al.* 2009). Recently, NMFS NEFSC developed a population viability analysis (PVA) to examine the influence of anthropogenic mortality reduction on the recovery prospects for the species (Pace, in review). The PVA evaluated several scenarios on how the populations would fare without entanglement mortalities compared to the status quo. Only 2 of 1000 projections (with the status quo simulation) ended with a smaller total population size than they started and zero projections resulted in extinction. As described above, the mean growth rate estimated in the latest stock assessment report, for the period 1990-2005, was 1.8% (Waring *et al.* 2009).

Reproductive Fitness
Healthy reproduction is critical for the recovery of the North Atlantic right whale (Kraus *et al.* 2007). Researchers have suggested that the population has been affected by a decreased reproductive rate (Best *et al.* 2001; Kraus *et al.* 2001). Kraus *et al.* (2007) reviewed reproductive parameters for the period 1983-2005, and estimated calving intervals to have changed from 3.5 years in 1990 to over five years between 1998-2003, and then decreased to just over 3 years in 2004 and 2005.

Factors that have been suggested as affecting the right whale reproductive rate include reduced genetic diversity (and/or inbreeding), contaminants, biotoxins, disease, and nutritional stress. Although it is believed that a combination of these factors is likely causing an effect on right whales (Kraus *et al.* 2007), there is currently no evidence available to determine their potential effect, if any. The dramatic reduction in the North Atlantic right whale population believed to have occurred due to commercial whaling may have resulted in a loss of genetic diversity which could affect the ability of the current population to successfully reproduce (*i.e.*, decreased conceptions, increased abortions, and increased neonate mortality). One hypothesis is that the low level of genetic variability in this species produces a high rate of mate incompatibility and unsuccessful pregnancies (Frasier *et al.* 2007). Analyses are currently under way to assess this relationship further as well as the influence of genetic characteristics on the potential for species recovery (Frasier *et al.* 2007). Studies by Schaeff *et al.* (1997) and Malik *et al.* (2000) indicate that western North Atlantic right whales are less genetically diverse than southern right whales. However, several apparently healthy populations of cetaceans, such as sperm whales and pilot whales, have even lower genetic diversity than observed for western North Atlantic right whales

(IWC 2001). Similarly, while contaminant studies have confirmed that right whales are exposed to and accumulate contaminants, researchers could not conclude that these contaminant loads were negatively affecting right whale reproductive success since concentrations were lower than those found in marine mammals proven to be affected by PCBs and DDT (Weisbrod *et al.* 2000). Another suite of contaminants (i.e. antifouling agents and flame retardants) that have been proven to disrupt reproductive patterns and have been found in other marine animals, have raised new concerns (Kraus *et al.* 2007). Recent data also support a hypothesis that chromium, an industrial pollutant, may be a concern for the health of the North Atlantic right whales and that inhalation may be an important exposure route (Wise *et al.* 2008). A number of diseases could be also affecting reproduction, however tools for assessing disease factors in free-swimming large whales currently do not exist (Kraus *et al.* 2007). Once developed, such methods may allow for the evaluation of disease effects on right whales. Impacts of biotoxins on marine mammals are also poorly understood, yet data is showing that marine algal toxins may play significant roles in mass mortalities of large whales (Rolland *et al.* 2007). Although there are no published data concerning the effects of biotoxins on right whales, researchers are now certain that right whales are being exposed to measurable quantities of paralytic shellfish poisoning (PSP) toxins and domoic acid via trophic transfer through the presence of these biotoxins in prey upon which they feed (Durbin *et al.* 2002, Rolland *et al.* 2007).

Data to indicate whether right whales are food-limited are difficult to evaluate (Kraus *et al.* 2007). Although North Atlantic right whales seem to have thinner blubber than right whales from the South Atlantic (Kenney 2002), there is no evidence at present to demonstrate that birth rates and calving intervals are related to food abundance. However, modeling work by Caswell *et al.* (1999) and Fujiwara and Caswell (2001) suggests that the North Atlantic Oscillation (NAO), a naturally occurring climatic event, does affect the survival of mothers and the reproductive rate of mature females, and it also seems to affect calf survival (Clapham *et al.* 2002). Greene *et al.* (2003) described the potential oceanographic processes linking climate variability to the reproduction of North Atlantic right whales. Climate-driven changes in ocean circulation have had a significant impact on the plankton ecology of the Gulf of Maine, including effects on *Calanus finmarchicus*, a primary prey resource for right whales. Researchers found that during the 1980s, when the NAO index was predominately positive, *C. finmarchicus* abundance was also high; when a record drop occurred in the NAO index in 1996, *C. finamarchicus* abundance levels also decreased significantly. Right whale calving rates since the early 1980s seem to follow a similar pattern, where stable calving rates were noted from 1982-1992, but then two major, multi-year declines occurred from 1993-2001, consistent with the drops in copepod abundance. It has been hypothesized that right whale calving rates are thus a function of food availability as well as the number of females available to reproduce (Greene et al 2003, Greene and Pershing 2004). Such findings suggest that future climate change may emerge as a significant factor influencing the recovery of right whales. Some believe the effects of increased climate variability on right whale calving rates should be incorporated into future modeling studies so that it may be possible to determine how sensitive right whale population numbers are to variable climate forcing (Greene and Pershing 2004).

Anthropogenic Mortality

There is general agreement that right whale recovery is negatively affected by anthropogenic mortality. From 2003-2007, right whales had the highest proportion of entanglement and ship strike events relative to the number of total events (mortality, entanglement or ship strike) for any species of large whale (Glass *et al.* 2009). Given the small population size and low annual reproductive rate of right whales, human sources of mortality may have a greater effect to relative population growth rate than for other large whale species (Waring *et al.* 2009). For the period 2003-2007, the annual mortality and serious injury rate for the North Atlantic right whale averaged 3.0 per year (2.2 in U.S. waters; 0.8 in Canadian waters) (Glass *et al.* 2009, Waring *et. al.* 2009). Twenty confirmed right whale mortalities were reported along the U.S. east coast and adjacent Canadian Maritimes from 2003-2007 (Glass *et al.* 2009). These numbers represent the minimum values for human-caused mortality for this period. Given the range and distribution of right whales in the North Atlantic, and the fact that positively buoyant species like right whales may become negatively buoyant if injury prohibits effective feeding for prolonged periods, it is highly unlikely that all carcasses will be observed (Moore *et. al.* 2004, Glass *et al.* 2009). Moreover, carcasses floating at sea often cannot be examined sufficiently and cause of death may be unknown if they are not towed to shore for further necropsy (Glass *et al.* 2009). Decomposed and/or unexamined animals represent lost data, some of which may relate to human impacts (Waring *et al.* 2009).

Considerable effort has been made to examine right whale carcasses for the cause of death (Moore *et al.* 2004). Because they exist in an ocean environment, examining right whale carcasses is often very difficult. Some carcasses are discovered floating at sea and cannot be retrieved. Others are in such an advanced stage of decomposition when discovered that a complete examination is not possible. Wave action and post-mortem predation by sharks can also damage carcasses and preclude a thorough examination of all body parts. It should also be noted that mortality and serious injury judgments are based upon the best available data and additional information may result in revisions (Glass *et al.* 2009). Of the 20 total, confirmed right whale mortalities (2003-2007) described in Glass *et al.* (2009), 3 were confirmed to be entanglement mortalities (1 adult female, 1 female calf, 1 male calf) and 9 were confirmed to be ship strike mortalities (6 adult females, 1 female of unknown age, 1 male calf, and 1 yearling male). Serious injury involving right whales was documented for 1 entanglement event (adult female) and 2 ship strike events (1 adult female and 1 yearling male).

Although disentanglement is either unsuccessful or not possible for the majority of cases, during the period of 2003-2007, there were at least 4 documented cases of entanglements for which the intervention of disentanglement teams averted a likely serious injury (Waring *et al.* 2009). Even when entanglement or vessel collision does not cause direct mortality, it may weaken or otherwise affect individuals so that further injury or death is likely (Waring *et. al* 2009). Some right whales that have been entangled were subsequently involved in ship strikes (Hamilton *et al.* 1998) suggesting that the animal may have become debilitated by the entanglement to such an extent that it was less able to avoid a ship. Similarly, skeletal fractures and/or broken jaws sustained during a vessel collision may heal, but then compromise a whale's ability to efficiently filter feed (Moore *et al.* 2007). A necropsy of right whale #2143 ("Lucky") found dead in January 2005 suggested the animal (and her near-term fetus) died after healed propeller wounds from a previous ship strike re-opened and became infected as a result of pregnancy (Moore *et al.* 2007, Glass *et al.* 2008). Sometimes, even with a successful disentanglement, an animal may die of injuries sustained by fishing gear (e.g. RW #3107) (Waring *et al.* 2009).

Entanglement records from 1990-2007 maintained by NMFS include 46 confirmed right whale entanglement events (Waring *et al.* 2009). Because whales often free themselves of gear following an entanglement event, scarification analysis of living animals may provide better indications of fisheries interactions rather than entanglement records (Waring *et al.* 2009). Data presented in Knowlton *et al.* 2008 indicate the annual rate of entanglement interaction remains at high levels. Four hundred and ninety-three (493) individual, catalogued right whales were reviewed and 625 separate entanglement interactions were documented between 1980 and 2004. Approximately 358 out of 493 animals (72.6% of the population) were entangled at least once; 185 animals bore scars from a single entanglement, however one (1) animal showed scars from six (6) different entanglement events. The number of male and female right whales bearing entanglement scars was nearly equivalent (142/202 females, 71.8%; 182/224 males, 81.3%), indicating that right whales of both sexes are equally vulnerable to entanglement. However, juveniles appear to become entangled at a higher rate than expected if all age groups were equally vulnerable. For all years but one (1998), the proportion of juvenile, entangled right whales exceeded their proportion within the population. Based on photographs of catalogued animals from 1935 through 1995, Hamilton *et al.* (1998) estimated that 6.4% of the North Atlantic right whale population exhibit signs of injury from vessel strikes. Reports received from 2003-2007 indicate that right whales had the greatest number of ship strike mortalities (n=9) and serious injuries (n=2) compared to other large whales in the Northwest Atlantic (Glass *et al.* 2009). In 2006 alone, four (4) reported mortalities and one (1) serious injury resulted from right whale ship strikes (Glass *et al.* 2009).

Summary of Right Whale Status

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

Previous models estimated that the right whale population in the Atlantic numbered 300 (+/- 10%) (Best *et al.* 2001). However, a review of the photo-ID database on October 10, 2008 indicated that 345 individually recognized right whales were known to be alive in 2005 (Waring *et al.* 2009). The 2000/2001 - 2007/2008 calving seasons have had relatively high calf production (31, 21, 19, 17, 28,19, and 23 calves, respectively) and have included additional first time mothers (*e.g.*, eight (8) new mothers in 2000/2001) (Waring *et al.* 2009). There are some indications that climate-driven ocean changes impacting the plankton ecology of the Gulf of Maine, may, in some manner, be affecting right whale fitness and reproduction. However, there is also general agreement that right whale recovery is negatively affected by human sources of mortality, which may have a greater impact on population growth rate given the small population size and low annual reproductive rate of right whales (Waring *et al.* 2009). Of particular concern is the death of mature females. Recent mortality records include at least six (6) adult females, three (3) of which were carrying near-term fetuses and four (4) of which were just starting to bear calves (Glass *et al.* 2009).

Over the five-year period 2003-2007, right whales had the highest proportion of entanglements and ship strikes relative to the number of reports for a species: of 58 reports involving right whales, 20 were confirmed entanglements and 17 were confirmed ship strikes. There were 20 verified right whale mortalities, three (3) due to entanglements, and nine due to ship strikes (Glass *et al.* 2009). This represents an absolute minimum number of the right whale mortalities for this period. Given the range and distribution of right whales in the North Atlantic, it is highly unlikely that all carcasses will be observed. Scarification analysis indicates that some whales do survive encounters with ships and fishing gear. However, the long-term consequences of these interactions are unknown.

A variety of modeling exercises and analyses indicate that survival probability declined in the 1990s (Best *et al.* 2001), and mortalities in 2004-2005, including a number of adult females, also suggested an increase in the annual mortality rate (Kraus *et al.* 2005). Nonetheless, a census of the minimum number of right whales alive based on the photo-ID catalog as it existed on October 10, 2008, indicates a positive trend in numbers for the years 1990-2005 (Waring *et al.* 2009). In addition, calving intervals appear to have declined to 3 years in recent years (Kraus *et al.* 2007), and calf production has been relatively high over the past several seasons. Based on the information currently available, for the purposes of this Opinion, NMFS believes that the western North Atlantic right whale subpopulation is increasing.

The draft 2010 SAR (Waring *et al.* 2010) for the western stock of North Atlantic right whales reports an increase in the minimum population size (361), the average annual calf production (17.2), and the average growth rate (2.1%). The Draft SAR also assigned a PBR of 0.7 to this stock of right whales. Overall documented serious injury and mortality to right whales decreased to an average rate of 2.8 per year. Incidental fishery entanglement records and ship strike records for the period 2004 through 2008 averaged of 0.8 (U.S. waters 0.6) and 2.0 (U.S. waters, 1.6) respectively per year. The preliminary data from the Draft 2010 SAR is consistent with the 2009 SAR and provides additional indications of an increasing population size of and positive growth rate for North Atlantic right whales.

3.1.2 Humpback whales

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

North Pacific, Northern Indian Ocean and Southern Hemisphere

Humpback whales in the North Pacific feed in coastal waters from California to Russia and in the Bering Sea. They migrate south to wintering destinations off Mexico, Central America, Hawaii, southern Japan, and the Philippines (Carretta *et al.* 2009). Although the IWC only considered one stock (Donovan 1991) there is evidence to indicate multiple populations migrating between their respective summer/fall feeding areas to winter/spring calving and mating areas within the North Pacific Basin (Angliss and Outlaw 2007, Carretta *et al.* 2008). Within the Pacific Ocean, NMFS recognizes three management units within the U.S. EEZ for the purposes of managing this species under the MMPA. These are: the eastern North Pacific stock (feeding areas off the US West coast), the central North Pacific stock (feeding areas from Southeast Alaska to the Alaska Peninsula) and the western North Pacific stock (feeding areas from the Aleutian Islands, the Bering Sea, and Russia) (Carretta *et al.* 2009). Because fidelity appears to be greater in feeding areas than in breeding areas, the stock structure of humpback whales is defined based on feeding areas (Carretta *et al.* 2009). Recent research efforts via the Structure of Populations, Levels of Abundance, and Status of Humpback Whales (SPLASH) Project estimate the abundance of humpback whales to be just under 20,000 whales for the entire North Pacific, a number which doubles previous population predictions (Calambokidis *et al.* 2008). There are indications that the eastern North Pacific stock was growing in the 1980's and early 1990's with a best estimate of 8% growth per year (Carretta *et al.* 2009). The minimum population for the eastern North Pacific stock is 1,391 whales (Carretta *et al.* 2009). The central North Pacific stock is minimally at 4,005 animals (Allen and Angliss 2010), and various studies report that it appears to have increased in abundance at rates between 6.6%-10% per year (Allen and Angliss 2010). Although there is no reliable population trend data for the western North Pacific stock, as surveys of the known feeding areas are incomplete and many feeding areas remain unknown, minimum population size is currently estimated at 367 whales (Allen and Angliss 2010).

The Northern Indian Ocean population of Humpback whales consists of a resident stock in the Arabian Sea, which apparently do not migrate (Minton *et al.* 2008). The lack of photographic matches with other areas suggests this is an isolated subpopulation. The Arabian Sea subpopulation of humpback whales is geographically, demographically and genetically isolated, reside year round in sub-tropical waters of the Arabian Sea (Minton *et al.* 2008). Although potentially an underestimate due to small sample sizes and insufficient spatial and temporal coverage of the population's suspected range, based on photo-identification, the abundance estimate off the coast of Oman is 82 animals [95% confidence interval (CI)](Minton *et al.* 2008).

The Southern Hemisphere population of humpback whales are known to feed mainly in the Antartic, although some have been observed feeding in the Benguela Current ecosystem on the migration route west of South Africa (Reilly *et al.* 2008a). The IWC Scientific Committee recognizes seven major breeding stocks, some of which are tentatively further subdivided into substocks. The seven major breeding stocks, with their repective breeding ground estimates in paranthesis, include Southwest Atlantic (6,251), Southeast Atlantic (1,594), southwestern Indian Ocean (5,965), southeastern Indian Ocean (10,032), Southwest Pacific (7,472), central South Pacific (not available), and southeast Pacific (2,917) (Reilly *et al.* 2008a). The total abundance estimate of 36,600 humpback whales for the Southern Hemisphere is negatively biased due to no available abundance estimate for the central South Pacific subpopulation and only a partial estimate for the Southeast Atlantic subpopulation. Additionally, these abundance estimates have been obtained on each subpopulations wintering grounds, and the possibility exists that the entire population does not migrate to the wintering grounds (Reilly *et al.* 2008a).

Like other whales, southern hemisphere humpback whales were heavily exploited for commercial whaling. Although they were given protection by the IWC in 1963, Soviet whaling data made available in the 1990's revealed that 48,477 southern hemisphere humpback whales were taken from 1947-1980, contrary to the original reports to the IWC which accounted for the take of only 2,710 humpbacks (Zemsky *et al.* 1995, IWC 1995, Perry *et al.* 1999).

Gulf of Maine (North Atlantic)
Humpback whales from most Atlantic feeding areas calve and mate in the West Indies and migrate to feeding areas in the northwestern Atlantic during the summer months. Most of the humpbacks that forage in the Gulf of Maine visit Stellwagen Bank and the waters of Massachusetts and Cape Cod Bays. Previously, the North Atlantic humpback whale population was treated as a single stock for management purposes, however due to the strong fidelity to the region displayed by many whales, the Gulf of Maine stock was reclassified as a separate feeding stock (Waring *et al.* 2009). The Gulf of St. Lawrence, Newfoundland/Labrador, western Greenland, Iceland and northern Norway are the other regions that represent relatively discrete subpopulations. Sightings are most frequent from mid-March through November between 41°N and 43°N, from the Great South Channel north along the outside of Cape Cod to Stellwagen Bank and Jeffrey's Ledge (CeTAP 1982) and peak in May and August. Small numbers of individuals may be present in this area year-round, including the waters of Stellwagen Bank. They feed on a number of species of small schooling fishes, particularly sand lance and Atlantic herring, targeting fish schools and filtering large amounts of water for their associated prey. It is hypothesized humpback whales may also feed on euphausiids (krill) as well as capelin (Waring *et al.* 2009, Stevick *et al.* 2006).

In winter, whales from waters off New England, Canada, Greenland, Iceland, and Norway migrate to mate and calve primarily in the West Indies where spatial and genetic mixing among these groups does occur (Waring *et al.* 2009). Various papers (Clapham and Mayo 1990; Clapham 1992; Barlow and Clapham 1997; Clapham *et al.* 1999) summarize information gathered from a catalogue of photographs of 643 individuals from the western North Atlantic population of humpback whales. These photographs identified reproductively mature western North Atlantic humpbacks wintering in tropical breeding grounds in the Antilles, primarily on Silver and Navidad Banks, north of the Dominican Republic. The primary winter range also includes the Virgin Islands and Puerto Rico (NMFS 1991b).

Humpback whales use the Mid-Atlantic as a migratory pathway to and from the calving/mating grounds, but it may also be an important winter feeding area for juveniles. Since 1989, observations of juvenile humpbacks in the Mid-Atlantic have been increasing during the winter months, peaking January through March (Swingle *et al.* 1993). Biologists theorize that non-reproductive animals may be establishing a winter feeding range in the Mid-Atlantic since they are not participating in reproductive behavior in the Caribbean. Swingle *et al.* (1993) identified a shift in distribution of juvenile humpback whales in the nearshore waters of Virginia, primarily in winter months. Identified whales using the Mid-Atlantic area were found to be residents of the Gulf of Maine and Atlantic Canada (Gulf of St. Lawrence and Newfoundland) feeding groups, suggesting a mixing of different feeding populations in the Mid-Atlantic region. Strandings of humpback whales have increased between New Jersey and Florida since 1985 consistent with the increase in Mid-Atlantic whale sightings. Strandings were most frequent during September through April in North Carolina and Virginia waters, and were composed primarily of juvenile humpback whales of no more than 11 meters in length (Wiley *et al.* 1995).

Photographic mark-recapture analyses from the Years of the North Atlantic Humpback (YONAH) project gave an ocean-basin-wide estimate of 11,570 animals during 1992/1993 and an additional genotype-based analysis yielded a similar but less precise estimate of 10,400 whales (95% CI = 8,000 - 13,600) (Waring *et al.* 2009). For management purposes under the MMPA, the estimate of 11,570 individuals is regarded as the best available estimate for the North Atlantic population (Waring *et al.* 2009). The best, recent estimate for the Gulf of Maine stock is 847 whales, derived from the 2006 aerial survey (Waring *et al.* 2009).

As is the case with other large whales, the major known sources of anthropogenic mortality and injury of humpback whales occur from fishing gear entanglements and ship strikes. For the period 2003 through 2007, the minimum annual rate of human-caused mortality and serious injury to the Gulf of Maine humpback whale stock averaged 4.4 animals per year (U.S. waters, 4.0; Canadian waters, 0.4) (Glass *et al.* 2009, Waring *et al.* 2009). Between 2003 and 2007 humpback whales were involved in 76 confirmed entanglement events and 11 confirmed ship strike events (Glass *et al.* 2009). Over the five-year period, humpback whales were the most commonly observed entangled whale species; entanglements accounted for 4 mortalities and 10 serious injuries (Glass *et al.* 2009). Although ship strikes were relatively uncommon, 8 of the 11 confirmed events were fatal (Glass *et al.* 2009). As of May 2009, all of the available information indicated that the events described here involved animals from the Gulf of Maine stock (Glass *et al.* 2009). There were also many carcasses that washed ashore or were spotted floating at sea for which the cause of death could not be determined. Decomposed and/or unexamined animals (e.g., carcasses reported but not retrieved or no necropsy performed) represent 'lost data' some of which may relate to human impacts (Glass *et al.* 2009, Waring *et al.* 2009).

Based on photographs taken between 2000-2002 of the caudal peduncle and fluke of humpback whales, Robbins and Mattila (2004) estimated that at least half (48-57%) of the sample (187 individuals) was coded as having a high likelihood of prior entanglement. Evidence suggests that entanglements have occurred at minimum rate of 8-10% per year. Scars acquired by Gulf of Maine stock of humpback whales between 2000 and 2002 suggest a minimum of 49 interactions with gear took place. Based on composite scar patterns, it was believed that male humpback whales were more vulnerable to entanglement than females. Males may be subject to other sources of injury that could affect scar pattern interpretation. Images were obtained from a humpback whale breeding ground; 24% exhibited raw injuries, presumable a result from agonistic interactions. However, current evidence suggests that breeding ground interactions alone cannot explain the higher frequency of healed scar patterns among Gulf of Maine stock male humpback whales (Robbins and Matilla 2004).

Humpback whales, like other baleen whales, may also be adversely affected by habitat degradation, habitat exclusion, acoustic trauma, harassment, or reduction in prey resources due to trophic effects resulting from a variety of activities including fisheries operations, vessel traffic, and coastal development. Currently, there is no evidence that these types of activities are affecting humpback whales. However, Geraci *et al.* (1989) provide strong evidence that a mass mortality of humpback whales from 1987-1988 resulted from the consumption of mackerel whose livers contained high levels of saxitoxin, a naturally occurring red tide toxin, the origin of which remains unknown. It has been suggested that the occurrence of a red tide event is related to an increase in freshwater runoff from coastal development, leading some observers to suggest that such events may become more common among marine mammals as coastal development continues (Clapham *et al.* 1999). There have been three additional known cases of a mass mortality involving large whale species along the East coast between 1998 and 2008. In the 2006 mass mortality event, 21 dead humpback whales were found between July 10 and December 31, 2006, triggering NMFS to declare an unusual mortality event (UME) for humpback whales in the Northeast United States. The UME was officially closed on December 31, 2007 after a review of 2007 humpback whale strandings and mortality showed that the elevated numbers were no longer being observed. The cause of the 2006 UME has not been determined to date, although investigations are ongoing.

Changes in humpback distribution in the Gulf of Maine have been found to be associated with changes in herring, mackerel, and sand lance abundance associated with local fishing pressures (Stevick *et al.* 2006, Waring *et al.* 2009). Shifts in relative finfish species abundance correspond to changes in observed humpback whale movements (Stevick *et al.* 2006). However, there is no evidence that humpback whales were adversely affected by these trophic changes.

Summary of Humpback Whale Status
The best available population estimate for humpback whales in the North Atlantic Ocean is 11,570 animals, and the best, recent estimate for the Gulf of Maine stock is 847 whales (Waring *et al.* 2009). Anthropogenic mortality associated with fishing gear entanglements and ship strikes remains significant. In the winter, mating and calving occurs in areas located outside of the United States where the species is afforded less protection. Despite all of these factors, current data suggest that the Gulf of Maine humpback stock is steadily increasing in size (Waring *et al.* 2009). Population modeling, using data obtained from photographic mark-recapture studies, estimates the growth rate of the Gulf of Maine stock to be at 6.5% for the period 1979-1991 (Barlow and Clapham 1997). More recent analysis for the period 1992-2000 estimated lower population growth rates ranging from 0% to 4.0%, depending on calf survival rate (Clapham *et al.* 2003 in Waring *et al.* 2009). However, it is unclear whether the apparent decline in growth rate is a bias result due to a shift in distribution documented for the period 1992-1995, or whether the population growth rates truly declined due to high mortality of young-of-the-year whales in US Mid-Atlantic waters (Waring *et al.* 2009). Regardless, calf survival appears to have increased since 1996, presumably accompanied by an increase in population growth (Waring *et al.* 2009). Stevick *et al.* (2003) calculated an average population growth rate of 3.1% in the North Atlantic population overall for the period 1979-1993. With respect to the species overall, there are also indications of increasing abundance for the eastern and central North Pacific stocks, and Southern Hemisphere stocks: Southwest Atlantic, Southeast Atlantic, Southwest Indian Ocean, Southeast Indian Ocean, and Southwest Pacific. Trend data is lacking for the western North Pacific stock, the central South Pacific and Southeast Pacific subpopulations of the southern hemisphere humpback whales, and the northern Indian Ocean humpbacks. Therefore, given the best available information, for the purposes of this biological opinion, NMFS believes the humpback whale population is increasing.

Compared to the final 2009 SAR, the draft 2010 SAR (Waring *et al.* 2010) for the Gulf of Maine stock of humpback whales reports the same minimum population size, average annual calf production, average growth rate, and PBR. Overall documented serious injury and mortality to humpback whales increased by 0.2 to an average rate of 4.6 per year over the time period 2004 through 2008. Incidental fishery entanglement records and ship strike records for the period 2004 through 2008 averaged of 3.0 (U.S. waters, 2.8) and 1.6 (U.S. waters, 1.6) respectively per year. Consistent with the 2009 final SAR, the draft 2010 SAR concludes that the Gulf of Maine humpback whale stock is steadily increasing in size.

3.1.3 Fin Whales

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

Pacific Ocean

Within US waters of the Pacific, fin whales are found seasonally off the coast of North America and Hawaii and in the Bering Sea during the summer (Allen and Angliss 2010). Although stock structure in the Pacific is not fully understood, NMFS recognizes three fin whale stocks in the US Pacific waters for the purposes of managing this species under the MMPA. These are: Alaska (Northeast Pacific), California/Washington/Oregon, and Hawaii (Carretta *et al.* 2009). Reliable estimates of current abundance for the entire Northeast Pacific fin whale stock are not available (Allen and Angliss 2010). A provisional population estimate of 5,700 was calculated for the Alaska stock west of the Kenai Peninsula by adding estimates from multiple surveys (Allen and Angliss 2010). This can be considered a minimum estimate for the entire stock because it was estimated from surveys that covered only a portion of the range of the species (Allen and Angliss 2010). An annual population increase of 4.8% between 1987-2003 was estimated for fin whales in coastal waters south of the Alaska Peninsula (Allen and Angliss 2010). This is the first estimate of population trend for North Pacific fin whales; however, it must be interpreted cautiously due to the uncertainty in the initial population estimate and the population structure (Allen and Angliss 2010). The best available estimate for the California/Washington/Oregon stock is 2,636, which is likely an underestimate (Carretta *et al.* 2009). The best available estimate for the Hawaii stock is 174, based on a 2002 line-transect survey (Carretta *et al.* 2009).

Stock structure for fin whales in the southern hemisphere is unknown. Prior to commercial exploitation, the abundance of southern hemisphere fin whales is estimated to have been at 400,000 (IWC 1979, Perry *et al.* 1999). There are no current estimates of abundance for southern hemisphere fin whales. Since these fin whales do not occur in US waters, there is no recovery plan or stock assessment report for the southern hemisphere fin whales.

North Atlantic
NMFS has designated one population of fin whale in US waters of the North Atlantic (Waring *et al.* 2009). This species is commonly found from Cape Hatteras northward. A number of researchers have suggested the existence of fin whale subpopulations in the North Atlantic based on local depletions resulting from commercial overharvesting (Mizroch and York 1984) or genetics data (Bérubé *et al.* 1998). Photo-identification studies in western North Atlantic feeding areas, particularly in Massachusetts Bay, have shown a high rate of annual return by fin whales, both within years and between years (Seipt *et al.* 1990) suggesting some level of site fidelity. The Scientific Committee of the International Whaling Commission (IWC) has proposed stock boundaries for North Atlantic fin whales. Fin whales off the eastern United States, Nova Scotia and southeastern coast of Newfoundland are believed to constitute a single stock of fin whales under the present IWC scheme (Donovan 1991). However, it is uncertain whether the proposed boundaries define biologically isolated units (Waring *et al.* 2009).

During 1978-1982 aerial surveys, fin whales accounted for 24% of all cetaceans and 46% of all large cetaceans sighted over the continental shelf between Cape Hatteras and Nova Scotia (Waring *et al.* 2009). Underwater listening systems have also demonstrated that the fin whale is the most acoustically common whale species heard in the North Atlantic (Clark 1995). The single most important area for this species appeared to be from the Great South Channel, along the 50m isobath past Cape Cod, over Stellwagen Bank, and past Cape Ann to Jeffrey's Ledge (Hain *et al.* 1992).

Like right and humpback whales, fin whales are believed to use North Atlantic waters primarily for feeding, and more southern waters for calving. However, evidence regarding where the majority of fin whales winter, calve, and mate is still scarce. Clark (1995) reported a general pattern of fin whale movements in the fall from the Labrador/Newfoundland region, south past Bermuda and into the West Indies, but neonate strandings along the US Mid-Atlantic coast from October through January suggest the possibility of an offshore calving area (Hain *et al.* 1992).

Fin whales achieve sexual maturity at 6-10 years of age in males and 7-12 years in females (Jefferson *et al.* 2008), although physical maturity may not be reached until 20-30 years (Aguilar and Lockyer 1987). Conception is believed to occur in tropical and subtropical areas during the winter with birth of a single calf after a 11-12 month gestation (Jefferson *et al.* 2008). The calf is weaned 6-11 months after birth (Perry *et al.* 1999). The mean calving interval is 2.7 years (Agler *et al.* 1993).

The predominant prey of fin whales varies greatly in different geographical areas depending on what is locally available (IWC 1992). In the western North Atlantic, fin whales feed on a variety of small schooling fish (*i.e.*, herring, capelin, sand lance) as well as squid and planktonic crustaceans (Wynne and Schwartz 1999).

Threats to fin whale recovery

The major known sources of anthropogenic mortality and injury of fin whales include entanglement in commercial fishing gear and ship strikes. The minimum annual rate of confirmed human-caused serious injury and mortality to North Atlantic fin whales from 2003-2007 was 2.8 (Glass *et al.* 2009). During this five year period, there were 13 confirmed entanglements (3 fatal; 3 serious injuries) and 11 ship strikes (8 fatal) (Glass *et al.* 2009). Fin whales are believed to be the cetacean most commonly struck by large vessels (Laist *et al.* 2001). In addition, hunting of fin whales continued well into the 20th century. Fin whales were given total protection in the North Atlantic in 1987 with the exception of aboriginal subsistence whaling hunt in Greenland (Gambell 1993, Caulfield 1993). However, Iceland reported a catch of 136 whales in the 1988/89 and 1989/90 seasons (Perry *et al.* 1999), and 7 in 2006/07. Fin whales may also be adversely affected by habitat degradation, habitat exclusion, acoustic trauma, harassment, or reduction in prey resources due to trophic effects resulting from a variety of activities.

Population Trends and Status
Various estimates have been provided to describe the current status of fin whales in western North Atlantic waters. One method used the catch history and trends in Catch Per Unit Effort to obtain an estimate of 3,590 to 6,300 fin whales for the entire western North Atlantic (Perry *et al.* 1999). Hain *et al.* (1992) estimated that about 5,000 fin whales inhabit the northeastern US continental shelf waters. The 2009 Stock Assessment Report (SAR) gives a best estimate of abundance for fin whales in the western North Atlantic of 2,269 (CV = 0.37). However, this estimate must be considered extremely conservative in view of the incomplete coverage of the known habitat of the stock and the uncertainties regarding population structure and whale movements between surveyed and unsurveyed areas (Waring *et al.* 2009). The minimum population estimate for the western North Atlantic fin whale is 1,678 (Waring *et al.* 2009). There are insufficient data at this time to determine population trends for the fin whale (Waring *et al.* 2009).

Summary of Fin Whale Status
Information on the abundance and population structure of fin whales worldwide is limited. NMFS recognizes three fin whale stocks in the Pacific for the purposes of managing this species under the MMPA. Reliable estimates of current abundance for the entire Northeast Pacific fin whale stock are not available (Angliss *et al.* 2001). Stock structure for fin whales in the southern hemisphere is unknown and there are no current estimates of abundance for southern hemisphere fin whales. As noted above, the best population estimate for the western North Atlantic fin whale is 2,269 and the minimum population estimate is 1,678. The 2009 SAR indicates that there are insufficient data at this time to determine population trends for the fin whale. Fishing gear appears to pose less of a threat to fin whales in the North Atlantic Ocean than to North Atlantic right or humpback whales. However, fin whales continue to be struck by large vessels and some level of whaling for fin whales in the North Atlantic still occurs.

The Draft 2010 SAR (Waring *et al.* 2010) for the western North Atlantic fin whale stock reports an increase in the estimated population size (3,985), minimum population size (3,269), and PBR (6.5). The Draft SAR reported an increase in overall documented serious injury and mortality to fin whales to an average rate of 3.2 per year. Incidental fishery entanglement records and ship strike records for the period 2004 through 2008 averaged of 1.2 (U.S. waters, 1.0) and 2.0 (U.S. waters, 1.4) respectively per year.

3.1.4 Sei Whales

Sei whales are a widespread species in the world's temperate, subpolar, subtropical, and even tropical marine waters. Sei whales reach sexual maturity at 5-15 years of age. The calving interval is believed to be 2-3 years (Perry *et al.* 1999).

North Pacific and Southern Hemisphere. The IWC only considers one stock of sei whales in the North Pacific (Donovan 1991), but for NMFS management purpose under the MMPA, sei whales in the eastern North Pacific are considered a separate stock (Carretta *et al.* 2008). There are no abundance estimates for sei whales in the entire eastern North Pacific. The best estimate of abundance for U.S. Pacific EEZ (California, Oregon, and Washington waters out to 300nmi) is 46 (CV=0.61) sei whales (Barlow and Forney 2007; Forney 2007; Carretta *et al.* 2008). No fishery related serious injuries or mortality have been documented from 2002 through 2006 in the North Pacific stock of sei whales (Carretta *et al.* 2008). During 2002-2006 there was one (1) reported ship strike mortality in Washington in 2003 (NMFS Northwest Regional Office, unpublished data).

The stock structure of sei whales in the southern hemisphere is unknown. Like other whale species, sei whales in the southern hemisphere were heavily impacted by commercial whaling, particularly in the mid-20th century as humpback, fin and blue whales became scarce. Sei whales were protected by the IWC in 1977 after their numbers had substantially decreased and they also became more difficult to find (Perry *et al.* 1999). Since southern hemisphere sei whales do not occur in U.S. waters, there is no stock assessment report for southern hemisphere sei whales.

North Atlantic. Sei whales occur in deep water throughout their range, typically over the continental slope or in basins situated between banks (NMFS 1998b). In the Northwest Atlantic, the whales travel along the eastern Canadian coast in June, July, and autumn on their way to and from the Gulf of Maine and Georges Bank where they occur in winter and spring. Within the U.S. Atlantic EEZ, the sei whale is most common on Georges Bank and into the Gulf of Maine/Bay of Fundy region during spring and summer, primarily in deeper waters. In years of reduced predation on copepods by other predators, and thus greater abundance of this prey source, sei whales are reported in more inshore locations (Waring *et al.* 2009).

Although sei whales may prey upon small schooling fish and squid in the action area, available information suggests that calanoid copepods and euphausiids are the primary prey of this species (Flinn *et al.* 2002). Sei whales are occasionally seen feeding in association with right whales in the southern Gulf of Maine and in the Bay of Fundy. However, there is no evidence to demonstrate interspecific competition between these species for food resources.

There is limited information on the stock identity of sei whales in the North Atlantic (Waring *et al.* 2009). For purposes of the Marine Mammal Stock Assessment Reports, and based on a proposed IWC stock definition, NMFS recognizes the sei whales occurring from the U.S. East coast to Cape Breton, Nova Scotia, and east to 42° W longitude as the "Nova Scotia stock" of sei whales (Waring *et al.* 2009).

The abundance estimate of 386 sei whales (CV=0.85), obtained from a line-transect sighting survey conducted during 12 June to 4 August 2004, by a ship and a plane covering 10,761 km of trackline in the region from the 100 m depth contour on the southern of Georges Bank to the lower Bay of Fundy is considered the best available for the Nova Scotia stock of sei whales according to the 2009 SAR (Waring *et al.* 2009). This estimate is considered extremely conservative in view of the known range of the sei whale in the entire western North Atlantic, and the uncertainties regarding population structure and whale movements between surveyed and unsurveyed areas. The minimum population estimate for this sei whale stock is 208 (Waring *et al.* 2009). Current and maximum net productivity rates are unknown for this stock. There are insufficient data to determine trends of the sei whale population (Waring *et al.* 2009).

Few instances of injury or mortality of sei whales due to entanglement or vessel strikes have been recorded in U.S. waters, possibly because sei whales typically inhabit waters further offshore than most commercial fishing operations, or perhaps entanglements do occur but are less likely to be observed. The records on file at NMFS of stranded, floating or injured sei whales for the period 2003 through 2007 show one (1) record with substantial evidence of fishery interactions causing serious injury in April 2006 (Glass *et al.* 2009). Between 2003 and 2007, three (3) ship strike mortalities have been confirmed. The first ship strike was in February 2003, an 11-meter male was discovered outside of Norfolk Naval Base in Norfolk, Virginia. Another ship strike mortality was reported in April 2006 when a fresh sei whale carcass was brought in on the bow of a ship to Baltimore, Maryland. In 2007, a ship strike mortality was recorded off Deer Island, Massachusetts (Waring *et al.* 2009). NMFS also has two (2) other human caused sei whale mortalities on record. One (1) incident occurred in 1994 when a carcass was brought in on the bow of a container ship in Charlestown, Massachusetts, and in May 2001 a 13-meter female sei whale carcass slid off the bow of a ship arriving in New York harbor (Waring *et al.* 2009).

Summary of Sei Whale Status

The best estimate of abundance for the Nova Scotia stock of sei whales is 386 (Waring *et al.* 2009). There are insufficient data to determine trends of the Nova Scotian sei whale population. One (1) sei whale serious injury from fishery interaction and three mortalities from ship strike has been recorded in U.S. waters between 2003-2007 (Glass *et al.* 2009). Information on the status of sei whale populations worldwide is similarly lacking. There are no abundance estimates for sei whales in the entire eastern North Pacific, however the best estimate of abundance for in U.S. Pacific EEZ is 46 (Carretta *et al.* 2008). The stock structure of sei whales in the southern hemisphere is unknown.

The Draft 2010 SAR (Waring *et al.* 2010) for the Nova Scotia stock of sei whales reports the same minimum population size and the PBR remained the same at 0.4. Overall documented serious injury and mortality to sei whales increased by 0.2 to an average rate of 1.0 per year. Incidental fishery entanglement records and ship strike records for the period 2004 through 2008 averaged of 0.6 and 0.4 respectively per year.

3.2 Status of Sea Turtles

Sea turtles continue to be affected by many activities occurring on the nesting beaches and in the marine environment. Poaching, habitat modification and destruction, and nesting predation affect eggs, hatchlings, and nesting females while on land. Fishery interactions, vessel interactions, marine pollution, and non-fishery operations (*e.g.,* dredging, military activities, oil and gas exploration), for example, affect sea turtles in the neritic zone, which is defined as the marine environment extending from mean low water down to 200 m (660 feet) in depth, generally corresponding to the continental shelf (Lalli and Parsons 1997; Encyclopedia Britannica 2010). Fishery interactions and marine pollution also affect sea turtles in the oceanic zone, which is defined as the open ocean environment where bottom depths are greater than 200 m (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 several decades ago.

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. Additional background information on the range-wide status of these species, as well as a description and life history of the species, can be found in a number of published documents, including sea turtle status reviews and biological reports (NMFS and USFWS 1995; Hirth 1997; Turtle Expert Working Group [TEWG] 1998, 2000, 2007, 2009; NMFS and USFWS 2007a, 2007b, 2007c, 2007d), and recovery plans for the loggerhead sea turtle (NMFS and USFWS 1998a, 2008), leatherback sea turtle (NMFS and USFWS 1992, 1998b), Kemp's ridley sea turtle (USFWS and NMFS 1992), and green sea turtle (NMFS and USFWS 1991, 1998c).

3.2.1 Loggerhead Sea Turtle

Loggerhead sea turtles are found in temperate and subtropical waters and occupy a range of habitats including offshore waters, continental shelves, bays, estuaries, and lagoons. The loggerhead is the most abundant species of sea turtle in U.S. waters. Genetic differences exist between loggerhead sea turtles that nest and forage in the different ocean basins (Bowen 2003; Bowen and Karl 2007). Differences in the maternally inherited mitochondrial DNA also exist between loggerhead nesting groups that occur within the same ocean basin (TEWG 2000; Pearce 2001; Bowen 2003; Bowen *et al.* 2005; Shamblin 2007). 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). Loggerhead sea turtles are currently listed under the ESA at the species level rather than as subspecies or distinct population segments (DPS). The ESA requires NMFS to ultimately conclude whether the action under consultation, in light of the Environmental Baseline (Section 4.0) and Cumulative Effects (Section 5.0), is likely to jeopardize the species as it is listed. Therefore, information on the range-wide status of the species is included as follows.

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

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 sea turtles at nesting colonies throughout the Pacific basin has declined dramatically over the past ten to twenty years. Loggerhead sea turtles in the Pacific Ocean are represented by a northwestern Pacific nesting group (located in Japan) and a smaller southwestern Pacific nesting group that occurs in eastern Australia and New Caledonia. Data from 1995 estimated the Japanese nesting group at 1,000 adult females (Bolten *et al.* 1996). More recent information suggests that nest numbers have increased gradually over the period of 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 indicate the presence of genetically distinct nesting colonies (Hatase *et al.* 2002).

In Australia, long-term census data have been collected at some rookeries since the late 1960s and early 1970s, and nearly all the data show marked declines in nesting since the mid-1980s. The nesting group in Queensland, Australia is now less than 500 adult females, which represents an 86% reduction in the size of the annual nesting population in 23 years (Limpus and Limpus 2003).

Pacific loggerhead sea turtles are captured, injured, or killed in numerous Pacific fisheries including gillnet, longline, pound net, and trawl fisheries in the western and/or eastern Pacific Ocean (NMFS and USFWS 2007a). In Australia, where sea turtles are taken in bottom trawl and longline fisheries, efforts have been made to reduce fishery bycatch (NMFS and USFWS 2007a). Loggerheads in the Pacific are also impacted by a reduction in nesting habitat from erosion and extensive beach use, predation (by humans and animals), boat strikes, and marine pollution.

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 predation 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. Each year an estimated 20,000-40,000 females nest at Masirah, the largest nesting site within Oman (Baldwin *et al.* 2003). In the eastern Indian Ocean, all known nesting sites are found in western Australia (Dodd 1988). Nesting numbers are disproportionate within the area with the majority of nesting occurring at a single location; Dirk Hartog Island hosts approximately 70%-75% of the nesting loggerheads in the southeastern Indian Ocean (Baldwin *et al.* 2003). The depletion of nesting at other western Australia sites may be the result of longstanding red fox predation on eggs (Baldwin *et al.* 2003).

Mediterranean Sea. Nesting in the Mediterranean Sea is confined almost exclusively to the eastern basin (Margaritoulis *et al.* 2003). The greatest numbers 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 nests per year (NMFS and USFWS 2007a). There is a long history of exploitation of loggerheads in the Mediterranean (Margaritoulis *et al.* 2003). Although much of this is now prohibited, some directed captures still occur (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 habitats and foraging areas for loggerheads within the Atlantic Ocean. Detailed information is also provided in the 5-year status review for loggerheads (NMFS and USFWS 2007a) and the final revised recovery plan for loggerheads in the Northwest Atlantic Ocean (NMFS and USFWS 2008), which is a second revision to the original recovery plan that was approved in 1984 and subsequently revised in 1991.

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 Atlantic 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), and Mexico, where several thousand nests are estimated to be laid each year. For example, the Yucatán nesting population had a range of 903-2,331 nests per year from 1987-2001 (Zurita *et al.* 2003; NMFS and USFWS 2008). In both the eastern and western Atlantic, waters as far north as 41°N to 42°N latitude are used for foraging by juveniles as well as adults (Shoop 1987; Shoop and Kenney 1992; Ehrhart *et al.* 2003; Mitchell *et al.* 2003).

In U.S. Atlantic waters, loggerheads commonly occur throughout the inner continental shelf from Florida to Cape Cod, Massachusetts and in the Gulf of Mexico from Florida to Texas, although their presence varies with the seasons due to changes in water temperature (Shoop and Kenney 1992; Epperly *et al.* 1995a, 1995b; Braun and Epperly 1996; Epperly and Braun-McNeill 2002; Mitchell *et al.* 2003). Loggerheads have been observed in waters with surface temperatures of 7° to 30°C, but water temperatures ≥11°C are most favorable (Shoop and Kenney 1992; Epperly *et al.* 1995b). The presence of loggerhead sea turtles in U.S. Atlantic waters is also influenced by water depth. Aerial surveys of continental shelf waters north of Cape Hatteras, North Carolina indicated that loggerhead sea turtles were most commonly sighted in waters with bottom depths ranging from 22 to 49 m deep (Shoop and Kenney 1992). However, more recent survey and satellite tracking data support that they occur in waters from the beach to beyond the continental shelf (Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; Blumenthal *et al.* 2006; Hawkes *et al.* 2006; McClellan and Read 2007).

Loggerhead sea turtles occur year round in ocean waters off North Carolina, South Carolina, Georgia, and Florida. In these areas of the South Atlantic Bight, water temperature is influenced by the proximity of the Gulf Stream. As coastal water temperatures warm in the spring, loggerheads begin to migrate to inshore waters of the Southeast U.S. (*e.g.*, Pamlico and Core Sounds) and also move up the U.S. Atlantic coast (Epperly *et al.* 1995a, 1995b, 1995c; Braun-McNeill and Epperly 2004), occurring in Virginia foraging areas as early as April/May and on the most northern foraging grounds in the Gulf of Maine in June (Shoop and Kenney 1992). The trend is reversed in the fall as water temperatures cool. The large majority leave the Gulf of Maine by mid-September but some turtles may remain in Mid-Atlantic and Northeast areas until late fall. By December, loggerheads have migrated from inshore and more northern coastal waters to waters offshore of North Carolina, particularly off of Cape Hatteras, and waters further south where the influence of the Gulf Stream provides temperatures favorable to sea turtles (Shoop and Kenney 1992; Epperly *et al.* 1995b; Epperly and Braun-McNeill 2002).

In the southeastern U.S., loggerheads mate from late March to early June, and eggs are laid throughout the summer, with a mean clutch size of 100-126 eggs (Dodd 1988). Individual females nest multiple times during a nesting season, with a mean of 4.1 nests per individual (Murphy and Hopkins 1984). Nesting migrations for an individual female loggerhead are usually on an interval of 2 to 3 years, but can vary from 1 to 7 years (Dodd 1988; NMFS and USFWS 2008). Age at sexual maturity for loggerheads has been estimated at 32 to 35 years (NMFS and USFWS 2008).

For the past decade or so, the scientific literature has recognized five distinct nesting groups, or subpopulations, of loggerhead sea turtles in the Northwest Atlantic, divided geographically as follows: (1) a northern group of nesting females that nest from North Carolina to Northeast Florida at about 29°N latitude; (2) a South Florida group of nesting females that nest from 29°N latitude on the East coast to Sarasota on the West coast; (3) a Florida Panhandle group of nesting females that nest around Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán group of nesting females that nest on beaches of the eastern Yucatán Peninsula, Mexico (Márquez 1990; TEWG 2000); and (5) a Dry Tortugas group that nests on beaches of the islands of the Dry Tortugas, near Key West, Florida (NMFS SEFSC 2001). Genetic analyses of mitochondrial DNA, which a sea turtle inherits from its mother, indicate that there are genetic differences between loggerheads that nest at and originate from the beaches used by each of the five identified nesting groups of females (TEWG 2000). However, analyses of microsatellite loci from nuclear DNA, which represents the genetic contribution from both parents, indicates little to no genetic differences between loggerheads originating from nesting beaches of the five Northwest Atlantic nesting groups (Pearce and Bowen 2001; Bowen 2003; Bowen *et al.* 2005; Shamblin 2007). These results suggest that female loggerheads have site fidelity to nesting beaches within a particular area, while males provide an avenue of gene flow between nesting groups by mating with females that originate from different nesting groups (Bowen 2003; Bowen *et al.* 2005). The extent of such gene flow, however, is unclear (Shamblin 2007).

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

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

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

From the beginning of standardized index surveys in 1989 until 1998, the PFRU, the largest nesting assemblage in the Northwest Atlantic by an order of magnitude, had a significant increase in the number of nests. However, from 1998 through 2008, there was a 41% decrease in annual nest counts from index beaches, which represent an average of 70% of the statewide nesting activity (NMFS and USFWS 2008). From 1989-2008, the PFRU had an overall declining nesting trend of 26% (95% CI: -42% to -5%; NMFS and USFWS 2008). In 2008, an increase in nest counts from the previous four years was reported, but this did not alter the declining trend. The Loggerhead Recovery Team acknowledged that this dramatic change in status for the PFRU is a serious concern and requires immediate attention to determine the cause(s) of this change and the actions needed to reverse it. The NRU, the second largest nesting assemblage of loggerheads in the U.S., has been declining at a rate of 1.3% annually since 1983 (NMFS and USFWS 2008). The NRU dataset included 11 beaches with an uninterrupted time series of coverage of at least 20 years; these beaches represent approximately 27% of NRU nesting (in 2008). Overall, there is strong statistical data to suggest the NRU has experienced a long-term decline. Evaluation of long-term nesting trends for the NGMRU is difficult because of changed and expanded beach coverage. However, the NGMRU has shown a significant declining trend of 4.7% annually since index nesting beach surveys were initiated in 1997 (NMFS and USFWS 2008). No statistical trends in nesting abundance can be determined for the DTRU because of the lack of long-term data. Similarly, statistically valid analyses of long-term nesting trends for the entire GCRU are not available because there are few long-term standardized nesting surveys representative of the region. Additionally, changing survey effort at monitored beaches and scattered and low-level nesting by loggerheads at many locations currently precludes comprehensive analyses (NMFS and USFWS 2008).

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

Unlike nesting surveys, in-water studies of sea turtles typically sample both sexes and multiple age classes. In-water studies have been conducted in some areas of the Northwest Atlantic and provide data by which to assess the relative abundance of loggerhead sea turtles and changes in abundance over time (Maier *et al.* 2004; Morreale *et al.* 2005; Mansfield 2006; Ehrhart *et al.* 2007; Epperly *et al.* 2007). The 2008 loggerhead recovery plan includes a full discussion of in-water population studies for which trend data have been reported, and a brief summary will be provided here. Maier *et al.* (2004) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the Southeast coast of the U.S. (Winyah Bay, South Carolina to St. Augustine, Florida) during the period 2000-2003. A comparison of loggerhead catch data from this study with historical values suggested that in-water populations of loggerhead sea turtles along the Southeast U.S. coast appear to be larger, possibly an order of magnitude higher than they were 25 years ago, but the authors caution a direct comparison between the two studies given differences in sampling methodology (Maier *et al.* 2004). A comparison of catch rates for sea turtles in pound net gear fished in the Pamlico-Albemarle Estuarine Complex of North Carolina between the years 1995-1997 and 2001-2003 found a significant increase in catch rates for loggerhead sea turtles for the latter period (Epperly *et al.* 2007). A long-term, on-going study of loggerhead abundance in the Indian River Lagoon System of Florida found a significant increase in the relative abundance of loggerheads over the last 4 years of the study (Ehrhart *et al.* 2007). However, there was no discernible trend in loggerhead abundance during the 24-year time period of the study (1982-2006) (Ehrhart *et al.* 2007). At St. Lucie Power Plant, data collected from 1977-2004 show an increasing trend of loggerheads at the power plant intake structures (FPL and Quantum Resources 2005).

In contrast to these studies, Morreale *et al.* (2005) observed a decline in the percentage and relative numbers of loggerhead sea turtles incidentally captured in pound net gear fished around Long Island, New York during the period 2002-2004 in comparison to the period 1987-1992, with only two (2) loggerheads (of a total 54 turtles) observed captured in pound net gear during the period 2002-2004. This is in contrast to the previous decade's study where numbers of individual loggerheads ranged from 11 to 28 per year (Morreale *et al.* 2005). No additional loggerheads were reported captured in pound net gear through 2007, although 2 were found cold-stunned on Long Island bay beaches in the fall of 2007 (Memo to the File, L. Lankshear, December 2007). Potential explanations for this decline include major shifts in loggerhead foraging areas and/or increased mortality in pelagic or early benthic stage/age classes (Morreale *et al.* 2005). Using aerial surveys, Mansfield (2006) also found a decline in the densities of loggerhead sea turtles in Chesapeake Bay over the period 2001-2004 compared to aerial survey data collected in the 1980s. Significantly fewer loggerheads ($p<0.05$) were observed in both the spring (May-June) and the summer (July-August) of 2001-2004 compared to those observed during aerial surveys in the 1980s (Mansfield 2006). A comparison of median densities from the 1980s to the 2000s suggested that there had been a 63.2% reduction in densities during the spring residency period and a 74.9% reduction in densities during the summer residency period (Mansfield 2006). The decline in observed loggerhead populations in Chesapeake Bay may be related to a significant decline in prey, namely horseshoe crabs and blue crabs, with loggerheads redistributing outside of Bay waters (NMFS and USFWS 2008).

The diversity of a sea turtle's life history leaves them susceptible to many natural and human impacts, including impacts while they are on land, in the neritic environment, and in the oceanic environment. Recent studies have established that the loggerhead's life history is more complex than previously believed. Rather than making discrete developmental shifts from oceanic to neritic environments, research is showing that both adults and (presumed) neritic stage juveniles continue to use the oceanic environment and will move back and forth between the two habitats (Witzell 2002; Blumenthal *et al.* 2006; Hawkes *et al.* 2006; McClellan and Read 2007). One of the studies tracked the movements of adult post-nesting females and found that differences in habitat use were related to body size with larger adults staying in coastal waters and smaller adults traveling to oceanic waters (Hawkes *et al.* 2006). A tracking study of large juveniles found that the habitat preferences of this life stage were also diverse with some remaining in neritic waters and others moving off into oceanic waters (McClellan and Read 2007). However, unlike the Hawkes *et al.* (2006) study, there was no significant difference in the body size of turtles that remained in neritic waters versus oceanic waters (McClellan and Read 2007). In either case, the research 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 and 2008 recovery plan provide a summary of natural as well as anthropogenic threats to loggerhead sea turtles (NMFS and USFWS 2007a, 2008). Amongst those of natural origin, hurricanes are known to be destructive to sea turtle nests. Sand accretion, rainfall, and wave action that result from these storms can appreciably reduce hatchling success. Other sources of natural mortality include cold stunning, biotoxin exposure, and native species predation..

Anthropogenic factors that impact hatchlings and adult females on land, or the success of nesting and hatching include: beach erosion, beach armoring, and nourishment; artificial lighting; beach cleaning; beach pollution; increased human presence; recreational beach equipment; vehicular and pedestrian traffic; coastal development/construction; exotic dune and beach vegetation;

removal of native vegetation; and poaching. An increased human presence at some nesting beaches or close to nesting beaches has led to secondary threats such as the introduction of exotic fire ants, feral hogs, dogs, and an increased presence of native species (*e.g.*, raccoons, armadillos, and opossums) which raid nests and feed on turtle eggs (NMFS and USFWS 2007a, 2008). Although sea turtle nesting beaches are protected along large expanses of the Northwest Atlantic coast (in areas like Merritt Island, Archie Carr, and Hobe Sound National Wildlife Refuges), other areas along these coasts have limited or no protection. Sea turtle nesting and hatching success on unprotected high density East Florida nesting beaches from Indian River to Broward County are affected by all of the above threats.

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

A 1990 National Research Council (NRC) report concluded that for juveniles, subadults, and breeders in coastal waters, the most important source of human caused mortality in U.S. Atlantic waters was fishery interactions. Of the many fisheries known to adversely affect loggerheads, the U.S. South Atlantic and Gulf of Mexico shrimp fisheries were considered to pose the greatest threat of mortality to neritic juvenile and adult age classes of loggerheads, accounting for an estimated 5,000 to 50,000 loggerhead 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 2002a; Lewison *et al.* 2003). Section 7 consultation on shrimp trawling in the southeastern U.S. was reinitiated in 2002, in part, to consider the effect of a new rulemaking that would require increasing the size of TED escape openings to allow larger loggerheads (as well as green and leatherback 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 3,948 turtles assuming that all TEDs were installed properly and that compliance was 100% (Epperly *et al.* 2002; NMFS 2002a). The total annual 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 loggerhead interactions (the total number of turtles that enter a shrimp trawl, which may then escape through the TED or fail to escape and be captured) with 3,948 of those takes being lethal (NMFS 2002a). On February 21, 2003, NMFS issued the final rule in the *Federal Register* to require the use of the larger opening TEDs (68 FR 8456, February 21, 2003). 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.

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

Loggerhead sea turtles are also known to interact with non-shrimp trawl, gillnet, longline, dredge, pound net, pot/trap, and hook and line fisheries. The NRC (1990) report stated that other U.S. Atlantic fisheries collectively accounted for 500 to 5,000 loggerhead deaths each year, but recognized that there was considerable uncertainty in the estimate. The first estimate of loggerhead sea turtle bycatch in U.S. Mid-Atlantic bottom otter trawl gear was completed in September 2006 and later updated in November 2008 (Murray 2006, 2008). Observers reported 66 loggerhead sea turtle interactions with bottom otter trawl gear from 1994-2004 of which 38 were reported as alive and uninjured and 28 were reported as dead, injured, resuscitated, or of unknown condition (Murray 2006, 2008). Fifty percent of observed sea turtle interactions occurred on vessels targeting summer flounder, 27% on vessels targeting Atlantic croaker, 11% on vessels targeting weakfish, 8% on vessels targeting long-finned squid, 3% on vessels targeting groundfish, and 1% on vessels targeting short-finned squid. Based on observed interactions and fishing effort as reported on VTRs, the average annual loggerhead bycatch in bottom otter trawl during 1996-2004 was estimated to be 616 sea turtles (CV=0.23, 95% CI over the 9 year period: 367-890) (Murray 2006, 2008).

The 2008 update also reported loggerhead bycatch from 2000-2004 by main species (fish or invertebrate) group landed. The average annual bycatch estimate of loggerhead sea turtles from 2000-2004 (based on the rate from 1994-2004) over FMP groups identified by NERO was 411 turtles, with an additional 77 estimated bycatch events unassigned. An estimated 192 (47%) takes occurred annually in the summer flounder/scup/black sea bass group, 62 (15%) in the Atlantic mackerel/squid/butterfish group, 43 (10%) in the Northeast multispecies group, and 41 (10%) in the Atlantic croaker group. A total of 20 loggerheads (4.8%) were estimated as having been taken annually in bottom otter trawl gear catching sea scallops, which is in addition to the estimated 81-191 loggerheads reported by Murray (2007) as being caught annually in trawl gear designed specifically to harvest scallops based on data from 2004-2005 (Murray 2008).

There have been several published estimates of the number of loggerheads taken annually as a result of the dredge fishery for Atlantic sea scallops, ranging from a low of zero in 2005 (Murray 2007) to a high of 749 in 2003 (Murray 2004). An estimate of the number of loggerheads taken annually in U.S. Mid-Atlantic gillnet fisheries has recently been published in Murray (2009a). From 1995-2006, the average annual bycatch of loggerheads in U.S. Mid-Atlantic gillnet gear was estimated to be 350 turtles (95% over the 12 year period CI: 234 to 504). Bycatch rates were correlated with latitude, sea surface temperature, and mesh size. The highest predicted bycatch rates occurred in warm waters of the southern Mid-Atlantic in large-mesh gillnets (Murray 2009a).

The U.S. tuna and swordfish longline fisheries that are managed under the Highly Migratory Species (HMS) FMP are estimated to capture 1,905 loggerheads (no more than 339 mortalities) for each 3-year period starting in 2007 (NMFS 2004a). NMFS has mandated gear changes for the HMS fishery to reduce sea turtle bycatch and the likelihood of death from those incidental takes that would still occur (Garrison *et al.* 2009). In 2008, there were 82 observed interactions between loggerhead sea turtles and longline gear used in the HMS fishery. All of the loggerheads were released alive, but the vast majority with injuries (Garrison *et al.* 2009). Most of the injured loggerheads had been hooked in the mouth or beak or swallowed the hook (Garrison *et al.* 2009). Based on the observed take, an estimated 771.6 (95% CI: 481.4-1236.6) loggerhead sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP in 2008 (Garrison *et al.* 2009). The 2008 estimate is higher than that in 2007 and is consistent with historical averages since 2001 (Garrison *et al.* 2009). This fishery represents just one of several longline fisheries operating in the Atlantic Ocean. Lewison *et al.* (2004) estimated that 150,000-200,000 loggerheads were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries as well as others).

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

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

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

Currently, there are no population estimates for loggerhead sea turtles in any of the ocean basins in which they occur. However, a recent loggerhead assessment prepared by NMFS states that the loggerhead adult female population in the western North Atlantic ranges from 20,000 to 40,000 or more, with a large range of uncertainty in total population size. However, 95% of the distribution of conservative estimates of the adult female population size fell between 18,333 (2.5 percentile) and 68,192 (97.5 percentile) individuals (NMFS SEFSC 2009).

Based on their 5-year status review of the species, NMFS and FWS determined that loggerhead sea turtles should not be delisted or reclassified as endangered. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPSs should be identified for the loggerhead (NMFS and USFWS 2007a). In 2008, NMFS and FWS established a Loggerhead Biological Review Team (BRT) to assess the global loggerhead population structure to determine whether DPSs exist and, if so, the status of each DPS. The BRT report was completed in August 2009 (Conant *et al.* 2009). In this report, the BRT identified the following nine loggerhead DPSs distributed globally: (1) North Pacific Ocean, (2) South Pacific Ocean, (3) North Indian Ocean, (4) Southeast Indo-Pacific Ocean, (5) Southwest Indian Ocean, (6) Northwest Atlantic Ocean, (7) Northeast Atlantic Ocean, (8) Mediterranean Sea, and (9) South Atlantic Ocean. According to an analysis using expert opinion in a matrix model framework used in the BRT report, all loggerhead DPSs have the potential to decline in the future. The BRT concluded that although some DPSs are indicating increasing trends at nesting beaches (Southwest Indian Ocean and South Atlantic Ocean), available information about anthropogenic threats to juveniles and adults in neritic and oceanic environments indicate possible unsustainable additional mortalities. According to the threat matrix analysis in the BRT report, the potential for future decline is greatest for the North Indian Ocean, Northwest Atlantic Ocean, Northeast Atlantic Ocean, Mediterranean Sea, and South Atlantic Ocean DPSs (Conant *et al.* 2009).

On March 16, 2010, NMFS and USFWS published a proposed rule in the Federal Register to divide the worldwide population of loggerhead sea turtles into nine DPSs, as described in the 2009 Status Review. Two of the DPSs are proposed to be listed as threatened and seven of the DPSs, including the Northwest Atlantic Ocean DPS, are proposed to be listed as endangered. NMFS and the USFWS accepted comments on the proposed rule through September 13, 2010 (75 FR 30769, June 2, 2010).

3.2.2 Leatherback Sea Turtle

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

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

Pacific Ocean. Leatherback nesting has been declining at all major Pacific basin nesting beaches for the last two decades (Spotila *et al.* 1996, 2000; NMFS and USFWS 1998b, 2007b; Sarti *et al.* 2000). In the western Pacific, major nesting beaches occur in Papua New Guinea, Indonesia, Solomon Islands, and Vanuatu, with an approximate 2,700-4,500 total breeding females, estimated from nest counts (Dutton *et al.* 2007). Leatherback sea turtles disappeared from India before 1930, have been virtually extinct in Sri Lanka since 1994, and appear to be approaching extinction in Malaysia (Spotila *et al.* 2000). For example, the nesting group on Terengganu (Malaysia) - which was one of the most significant nesting sites in the western Pacific Ocean - declined severely from an estimated 3,103 females in 1968 to 2 nesting females in 1994 (Chan and Liew 1996). Nesting groups of leatherback sea 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, leatherback sea turtles have only been known to nest in low densities and scattered colonies.

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

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

In the eastern Pacific Ocean, major leatherback nesting beaches are located in Mexico and Costa Rica, where nest numbers have been declining. According to reports from the late 1970s and early 1980s, beaches located on the Mexican Pacific coasts of Michoacán, Guerrero, and Oaxaca sustained a large portion, perhaps 50%, of all global nesting by leatherbacks (Sarti *et al.* 1996). A dramatic decline has been seen on nesting beaches in Pacific Mexico, where aerial survey data was used to estimate that tens of thousands of leatherback nests were laid on the beaches in the 1980s (Pritchard 1982), but a total of only 120 nests on the four primary index beaches (combined) were counted in the 2003-2004 season (Sarti Martinez *et al.* 2007). Since the early 1980s, the Mexican Pacific population of adult female leatherback turtles has declined to slightly more than 200 during 1998-1999 and 1999-2000 (Sarti *et al.* 2000). Spotila *et al.* (2000) reported the decline of the leatherback nesting at Playa Grande, Costa Rica, which had been the fourth largest nesting group in the world and the most important nesting beach in the Pacific. Between 1988 and 1999, the nesting group declined from 1,367 to 117 female leatherback sea turtles. Based on their models, Spotila *et al.* (2000) estimated that the group could fall to less than 50 females by 2003-2004. An analysis of the Costa Rican nesting beaches indicates a decline in nesting during 15 years of monitoring (1989-2004) with approximately 1,504 females nesting in 1988-1989 to an average of 188 females nesting in 2000-2001 and 2003-2004 (NMFS and USFWS 2007b).

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

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

Mediterranean Sea. Casale *et al.* (2003) reviewed the distribution of leatherback sea turtles in the Mediterranean. Among the 411 individual records of leatherback sightings in the Mediterranean, there were no nesting records. Nesting in the Mediterranean is not known or is believed to be extremely rare. Leatherbacks found in Mediterranean waters originate from the Atlantic Ocean (P. Dutton, NMFS, unpublished data).

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 (*e.g.*, *Stomolophus, Chryaora*, and *Aurelia* species) and tunicates (*e.g.*, salps, pyrosomas) in oceanic habitats (Rebel 1974; Davenport and Balazs 1991). However, leatherbacks are also known to use coastal waters of the U.S. continental shelf (James *et al.* 2005a; Eckert *et al.* 2006; Murphy *et al.* 2006) as well as the European continental shelf on a seasonal basis (Witt *et al.* 2007). The waters adjacent to Sandy Point, St. Croix, U.S. Virgin Islands have been designated as critical habitat for the leatherback sea turtle.

The CETAP aerial survey of the outer Continental Shelf from Cape Hatteras, North Carolina to Cape Sable, Nova Scotia conducted between 1978 and 1982 showed leatherbacks to be present throughout the area with the most numerous sightings made from the Gulf of Maine south to Long Island. Leatherbacks were sighted in water depths ranging from 1 to 4,151 m, but 84.4% of sightings were in waters less than 180 m (Shoop and Kenney 1992). Leatherbacks were sighted in waters within a sea surface temperature range similar to that observed for loggerheads; from 7°-27.2°C (Shoop and Kenney 1992). However, leatherbacks appear to have a greater tolerance for colder waters in comparison to loggerhead sea turtles since more leatherbacks were found at the lower temperatures (Shoop and Kenney 1992). This aerial survey estimated the summer 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 underestimated the leatherback population for the northeastern U.S. at the time of the survey. Estimates of leatherback abundance of 1,052 turtles (C.V. = 0.38) and 1,174 turtles (C.V. = 0.52) were obtained from surveys conducted from Virginia to the Gulf of St. Lawrence in 1995 and 1998, respectively (Palka 2000). However, since these estimates were also based on sightings of leatherbacks at the surface, the author considered the estimates to be negatively biased and the true abundance of leatherbacks may be 4.27 times the estimates (Palka 2000). Studies of satellite tagged leatherbacks suggest that they spend 10%-41% of their time at the surface, depending on the phase of their migratory cycle (James *et al.* 2005b). The greatest amount of surface time (up to 41%) was recorded when leatherbacks occurred in continental shelf and slope waters north of 38°N (James *et al.* 2005b).

Leatherbacks are a long lived species (>30 years). They were originally believed to mature at a younger age than loggerhead sea turtles, with a previous estimated age at sexual maturity of about 13-14 years for females with 9 years reported as a likely minimum (Zug and Parham 1996) and 19 years as a likely maximum (NMFS SEFSC 2001). However, new sophisticated analyses suggest that leatherbacks in the Northwest Atlantic may reach maturity at 24.5-29 years of age (Avens *et al.* 2009). In the 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 the total number of eggs produced per season. As is the case with other sea turtle species, leatherback hatchlings enter the water soon after hatching. Based on a review of all sightings of leatherback sea turtles of <145 centimeters (cm) curved carapace length (CCL), Eckert (1999) found that leatherback juveniles remain in waters warmer than 26°C until they exceed 100 cm 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 (TEWG 2007). In the U.S., the Florida Statewide Nesting Beach Survey program has documented an increase in leatherback nesting numbers from 98 nests in 1988 to between 800 and 900 nests in the early 2000s (NMFS and USFWS 2007b). An analysis of Florida's index nesting beach sites from 1989-2006 shows a substantial increase in leatherback nesting in Florida during this time, with an annual growth rate of approximately 1.17 (TEWG 2007). The TEWG reports an increasing or stable nesting trend for all of the seven populations or groups of populations with the exception of the Western Caribbean and West Africa. In St. Croix, for example, researchers have noted a declining presence of neophytes (first-time nesters) since 2002 (Garner and Garner 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 (TEWG 2007), and represents more than half of total nesting by leatherback sea turtles worldwide (Hilterman and Goverse 2004). Nest numbers in Suriname have shown an increase and the long-term trend for the Suriname and French Guiana nesting group seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests for Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). The TEWG (2007) report indicates that using nest numbers from 1967-2005, a positive population growth rate was found over the 39-year period for French Guinea and Suriname, with a 95% probability that the population was growing. Nevertheless, given the magnitude of leatherback nesting in this area compared to other nest sites, impacts to this area that negatively affect leatherback sea turtles could have profound impacts on the species, overall.

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

The 5-year status review (NMFS and USFWS 2007b) and TEWG (2007) report provide summaries of natural as well as anthropogenic threats to leatherback sea turtles. Of the Atlantic sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear, trap/pot gear in particular. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), 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. Leatherbacks entangled in fishing gear generally have a reduced ability to feed, dive, surface to breathe, or perform any other behavior essential to survival (Balazs 1985). In addition to drowning from forced submergence, they may be more susceptible to boat strikes if forced to remain at the surface, and entangling lines can constrict blood flow resulting in tissue necrosis.

Leatherbacks have been documented interacting with longline, trap/pot, trawl, and gillnet fishing gear. For instance, according to observer records, an estimated 6,363 leatherback sea turtles were documented as caught by the U.S. Atlantic tuna and swordfish longline fisheries between 1992-1999 (NMFS SEFSC 2001). Currently, the U.S. tuna and swordfish longline fisheries managed under the HMS FMP are estimated to capture 1,764 leatherbacks (no more than 252 mortalities) for each 3-year period starting in 2007 (NMFS 2004a). In 2008, there were 90 observed interactions between leatherback sea turtles and longline gear used in the HMS fishery. Four (4) of the leatherbacks were dead upon release and one (1) was in unknown condition. The vast majority of leatherbacks that were released alive had injuries due to external hooking (Garrison *et al.* 2009). Based on the observed take, an estimated 381.3 (95% CI: 288.7-503.7) leatherback sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP in 2008 (Garrison *et al.* 2009). The 2008 estimate is consistent with the annual numbers since 2005 and remains well below the average prior to implementation of gear regulations (Garrison *et al.* 2009). Since the U.S. fleet accounts for only 5%-8% of the longline hooks fished in the Atlantic Ocean, adding up the under-represented observed takes of the other 23 countries actively fishing in the area would likely result in annual take estimates of thousands of leatherbacks over different life stages (NMFS SEFSC 2001). Lewison *et al.* (2004) estimated that 30,000-60,000 leatherbacks were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries as well as others).

Leatherbacks are susceptible to entanglement in the lines associated with trap/pot gear used in several fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine (Dwyer *et al.* 2002). Additional leatherbacks stranded wrapped in line of unknown origin or with evidence of a past entanglement (Dwyer *et al.* 2002). More recently, from 2002 to 2007, NMFS received 144 reports of entangled sea turtles in vertical lines from Maine to Virginia, with 96 events confirmed (verified by photo documentation or response by a trained responder; NMFS 2008a). Of the 96 confirmed events during this period, 87 events involved leatherbacks. NMFS identified the gear type and fishery for 42 of the 96 confirmed events, which included lobster, whelk, sea bass, crab, and research pot gear. 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 (2) 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 U.S., 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 (1) of five (5) 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 2002a). Leatherbacks are likely to encounter shrimp trawls working in the coastal waters off the U.S. Atlantic coast (from Cape Canaveral, Florida through North Carolina) as they make their annual spring migration north. For many years, TEDs that were required for use in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries were less effective for leatherbacks as compared to the smaller, hard-shelled turtle species, because the TED openings were too small to allow leatherbacks to escape. To address this problem, NMFS issued a final rule on February 21, 2003 to amend the TED regulations (68 FR 8456, February 21, 2003). Modifications to the design of TEDs are now required in order to exclude leatherbacks as well as large benthic immature and sexually mature loggerhead and green sea turtles (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 currently required in this fishery. In November 2007, fisheries observers reported the capture of a leatherback sea turtle in bottom otter trawl gear fishing for summer flounder.

Gillnet fisheries operating in the waters of the Mid-Atlantic states are also known to capture, injure, and/or kill leatherbacks when these fisheries and leatherbacks co-occur. Data collected by the NEFSC Fisheries Observer Program from 1994-1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54%-92%. In North Carolina, six (6) additional leatherbacks were reported captured in gillnet sets in the spring (NMFS SEFSC 2001). In addition to these, in September 1995, two dead leatherbacks were removed from an 11-inch (28.2-cm) monofilament shark gillnet set in the nearshore waters off of Cape Hatteras (STSSN unpublished data reported in NMFS SEFSC 2001).

Fishing gear interactions can occur throughout the range of leatherbacks. Entanglements occur in Canadian waters where Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland/Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line, and crab pot line. Leatherbacks are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo *et al.* 1994; Graff 1995). Gillnets are one of the suspected causes for the decline in the leatherback sea turtle population in French Guiana (Chevalier *et al.* 1999), and gillnets targeting green and hawksbill sea turtles in the waters of coastal Nicaragua also incidentally catch leatherback sea turtles (Lagueux *et al.* 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six (6) leatherbacks from 13,600 trawls (Marcano and Alio-M. 2000). An estimated 1,000 mature female leatherback sea turtles are caught annually in fishing nets off of Trinidad and Tobago with mortality estimated to be between 50%-95% (Eckert and Lien 1999). Many of the sea turtles do not die as a result of drowning, but rather because the fishermen cut them out of their nets (NMFS SEFSC 2001).

Leatherbacks may be more susceptible to marine debris ingestion than other sea turtle species due to the tendency of floating debris to concentrate in convergence zones that juveniles and adults use for feeding 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 (e.g., jellyfish) and plastic debris (Mrosovsky 1981). Balazs (1985) speculated that plastic objects may resemble food items by their shape, color, size, or even movements as they drift about, and induce a feeding response in leatherbacks.

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

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

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

3.2.3 Kemp's ridley Sea Turtles

The Kemp's ridley is one of the least abundant of the world's sea turtle species. In contrast to loggerhead, leatherback, and green sea turtles, which are found in multiple oceans of the world, Kemp's ridleys typically occur only in the Gulf of Mexico and the northwestern Atlantic Ocean (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 fewer than 250 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% CI slope = 0.096-0.130) per year (TEWG 2000). An estimated 5,500 females nested in the State of Tamaulipas over a 3-day period in May 2007 and over 4,000 of those nested at Rancho Nuevo (NMFS and USFWS 2007c). There is limited nesting in the U.S., most of which is located in South Texas. In 2006, approximately 100 nests were laid in Texas (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 nesting beach, neonates presumably enter the Gulf of Mexico where they feed on available S*argassum* and associated infauna or other epipelagic species (USFWS and NMFS 1992). The presence of juvenile turtles along both the U.S. Atlantic and Gulf of Mexico coasts, where they are recruited to the coastal benthic environment, indicates that post-hatchlings are distributed in both the Gulf of Mexico and Atlantic Ocean (TEWG 2000).

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

Foraging areas documented along the U.S. Atlantic coast include Charleston Harbor, Pamlico Sound (Epperly *et al.* 1995c), Chesapeake Bay (Musick and Limpus 1997), Delaware Bay, and Long Island Sound (Morreale and Standora 1993). For instance, in the Chesapeake Bay, where the seasonal juvenile population of Kemp's ridley sea turtles is estimated to be 211-1,083 individuals, Kemp's ridleys frequently forage in submerged aquatic grass beds for crabs (Musick and Limpus 1997). Upon leaving Chesapeake Bay in autumn, juvenile Kemp's ridleys migrate down the coast, passing Cape Hatteras in December and January (Musick and Limpus 1997). These larger juveniles are joined by juveniles of the same size from North Carolina sounds and smaller juveniles from New York and New England to form one of the densest concentrations of Kemp's ridleys outside of the Gulf of Mexico (Epperly *et al.* 1995a, 1995b; Musick and Limpus 1997).

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

Kemp's ridleys face many of the same natural threats as loggerheads, including destruction of nesting habitat from storm events, natural predators, 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 sea 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 U.S. waters in a given year, oceanographic conditions, and the occurrence of storm events in the late fall. Although many cold-stunned turtles can survive if found early enough, cold-stunning events can represent a significant cause of natural mortality.

Like other sea turtle species, the severe decline in the Kemp's ridley population appears to have been heavily influenced by a combination of exploitation of eggs and impacts from fishery interactions. From the 1940s through the early 1960s, nests from Ranch Nuevo were heavily exploited, but beach protection in 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 adult Kemp's ridley sea turtles occur. Information from fishermen helped to demonstrate the high number of turtles taken in these shrimp trawls (USFWS and NMFS 1992). Subsequently, NMFS has worked with the industry to reduce sea turtle takes in shrimp trawls and other trawl fisheries, including the development and use of 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 2002a; Lewison *et al.* 2003). The Biological Opinion on shrimp trawling in the southeastern U.S. 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 2002a).

Although modifications to shrimp trawls have helped to reduce mortality of Kemp's ridleys, this species is also affected by other sources of anthropogenic impacts (fishery and non-fishery related) similar to those discussed above. For example, in the spring of 2000, a total of five (5) Kemp's ridley carcasses were recovered from the same North Carolina beaches where 275 loggerhead carcasses were found. The cause of death for most of the turtles recovered was unknown, but the mass mortality event was suspected by NMFS to have been from a large-mesh gillnet fishery for monkfish and dogfish operating offshore in the preceding weeks (67 FR 71895, December 3, 2002). The five (5) Kemp's ridley carcasses that were found are likely to have been only a minimum count of the number of Kemp's ridleys that were killed or seriously injured as a result of the fishery interaction, since it is unlikely that all of the carcasses washed ashore.

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 1990s (NMFS and USFWS 2007c). Based on the number of nests laid in 2006 and the remigration interval for Kemp's ridley sea turtles (1.8-2 years), there were an estimated 7,000-8,000 adult female Kemp's ridley sea turtles in 2006 (NMFS and USFWS 2007c). The number of adult males in the population is unknown, but sex ratios of hatchlings and immature Kemp's ridleys suggest that the population is female biased, suggesting that the number of adult males is less than the number of adult females (NMFS and USFWS 2007c).

As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like dredging, pollution, and habitat destruction account for an unknown level of other mortality. Based on their 5-year status review of the species, NMFS and USFWS (2007c) determined that Kemp's ridley sea turtles should not be reclassified as threatened under the ESA.

3.2.4 Green Sea Turtles

Green sea turtles are distributed circumglobally, and can be found in the Pacific, Indian, and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 1991; 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 sea turtles occur in the western, central, and eastern Pacific. Foraging areas are also found throughout the Pacific and along the southwestern U.S. coast (NMFS and USFWS 1998c). In the western Pacific, major nesting rookeries at four sites including Heron Island (Australia), Raine Island (Australia), Guam, and Japan were evaluated and determined to be increasing in abundance, with the exception of Guam which appears stable (NMFS and USFWS 2007d). In the central Pacific, nesting occurs on French Frigate Shoals, Hawaii, which has also been reported as increasing with a mean of 400 nesting females from 2002-2006 (NMFS and USFWS 2007d). The main nesting sites for the green sea turtle in the eastern Pacific are located in Michoacan, Mexico and in the Galapagos Islands, Ecuador (NMFS and USFWS 2007d). The number of nesting females per year exceeds 1,000 females at each site (NMFS and USFWS 2007d). However, historically, greater than 20,000 females per year are believed to have nested in Michoacan alone (Cliffton *et al.* 1982; NMFS and USFWS 2007d). Thus the current number of nesting females is still far below what has historically occurred. Again, the Pacific Mexico green turtle nesting population (also called the black turtle) is considered endangered.

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

Indian Ocean. There are numerous nesting sites for green sea turtles in the Indian Ocean. One of the largest nesting sites for green sea turtles worldwide occurs on the beaches of Oman where an estimated 20,000 green sea turtles nest annually (Hirth 1997; Ferreira *et al.* 2006). Based on a review of the 32 Index Sites used to monitor green sea turtle nesting worldwide, Seminoff (2004) concluded that declines in green sea turtle nesting were evident for many of the Indian Ocean Index Sites. While several of these had not demonstrated further declines in the more recent past, only the Comoros Island Index Site in the western Indian Ocean showed evidence of increased nesting (Seminoff 2004).

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

Atlantic Ocean. As has occurred in other oceans of its range, green sea turtles were once the target of directed fisheries in the U.S. and throughout the Caribbean. In 1890, over one million lbs of green sea 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 sea turtles occur seasonally in Mid-Atlantic and Northeast waters such as Chesapeake Bay and Long Island Sound (Musick and Limpus 1997; Morreale and Standora 1998; Morreale *et al.* 2005), which serve as foraging and developmental habitats.

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

Age at maturity for green sea turtles is estimated to be 20-50 years (Balazs 1982; Frazer and Ehrhart 1985; Seminoff 2004). As is the case with the other sea turtle species described above, adult females may nest multiple times in a season (average 3 nests/season with approximately 100 eggs/nest) and typically do not nest in successive years (NMFS and USFWS 1991; 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 threatened green sea turtle nesting in the Atlantic/Caribbean, and reviewed the trend in nest count data for each (NMFS and USFWS 2007d). These include: (1) Yucatán Peninsula, Mexico, (2) Tortuguero, Costa Rica, (3) Aves Island, Venezuela, (4) Galibi Reserve, Suriname, (5) Isla Trindade, Brazil, (6) Ascension Island, United Kingdom, (7) Bioko Island, Equatorial Guinea, and (8) Bijagos Achipelago, Guinea-Bissau (NMFS and USFWS 2007d). Nesting at all of these sites is considered to be stable or increasing with the exception of Bioko Island, which may be declining, and the Bijagos Archipelago, which may be stable; however, the lack of sufficient data precludes a meaningful trend assessment for either site (NMFS and USFWS 2007d).

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

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

The status of the endangered Florida breeding population was also evaluated in the 5-year review (NMFS and USFWS 2007d). The pattern of green sea turtle nesting shows biennial peaks in abundance, with a generally positive trend since establishment of the Florida index beach surveys in 1989 to 2006. This is perhaps due to increased protective legislation throughout the Caribbean (Meylan *et al.* 1995), as well as protections in Florida and throughout the U.S. (NMFS and USFWS 2007d).

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

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

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

Summary of Status of Green Sea Turtles
A review of 32 Index Sites[5] distributed globally revealed a 48%-67% decline in the number of mature females nesting annually over the last three 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 threatened nesting groups assessed in that report for which nesting abundance trends could be determined, 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). However, nesting populations were determined to be doing relatively poorly in Southeast Asia, Eastern Indian Ocean, and perhaps the Mediterranean. Overall, based on mean annual reproductive effort, the report estimated that 108,761 to 150,521 females nest each year among the 46 threatened and endangered nesting sites included in the evaluation (NMFS and USFWS 2007d). However, given the late age to maturity for green sea turtles, caution is urged regarding the status for any of the nesting groups since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d).

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

However, the 5-year review also noted that the Tortuguero nesting stock continued to be affected by ongoing directed take at their primary foraging area in Nicaragua (NMFS and USFWS 2007d). The endangered breeding population in Florida appears to be increasing based upon index nesting data from 1989-2006 (NMFS and USFWS 2007d).

[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

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

4.0 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 right, humpback, fin and sei whales, as well as 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 whales and/or turtles ability to function, and (3) recovery activities associated with reducing impacts to ESA-listed sea turtles and/or cetaceans.

Many of the fisheries and other activities causing death or injury to cetaceans and/or 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 sea turtle nesting habitat), others have been undertaken more recently following new information on the impact of certain activities on the species.

The overall impacts that each state, Federal, and private action or other human activity in the action area had on ESA-listed species is unknown. However, to the extent they have manifested themselves at the population level, such past impacts are subsumed in the information presented on the status of each species considered in this Opinion, recognizing that the benefits to each species as a result of recovery activities already implemented may not be evident in the status of the respective population for years given the relatively late age the species reach maturity, and depending on the age class(es) affected.

4.1 Fishery Operations

4.1.1 Federal fisheries

ESA section 7 consultation has been conducted on all federal fisheries authorized under a federal fishery management plan. The action area of the Monkfish FMP overlaps areas of other fishery activity that may adversely affect threatened and endangered species, these fisheries include American lobster, Atlantic bluefish, Atlantic herring, Atlantic mackerel/squid/Atlantic butterfish, Atlantic sea scallop, highly migratory species, Northeast multispecies, red crab, skate, spiny dogfish, summer flounder/scup/black sea bass, and tilefish. Given the broad action area for this consultation, and the broad area of operation for the fisheries, a portion of the fishing effort for each of these previously mentioned fisheries is expected to occur within the action area of this consultation.

ESA-listed cetaceans and sea turtles are known to be killed and injured as a result of being struck by vessels on the water. However, the operation of fishing vessels used in the aforementioned fisheries will have discountable effects on these species. Fishing vessels operate at relatively slow speeds, particularly when towing or hauling gear. Thus, large cetaceans and sea turtles in the path of a fishing vessel would be more likely to have time to move away before being struck.

Gear used in the federal fisheries described below is expected to have an insignificant effect on cetacean or turtle prey. As described in section 3.0, right whales and sei whales feed on copepods (Horwood 2002; Kenney 2002). Copepods are very small organisms that will pass through fishing gear rather than being captured in it. Humpback whales and fin whales also feed on krill as well as small schooling fish (*e.g.*, sand lance, herring, mackerel) (Aguilar 2002; Clapham *et al.* 2002). Some fisheries described below do target fish (*i.e.*, herring, mackerel) that are food items for humpback and fin whales. Nevertheless, given the diversity of their diet, the harvesting of some humpback and fin whale prey as part of commercial fishery operations is not expected to have a significant effect on the availability of humpback and fin whale prey species.

Sea turtle prey items such as horseshoe crabs, other crabs, whelks, and fish are removed from the marine environment as fisheries bycatch in one or more of the aforementioned fisheries. 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 occur in continental shelf waters where the fisheries operate) (Rebel 1974; Mortimer 1982; Bjorndal 1985; USFWS and NMFS 1992; Bjorndal 1997). Therefore, the aforementioned fisheries 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 numerous fisheries (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. Injured or deceased bycatch would still be available as prey for sea turtles, particularly loggerheads, which are known to eat a variety of live prey as well as scavenge dead organisms (Keinath *et al.* 1987; Lutcavage and Musick 1985; Dodd 1988; Burke *et al.* 1993; Morreale and Standora 2005). Additionally, with respect to Kemp's ridley sea turtles, increased nesting by this species for the last several years strongly suggests that the species is not food limited. Given the time it takes for Kemp's ridley sea turtles to mature and nest, fishing effort was likely greater during the time that current nesters were maturing then it is presently. Therefore, any effects of the fisheries on the availability of Kemp's ridley prey should be evident at this time if such were occurring.

Gear used in the federal fisheries described below is believed to have the potential to adversely effect bottom habitat in the action area (NMFS 2003a). A panel of experts have previously concluded that the effects of even light weight otter trawl gear would include: (1) scraping or plowing of the doors on the bottom, sometimes creating furrows along their path; (2) sediment suspension resulting from the turbulence caused by the doors and the ground gear on the bottom; (3) removal or damage to benthic or demersal species; and (4) removal or damage to structure forming biota. The panel also concluded that the greatest impacts from otter trawls occur in high and low energy gravel habitats and in hard clay outcroppings, and that sand habitats were the least likely to be impacted (NREFHSC 2002). The action area does not include hard clay outcroppings, although gravel habitats may occur. The foraging distribution of Kemp's ridley, loggerhead, and green sea turtles in Mid-Atlantic and New England waters as far north as approximately Cape Cod, do not typically occur in gravel habitats. Leatherback sea turtles have a broader distribution in New England waters, which more likely includes clay outcroppings, but are pelagic feeders which should be less impacted by alterations to benthic habitat. For these reasons and the lack of any evidence that fishing practices affect habitats in degrees that harm or harass ESA-listed species, NMFS finds that while continued monkfish fishing efforts may potentially alter benthic habitats, these alterations will be insignificant to ESA-listed species.

Factors affecting food availability for leatherbacks are likely to be oceanographic conditions rather than bottom habitat. As is the case of leatherback sea turtles, prey availability (*i.e.* copepods, schooling fish) for foraging right, humpback, fin and sei whales is associated with oceanographic conditions rather than bottom habitat (Baumgartner *et al.* 2003, IWC 1992, Pace and Merrick 2008, Perry *et al.* 1999) that may be temporarily disturbed by the use of bottom fishing gear.

The American lobster, Atlantic bluefish, Atlantic mackerel/squid/butterfish, Atlantic sea scallop, highly migratory species, multispecies, red crab, skate, spiny dogfish, summer flounder/scup/black sea bass and tilefish fisheries employ gear in a time/area/manner that has been known to capture, injure, and kill sea turtles. Some of these fisheries also use gear known to injure and/or kill right, humpback, fin, or sei whales as a result of entanglements in the gear (Johnson *et al.* 2005; Waring *et al.* 2009; Glass *et al.* 2009). A summary of the impacts of each of these fisheries that has been subject to section 7 consultation is provided below.

The only fishery that has been determined by NMFS to reduce the reproduction, numbers, or distribution of ESA-listed sea turtles, and reduce appreciably their likelihood of survival and recovery, is the pelagic longline component of the Atlantic highly migratory species fishery. On June 14, 2001, NMFS released an Opinion that found that the continued operation of the Atlantic pelagic longline fishery was likely to jeopardize the continued existence of both loggerhead and leatherback sea turtles. To avoid jeopardy to these species, a Reasonable and Prudent Alternative (RPA) was developed. The RPA required the closure of the Northeast Distant (NED) Statistical Area of the Atlantic Ocean to pelagic longlining and the enactment of a research program to develop or modify fishing gear and techniques to reduce sea turtle interactions and mortality associated with such interactions. On June 1, 2004, NMFS released another Opinion on the Atlantic pelagic longline fishery which stated that the fishery was still likely to jeopardize the continued existence of leatherback sea turtles. Another RPA was then developed to attempt to remove jeopardy. The RPA required that NMFS (1) reduce post-release mortality of leatherbacks, (2) improve monitoring of the effects of the fishery, (3) confirm the effectiveness of the hook and bait combinations that are required as part of the proposed action, and (4) take management action to avoid long-term elevations in leatherback takes or mortality. NMFS stated in the Opinion that this RPA must be implemented in its entirety to avoid jeopardy. The Opinion specified an RPA that allows the continuation of the Atlantic highly migratory species fishery without jeopardizing ESA-listed species.

As described in Sections 1.0 and 2.1, consultation has also been previously conducted on the continued operation of the monkfish fishery. Gear types used in the monkfish fishery are known to capture or entangle ESA-listed cetaceans and sea turtles, with some events resulting in injuries and death. Therefore, the environmental baseline for this action also includes the effects of the past operation of the monkfish fishery.

The *American lobster fishery* has been identified as causing injuries to and mortality of loggerhead and leatherback sea turtles as a result of entanglement in buoy lines of the pot/trap gear (NMFS 2002b). Loggerhead or leatherback sea turtles caught/wrapped in the buoy lines of lobster pot/trap gear can die as a result of forced submergence or incur injuries leading to death as a result of severe constriction of a flipper from the entanglement. Given the seasonal distribution of loggerhead sea turtles in Mid-Atlantic and New England waters and the operation of the lobster fishery, loggerhead sea turtles are expected to overlap with the placement of lobster pot/trap gear in the fishery during the months of May through October in waters off of New Jersey through Massachusetts. Compared to loggerheads, leatherback sea turtles have a similar seasonal distribution in Mid-Atlantic and New England waters, but with a more extensive distribution in the Gulf of Maine (Shoop and Kenney 1992; James *et al.* 2005a). Therefore, leatherback sea turtles are expected to overlap with the placement of lobster pot/trap gear in the fishery during the months of May through October in waters off of New Jersey through Maine.

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. The most recent Opinion for this fishery, completed on October 31, 2002, concluded that operation of the Federally-regulated portion of the lobster trap fishery may adversely affect loggerhead and leatherback sea turtles as a result of entanglement in the groundlines and/or buoy lines associated with this type of gear. An ITS was issued with the 2002 Opinion, exempting the annual incidental take (lethal or non-lethal) of 2 loggerhead sea turtles and the biennial incidental take (lethal or non-lethal) of 9 leatherback sea turtles.

Pot/trap gear has also been identified as a gear type causing injuries and mortality of right, humpback, and fin whales (Johnson *et al.* 2005; Waring *et al.* 2009; Glass *et al.* 2009; 73 FR 73032, December 1, 2008). Large whales are known to become entangled in lines associated with multiple gear types. For pot/trap gear, vertical lines attach buoys to the gear while groundline attach the pots/traps in series. Lines wrapped tightly around an animal can cut into the flesh that can lead to injuries, infection and death (Moore *et al.* 2004).

A right whale entanglement in pot/trap gear used in the inshore lobster fishery resulting in death occurred in 2001 (Waring *et al.* 2007). A mortality of a humpback whale in pot/trap gear in the state lobster fishery occurred in 2002 (Waring *et al.* 2007). Other mortalities and serious injuries to ESA-listed cetaceans as a result of pot/trap gear consistent with that used in the lobster fishery have occurred as reported in Moore *et al.* (2004), Johnson *et al.* (2005), Glass *et al.* (2009).

However, it cannot be determined in all cases whether the gear was set in state waters as part of a state lobster fishery or in federal waters. In all waters regulated by the ALWTRP, pot/trap gear set by the American lobster fishery is required to follow regulations set by the plan.

American lobster occurs 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, constituting 76% of the U.S. landings between 1981 and 2007, and 87% since 2002. 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 ASMFCs Interstate Fishery Management Plan (ISFMP). The ISFMP includes measures to constrain or reduce fishing effort in the lobster fishery. In fact, the ASFMC is currently evaluating additional management options to address a May 2010, technical committee report that determined there is a lobster recruitment failure in the SNE stock area. Potential management options under consideration could further reduce fishing effort in the SNE stock area by an additional 75% over current levels. Such measures are of benefit to large whales and sea turtles by reducing the amount of gear (specifically buoy lines) in the water where whales and sea turtles also occur. Due to modifications in the ALWTRP, including replacing the SAM and DAM programs with broad based gear modifications, section 7 consultation has been reinitiated and is currently ongoing.

The *Atlantic bluefish fishery* has been operating in the U.S. Atlantic for at least the last half century, although its popularity did not heighten until the late 1970s and early 1980s (MAFMC and ASMFC 1998).

The most recent formal consultation on the bluefish fishery was completed on July 2, 1999. An ITS was provided with the 1999 Opinion along with non-discretionary RPMs to minimize the impacts of incidental take. As described in the ITS, up to 6 loggerheads, 6 Kemp's ridleys, and 1 shortnose sturgeon were anticipated to be injured or killed annually as a result of the continued operation of the bluefish fishery. Of the incidental takes exempted by the ITS, no more than 3 loggerheads were anticipated to be killed per year. At the time of the 1999 Opinion, no takes of ESA-listed whales were expected to occur in the bluefish fishery.

The anticipated incidental take of ESA-listed sea turtles and shortnose sturgeon in bluefish fishing gear exempted by the 1999 Opinion was based on observed interactions from Sea Sampling data for gear types targeting or capable of catching bluefish (NMFS 1999). At the time of the 1999 Opinion, the bluefish fishery was believed to interact with these species given the time and locations where the fishery occurred. Although no incidental takes of ESA-listed sea turtles had been reported in bottom otter trawl gear for trips that were 'targeting' bluefish (where greater than 50% of the catch was bluefish), incidental takes of loggerhead and Kemp's ridley sea turtles were observed in bottom otter trawl gear where bluefish were caught but constituted less than 50% of the catch (NMFS 1999).

An estimate of loggerhead sea turtle bycatch in bottom otter trawl gear used in the bluefish fishery has been published in a NMFS NEFSC Reference Document (Murray 2008). Using

Vessel Trip Report (VTR) data from 2000-2004 and the average annual bycatch of sea 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 per year (Murray 2008). The 1999 Opinion anticipated the annual incidental take of 6 loggerhead sea turtles. At the time of its publication, the information presented by Murray (2006) was not believed to represent new information on the effects of the bluefish fishery on loggerheads. However, NMFS has received additional information on the effects of the fishery on sea turtles. The captures of two (2) leatherback sea turtles and one (1) unidentified hard-shelled sea turtle were reported in gillnet gear used in the bluefish fishery in 2003 and 2004, records of which were verified by NMFS in 2007. Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the bluefish fishery, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear used in the bluefish fishery, based on VTR data from 2002-2006, was estimated to be 48 per year with a 95% CI of 23-79 (Murray 2009b). Both the trawl and gillnet bycatch estimates described above represent new information on the effects of the bluefish fishery on ESA-listed sea turtles.

Although NMFS was not aware until 2003 that sea turtle interactions with fishing gear targeting bluefish were likely to occur, there is no information to suggest that sea turtle interactions with bluefish 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 bluefish fishing gear were insufficient prior to increased observer coverage in recent years. In addition, there have been no known changes to the seasonal distribution of loggerhead sea turtles in the U.S. Atlantic (CeTAP 1982; Lutcavage and Musick 1985; Keinath *et al.* 1987; Thompson 1988; Shoop and Kenney 1992; Burke *et al.* 1993, 1994) with the exception of recent studies (Morreale *et al.* 2005; Mansfield 2006), which suggest a decrease rather than an increase in the use of some Mid-Atlantic loggerhead foraging areas for unknown reasons.

The commercial bluefish fishery does not typically operate in areas where and at times when large whales occur, however interactions between the whales and bluefish fishery are possible. Right, humpback, and fin whales are known to have been seriously injured and/or killed by gear types used by the bluefish fishery, specifically gillnet gear. Although the gillnet gear has never been traced back to the bluefish fishery specifically, often times the gear responsible can not be identified. The fishery's gear is required to follow regulations set by the ALWTRP.

As a result of the information discussed above, formal consultation on the bluefish fishery was reinitiated on December 18, 2007 to reevaluate the effects of the fishery on ESA-listed whales and sea turtles. The consultation is ongoing.

Sea turtle interactions with gear used in the *Atlantic herring fishery* have not been reported or observed by NMFS observers. However, in past consultations, NMFS concluded that sea turtle takes in fishing gear used in the fishery are reasonably likely to occur due to 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 2006). However, the gear type accounting for the majority of herring landings changed over the ten-year period from 1995-2005 (NEFMC 2006). 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 2006). Murray (2008) did not report an estimate of loggerheads in bottom otter trawl gear targeting herring because of the low number of reported VTR trips in this component of the fishery. An ITS was issued in the September 17, 1999 Biological Opinion anticipating the take of 6 (no more than 3 lethal) loggerheads, 1 leatherback, 1 green, and 1 Kemp's ridley.

An 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. 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 (NEFMC 2006). As a result of these changes, effort in the fishery is expected to be reduced or constrained. 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. The most recent reinitiated (due to the Atlantic salmon listing) consultation on the herring fishery was completed on Feb. 9, 2010. After review and evaluation of observer data (no observed takes of ESA-listed species, despite increased observer coverage in recent years) and information on where and when the fishery operates, NMFS concluded the consultation informally due to the discountable nature of sea turtle or Atlantic salmon interactions.

The *Atlantic mackerel/squid/butterfish fisheries* are managed under a single FMP that includes both the short-finned squid (*Illex illecebrosus*) and long-finned squid (*Loligo pealei*) fisheries. Bottom otter trawl gear is the primary gear type used to land *Loligo* and *Illex* squid. Based on NMFS dealer reports, the majority of *Loligo* and *Illex* squid are fished in the Mid-Atlantic including waters within the action are of this consultation where loggerheads also occur. While squid landings occur year round, the majority of *Loligo* squid landings occur in the fall through winter months while the majority of *Illex* landings occur from June through October (MAFMC 2007a); time periods that overlap in whole or in part with the distribution of loggerhead sea turtles in Mid-Atlantic waters. Gillnets account for a small amount of landings in the mackerel fishery, and all gillnet gear use by this fishery is subject to the requirements of the ALWTRP.

Loggerhead sea turtles are captured in bottom-otter trawl gear used in the *Loligo* and *Illex* squid fisheries, and gillnet gear used by the mackerel fishery and may be injured or killed as a result of forced submergence in the gear. In the latest Opinion, the Atlantic mackerel/squid/butterfish fishery was issued an ITS of 6 (no more than 3 lethal) loggerheads, 2 green, 2 Kemp's ridley, and 1 leatherback sea turtle. In 2008, the NEFSC, using VTR data from 2000-2004, estimated the average annual take (capture) of loggerhead sea turtles in bottom otter trawl gear targeting Atlantic mackerel, squid, butterfish fisheries to be 62 loggerhead sea turtles a year (Murray 2008). NMFS has reinitiated section 7 consultation on the continued operation of the mackerel, squid, butterfish fisheries under the Atlantic Mackerel, Squid, Butterfish FMP in light of this information on the capture of loggerhead sea turtles in bottom otter trawl gear used in the fisheries. That consultation is on-going.

Atlantic pelagic fisheries for swordfish, tuna, sharks, and billfish (highly migratory species) are known to incidentally capture large numbers of sea turtles, particularly in the pelagic longline component. Pelagic longline, pelagic driftnet, bottom longline, and/or purse seine gear have all been documented to hook, capture, or entangle sea turtles. 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. NMFS reinitiated consultation on the pelagic longline component of this fishery as a result of exceeded incidental take levels for loggerhead and leatherback sea turtles (NMFS 2004a). The resulting Opinion stated the long-term continued operation of the pelagic longline fishery for tuna and swordfish was likely to jeopardize the continued existence of leatherback sea turtles, but RPAs were implemented allowing for the continued authorization of the fishery that would not jeopardize leatherbacks. In 2006, the Atlantic HMS pelagic longline fishery had an estimated 771.6 interactions with loggerhead sea turtles and 381.3 interactions with leatherback sea turtles (Garrison *et al.* 2009).

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). Landings from Georges Bank and the Mid-Atlantic dominate the fishery (NEFSC 2007). On Georges Bank and in the Mid-Atlantic, sea scallops are harvested primarily at depths of 30-100 m, while the bulk of landings from the Gulf of Maine are from relatively shallow nearshore waters (<40 m) (NEFSC 2007). Effort (in terms of days fished) in the Mid-Atlantic is about half of what it was prior to implementation of Amendment 4 to the Scallop FMP in the 1990s (NEFSC 2007).

The Scallop FMP was originally implemented on May 15, 1982 (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. EEZ (NEFSC 2007). The limited access program, first established under Amendment 4, remains the basic effort control measure for the scallop fishery. From 2004 through 2008, vessels that did not qualify for a full-time, part-time, or occasional limited access permit could have obtained an open access, general category scallop permit.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 from 2000-2005 there was an increasing percentage of general category landings 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). In 2008, the implementation of Amendment 11 established a limited access general category program consisting of three permit types: Northern Gulf of Maine (NGOM), Incidental, and individual fishing quota (IFQ). The IFQ program became effective March 1, 2010. The implementation of the LAGC fleet contributes to to the management objectives of the fishery by reducing or constraining effort in the general category sector.

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 occurred, 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. Total estimated bycatch of loggerhead turtles in the sea scallop dredge fishery operating in the Mid-Atlantic region from June through November 2003 was 749 turtles (Murray 2004). Estimates for the same time period in 2004 and 2005 were 180 and zero respectively (Murray 2007). Loggerhead annual bycatch estimates in 2004 and 2005 in Mid-Atlantic scallop trawl gear ranged from 81-191 turtles, depending on the estimation methodology used (Murray 2007). 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, 1994) with the exception of recent studies (Morreale *et al.* 2005; 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 8 or so years, has been present for decades.

Formal section 7 consultation on the continued authorization of the scallop fishery was last reinitiated on April 3, 2007, with an Opinion issued by NMFS on March 14, 2008. The ITS for the Opinion was amended on February 4, 2009. In this Opinion, NMFS determined that the continued authorization of the Scallop FMP (including the seasonal use of chain mat modified scallop dredge gear in Mid-Atlantic waters) may adversely affect but was not likely to jeopardize the continued existence of loggerhead, leatherback, Kemp's ridley, and green sea turtles. Of the four species of sea turtles considered in the Opinion, loggerheads are expected to be the most frequently captured in the fishery. The ITS provided with the Opinion exempts the anticipated incidental take of up to 929 loggerheads biennially (up to 595 may be lethal) in scallop dredge gear and 154 loggerheads annually (up to 20 may be lethal) in scallop trawl gear. 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 incidental 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. RPMs to minimize the impact of these incidental takes are also included in the Opinion, including an RPM to limit scallop dredge fishing effort in the Mid-Atlantic area (NMFS 2008b), to be in effect by FY 2010. Measures to minimize the impact of turtle takes were implemented for FY 2010 through through Framework 21 to the Scallop FMP and will be re-evaluated in future Frameworks.

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 900 meters with concentrations between 70 and 100 meters and at 190 meters. The directed monkfish fishery uses several gear types that may entangle protected species, including gillnet and trawl gear.

Gillnet gear used in the monkfish fishery is known to capture ESA-listed sea turtles. Two unusually large stranding events occurred in April and May 2000 during which 280 sea turtles

(275 loggerheads and 5 Kemp's ridleys) washed ashore on ocean facing beaches in North Carolina. Although there was not enough information to specifically determine the cause of the sea turtle deaths, there was information to suggest that the turtles died as a result of entanglement with large-mesh gillnet gear. The monkfish gillnet fishery, which uses a large-mesh gillnet, was known to be operating in waters off of North Carolina at the time the stranded turtles would have died. As a result, 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; March 21, 2002) and were implemented to reduce the impact of the monkfish and other large-mesh gillnet fisheries on endangered and threatened species of 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.

A section 7 consultation conducted in 2001 concluded that the operation of the fishery may adversely affect sea turtles, but was not likely to jeopardize their continued existence. In 2003, proposed changes to the Monkfish FMP led to reinitiation of consultation to determine the effects of those actions on ESA-listed species. The resulting biological opinion concluded the continued operation of the fishery under the proposed changes was likely to adversely affect green, Kemp's ridley, loggerhead and leatherback sea turtles, but was not likely to jeopardize their continued existence (NMFS 2003b). The ITS issued with the 2003 Opinion exempted the annual incidental take of three (3) loggerheads and one (1) non-loggerhead sea turtle in monkfish gillnet gear and one (1) sea turtle (either loggerhead, leatherback, Kemp's ridley, or green) in monkfish trawl gear.

An estimate of loggerhead sea turtle bycatch in bottom otter trawl gear used in the monkfish fishery has been published in a 2008 NEFSC Reference Document (Murray 2008). Using VTR data from 2000-2004 and the average annual bycatch of sea turtles as described in Murray (2006), the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the monkfish fishery was estimated to be 2 loggerhead sea turtles a year (Murray 2008). This information represents new information on the capture of loggerhead sea turtles in the monkfish fishery. As a result, this information contributed to NMFS reinitiating formal section 7 consultation on the continued operation of the monkfish fishery under the Monkfish FMP on April 2, 2008. Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the monkfish fishery, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear used in the monkfish fishery, based on VTR data from 2002-2006, was estimated to be 118 per year with a 95% CI of 68-171 (Murray 2009b). A thorough analysis of sea turtle interactions with gillnet gear is being included in the ongoing consultation.

Use of gillnet gear in the fishery is also affected by measures implemented under the ALWTRP. In the June 2001 Opinion, NMFS determined that the continued operation of the fishery would jeopardize the continued existence of right whales as a result of entanglement in gillnet gear used in the fishery, causing serious injury or death. The RPA issued to the monkfish fishery in the 2001 Opinion, and reissued in the 2003 Opinion, included the SAM and DAM programs under the ALWTRP. There have been no confirmed entanglements of right whales in gillnet gear set to target monkfish. However, right, humpback and fin whale entanglements in gillnet gear of unidentified origin have occurred (Johnson *et al.* 2005; Waring *et al.* 2009). The SAM and DAM programs have been replaced with broad based gear modifications under the ALWTRP. Section 7 consultation has been reinitiated with the monkfish fishery due to new information received on sea turtle takes in bottom trawl gear and changes in management of interactions between endangered whales and commercial fishing gear.

The *Northeast multispecies fishery* operates throughout the year, with peaks in the spring and from October through February. Multiple gear types are used in the fishery including sink gillnet, trawl, and pot/trap gear, which are known to be a source of injury and mortality to right, humpback, and fin whales as well as loggerhead and leatherback sea turtles as a result of entanglement and capture in the gear (NMFS 2001a). The Northeast multispecies sink gillnet fishery has historically occurred from the periphery of the Gulf of Maine to Rhode Island in water as deep as 360 feet. In recent years, more of the effort in the fishery has occurred in offshore waters and into the Mid-Atlantic. 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 fishery. The exact relationship between multispecies fishing effort and the number of endangered species interactions with gear used in the fishery is unknown. However, in general, less fishing effort results in less time that gear is in the water and therefore less opportunity for sea turtles or cetaceans to be captured or entangled in multispecies fishing gear. Using VTR data from 2000-2004 and the average annual bycatch of sea 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 (Murray 2008). This information represents new information on the capture of loggerhead sea turtles in the Northeast multispecies fishery. Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the Northeast multispecies fishery, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear targeting 'other species' which includes gillnet gear used in the Northeast multispecies fishery, based on VTR data from 2002-2006, was estimated to be 3 per year (Murray 2009b). A thorough analysis of sea turtle interactions with gillnet gear is being included in the ongoing consultation.

Gillnet and trap/pot gear in the fishery is also affected by measures implemented under the ALWTRP. In the June 2001 Northeast multispecies Biological Opinion, NMFS determined that the continued operation of the fishery would jeopardize the continued existence of right whales as a result of entanglement in gillnet gear used in the fishery, causing serious injury or death. The RPA issued in the 2001 Opinion led to implementation of the SAM and DAM programs under the ALWTRP. Recently, the SAM and DAM programs have been replaced with broad based gear modifications under the ALWTRP. Given this new information on sea turtle takes and the new ALWTRP management measures which may affect ESA-listed species in a manner or to an extent not previously considered, NMFS has, reinitiated formal section 7 consultation on the continued authorization of the multispecies fishery under the Northeast Multispecies FMP.

Section 7 consultation was completed on the *red crab fishery* during the proposed implementation of the Red Crab FMP (NMFS 2002c). The Opinion concluded that the action was not likely to result in jeopardy to any ESA-listed species under NMFS' jurisdiction. The 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 1,300-2,600 feet along the continental shelf in the Northeast region, and is limited to waters north of 35°15.3'N (Cape Hatteras, North Carolina) and south of the Hague Line. Following concerns that red crab could be overfished, an FMP was developed and became effective on October 21, 2002. In the 2002 biological opinion, an ITS was provided for leatherback and loggerhead sea turtles, which exempts the incidental take of 1 loggerhead and 1 leatherback sea turtle annually as a result of entanglement in lines associated with the pot/trap gear utilized in the fishery. Right, humpback, fin and sei whales are also at risk of entanglement in gear used by the red crab fishery. Gear used by this fishery is required to be in compliance with the ALWTRP. One exemption from the ALWTRP that affects the red crab fishery is the deep water exemption. The sinking groundline requirement is not required for gear that is fished at depths greater than 280 fathoms. Whales and sea turtles in the action are not known to commonly dive to depths greater than 275 fathoms. Therefore, this exemption is unlikely to have an adverse impact on entanglement risks.

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. Section 7 consultation on the Skate FMP was completed July 24, 2003, and concluded that operation of the skate fishery may adversely affect ESA-listed sea turtles as a result of interactions with (capture in) gillnet and trawl gear. Subsequently, the NEFSC, using VTR data from 2000-2004, estimated the average annual take (capture) of loggerhead sea turtles in bottom otter trawl gear used in the directed skate to be 24 loggerhead sea turtles a year (Murray 2008). This information represents new information on sea turtle takes in the skate fishery and NMFS has reinitiated section 7 consultation on the continued operation of the skate fishery. Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the skate fishery, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear used in the skate fishery, based on VTR data from 2002-2006, was estimated to be 9 per year with a 95% CI of 5-15 (Murray 2009b). A thorough analysis of sea turtle interactions with gillnet gear is being included in the ongoing consultation.

ESA-listed cetaceans have also been known to interact with gillnet gear, thus interaction may occur where the gear and the cetacean distributions overlap. The 2003 Biological Opinion concluded that the skate fishery was not likely to jeopardize the continued existence of any ESA-listed species under NMFS jurisdiction. Gillnet gear used in the skate fishery is required to be in compliance with the ALWTRP.

The *spiny dogfish fishery* in the U.S. EEZ is managed under the Spiny Dogfish FMP. The primary gear types for the spiny dogfish fishery are sink gillnets, otter trawls, bottom longline, and driftnet gear (NMFS NEFSC 2003). The predominance of any one gear type has varied over time (NMFS NEFSC 2003). In 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) (NMFS NEFSC 2006). More recently, data from fish dealer reports in FY 2008 indicate that spiny dogfish landings came mostly from sink gill nets (68.2%), and hook gear (15. 2%), bottom otter trawls (4.9%), as well as unspecified (7.7%) or other gear (3.9%) (MAFMC 2010). Sea turtles can be incidentally captured in all gear sectors of the spiny dogfish fishery, which can lead to injury and death as a result of forced submergence in the gear. ESA-listed cetaceans are also known to be seriously injured or killed from interaction with sink gillnet gear.

NMFS reinitiated section 7 consultation on the Spiny Dogfish FMP on May 4, 2000, to reevaluate the effects of the spiny dogfish gillnet fishery on sea turtles and cetaceans following the death of a right whale in 1999 as a result of entanglement in gillnet gear that may have originated from the spiny dogfish fishery (NMFS 2001b). The FMP for spiny dogfish called for a 30% reduction in quota allocation levels for 2000 and a 90% reduction in 2001. Although there were delays in implementing the plan, quota allocations were substantially reduced over the 4.5 year rebuilding schedule; this has resulted in a substantial decrease in effort directed at spiny dogfish. The reduction in effort has likely benefited protected species by reducing the likelihood that gear interactions would occur. As a result, the June 14, 2001 Opinion on the fishery concluded that its authorization under the Spiny Dogfish FMP may adversely affect but was not likely to jeopardize the continued existence of ESA-listed sea turtles. A new ITS was provided for the incidental take of sea turtles in the fishery. It exempted the annual incidental take of 3 loggerheads (no more than 2 lethal), 1 leatherback, 1 Kemp's ridley, and 1 green sea turtle in gear used in the fishery.

The NEFSC, using VTR data from 2000-2004, estimated the average annual take (capture) of loggerhead sea turtles in bottom otter trawl gear used in the directed spiny dogfish to be 1 loggerhead sea turtle a year (Murray 2008). Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the spiny dogfish fishery, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear used in the spiny dogfish fishery, based on VTR data from 2002-2006, was estimated to be 1 per year with a 95% CI of 0-1 (Murray 2009b). A thorough analysis of sea turtle interactions with gillnet and trawl gear is being included in the ongoing consultation.

The same Opinion also concluded that the continued operations of the spiny dogfish fishery would adversely affect North Atlantic right whales. The Opinion provided RPA which included components to minimize the overlap of right whales and spiny dogfish gillnet gear (e.g., SAM and DAM program introduced to the ALWTRP), expand gear modifications to the Mid-Atlantic and southeastern U.S. waters, continued gear research, and monitor the implementation and effectiveness of the RPA. In 2008, Section 7 consultation on the continued authorization of the spiny dogfish fishery was reinitiated by NMFS due to replacing the SAM and DAM programs with broad based gear modifications under the ALWTRP, which represents new information not previously considered, on the effects the fishery may have on ESA-listed whales.

The *summer flounder, scup, and black sea bass fisheries* are managed under one FMP. Bottom otter and beam trawl gear are used most frequently in the commercial fisheries for all three species (MAFMC 2007b). Gillnets, handlines, dredges, and pots/traps are also occasionally used (MAFMC 2007b). 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. It currently exempts the annual incidental take of up to 19 loggerhead or Kemp's ridley sea turtles and 2 green sea turtles (NMFS 2001c). 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).

Significant measures have been developed to reduce the incidental take of sea turtles in summer flounder trawls and trawls that meet the definition of a summer flounder trawl (which includes fisheries for other species like scup and black sea bass). TEDs are required throughout the year for trawl nets fished from the North Carolina/South Carolina border to Oregon Inlet, North Carolina, and seasonally (March 16-January 14) for trawl vessels fishing between Oregon Inlet, North Carolina, and Cape Charles, Virginia. Effort in the summer flounder, scup, and black sea bass fisheries has also declined since the 1980s and since each fishery became managed under the FMP. Therefore, effects to sea turtles are expected, in general, to have declined as a result of the decline in fishing effort. Nevertheless, the fisheries primarily operate in Mid-Atlantic waters in areas and times when sea turtles occur. Thus, there is a continued risk of sea turtle captures causing injury and death in summer flounder, scup, and black sea bass fishing gear.

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 192 loggerhead sea turtles (Murray 2008). This information represents new information on the capture of loggerhead sea turtles in the summer flounder, scup, and black sea bass fisheries. NMFS has, therefore, reinitiated section 7 consultation on the continued authorization of the summer flounder, scup, and black sea bass fisheries. Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the summer flounder, scup, and black sea bass fisheries, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear used in the summer flounder, scup, and black sea bass fisheries, based on VTR data from 2002-2006, was estimated to be 6 per year with a 95% CI of 2-11 (Murray 2009b). A thorough analysis of ESA-listed whales and sea turtle interactions with gillnet gear is being included in the ongoing consultation. All gillnet and pot/trap gear used by the summer flounder, scup, and black sea bass fishery are subject to complying with the ALWTRP.

A summary of the current *tilefish fishery* is provided in the 48th Northeast Regional Stock Assessment Report (NMFS NEFSC 2009). 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. Tilefish have some unique habitat characteristics, and are found in a warm water band (9°-14°C) approximately 250 to 1,200 feet deep on the outer continental shelf and upper slope of the U.S. Atlantic coast. Because of their restricted habitat and low biomass, the tilefish fishery in recent years has occurred in a relatively small area in the Mid-Atlantic Bight, south of New England and west of New Jersey. Bottom longline gear equipped with circle hooks is the primary gear type used in the tilefish fishery.

The effects of the Northeast and Mid-Atlantic tilefish fishery on ESA-listed species were considered during formal section 7 consultation on the implementation of a new Tilefish FMP, concluded on March 13, 2001, with the issuance of a non-jeopardy biological opinion. The Opinion included an ITS for loggerhead and leatherback sea turtles, exempting the annual incidental take of 6 loggerheads and 1 leatherback as a result of capture, entanglement, or hooking in bottom longline and/or bottom trawl gear associated with the fishery (NMFS 2001d).

On December 2, 2002, NMFS completed an Opinion for *shrimp trawling in the southeastern U.S.* under proposed revisions to the TED regulations (68 FR 8456, February 21, 2003). This Opinion determined that the shrimp trawl fishery under the revised TED regulations may adversely affect but would not jeopardize the continued existence of any sea turtle species (NMFS 2002a). This determination was based, in part, on the Opinion's analysis that showed that the revised TED regulations were expected to reduce shrimp trawl related mortality by 94% for loggerheads and 97% for leatherbacks. The ITS included with the Opinion exempted the annual incidental take of up to 163,160 loggerheads (3,948 mortalities), 3,090 leatherbacks (80 mortalities), 155,503 Kemp's ridleys (4,208 mortalities), and 18,757 greens (514 mortalities).

Recently, however, NMFS has estimated that the annual take levels and mortalities of sea turtles in the Gulf of Mexico shrimp fishery are significantly lower than what is exempted by the 2002 Opinion. In addition to improvements in TED designs and TED enforcement, interactions between sea turtles and the shrimp fishery have also been declining because of reductions in fishing effort unrelated to fisheries management actions. The 2002 Opinion take estimates are based in part on fishery effort levels. In recent years, low shrimp prices, rising fuel costs, competition with imported products, and the impacts of recent hurricanes in the Gulf of Mexico have all impacted the shrimp fleets; in some cases reducing fishing effort by as much as 50% for offshore waters of the Gulf of Mexico (GMFMC 2007). As a result, sea turtle interactions and mortalities in the Gulf of Mexico, most notably for loggerheads and leatherbacks, have been substantially less than projected in the 2002 Opinion. For the U.S. south Atlantic shrimp fishery, there is currently no new information on the number of takes and mortalities occurring annually, although NMFS is currently researching this as well.

On August 16, 2010, NMFS reinitiated formal section 7 consultation on the shrimp trawl fishery in the southeastern U.S. to reanalyze its effects on sea turtles. This was primarily due to the after-effects of the April 20, 2010 BP Deepwater Horizon oil spill, from which NMFS has documented extraordinarily high numbers of sea turtle strandings in the Gulf of Mexico, particularly Mississippi Sound. NMFS suspects that much of the increased level of strandings is attributable to shrimp fishing activity as there is recent evidence of a lack of compliance with TED regulations and tow time provisions. In addition, there is also new information that trawl CPUE of sea turtles in Louisiana nearshore waters is elevated. That consultation is ongoing.

4.1.2 Non-federally regulated fisheries

Several trap/pot fisheries, gillnet and trawl fisheries for non-federally regulated species do occur in the action area. The amount of gear contributed to the environment by these fisheries is unknown. In most cases, there is no observer coverage of these fisheries and the extent of interactions with ESA-listed species is unknown.

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 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 to take 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 (NMFS 2004b). In North Carolina, a large-mesh gillnet fishery for summer flounder in the southern portion of Pamlico Sound was found to take of sea turtles in gillnet gear. A Section 10 incidental take permit was issued to this fishing in 2001 based on take levels set by NMFS during the 2000 fishing season for large mesh gillnet fisheries in both shallow and deep water. The annual estimated lethal and live takes for the 2002-2004 fishing seasons was 24 lethal and 164 live takes of each Kemp's ridley, green, and loggerhead sea turtles. The permit was renewed for the 2005-2010 fishing years and a new take estimates were derived from the 2001-2004 at-sea monitoring program. The new ITS exempted the take of 41, 168, and 41 for Kemp's ridley, green, and loggerhead turtles respectively.

An *Atlantic croaker fishery* using trawl and gillnet gear also occurs within the action area and turtle takes have been observed in the fishery. The average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the Atlantic croaker fishery was estimated to be 41 loggerhead sea turtles (Murray 2008). Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the Atlantic croaker fishery, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear used in the Atlantic croaker fishery, based on VTR data from 2002-2006, was estimated to be 11 per year with a 95% CI of 3-20 (Murray 2009b). ESA-listed cetaceans have also been known to interact with gillnet gear, thus interaction may occur where the gear and the cetacean distributions overlap.

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, sea turtle bycatch in the weakfish fishery has occurred (Murray 2008, 2009a, 2009b). The average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the weakfish fishery was estimated to be 4 loggerhead sea turtles (Murray 2008). Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the weakfish fishery, has also been recently published by Murray (2009a, 2009b). The average annual bycatch of loggerhead sea turtles in gillnet gear used in the weakfish fishery, based on VTR data from 2002-2006, was estimated to be 1 per year with a 95% CI of 0-1 (Murray 2009b). ESA-listed cetaceans have also been known to interact with gillnet gear, thus interaction may occur where the gear and the cetacean distributions overlap.

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

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.* 2005), commensurate with noted declines in the abundance of horseshoe crab and other crab species raises concerns that crab fisheries may be impacting the forage base for loggerheads in some areas of their range.

Sea turtle takes in the *Virginia pound net fishery* have been observed. 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 4.4.3.4 below, NMFS has taken regulatory action to address turtle takes in the Virginia pound net fishery. Although no incidental captures have been documented from fish traps set off North Carolina, they are another potential anthropogenic impact to loggerheads and other sea turtles (NMFS SEFSC 2001).

Observations of state recreational fisheries have shown that loggerhead, leatherback, and green sea turtles are known to bite baited hooks, and loggerheads frequently ingest the hooks. Hooked sea turtles have been reported by the public fishing from boats, piers, beaches, banks, and jetties, and from commercial fishermen fishing for snapper, grouper, and sharks with both single rigs and bottom longlines (NMFS SEFSC 2001). A summary of known impacts of hook-and-line incidental captures to loggerhead sea turtles can be found in the TEWG (1998, 2000) reports. Although no incidental captures have been documented from fish traps set off North Carolina, they are another potential anthropogenic impact to loggerheads and other sea turtles (NMFS SEFSC 2001).

4.2 Military Vessel Activity and Operations

Potential sources of adverse effects to sea turtles from Federal vessel operations in the action area include operations of the U.S. Navy (USN), U.S. Coast Guard (USCG), Environmental Protection Agency (EPA), Army Corps of Engineers (ACOE), and NOAA. NMFS has previously conducted formal consultations with the USN, USCG, and NOAA on their vessel-based operations. NMFS has also conducted section 7 consultations with the Minerals Management Service (MMS), Federal Energy Regulatory Commission (FERC), and Maritime Administration (MARAD) on vessel traffic related to energy projects in the Northeast Region and has implemented conservation measures. Through the section 7 process, where applicable, NMFS has and will continue to identify conservation measures for all these agency vessel operations to avoid or minimize adverse effects to listed species.

Several Opinions for USN activities (NMFS 1996, 1997, 2006b, 2008c, 2009a,b) and USCG (NMFS 1995, 1998c) contain details on the scope of vessel operations for these agencies and the conservation measures that are being implemented as standard operating procedures. In the U.S. Atlantic, the operation of USCG boats and cutters is not expected to jeopardize the continued existence of the ESA-listed species while operating with an estimated take of no more than one individual sea turtle, of any species, per year (NMFS 1995, 1998c).

In June 2009, NMFS prepared an Opinion on USN activities in each of their four training range complexes along the U.S. Atlantic coast—Northeast, Virginia Capes, Cherry Point, and Jacksonville (NMFS 2009b). That Opinion found that no whales are likely to die or be wounded as a result of their exposure to U.S. Navy training in the Atlantic Ocean. However, the Virginia Capes Range Complex was assigned potential take in the form of harassment of fin, sei and humpback whales. Regarding impacts to sea turtles, the Virginia Capes Range Complex and Jacksonville Range Complex were attributed with potential harassment of leatherback sea turtles and hard shell turtles and the Virginia Capes Range Complex has been characterized as having the potential to harm loggerhead and Kemp's ridley turtles.

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

NMFS has also conducted more recent section 7 consultations on USN explosive ordnance disposal, mine warfare, sonar testing (e.g., AFAST, SURTASS LFA), and other major training exercises (e.g., bombing, Naval gunfire, combat search and rescue, anti-submarine warfare, and torpedo and missile exercises) in the Atlantic Ocean. These consultations have determined that the proposed USN activities may adversely affect but would not jeopardize the continued existence of ESA-listed marine mammals and sea turtles (NMFS 2008c, 2009a,b). NMFS estimated that five (5) loggerhead and six (6) Kemp's ridley sea turtles are likely to be harmed as a result of training activities in the Virginia Capes Range Complex from June 2009 to June 2010, and that nearly 1,500 sea turtles, including 10 leatherbacks, are likely to experience harassment (NMFS 2009b).

Similarly, operations of vessels by other Federal agencies within the action area (NOAA, EPA, and ACOE) may adversely affect ESA-listed marine mammals and sea turtles. However, vessel activities of those agencies are often limited in scope, as they operate a limited number of vessels or are engaged in research/ operational activities that are unlikely to contribute a large amount of risk. For example, NOAA research vessels conducting fisheries surveys for the NEFSC are estimated to take no more than nine sea turtles per year (eight (8) alive, one (1) dead). This includes up to seven (7) loggerheads as well as an additional loggerhead, leatherback, Kemp's ridley, or green sea turtle per year during bottom trawl surveys and one (1) loggerhead, leatherback, Kemp's ridley, or green sea turtle per year during scallop dredge surveys (NMFS 2007b).

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 15 loggerhead turtles or one (1) Kemp's ridley or green turtle throughout the duration of the project. 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 (5) loggerheads and one (1) Kemp's ridley or green turtle due to the lack of observed takes in the previous cycle, along with information on the levels of anticipated and observed take in hopper dredging projects in nearby locations.

In January 1996 NMFS completed section 7 consultation on the Navy's Dam Neck Annex beach nourishment project, 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 (10) loggerheads and one (1) 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 (4) loggerheads and one (1) 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 ESA-listed species. 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. Larger oil spills may result from accidents, although these events would be rare. No direct adverse effects on listed species resulting from fishing vessel fuel spills have been documented.

4.3.3 Pollution

Anthropogenic sources of marine pollution, while difficult to attribute to a specific Federal, state, local or private action, may affect ESA-listed species in the action area. 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. Marine debris (e.g., discarded fishing line or lines from boats) can entangle cetaceans or sea turtles causing serious injury or mortality. Turtles commonly ingest plastic or mistake debris for food, as observed with the leatherback sea turtle. Jellyfish are a preferred prey for leatherbacks, and similar looking plastic bags are often found in the turtles stomach contents (Magnuson *et al.* 1990).

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 affect ESA-listed species if the pollution reduces the food available to marine animals.

4.3.4 Coastal development

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

4.3.5 Catastrophic events

Commercial vessel traffic/shipping imposes the potential for oil/chemical spills. With human population rising and commerce becoming increasingly globalized, so too does the demand for more ships. The pathological effects of oil spills have been documented in laboratory studies of marine mammals and sea turtles (Vargo *et al.* 1986). There have been a number of documented oil spills in the northeastern U.S. Oil spills outside the action area also have the potential to affect ESA-listed species that occur within the action area. For instance, on April 20, 2010 the Deepwater Horizon oil spill occurred in the Gulf of Mexico off the coast of Louisiana. As ESA-listed species (e.g., loggerhead and Kemp's ridley sea turtles) are known to migrate through, forage, and/or nest along the coastal waters of the Gulf of Mexico, the oil spill is likely to affect their populations; however, because all the information on sea turtle and other ESA-listed species' stranding, deaths, and recoveries has not yet been documented, the effects of the oil spill on their populations cannot be determined at this time.

4.3.6 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. 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.

Sea Turtles

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). It is believed that increases in sea level, approximately 4.2 mm per year until 2080, have the potential to remove available nesting beaches, particularly on narrow low lying coastal and inland beaches and on beaches where coastal development has occurred (Church *et al.* 2001; IPCC 2007; Nicholls 1998; Fish *et al.* 2005; Baker *et al.* 2006; Jones *et al.* 2007; Mazaris *et al.* 2009). Additionally, global climate change may affect the severity of extreme weather (e.g., hurricanes), with more intense storms expected, which may result in the loss/erosion of or damage to shorelines, and therefore, the loss of potential sea turtle nests and/or nesting sites (Goldenburg *et al.* 2001; Webster *et al.* 2005; IPCC 2007). The cyclical loss of nesting beaches resulting from extreme storm events may then result in a decrease in hatching success and hatchling emergence (Martin 1996; Ross 2005; Pike and Stiner 2007; Prusty *et al.* 2007; Van Houton and Bass 2007). However, there is evidence that, depending on the species, sea turtles species with lower nest site fidelity (i.e., leatherbacks) would be less vulnerable to storm related threats than those with a higher site fidelity (i.e., loggerheads). In fact, it has been reported that sea turtles in Guiana are able to maintain successful nesting despite the fact that between nesting years some beaches they once nested on have disappeared, suggesting that sea turtle species may be able to behavioral adapt to such changes (Pike and Stiner 2007; Witt *et al.* 2008; Plaziat and Augustinius 2004; Girondot and Fretey 1996; Rivalan *et al.* 2005; Kelle *et al.* 2007).

Changes in water temperature are also expected as a result of global climate change. Changes in water temperature are expected affect water circulation patterns perhaps even to the extent that the Gulf Stream is disrupted, which would have profound effects on every aspect of sea turtle life history from hatching success, oceanic migrations at all life stages, foraging, and nesting. (Gagosian 2003; NMFS and USFWS 2007a; 2007b; 2007c; 2007d; Rahmstorf 1997, 1999; Stocker and Schmittner 1997). Thermocline circulation patterns are expected to change in intensity and direction with changes in temperature and freshwater input at the poles (Rahmstorf 1997; Stocker and Schmittner 1997), which will potentially affect not only hatchlings, which rely on passive transport in surface currents for migration and dispersal but also pelagic adults (i.e., leatherbacks) and juveniles, which depend on current patterns and major frontal zones in obtaining suitable prey, such as jellyfish (Hamann *et al.* 2007; Hawkes *et al.* 2009).

Changes in water temperature may also affect prey availability for species of sea turtles. Herbivorous species, such as the green sea turtle, depend primarily on seagrasses as their forage base. Seagrasses could ultimately be negatively affected by increased temperatures, salinities, and acidification of coastal waters (Short and Neckles 1999; Bjork 2008), as well as increased runoff due the expected increase in extreme storm events as a result of global climate change. These alterations of the marine environment due to global climate change could ultimately affect the distribution, physiology, and growth rates of seagrasses, potentially eliminating them from particular areas. However, the magnitude of these effects on seagrass beds, and therefore green sea turtles, are difficult to predict, although some populations of green sea turtles appear to specialize in the consumption of algae (Bjorndal 1997) and mangroves (Limpus and Limpus 2000) and as such, green sea turtles may be able to adapt their foraging behavior to the changing availability of seagrasses in the future. Omnivorous species, such as Kemp's ridley and loggerhead sea turtles, may face changes to benthic communities as a result of changes to water temperature; however, these species are probably less likely to suffer shortages of prey than species with more specific diets (*i.e.,* green sea turtles) (Hawkes *et al.* 2009).

Several studies have also 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 (Hays *et al.* 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 done of loggerhead sea turtles, 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 (i.e., greater than 35° C) resulting in death (Hawkes *et al.* 2007). Glen et al. (2003) also reported that, for green sea turtles, incubation temperatures also appeared to affect hatchling size with smaller turtles produced at higher incubation temperatures; however, it is unknown whether this effect is species specific and what impact it has on the survival of the offspring. 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. (Hawkes et al. 2007; Hamann *et al.* 2007). 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, including the action area; however; variation of sex ratios to incubation temperature between individuals and populations is not fully understood and as such, it is unclear whether sea turtles will (or can) adapt behaviorally to alter incubation conditions to counter potential feminization or death of clutches associated with water temperatures (*e.g.*, choosing nest sites that are located in cooler areas, such as shaded areas of vegetation or higher latitudes; nesting earlier or later during cooler periods of the year) (Hawkes *et al.* 2009).

Ocean acidification related to global warming would also reasonably be expected to negatively affect sea turtles. The term "ocean acidification" describes the process of ocean water becoming corrosive as a result of carbon dioxide (CO_2) being absorbed from the atmosphere. The absorption of atmospheric CO_2 into the ocean lowers the pH of the waters. Evidence of corrosive water caused by the ocean's absorption of CO_2 was found less than 20 miles off the West coast of North America during a field study from Canada to Mexico in the summer of 2007 (Feely *et al.* 2008). This was the first time "acidified" ocean water was found on the continental shelf of western North America. While the ocean's absorption of CO_2 provides a great service to humans by significantly reducing the amount of greenhouse gases in the atmosphere and decreasing the effects of global warming, the resulting change in ocean chemistry could adversely affect marine life, particularly organisms with calcium carbonate shells such as corals, mussels, mollusks, and small creatures in the early stages of the food chain (*e.g.*, plankton). A number of these organisms serve as important prey items for sea turtles.

Although potential effects of climate change on sea turtle species are currently being addressed, fully understanding the effects of climate change on listed species of sea turtles will require development of conceptual and predictive models of the effects of climate change on sea turtles, which to date are still being developed and will depend greatly on the continued acquisition and maintenance of long-term data sets on sea turtle life history and responses to environmental changes. Until such time, the type and extent of effects to sea turtles as a result of global climate change are will continue to be speculative and as such, the effects of these changes on sea turtles cannot, for the most part, be accurately predicted at this time.

Marine Mammals

Marine mammals are also expected to be affected by global climate change. The impact of climate change on cetaceans is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats and the potential decline of forage.

Of the main factors affecting distribution of cetaceans, water temperature appears to be the main influence on geographic ranges of cetacean species (Macleod 2009). Humpback and fin whales are distributed in all water temperature zones, therefore, it is unlikely that their range will be directly affected by an increase in water temperature.

The North Atlantic right whale currently has a range of sub-polar to sub-tropical waters. An increase in water temperature would likely result in a northward shift of range, with both the northern and southern limits moving poleward. The northern limit, which may be determined by feeding habitat and the distribution of preferred prey, may shift to a greater extent than the southern limit, which requires ideal temperature and water depth for calving. This may result in an unfavorable affect on the North Atlantic right whale due to an increase in the length of migrations (Macleod 2009) or a favorable effect by allowing them to expand their range.

Sei whales currently range from sub-polar to tropical waters. An increase in water temperature may be a favorable affect on sei whales, allowing them to expand their range into higher latitudes (Macleod 2009).

Cetaceans are unlikely to be directly affected by sea level rise, although important coastal bays for humpback breeding could be affected (IWC 1997). The indirect effects to marine mammals, that may be associated with sea level rise, is the construction of sea-wall defenses and protective measures for coastal habitats, which may impact coastal marine species and may interfere with migration (Learmonth *et al.* 2006). The effect of sea level rise to cetaceans is likely negligible.

The direct effects of increased CO2 concentrations, and associated decrease in pH (ocean acidification), on marine mammals are unknown (Learmonth *et al.* 2006). Marine plankton is a vital food source for many marine species. Studies have demonstrated adverse impacts from ocean acidification on a reduction in the ability of marine algae and free-swimming zooplankton to maintain protective shells as well as a reduction in the survival of larval marine species. A decline in the marine plankton could have serious consequences for the marine food web.

There are many direct and indirect effects that global climate change may have on marine mammal prey species. More information is needed in order to determine the potential impacts global climate change will have on the timing and extent of population movements, abundance, recruitment, distribution and species composition of prey (Learmonth *et al.* 2006). Changes in climate patterns, ocean currents, storm frequency, rainfall, salinity, melting ice, and an increase in river inputs/runoff (nutrients and pollutants) will all directly affect the distribution, abundance and migration of prey species (Waluda *et al.* 2001; Tynan & DeMaster 1997; Learmonth *et al.* 2006). These changes will likely have several indirect effects on marine mammals, which may include changes in distribution including displacement from ideal habitats, decline in fitness of individuals, population size due to the potential loss of foraging opportunities, abundance, migration, community structure, susceptibility to disease and contaminants, and reproductive success (Macleod 2009). Global climate change may also result in changes to the range and abundance of competitors and predators which will also indirectly affect marine mammals (Learmonth *et al.* 2006). Similarly to sea turtles, a decline in the reproductive fitness as a result of global climate change could have profound effects on the abundance and distribution of large whales in the Atlantic. However, fully understanding the effects of climate change on listed species of marine mammals will require development of conceptual and predictive models of the effects of climate change on marine mammals, which to date are still being developed and will depend greatly on the continued aquistion and maintenance of long-term data sets on marine mammal life history and responses to environmental changes. Until such time, the type and extent of effects to marine mammals as a result of global climate change are will continue to be speculative and as such, the effects of these changes on marine mammals cannot, for the most part, be accurately predicted at this time.

4.4 Reducing Threats to ESA-listed Whales and Sea Turtles

4.4.1 Education and Outreach Activities

Education and outreach activities are considered one of the primary tools to reduce the threats to all protected species. For example, NMFS has been active in public outreach to educate fishermen regarding sea turtle handling and resuscitation techniques, as well as guidelines for recreational fishermen and boaters to avoid the likelihood of interactions with marine mammals. NMFS is engaged in a number of education and outreach activities aimed specifically at increasing mariner awareness of the threat of ship strike to right whales. NMFS intends to continue these outreach efforts in an attempt to reduce interactions with protected species, and to reduce the likelihood of injury to protected species when interactions do occur.

4.4.2 Sea Turtle Stranding and Salvage Network (STSSN)

There is an extensive network of STSSN participants along the Atlantic and Gulf of Mexico coasts which not only collects data on dead sea turtles, but also rescues and rehabilitates live stranded turtles. 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.

4.4.3 Regulatory Measures for Sea Turtles

4.4.3.1 Large-Mesh Gillnet Requirements in the Mid-Atlantic

Since 2002, NMFS has regulated the use of large mesh gillnets in Federal waters off North Carolina and Virginia (67 FR 13098, March 21, 2002) to reduce the impact of these fisheries on ESA-listed sea turtles. Currently, gillnets with stretched mesh size 7-inches (17.8 cm) or larger are prohibited in the Exclusive Economic Zone (as defined in 50 CFR 600.10) during the following times and in the following areas: (1) north of the NC/SC border to Oregon Inlet at all times, (2) north of Oregon Inlet to Currituck Beach Light, NC from March 16 through January 14, (3) north of Currituck Beach Light, NC to Wachapreague Inlet, VA from April 1 through January 14, and (4) north of Wachapreague Inlet, VA to Chincoteague, VA from April 16 through January 14. 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 72E30'W longitude) from February 15-March 15, annually.

NMFS has also issued regulations to address the take of sea turtles in gillnet gear fished in Pamlico Sound, NC. Waters of Pamlico Sound are closed to fishing with gillnets with a stretched mesh size larger than 4 ¼ inch (10.8 cm) from September 1 through December 15 each year to protect sea turtles. The closed area includes all inshore waters of Pamlico Sound, and all contiguous tidal waters, south of 35E 46.3' N. lat., north of 35E00' N. lat., and east of 76E 30' W. long.

4.4.3.2 TED Requirements in Trawl Fisheries

Turtle Excluder Devices (TEDs) are required in the shrimp and summer flounder fisheries. TEDs allow sea turtles to escape the trawl net, reducing injury and mortality resulting from capture in the net. Approved TEDs are required in the shrimp trawl fishery operating in the Atlantic and Gulf Areas unless the trawler is fishing under one of the exemptions (e.g., skimmer trawl, try net) and all requirements of the exemption (50 CFR 223.206) are met. On February 21, 2003, NMFS issued a final rule to amend the TED regulations to enhance their effectiveness in reducing sea turtle mortality resulting from shrimp trawling in the Atlantic and Gulf Areas of the southeastern United States by requiring an escape opening designed to exclude leatherbacks as well as large loggerhead and green turtles (68 FR 8456; February 21, 2003).

TEDs are also required for summer flounder trawlers in the summer flounder fishery-sea turtle protection area. This area is bounded on the north by a line extending along 37° 05'N latitude (Cape Charles, VA) and on the south be a line extending out from the North Carolina-South Carolina border. Vessels north of Oregon Inlet, NC are exempt from the TED requirement from

January 15 through March 15 each year (50 CFR 223.206). The TED requirements for the summer flounder trawl fishery do not require the use of the larger escape opening. NMFS is considering increasing the size of the TED escape opening currently required in the summer flounder fishery and implementing sea turtle conservation requirements in other trawl fisheries and in other areas (72 FR 7382, February 15, 2007; 74 FR 21630, May 8, 2009).

4.4.3.3 Sea Turtle Conservation Requirements in the Virginia Pound Net Fishery

NMFS has issued several regulations to help protect sea turtles from entanglement in and impingement on Virginia pound net gear (66 FR 33489, June 22 2001; 67 FR 41196; June 17, 2002; 68 FR 41942, July 16, 2003; 69 FR 24997, May 5, 2004). Currently, all offshore pound leaders in Pound Net Regulated Area I must meet the definition of a modified pound net from May 6 through July 15. The modified leader has been found to be effective in reducing sea turtle interactions as compared to the unmodified leader. Pound Net Regulated Area I includes Virginia waters of the mainstream Chesapeake Bay, south of 37E 19' N and west of 76E 13' W, and all waters south of 37E 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. Nearshore pound net leaders in Pound Net Regulated Area I and all pound net leaders in Pound Net Regulated area II must have mesh size less than 12 inches (30.5 cm) stretched mesh and may not employ stringers (50 CFR 223.206) from May 6 through July 15 each year. Regulated Area II includes Virginia waters of the Chesapeake Bay outside of Pound Net Regulated Area I defined above, extending from the Maryland-Virginia State line and the Great Wicomico River, Rappahannock River, and Piankatank Rivers downstream of the first bridge in each tributary to the COLREGS line at the mouth of the Chesapeake Bay. Applicable to the 2010 fishing season and beyond, the state of Virginia required modified pound net leaders (as defined by Federal regulations) east of the Chesapeake Bay Bridge year round, and in offshore leaders in Regulated Area I (also as defined by Federal regulations) from May 6 to July 31. This is a 16 day extension of the Federal regulations in this area. In addition, there are monitoring and reporting requirements in this fishery (50 CFR 223.206).

4.4.3.4 Sea Turtle Conservation Requirements in the HMS Fishery

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

4.4.3.5 Modified Gear in the Atlantic Sea Scallop Fishery

To reduce serious injury and mortality to sea turtles resulting from capture in the sea scallop dredge bag, NMFS has required the use of a chain-mat modified dredge in the Atlantic sea scallop fishery since 2006 (71 FR 50361, August 25, 2006; 71 FR 66466, November 15, 2006; 73 FR 18984, April 8, 2008; 74 FR 20667, May 5, 2009). Federally permitted scallop vessels south of 41°09'N lat. from the shoreline to the outer boundary of the EEZ are required to modify their dredge gear by adding an arrangement of horizontal and vertical chains (hereafter referred to as a "chain mat") over the opening of the dredge bag during the period of May 1-November 30 each year. In general, the chain mat gear modification is expected to reduce the severity of some sea turtle interactions with scallop dredge gear. However, this modification is not expected to reduce the overall number of sea turtle interactions with the gear.

4.4.3.6 Sea Turtle Handling and Resuscitation Requirements

NMFS published as a final rule in the *Federal Register* (66 FR 67495, December 31, 2001) 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 regulations (50 CFR 223.206). These measures help to prevent mortality of turtles caught in fishing or scientific research gear.

4.4.3.7 Sea Turtle Entanglements and Rehabilitation

Any agent or employee of NMFS, the USFWS, the U.S. Coast Guard, or any other Federal land or water management agency, or any agent or employee of a state agency responsible for fish and wildlife, when acting in the course of his or her official duties, is allowed to take threatened or endangered sea turtles encountered in the marine environment if such taking is necessary to aid a sick, injured, or entangled endangered sea turtle, or dispose of a dead endangered sea turtle, or salvage a dead endangered sea turtle that may be useful for scientific or educational purposes (50 CFR 223.206(b); 70 FR 42508, July 25, 2005; 50 CFR 222.310).

4.4.4 Atlantic Large Whale Take Reduction Plan

The Atlantic Large Whale Take Reduction Plan (ALWTRP) reduces the risk of serious injury to or mortality of large whales due to incidental entanglement in U.S. commercial fishing gear. The ALWTRP focuses on the critically endangered North Atlantic right whale, but is also intended to reduce entanglement of endangered humpback and fin whales. The plan is required by the Marine Mammal Protection Act (MMPA) and has been developed by NOAA's National Marine Fisheries Service (NMFS). The ALWTRP covers the U.S. Atlantic Exclusive Economic Zone (EEZ) from Maine through Florida (26°46.5'N lat.). The requirements are year-round in the Northeast, and seasonal in the Mid and South Atlantic.

The plan has been developing in collaboration with the Atlantic Large Whale Take Reduction Team (ALWTRT), which consists of fishing industry representatives, environmentalists, state and federal officials, and other interested parties. The ALWTRP is an evolving plan that changes as NMFS and the ALWTRT learn more about why whales become entangled and how fishing practices might be modified to reduce the risk of entanglement. Regulatory actions are directed at reducing serious entanglement injuries and mortality of right, humpback and fin whales from fixed gear fisheries (*i.e.*, trap and gillnet fisheries). The non-regulatory component of the ALWTRP is composed of four principal parts: (1) gear research and development, (2) disentanglement, (3) the Sighting Advisory System (SAS), and (4) education/outreach. These components will be discussed in more detail below. The first ALWTRP went into effect in 1997.

4.4.4.1 Regulatory Measures to Reduce the Threat of Entanglement on Whales

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

The ALWTRT initially concluded that all parts of gillnet and trap/pot gear can and have caused entanglements. Initial measures in the ALWTRP addressed both parts of the gear, and since then, the ALWTRT has identified the need to further reduce risk posed by both vertical and horizontal portions of gear. Research and testing has been ongoing to identify risk reduction measures that are feasible. The regulations recently placed in effect focused on horizontal lines.

The ALWTRP measures vary by designated area that roughly approximate the Federal Lobster Management Areas (FLMAs) designated in the Federal lobster regulations. The major requirements of the ALWTRP are:

- No buoy line floating at the surface.
- No wet storage of gear (all gear must be hauled out of the water at least once every 30 days).
- Surface buoys and buoy line need to be marked to identify the vessel or fishery.
- All buoys, floatation devices and/or weights must be attached to the buoy line with a weak link. This measure is designed so that, if a large whale does become entangled, it could extert enough force to break the weak link and break free of the gear reducing the risk of injury or mortality.
- All groundline must be made of sinking line.

In addition to gear modification requirements, the ALWTRP prohibits all trap/pot fishing in The Great South Channel from April 1 – June 30.

In addition to the regulatory measures recently implemented to reduce the risk of entanglement in horizontal/ground lines, NMFS, in collaboration with the ALWTRT, has developed a strategy to further reduce risk associated with vertical lines.

It is anticipated that the final regulations implementing the vertical line strategy will prioritize risk reduction in areas where there is the greatest co-occurrence of vertical lines and large whales. There are two ways to achieve a reduced risk: (1) maintain the same number of active lines but decrease the risk from each one (not currently feasible), or (2) reduce the number of lines in the water column.

Whale distribution data will be used to help prioritize areas for implementation of future vertical line action(s). These data will be overlaid with the vertical line distribution data to look at the combined densities by area. A model is being developed and constructed to allow gear configurations to be manipulated and determine what relative co-occurrence reductions (as a proxy for risk) can be achieved by gear configuration changes and/or effort reductions by area. This co-occurrence analysis is an integral component of the vertical line strategy that will further minimize the risk of large whale entanglement and associated serious injury and death. The actions and timeframe for the implementation of the vertical line strategy is as follows:

- Vertical line model development over the next year for all areas to gather as much information as possible regarding the distribution and density of vertical line fishing gear. Time frame: Northeast, Southeast, and Mid-Atlantic by April 2011;

- Compile and analyze whale distribution and density data in a manner to overlay with vertical line density data. Time frame: Northeast, Southeast and Mid-Atlantic by April 2011;

- Development of vertical line and whale distribution co-occurrence overlays. Time frame: by April 2011;

- Develop and publish proposed rule to implement risk reduction from vertical lines. Time frame: by April 2013;

- Develop and publish final rule to implement risk reduction from vertical lines. Time frame: by April 2014;

- Implement final rule to implement risk reduction from vertical lines. Time frame: by January 1, 2015;

- Develop an ALWTRP monitoring plan designed to track implementation of vertical line strategy, including risk reduction. Time frame: Adopt plan by January 2012, with annual interim reports beginning in July 2012.

4.4.4.2 Non-regulatory components of the ALWTRP

4.4.4.2.1 Gear Research and Development

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

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

4.4.4.2.2 Large Whale Disentanglement Program

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

4.4.4.2.3 Sighting Advisory System (SAS)

Although the Sighting Advisory System (SAS) was developed primarily as a method of locating right whales and alerting mariners to right whale sighting locations in a real time manner, the SAS also addresses entanglement threats. Fishermen can obtain SAS sighting reports and make necessary adjustments in operations to decrease the potential for interactions with right whales. Some of these sighting efforts have resulted in successful disentanglement of right whales. The SAS is discussed further in section 4.4.6.5.

4.4.4.2.4 Educational Outreach

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

4.4.5 Ship Strike Reduction Program

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

4.4.6 Regulatory Measures to Reduce Vessel Strikes to Large Whales

4.4.6.1 Restricting vessel approach to right whales

In one recovery action aimed at reducing vessel-related impacts, including disturbance, NMFS published a proposed rule in August 1996 restricting vessel approach to right whales (61 FR 41116, August 7, 1996) to a distance of 500 yards. The Recovery Plan for the North Atlantic right whale identified anthropogenic disturbance as one of many factors which had some potential to impede right whale recovery (NMFS 2005a). Following public comment, NMFS published an interim final rule in February 1997 codifying the regulations. With certain exceptions, the rule prohibits both boats and aircraft from approaching any right whale closer than 500 yds. Exceptions for closer approach are provided for the following situations, when: (a) compliance would create an imminent and serious threat to a person, vessel, or aircraft; (b) a vessel is restricted in its ability to maneuver around the 500-yard perimeter of a whale; (c) a vessel is investigating or involved in the rescue of an entangled or injured right whale; or (d) the vessel is participating in a permitted activity, such as a research project. If a vessel operator finds that he or she has unknowingly approached closer than 500 yds, the rule requires that a course be steered away from the whale at slow, safe speed. In addition, all aircraft, except those involved in whale watching activities, are exempted from these approach regulations. This rule is expected to reduce the potential for vessel collisions and other adverse vessel-related effects in the environmental baseline.

4.4.6.2 Mandatory Ship Reporting System (MSR)

In April 1998, the USCG submitted, on behalf of the US, a proposal to the International Maritime Organization (IMO) requesting approval of a mandatory ship reporting system (MSR) in two areas off the East coast of the U.S., the right whale feeding grounds in the Northeast, and the right whale calving grounds in the Southeast. The USCG worked closely with NMFS and other agencies on technical aspects of the proposal. The package was submitted to the IMO's Subcommittee on Safety and Navigation for consideration and submission to the Marine Safety Committee at IMO and approved in December 1998. The USCG and NOAA play important roles in helping to operate the MSR system, which was implemented on July 1, 1999. Ships entering the Northeast and Southeast MSR boundaries are required to report the vessel identity, date, time, course, speed, destination, and other relevant information. In return, the vessel receives an automated reply with the most recent right whale sightings or management areas in the area and information on precautionary measures to take while in the vicinity of right whales.

4.4.6.3 Vessel Speed Restrictions

A key component of NOAA's right whale ship strike reduction program is the implementation of speed restrictions for vessels transiting the US Atlantic in areas and seasons where right whales predictably occur in high concentrations. The Northeast Implementation Team (NEIT)-funded "Recommended Measures to Reduce Ship Strikes of North Atlantic Right Whales" found that seasonal speed and routing measures could be an effective means of reducing the risk of ship strike along the US East coast. Based on these recommendations, NMFS published an Advance Notice of Proposed Rulemaking (ANPR) in June 2004 (69 FR 30857; June 1, 2004), and subsequently published a proposed rule on June 26, 2006 (71 FR 36299; June 26, 2006). NMFS published regulations on October 10, 2008 to implement a 10-knot speed restriction for all vessels 65 ft (19.8 m) or longer in Seasonal Management Areas (SMAs) along the East coast of the U.S. Atlantic seaboard at certain times of the year (73 FR 60173; October 10, 2008). The rule will expire five years from the date of effectiveness. During the five-years the rule is in effect, NOAA will analyze data on ship-whale interactions and review the economic consequences to determine further steps regarding the rule.

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

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

Through a joint effort between NOAA and the USCG, the US also submitted a proposal to the IMO to shift the northern leg of the existing Boston Traffic Separation Scheme (TSS) 12 degrees to the north. Overlaying sightings of right whales and all baleen whales on the existing TSS revealed that the existing TSS directly overlaps with areas of high whale densities, while an area slightly to the north showed a considerable decrease in sightings. Separate analyses by the SBNMS and the NEFSC both indicated that the proposed TSS would overlap with 58% fewer right whale sightings and 81% fewer sightings of all large whales, thus considerably reducing the risk of collisions between ships and whales. The proposal was submitted to the IMO in April 2006, and was adopted by the Maritime Safety Committee in December 2006. The shift took effect on July 1, 2007. In 2009 this TSS was modified by narrowing the width of the north-south portion by one mile to reduce the threat of ship collisions with endangered right whales and other whale species.

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

4.4.6.5 Sighting Advisory System (SAS)

The right whale Sighting Advisory System (SAS) was initiated in early 1997 as a partnership among several federal and state agencies and other organizations to conduct aerial and ship board surveys to locate right whales and to alert mariners to right whale sighting locations in a near real time manner. The SAS surveys and opportunistic sightings reports document the presence of right whales and are provided to mariners via fax, email, NAVTEX, Broadcast Notice to Mariners, NOAA Weather Radio, several web sites, and the Traffic Controllers at the Cape Cod Canal. Fishermen and other vessel operators can obtain SAS sighting reports, and make necessary adjustments in operations to decrease the potential for interactions with right whales. The SAS has also served as the only form of active entanglement monitoring in the Cape Cod Bay and Great South Channel feeding areas. Some of these sighting efforts have resulted in successful disentanglement of right whales. SAS flights have also contributed sightings of dead floating animals that can occasionally be retrieved to increase our knowledge of the biology of the species and effects of human impacts.

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

4.4.6.6 Dynamic Management Area (DMA) Program

The DMA program was initiated in December 2008 as a supplement to the ship speed regulations discussed above. The program implements dynamic vessel traffic management zones in order to provide protection for unpredictable aggregations of right whales that occur outside of SMAs. When NOAA aerial surveys or other reliable sources report aggregations of 3 or more right whales in a density that indicates the whales are likely to persist in the area, NOAA calculates a buffer zone around the aggregation and announces the boundaries of the zone to mariners via various mariner communication outlets, including NOAA Weather Radio, USCG Broadcast Notice to Mariners, MSR return messages, email distribution lists, and the Right Whale Sighting Advisory System (SAS). NOAA requests mariners to route around these zones or transit through them at 10 knots or less. Compliance with these zones is voluntary.

4.4.7 Marine Mammal Health and Stranding Response Program (MMHSRP)

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

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

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

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

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

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

4.4.8 Harbor Porpoise Take Reduction Plan (HPTRP)

NMFS has implemented the HPTRP to decrease interactions between harbor porpoise and commercial gillnet gear in the Gulf of Maine and the Mid-Atlantic. The HPTRP includes time and area closures, some of which are complete closures. Some areas are closed to gillnet fishing unless pingers are used. The pingers act as an acoustic deterrent device that broadcasts a 10 kHz (+/- 2 kHz) sound underwater at 132 dB(+/- 4 dB) re 1 micropascal at 1 m, lasting 300 milliseconds (+/- 15 milliseconds), and repeating every 4 seconds (+/- 0.2 seconds). Time and area closures implemented by the HPTRP may decrease the chance of interactions between ESA-listed species that are present in the area at the time of the closure and gillnet gear. Pingers may also help deter large whales away from gillnets, but more research is needed to confirm this. The HPTRP is an evolving plan and changes are made as members of the take reduction team identify the need for improvements by monitoring the progress of the plan and learning more about harbor porpoise abundance and bycatch rates. NMFS published a final rule for the HPTRP on February 19, 2010. In New England, new measures include the expansion of seasonal and temporal requirements within HPRTP management areas, incorporation of additional management areas, and establishment of a consequence closure area strategy to increase compliance and reduce bycatch levels within select management areas with historically high levels of harbor porpoise bycatch. In the Mid-Atlantic, new measures include the establishment of an additional management area, and modification to the current tie-down requirement for large mesh gillnet gear. The final rule also incorporates a research provision and finalizes regulatory text corrections and clarifications. For more information on the HPTRP including time and area closures visit: http://www.nero.noaa.gov/prot_res/porptrp/

4.4.9 Bottlenose Dolphin Take Reduction Plan (BDTRP)

Gear restrictions are currently implemented under the BDTRP, affecting small, medium, and large-mesh gillnets, along the Atlantic coast from New Jersey to Florida. The regulatory recommendations seek to reduce soak times and modify fishing practices to limit bycatch of bottlenose dolphins. These regulations may also benefit ESA-listed species that are present in the area during BDTRP regulatory measures. The take reduction team meets periodically to monitor implementation and effectives of the plan. For more information on the BDTRP visit: http://www.nmfs.noaa.gov/pr/interactions/trt/bdtrp.htm

4.4.10 Atlantic Trawl Gear Take Reduction Strategy (ATGTRS)

NMFS convened an Atlantic Trawl Gear Take Reduction Team (ATGTRT) in 2006 to address the incidental mortality and serious injury of long-finned pilot whales (*Globicephala melas*), short-finned pilot whales (*Globicephala macrorhynchus*), common dolphins (*Delphinus delphis*), and white sided dolphins (*Lagenorhynchus acutus*) incidental to bottom and mid-water trawl fisheries operating in both the Northeast and the Mid-Atlantic regions. Because none of the marine mammal stocks of concern to the ATGTRT are classified as a "strategic stock," nor do they currently interact with a Category I fishery it was determined that development of a take reduction plan was currently not necessary.

In lieu of a take reduction plan, the ATGTRT agreed to develop an ATGTRS. The ATGTRS identifies informational and research tasks as well as education and outreach needs the ATGTRT believes are necessary to provide the basis decreasing mortalities and serious injuries of marine mammals to insignificant levels approaching zero mortality and serious injury rates. The ATGTRS also identifies several potential voluntary measures that can be adopted by certain trawl fishing sectors to potentially reduce the incidental capture of marine mammals. These voluntary measures are as follows:

- reducing the numbers of turns made by the fishing vessel and tow times while fishing at night; and
- increasing radio communications between vessels about the presence and/or incidental capture of a marine mammal to alert other fishermen of the potential for additional interactions in the area.

While these measures have been recommended to reduce take of the four species of marine mammals listed above, ESA-listed species may also benefit from implementation of these measures.

4.4.11 Magnuson-Stevens Fishery Conservation and Management Act

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

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 cetaceans and sea 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, pollution, global climate change, coastal development, and catastrophic events. While the combination of these activities may affect populations of ESA-listed cetaceans and 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.

Right and humpback whale entanglements in gear set for state fisheries are also known to have occurred. As described above, recent entanglements include entanglements in gear set for the state lobster pot/trap fishery, and entanglement in croaker sink gillnet gear (Waring *et al.* 2007; Glass *et al.* 2008). Actions have been taken to reduce the risk of entanglement to large whales, although more information is needed on the effectiveness of these actions. State water fisheries continue to pose a risk of entanglement to large whales to a level that cannot be quantified.

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

Collisions of ESA-listed right, humpback, fin and sei whales with large vessels are known to occur, and are a source of serious injury and mortality for these species. As described in Section 4.4.7, NMFS has implemented a ship strike reduction program to reduce the number of right whale strikes by large vessels causing serious injuries and death. The program consists of both regulatory and non-regulatory components, such as requiring vessels to reduce speed in certain areas at certain times when right whales are likely to be present. The program is not specific to areas or times when other species of large whales are likely to be present in the vicinity of large ports of shipping lanes. The program does not require reduced speeds in all areas where right whales may occur. Although these measures are designed to reduce take of ESA-listed whales as a result of vessel interaction, the risk of takes has not been fully removed since interactions may still occur at times when large whales and vessels occupy the same areas.

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 cetaceans and 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 areas (Ruben and Morreale 1999).

Contaminant studies have confirmed that right whales are exposed to and accumulate contaminants. Antifouling agents and flame retardants that have been proven to disrupt reproductive patterns and have been found in other marine animals, have raised new concerns for their effects on right whales (Kraus *et al.* 2007). Recent data also support a hypothesis that chromium, an industrial pollutant, may be a concern for the health of the North Atlantic right whales and that inhalation may be an important exposure route (Wise *et al.* 2008). The impacts of biotoxins on marine mammals are also poorly understood, yet data is showing that marine algal toxins may play significant roles in mass mortalities of these animals (Rolland *et al.* 2007). Although there are no published data concerning the effects of biotoxins on right whales, researchers have discovered that right whales are being exposed to measurable quantities of paralytic shellfish poisoning (PSP) toxins and domoic acid via trophic transfer through the copepods upon which they feed (Durbin *et al.* 2002, Rolland *et al.* 2007, Leandro *et al.* 2009). Other large whales are likely similarly affected. Between November 1987 and January 1988, at least 14 humpback whales died after consuming Atlantic mackerel containing a dinoflagellate saxitoxin (Geraci *et al.* 1989; Waring *et al.* 2009). In July 2003, dead humpback whales tested positive for low levels of domoic acid (Waring *et al.* 2009). However, the cause of death could not be confirmed to be due to domoic acid poisoning (Waring *et al.* 2009).

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

Global climate change is likely to negatively affect sea turtles and large whales. 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 effects on ESA-listed species are unknown at this time. There are multiple hypothesized affects to sea turtles and cetaceans including changing the range and distribution of ESA-listed species as well as their prey distribution and/or abundance due to water temperature changes. Ocean acidification may also negatively affect marine life particularly organisms with calcium carbonate shells which serve as important prey items for many species. Global climate change may also affect reproductive behavior in sea turtles including earlier onset of nesting, shorter internesting intervals, and a decrease in the length of nesting season. Additionally, air temperature may affect the sex ratio of sea turtle offspring. Water temperature is a main factor affecting the distribution of cetaceans, and with global climate change the range of cetaceans may be altered. Ocean acidification may have an adverse impact on the prey for baleen whales which may result in serious consequences for the marine food web. A decline in reproductive fitness as a result of global climate change could have profound effects on the abundance and distribution of sea turtles and cetaceans in the Atlantic.

Coastal development – Along the Mid-Atlantic coastline, beachfront development, lighting, and beach erosion potentially reduce or degrade sea turtle nesting habitats or interfere with hatchlings movement to sea. Nocturnal human activities along nesting beaches may also discourage sea turtles from nesting sites. Coastal counties are presently adopting stringent protective measures to protect hatchling sea turtles from the disorienting effects of beach lighting. Some of these measures were drafted in response to lawsuits brought against the counties by concerned citizens who charged the counties with failing to uphold the ESA by allowing unregulated beach lighting that results in takes of hatchlings.

Catastrophic events- An increase in commercial vessel traffic/shipping increases the potential for oil/chemical spills. The pathological effects of oil spills have been documented in laboratory studies of marine mammals and sea turtles (Vargo *et al.* 1986). There have been a number of documented oil spills in the northeastern U.S.

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 operation of the monkfish fishery within the constraints of the current Monkfish FMP are analyzed to determine whether the action is likely to jeopardize the continued existence of listed species in the action area. Past effects of the monkfish fishery are included in this "baseline." To the extent available information allows, this baseline (which does not include the future effects of the monkfish fishery) would be compared to the baseline plus the effects of the continued operation of the fishery under the FMP from now into the future. The difference in the two trajectories would be reviewed to determine whether the continued operation of the fishery, within the constraints of the current Monkfish FMP, is likely to jeopardize the continued existence of these species. This section synthesizes the *Status of the Species*, *Environmental Baseline*, and *Cumulative Effects* sections as best as possible given that some information on ESA-listed species is quantified, yet much remains qualitative or unknown.

North Atlantic right whales, humpback whales, fin whales, sei whales, leatherback sea turtles 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.

North Atlantic right whales are listed as "endangered" under the ESA. The International Whaling Commission (IWC) recognizes two right whale populations in the North Atlantic: a western and eastern population (IWC 1986). However, sighting surveys from the eastern Atlantic Ocean suggest that right whales present in this region are rare (Best *et al.* 2001) and it is unclear whether a viable population in the eastern North Atlantic still exists (Brown 1986; NMFS 2005a). In the western Atlantic, North Atlantic right whales generally occur from the Southeast U.S. (waters off of Georgia, Florida) to Canada (*e.g.*, Bay of Fundy and Scotian Shelf) (Kenney 2002; Waring *et al.* 2009). Research results suggest the existence of six major habitats or congregation areas for western North Atlantic right whales. Results from telemetry studies and photo-id studies have shown extensive right whale movements: (a) over the continental shelf during the summer foraging period (Mate *et al.*1992; Mate *et al.* 1997; Bowman *et al.* 2003; Baumgartner and Mate 2005), (b) between known calving/nursery areas and foraging areas in the winter (Brown and Marx 2000; Waring *et al.* 2009), and (c) into deep water off of the continental shelf (Mate *et al.* 1997).

As of October 10, 2007, there were 345 minimum number of right whales alive as calculated from the sightings database which indicates a significant increase in the number of catalogued whales (Waring *et al.* 2009). Based on counts of animals alive from the sightings database as of 10 October 2008, for the years 1990-2004, the mean growth rate for the period was 1.8% (Waring *et al.* 2009). However, there was significant variation in the annual growth rate due to apparent losses exceeding gains during 1998-1999 and the number of photo-identified and catalogued female North Atlantic right whales numbers less than 200 whales (Waring *et al.* 2007). The current estimate of breeding females is 97 (Schick *et al.* 2009).

There is general agreement that right whale recovery is negatively affected by anthropogenic mortality. Fifty-four (54) right whale mortalities were reported from Florida to the Canadian Maritimes during the period 1970-2002 (Moore *et al.* 2004). For the more recent period of 2003-2007, 20 right whale mortalities were confirmed, three (3) due to entanglements, nine due to ship strikes (Glass *et al.* 2009). Serious injury was documented for an additional three (3) right whales during that timeframe. These numbers represent the minimum values for human-caused mortality for this period since it is unlikely that all carcasses will be observed (Moore *et. al.* 2004, Glass *et al.* 2009). Given the small population size and low annual reproductive rate of right whales, human sources of mortality may have a greater effect to relative population growth rate than for other large whale species (Waring *et al.* 2009). Other negative effects to the species may include changes to the environment as a result of global climate change, contaminants, and loss of genetic diversity.

In light of the above NMFS considers the trend for North Atlantic right whales to be increasing. Although the right whale population is believed to be increasing, caution is exercised in considering the overall effect to the species given the many on-going negative impacts to the species across all areas of its range and to all age classes, and information to support that there are fewer than 200 female right whales total (of all age classes) in the population. New measures recently implemented into the ALWTRP and ship strike reduction program are expected to reduce the risk of anthropogenic serious injury and mortality to right whales. The programs are evolving plans and will continue to undergo changes based on available information to reduce the serious injury and mortality risk to large whales.

Humpback whales are listed as "endangered" under the ESA. Humpback whales range widely across the North Pacific during the summer months (Johnson and Wolman 1984, Perry *et al.* 1999). Although the IWC only considered one stock (Donovan 1991) there is evidence to indicate multiple populations migrating between their respective summer/fall feeding areas to winter/spring calving and mating areas within the North Pacific Basin (Anglis and Outlaw 2007, Carretta *et al.* 2007). Recent research efforts via the Structure of Populations, Levels of Abundance, and Status of Humpback Whales (SPLASH) Project estimate the abundance of humpback whales to be just under 20,000 whales for the entire North Pacific, a number which doubles previous population predictions obtained for 1991-1993 in a previous study (Calambokidis *et al.* 2008). There are indications that some stocks of North Pacific humpback whales increased in abundance between the 1980's -1990's (Anglis and Outlaw 2007; Carretta *et al.* 2009). The abundance estimate for the northern Indian Ocean population of Humpback whales is 82 (Minton *et al.* 2008). The total abundance estimate for the Southern Hemisphere humpback whale population is 36,600 although it is negatively biased due to no available abundance estimates for two stocks. Although they were given protection by the IWC in 1963, Soviet whaling data made available in the 1990's revealed that southern hemisphere humpbacks continued to be hunted through 1980 (Zemsky *et al.* 1995, IWC 1995, Perry *et al.* 1999).

Photographic mark-recapture analyses from the Years of the North Atlantic Humpback (YONAH) project gave an ocean-basin-wide estimate of 11,570 animals during 1992/1993 and an additional genotype-based analysis yielded a similar but less precise estimate of 10,400 whales (95% CI = 8,000 - 13,600) (Waring *et al.* 2009). For management purposes under the MMPA, the estimate of 11,570 individuals is regarded as the best available estimate for the North Atlantic population (Waring *et al.* 2007). Previously, the North Atlantic humpback whale population was treated as a single stock for management purposes, however due to the strong fidelity to the region displayed by many whales, the Gulf of Maine stock was reclassified as a separate feeding stock (Waring *et al.* 2009). The best, recent estimate for the Gulf of Maine stock is 847 whales, derived from the 2006 aerial survey (Waring *et al.* 2009). Population modeling estimates the growth rate of the Gulf of Maine stock to be at 6.5% (Barlow and

Clapham 1997). Current productivity rates for the North Atlantic population overall are unknown, although Stevick *et al.* (2003) calculated an average population growth rate of 3.1% for the period 1979-1993 (Waring *et al.* 2009).

As is the case with other large whales, the major known sources of anthropogenic mortality and injury of humpback whales occur from fishing gear entanglements and ship strikes. There were 76 confirmed entanglement events and 11 confirmed ship strike events for humpback whales in the Atlantic between 2003-2007, resulting in a total of 12 confirmed mortalities and 10 serious injury determinations (Glass *et al.* 2009). These numbers are expected to be a minimum account of what actually occurred given the range and distribution of humpbacks in the Atlantic. In addition to their potential for being negatively affected by other human related effects such as global climate change and contaminants, humpbacks may be susceptible to consumption of lethal levels of toxic dinoflagellates that can become concentrated in humpback prey such as mackerel. In addition, humpback prey in the Atlantic includes fish species targeted in commercial fishing operations (*i.e.*, herring and mackerel). There is no evidence that current levels of fishing for these species has an effect on humpback survival. However, changes in humpback distribution in the Gulf of Maine have been found to be associated with changes in herring, mackerel, and sand lance abundance associated with local fishing pressures (Stevick *et al.* 2003, Waring *et al.* 2009).

Fin whales are listed as "endangered" under the ESA. NMFS recognizes three fin whale stocks in the Pacific for the purposes of managing this species under the MMPA. These are: Alaska (Northeast Pacific), Hawaii, and California/Washington/Oregon (Angliss *et al.* 2001). Reliable estimates of current abundance for the entire Northeast Pacific fin whale stock are not available (Angliss *et al.* 2001). Stock structure for fin whales in the southern hemisphere is unknown. Prior to commercial exploitation, the abundance of southern hemisphere fin whales is estimated to have been at 400,000 (IWC 1979, Perry *et al.* 1999). There are no current estimates of abundance for southern hemisphere fin whales.

NMFS recognizes fin whales off the eastern United States, Nova Scotia and the southeastern coast of Newfoundland as a single stock in the Atlantic for the purposes of managing this species under the MMPA (Waring *et al.* 2009). Various estimates have been provided to describe the current status of fin whales in western North Atlantic waters. One method used the catch history and trends in Catch Per Unit Effort to obtain an estimate of 3,590 to 6,300 fin whales for the entire western North Atlantic (Perry *et al.* 1999). Hain *et al.* (1992) estimated that about 5,000 fin whales inhabit the northeastern United States continental shelf waters. Previous abundance estimates of fin whales in the western North Atlantic were 2,200 (Palka 1995), 2,814 (Palka 2000), 2,933 (Palka 2006), and 1,925 (Palka 2006) in 1995, 1999, 2002, and 2004 respectively. The 2009 Stock Assessment Report (SAR) gives a best estimate of abundance for the western North Atlantic stock of fin whales as 2,269 (C.V. = 0.37), derived from an aerial survey in 2006 (Waring *et al.* 2009). This estimate is considered extremely conservative in view of the incomplete coverage of the known habitat of the stock and the uncertainties regarding population structure and whale movements between surveyed and unsurveyed areas (Waring *et al.* 2009). There are insufficient data to determine population trends for this species. Current and maximum net productivity rates are unknown for this stock (Waring *et al.* 2009).

Like right whales and humpback whales, anthropogenic mortality and injury of fin whales include entanglement in commercial fishing gear and ship strikes. From 1999-2003, fin whales had a low proportion of entanglements; of 40 reported events[7], only seven (7) were of entanglements (all confirmed), two (2) of which were fatal (Cole *et al.* 2005). Ten (10) ship strikes were reported, five (5) of which were confirmed and proved fatal. Of 61 fin whale events recorded between 2003 and 2007, eight (8) mortalities were associated with vessel interactions, and three (3) mortalities were attributed to entanglements (Glass *et al.* 2009). In addition to their potential for being negatively affected by other human related effects, commercial whaling, global climate change and contaminants may also adversely affect fin whales.

Sei whales are listed as "endangered" under the ESA. The IWC only considers one stock of sei whales in the North Pacific (Donovan 1991), but for NMFS management purposes under the MMPA, sei whales in the eastern North Pacific are considered a separate stock (Carretta *et al.* 2008). The best estimate of abundance for U.S. Pacific EEZ (California, Oregon, and Washington waters out to 300nmi) is 46 (CV=0.61) sei whales (Barlow and Forney 2007; Forney 2007, Carretta *et al.* 2008). The stock structure and abundance of sei whales in the southern hemisphere is unknown. Like other whale species, sei whales in the southern hemisphere were heavily impacted by commercial whaling (Perry *et al.* 1999).

There is limited information on the stock identity of sei whales in the North Atlantic (Waring *et al.* 2009). For purposes of the Marine Mammal Stock Assessment Reports, and based on a proposed IWC stock definition, NMFS recognizes the sei whales occurring from the U.S. East coast to Cape Breton, Nova Scotia, and east to 42° W longitude as the "Nova Scotia stock" of sei whales (Waring *et al.* 2009). The abundance estimate of 386 sei whales (CV=0.85), obtained from a sighting survey conducted in 2004, is considered the best available for the Nova Scotia stock of sei whales (Waring *et al.* 2009). However, this estimate is considered extremely conservative in view of the known range of the sei whale in the entire western North Atlantic, and the uncertainties regarding population structure and whale movements between surveyed and unsurveyed areas (Waring *et al.* 2009). Current and maximum net productivity rates are unknown for this stock, and there are insufficient data to determine trends of the sei whale population (Waring *et al.* 2009).

Few instances of injury or mortality of sei whales due to entanglement or vessel strikes have been recorded in U.S. waters. Of the eight (8) reported events for sei whales in the Atlantic from 2003-2007, one (1) was confirmed as a serious injury resulting from entanglement in unidentified gear (Glass *et al.* 2009). The remaining seven (7) events were mortalities with two of these confirmed to be due to ship strikes. In an additional ship strike event, it could not be determined if the strike occurred pre or post-mortem (Glass *et al.* 2009). Global climate change and contaminants may also adversely affect sei whales.

Loggerhead sea turtles are listed as "threatened" under the ESA. Loggerhead nesting occurs on beaches of the Pacific, Indian, and Atlantic Oceans, and the 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;

[7] A large whale event includes entanglements, ship strikes, and mortalities.

Bowen and Karl 2007). The BRT has recently identified the following nine loggerhead DPSs distributed globally: (1) North Pacific Ocean, (2) South Pacific Ocean, (3) North Indian Ocean, (4) Southeast Indo-Pacific Ocean, (5) Southwest Indian Ocean, (6) Northwest Atlantic Ocean, (7) Northeast Atlantic Ocean, (8) Mediterranean Sea, and (9) South Atlantic Ocean. It should be noted, however, that DPSs can only be designated for regulatory uses through the formal ESA listing process.

It takes decades for loggerhead sea turtles to reach maturity. Once they have reached maturity, females typically lay multiple clutches of eggs within a season, but do not typically lay eggs every season (NMFS and USFWS 2008). There are many natural and anthropogenic factors affecting the survival of loggerheads prior to their reaching maturity as well as for those adults who have reached maturity. As described in 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.

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, there is a large uncertainty associated with using nest counts to estimate the total population size of a nesting group or trends in the number of nests laid as an indicator of the population (Meylan 1982; Ross 1996; Zurita *et al.* 2003; Hawkes *et al.* 2005; letter to J. Lecky, NMFS Office of Protected Resources, from N. Thompson, NMFS Northeast Fisheries Science Center, December 4, 2007; TEWG 2009).

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 reflects the reproductive output of the nesting group each year, and also provides 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, the PFRU, occurs along the Southeast coast of the U.S. from the Florida/Georgia border through Pinellas County on Florida's West coast, where approximately 15,735 females nest per year (based on a mean of 64,513 nests laid per year from 1989-2007; NMFS and USFWS 2008). The world's third largest loggerhead nesting group also occurs in the U.S., from the Florida/Georgia border through southern Virginia. However, the approximate number of females nesting annually is 1,272 (based on a mean number of 5,215 nests laid per year from 1989-2008; NMFS and USFWS 2008), which is less than 1/10th the size of the PFRU. 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, 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. The 2008 revised recovery plan by NMFS and FWS identified five unique recovery units of loggerheads in the Northwest Atlantic. Based on the most recent information, a decline in annual nest counts has been measured or suggested for three of the five recovery units. These include nesting for the PFRU – the second largest loggerhead nesting group in the world and the largest of all of the loggerhead nesting groups in the Atlantic (Meylan *et al.* 2006; NMFS and USFWS 2008). The final revised plan reviews and discusses the species' ecology, population status and trends, and identifies the many threats to loggerhead sea turtles in the Northwest Atlantic Ocean. It lays out a recovery strategy to address the threats, based on the best available science, and includes recovery goals and criteria. In addition, the plan identifies substantive actions needed to address the threats to the species and achieve recovery. In 2009, the TEWG indicated that it could not determine whether or not the decreasing annual numbers of nest amount the Northwest Atlantic loggerhead subpopulations were due to stochastic processes resulting in few nests, a decreasing average reproductive output of adult females, decreasing number of adult females, or a combination of these factors. The TEWG noted there were likely several factors contributing to the decline. These factors include incidental capture (in fisheries, power plant intakes, and dredging operations), lower adult female survival rates, increases in the proportion of first-time nesters, continued directed harvest, and increases in mortality due to disease. The current levels of hatchling output will no doubt result in depressed recruitment to subsequent life stages over the coming decades (TEWG 2009).

Although there is an increasing trend at some nesting beaches (Southwest Indian Ocean and South Atlantic Ocean), available information about anthropogenic threats to juveniles and adults in neritic and oceanic environments indicate possible unsustainable additional mortalities. 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. According to the threat matrix analysis in the BRT report, the potential for future decline is greatest for the North Indian Ocean, Northwest Atlantic Ocean, Northeast Atlantic Ocean, Mediterranean Sea, and South Atlantic Ocean DPSs (Conant *et al.* 2009).

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

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 (TEWG 2007; NMFS and USFWS 2007b).

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

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

In addition, Atlantic and Pacific leatherbacks are impacted by different activities (NMFS and USFWS 1992, 1998a). However, the ESA-listing of leatherbacks as a single species means that the effects of a proposed action must, ultimately, be considered at the species level for section 7 consultations. NMFS recognizes that the nest count data available for 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 number of nests laid is not a reflection of the overall leatherback population given that the proportion of adult males to females and the age structure of the population(s) are unknown.

Kemp's Ridley sea turtles are listed as a single species classified as "endangered" under the ESA. Kemp's ridleys occur in the Atlantic Ocean and Gulf of Mexico. The only major nesting site for Kemp's ridleys is a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; USFWS and NMFS 1992; NMFS and USFWS 2007c). 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 sea 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 negatively impact 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 (Meylan 1982; Ross 1996; Zurita *et al.* 2003; Hawkes *et al.* 2005; letter to J. Lecky, NMFS Office of Protected Resources, from N. Thompson, NMFS Northeast Fisheries Science Center, December 4, 2007). Nevertheless, the nesting data does provide valuable information on the extent of Kemp's ridley nesting and the trend in the number of nests laid. Estimates of the adult female nesting population reached a low of approximately 250-300 in 1985 (USFWS and NMFS 1992; TEWG 2000). From 1985 to 1999, the number of nests observed at Rancho Nuevo and nearby beaches increased at a mean rate of 11.3% per year (TEWG 2000). Current estimates suggest an adult female population of 7,000-8,000 Kemp's ridleys (NMFS and USFWS 2007c).

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

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

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 sea 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 sea turtle species, green sea turtles face numerous threats on land and in the water that affect the survival of all age classes.

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

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

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

Pursuant to Section 7(a)(2) of the ESA (16 USC 1536), Federal agencies are directed to ensure that their activities are not likely to jeopardize the continued existence of any listed species or result in the destruction or adverse modification of critical habitat. This biological opinion examines the likely effects of the proposed action on listed species within the action area to determine if continued authorization of the Monkfish FMP is likely to jeopardize the continued existence of listed species. This analysis is done after careful review of the listed species status and the factors that affect the survival and recovery of that species, as described above.

In this section of a biological opinion, NMFS assesses the direct and indirect effects of the proposed action on threatened and endangered species. The purpose of the assessment is to determine if it is reasonable to conclude that the fishery is likely to have direct or indirect effects on threatened and endangered species that appreciably reduce their likelihood of surviving and recovering in the wild by reducing their reproduction, numbers, or distribution. Since the proposed action is not expected to affect designated critical habitat, this Opinion will focus only on the jeopardy analysis.

6.1 Approach to the Assessment

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

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

(1979)], jeopardy analyses are designed to avoid concluding that actions had no effect on listed species or critical habitat when, in fact, there was an effect.

In order to identify, describe, and assess the effects to ESA-listed cetaceans and sea turtles resulting from fishing gear used in the monkfish fishery, NMFS is using: (1) Information on entanglement of right, humpback, fin, and sei whales in fishing gear of known and/or unknown origin (Johnson *et al.* 2005; Waring *et al.* 2009; Glass *et al.* 2009), (2) captures of loggerhead sea turtles in bottom otter trawl gear and sink gillnet gear where effort in the monkfish fishery and sea turtle distribution overlap (Murray 2008; Murray 2009a), (3) information on the capture of other sea turtle species in bottom otter trawl gear in other fisheries where the monkfish fishery also operates, (4) life history information for cetaceans and sea turtles, and (5) the effects of fishing gear entanglements on cetaceans and sea turtles that has been published in a number of documents. These sources include status reviews and biological reports (NMFS and USFWS 2007a, 2007b, 2007c, 2007d; TEWG 2000, 2007, 2009; NMFS SEFSC 2001; Moore *et al.* 2004; Johnson *et al.* 2005; Waring *et al.* 2009; Glass *et al.* 2009), recovery plans (NMFS 1991a,b, 2005a; NMFS and USFWS 1991, 1992, 2008; USFWS and NMFS 1992), commercial fishery databases (NMFS fisheries statistics database) and numerous other sources of information from the published literature as cited within this Opinion.

6.1.1 Description of the Gear

Although the proportion of commercial landings by gear type varies by management area, overall, landings of monkfish in the directed fishery are fairly evenly split between gillnets and otter trawls, which together account for up to 95% of landings (according to the fishing vessel trip report database, 2000-2004). No other gear types account for more than trace landings of monkfish, and there is no recreational component to this fishery (NMFS 2007a). Bottom otter trawl represented the predominant gear type in the northern management area, representing 80% of the landings. Gillnet gear is the most commonly identified gear type in the southern management area.

As previously described, ESA-listed cetaceans are not reasonably likely to be captured in bottom otter trawl gear. ESA-listed cetaceans may, however, become entangled in lines associated with anchored sink gillnet gear. ESA-listed sea turtles are reasonably likely to be captured in bottom otter trawl gear as well as sink gillnet gear if gear overlaps with the distribution of sea turtles.

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). Turtle excluder devices (TEDs) are not required to be used in trawl gear targeting monkfish.

The Northeast Sink Gillnet Fishery, which targets monkfish, spiny dogfish, skates, and species managed by the NE Multispecies FMP, has been dominated by bottom-tending (sink) nets. Alternatively, less than 1% of the fishery utilizes gillnets that either is anchored floating or drift (Waring *et al.* 2009).

While targeting monkfish, individual gill nets are typically 300 feet long and are usually fished as a series of 5-15 nets attached end-to-end (NEFMC 2003). Sink gillnets are comprised of a leadline (weighted line) to help hold the net panels on or near the bottom, net panels that entangle the fish, and a floatline at the top of the net panel to help maintain the position of the net panel in the water column (NEFMC 2003; Johnson *et al.* 2005). Monofilament is the dominant material used with stretched mesh sizes ranging from 6 to 12 inches (Waring *et al.* 2009). However, monkfish gillnet gear has a minimum mesh size of 10-inches. String lengths range from 600 to 10,500 feet long. The mesh size and string length vary by the primary fish species targeted for catch. Nets are anchored at each end to keep them in place (NEFMC 2003). The frequency at which nets are tended varies by fishery. For monkfish, frequency of tending ranges from daily to bi-weekly (NREFHSC 2002).

6.1.2 Description of Incidental takes of Cetaceans

Table 1 summarizes documented fishing gear interactions with large whales in the Atlantic for 1999-2008, showing the number of documented entanglements, and how many of those have led to serious injury or mortality (NMFS NERO 2010). Serious injury has been defined in 50 CFR part 229.2 as an injury that is likely to lead to mortality. Trawl gear is not known to result in serious injury or mortality to right, humpback, fin, or sei whales and there have been no documented interactions between ESA-listed marine mammals and the North Atlantic bottom trawl fishery (NMFS 2006a). Their great size and mobility presumably allows them to avoid interactions with the relatively slow moving trawl gear.

Between April 1999 through December 2008, one (1) right whale and 11 humpback whales were verified to have been entangled in sink gillnet gear that was assessed to originate from U.S. fisheries or the country of origin was not definable (NMFS NERO 2010). Three (3) of the 12 sink gillnet interactions resulted in serious injuries or mortalities. Within the same time period, an additional 14 large whale entanglements were documented with gillnets without specific classification of the type of gillnet (NMFS NERO 2010). Since many entanglement events go unobserved and because the gear type, fishery, and/or country of origin for observed entanglement events are often not traceable; the list of identified entanglement events is assumed to be an under-representation of actual numbers of entanglements.

Because whales often free themselves of gear following an entanglement event, scarring may be a better indicator of fisheries interaction than entanglement records. In an analysis of the scarification of right whales, 338 of 447 (75.6%) whales examined during 1980-2002 were scarred at least once by fishing gear (Knowlton *et al.* 2005). As an example, in six records of right whales becoming entangled in groundfish gillnet gear in the Bay of Fundy and Gulf of Maine between 1975 and 1990, the whales were either released or escaped on their own, although several whales were observed carrying net or line fragments (Read 1994). Further research using the North Atlantic Right Whale Catalogue has indicated that, annually, between 14% and 51% of right whales are involved in entanglements (Knowlton *et al.* 2005).

Table 1. NMFS gear analysis for entangled/entrapped North Atlantic right whales, humpback whales, fin whales, and sei whales for the years 1999-2008. For the purposes of this evaluation, entanglement/entrapment events with gear determined to be from Canadian fisheries were not included. Results of gear analyses were the criteria used to categorize these events to U.S., Canada, or undefined origin; where not known, the NOAA Stock Assessment Reports for Marine Mammals use the location the animal was first sighted, which may be quite a distance from the original location of entanglement. For this analysis, animals entangled in gear of undefined origin are assumed to be entangled in gear from U.S. fisheries. Confirmed serious injury/mortality (SI/M) events are presented in parentheses.

	Entanglement events with gear of U.S. and unidentified origins	# of North Atlantic right whale events	Mean annual North Atlantic right whale events	# of humpback whale events	Mean annual humpback whale events	# of fin whale events	Mean annual fin whale events	# of sei whale events	Mean annual sei whale events
Sink gillnet gear	12 (3)	1 (1)	0.1 (0.1)	11 (2)	1.1 (0.2)	0	0	0	0
Unspecified gillnet gear	14 (3)	1 (1)	0.1 (0.1)	13 (2)	1.3 (0.2)	0	0	0	0
Lobster gear	19 (3)	6 (1)	0.6 (0.1)	13 (2)	1.3 (0.2)	0	0	0	0
Other pot/trap gear	4	0	0	4	0.4	0	0	0	0
Hook and line	6	0	0	6	0.6	0	0	0	0
Bottom longline	1	1	0.1	0	0	0	0	0	0
Purse seine	1	0	0	1	0.1	0	0	0	0
Unidentified gear	182 (46)	42 (7)	4.2 (0.7)	116 (28)	11.6 (2.8)	21 (8)	2.1 (0.8)	3 (3)	0.3 (0.3)
Totals	239 (55)	51 (10)	5.1 (1.0)	164 (34)	16.4 (3.4)	21 (8)	2.1 (0.8)	3 (3)	0.3 (0.3)

As noted previously, observed entanglement events do not provide a complete count of all of the entanglements that occur on an annual basis and we do not currently have an accepted method to extrapolate those observed events to obtain a complete count. For that reason, the observed entanglement events (and therefore the number of entanglement related serious injuries or mortalities) are an underestimate. Recently a methodology has been proposed for humpback whales that uses scar-based entanglement rates to extrapolate total entanglement mortality (Robbins *et al.* 2009). Robbins *et al.* (2009) used scar-based inference to estimate the annual frequency of non-lethal entanglement in the Gulf of Maine humpback whale population. For the period 1997-2006, annual estimates averaged 12.1%. The fraction of entanglements that were non-lethal was calculated using NMFS serious injury and mortality determinations. From 2002-2006 there were 49 (76.6%) non-lethal entanglements documented and 15 (23.4%) that were considered serious injuries or mortalities. Robbins *et al.* (2009) assumed a minimum population estimate of 549 whales and a scar based entanglement rate of 18.8% to calculate that approximately 103 Gulf of Maine humpback whales survived entanglement in 2003. If the survivors represented 76.6% of the entanglements that occurred that year then there were an additional approximately 32 entanglements that resulted in serious injury or mortality. While documented entanglement related serious injuries or mortalities are approximately 3%, this method for estimating actual entanglement related serious injuries or mortalities results in an estimate of 23.4%, which is significantly higher. The authors note that it is a crude, preliminary estimate of entanglement mortality and state that the approach and its input values require further examination and refinement. While we find this approach interesting, given its preliminary nature and questions regarding the approach and the input values, we have not utilized the results for humpbacks in this Opinion and furthermore have not attempted to apply the approach to North Atlantic right whales or other large whales. While we are not utilizing this approach for attempting to estimate the overall number or rate of serious injuries or mortalities caused by entanglement, we recognize the importance of attempting to calculate a reasonable and scientifically supportable estimate. We also note that the estimate using this approach indicates that the magnitude of the impact may be significantly higher than is documented and provides further support for ongoing efforts to implement and enhance risk reduction measures.

6.1.3 Incidental takes of Sea Turtles

Sea turtles incidentally taken in fishing gear must be reported to NMFS on Vessel Trip Reports (VTRs) that are required for the monkfish fishery and other Federal fisheries. Compliance with the Federal requirement for federally permitted fishermen to report sea turtle interactions on their VTRs is very low. Without reliable VTR reporting of sea turtle takes, NMFS is using information collected through the Northeast Fisheries Observer Program (NEFOP), which collects, processes and manages data and biological samples obtained by trained observers during a subset of commercial fishing trips throughout the New England and the Mid-Atlantic regions.

The discussion of sea turtle takes in monkfish gear that follows will focus on trawl gear and sink gillnet gear. Past observed takes of ESA-listed species in trawl gear and sink gillnets were reviewed in the April 14, 2003 Opinion for the monkfish fishery. Updated information is provided herein. It is difficult to ascertain gear types responsible for entanglements when only portions of the gear or injuries resulting from entanglements are observed. Additionally it is important to note that the reported takes are likely a fraction of the total takes, which are unknown.

The majority of interactions between sea turtles and bottom trawl fisheries of the Atlantic coast have occurred south of the New England region since the distribution of sea turtles correlates with warmer water temperatures, resulting in greater densities of sea turtles south of New England. The spatial distribution of turtles in southern New England and the Mid-Atlantic is coincident with several fisheries.

Loggerhead sea turtles represent the majority of sea turtles species observed incidentally taken in trawl and gillnet gear in the action area. Observers reported 66 loggerhead sea turtle interactions with bottom otter trawl gear from 1994-2004 (Murray 2008). Of the 66 documented loggerhead interactions, 38 (57%) were alive and uninjured, and 28 (43%) were dead, injured, resuscitated, or of unknown condition. From 1995-2006, 41 loggerhead turtles were documented as incidentally caught in Mid-Atlantic sink gillnet gear (Murray 2009a). Approximately 80% of the loggerheads taken in gillnet gear were determined to be juveniles; approximately 40% of the loggerheads taken were dead (Murray 2009a). Documented trawl and gillnet gear takes of loggerheads after the time periods analyzed in Murray (2008, 2009a) are presented in Table 2.

Table 2. Documented incidental captures of loggerhead sea turtles (excluding moderately and severely decomposed turtles) in bottom otter trawl (scallop, fish, and twin[9]) from 2005-2007 and gillnet gear from 2007-2009 along with the most abundant (by weight) abundant commercial species landed per trip. Gillnet gear includes anchored sink gillnets and drift sink gillnets. Source: NEFSC FSB database.

	Bottom Otter Trawl										Gillnet			
Most Landed Species (by weight)	Summer Flounder	Monkfish	Little Skate	Atlantic Croaker	Squid	Smooth Dogfish	Winter Flounder	Horseshoe Crab	Atlantic Sea Scallop	Unassigned	Sandbar Shark	Southern Flounder	Atlantic Croaker	Monkfish
Loggerhead takes	13	1	1	56	5	1	1	3	11	1	1	4	1	1
Years	2005-2009										2007-2009			

The estimates of loggerhead sea turtle bycatch in bottom otter trawl gear and gillnet gear published in Murray (2008, 2009a) represent the best available information and analysis for loggerhead bycatch in mid and North Atlantic commercial fisheries. Such estimates are not available for leatherback, Kemp's ridley, and green sea turtle takes. Therefore, observer data for these species represents the best available information.

The NEFOP has documented the most landed kept species (by weight) when an incidental take occurs (among many other variables), and that information has been used to provide a look at which commercial species most correspond to the incidental takes for leatherback, Kemp's ridley, and green sea turtles (Table 3).

Table 3. Documented incidental captures of leatherback, Kemp's ridley, green, and unidentified sea turtles (excluding moderately and severely decomposed turtles) in bottom otter trawl (scallop and fish) and gillnet gear from 2000-2009 along with the most landed species (by weight) per trip. Gillnet gear includes anchored sink gillnets and drift sink gillnets. Source: NEFSCFSB database.

	Bottom Otter Trawl						Gillnet								
Most Landed (by weight)	Longfin Squid	Summer Flounder	Atlantic Croaker	Silver Hake	Pollock	Atlantic Sea Scallop	Spanish Mackerel	Summer Flounder	Southern Flounder	Smooth Dogfish	Spiny Dogfish	Bluefish	Winter Skate	King Mackerel	Monkfish
Leatherback	1	1	0	1	0	0	1	0	0	0	0	1	1	0	0

[9] Twin trawl gear only accounted for one loggerhead capture with summer flounder as the most landed kept species (by weight).

Kemp's	0	1	1	0	0	0	1	0	5[10]	1	1	0	0	0	0
Green	1	0	0	0	0	0	1	2	12[11]	0	0	0	0	0	0
Unidentified	0	0	3	0	1	1	0	0	0	1	0	0	0	3	5

While it may be informative to look at the number of leatherback, Kemp's ridley and green sea turtles observed to have been taken on bottom otter trawl and gillnet trips when the majority of the landings were monkfish, using this number as the estimated take would be an underestimate in two ways. First, sea turtle takes could have occurred on trips where monkfish was part of the catch, but constituted less than the majority of the catch. Second, these takes are only observed takes and we are not currently able to use them to generate an estimate of total takes. In order to compensate for this underestimate, for the purposes of estimating incidental take of leatherback, Kemp's ridley and green sea turtles in fishing gear authorized under the Monkfish FMP we are going to look at takes by gear type as illustrated in the table below (Table 4).

Table 4. Documented incidental takes of leatherback, Kemp's ridley, green, and unidentified sea turtles (excluding moderately and severely decomposed turtles) in bottom otter trawl (BOT, scallop and fish) and gillnet gear from 2000-2009. Gillnet gear includes anchored sink gillnets and drift sink gillnets. Source: NEFSC Fisheries Sampling Branch (FSB) database.

	Documented # of incidental takes in BOT gear	**Documented # of incidental takes/year in BOT gear**	**Documented # of incidental takes in gillnet gear**	**Documented # of incidental takes/year in gillnet gear**
Leatherback sea turtle	3	0.3	3	0.3
Kemp's ridley sea turtle	2	0.2	8	0.8
Green sea turtle	1	0.1	15	1.5
Unidentified sea turtle	5	0.5	9	0.9

The NEFSC conducts trawl surveys to monitor marine resources and their habitats. During spring and fall bottom otter trawl surveys conducted by the NEFSC from 1963-2008, a total of 71 loggerhead sea turtles were observed captured. The NEFSC trawl survey tows are approximately 30 minutes in duration. In contrast, commercial fisheries typically tow bottom otter trawl gear in excess of one hour (Murray 2006).

[10] Twelve (12) green and five (5) Kemp's ridley sea turtles were observed incidentally taken in 2009 by a state fishery targeting southern flounder with sink gillnet gear in Pamlico Sound. Although Pamlico Sound is located at the southern most point of the action area, with a large presence south of the action area, these takes were documented by the NEFOP and will be included in this Opinion.

[11] Ibid

Observations of takes in bottom otter trawls indicate that fisheries using this gear type are capable of incidentally taking sea turtles and that some of these interactions are lethal. 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 trawl. 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). NMFS anticipates that green sea turtles will interact with trawl gear in the same manner as loggerhead sea turtles *(i.e.,* both on the bottom and in the water column). Therefore, if loggerhead, Kemp's ridley, and green sea turtles are foraging in areas where the monkfish trawl fishery operates, the turtles would be at greater risk.

Tagging studies have shown that leatherbacks, occurring seasonally for foraging in western North Atlantic continental shelf waters where the monkfish 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 dive-depth information on leatherback use of western North Atlantic continental shelf waters (James *et al.* 2005a; 2005b), leatherbacks may spend more time in the water column than on the bottom. Given that leatherbacks forage within the water column rather than on the bottom, interactions between leatherback sea turtles and bottom otter trawl gear are expected to occur when the gear is traveling through the water column versus on the bottom. Given that sea turtle interactions have been observed in the monkfish fishery as well as known distribution patterns of sea turtles in the water column along the Atlantic coast, bottom trawl interactions with sea turtles are expected to occur in the monkfish fishery.

Potential sea turtle interactions with sink gillnets are most likely to occur with loggerhead, Kemp's ridley, and green sea turtles since these species are more likely to be found near the bottom (NMFS 2001b). Theoretically, sea turtles could become entangled in either the net, buoy lines or surface system of sink gillnets. Sea turtles are unlikely to be able to break off entangling fishing gear. Turtles are vulnerable to drowning from forced submergence, although some turtles have been recovered alive in sink gillnet gear.

Fishing trips that target monkfish typically use 10 inch mesh, which is about the largest size mesh used across all the gillnet fisheries in the action area. Murray (2009b) found that gillnet mesh size explained the largest amount of variation in loggerhead bycatch rates among the variables examined, and that estimated bycatch rates of loggerheads in sink gillnet gear were significantly higher in large mesh sizes.

6.1.4 Factors Affecting Cetacean Takes in Monkfish Fishing Gear

As stated in Section 6.1.2, there have been no documented interactions between ESA-listed marine mammals and the North Atlantic bottom trawl fishery (NMFS 2006a). Their great size and mobility presumably allows them to avoid interactions with the relatively slow moving trawl gear. Sink gillnets are left in the water for a discrete period, after which time the nets are hauled and their catch retrieved. While the gear is in the water, whales may become incidentally entangled in the lines and nets that comprise gillnet fishing gear. The effects of entanglement can range from no injury to death.

Atlantic large whales are at risk of becoming entangled in fishing gear because the whales feed, travel and breed in many of the same ocean areas utilized in the monkfish fishery action area. As described in detail in sections 3.1.1-3.1.4 North Atlantic right whales, humpback whales, sei 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 (Waring *et al.* 2009). 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).

Western North Atlantic right whales occur from the southeastern U.S. (waters off of Georgia and Florida) to Canada (Kenney 2002, Waring *et al.* 2009). Generally, they follow an annual pattern of migration from foraging areas to calving areas in Florida. However, only a portion of the known North Atlantic right whale population has been observed on the calving grounds. Results from winter surveys and passive acoustic studies suggest that animals may be dispersed in several areas including Cape Cod Bay (Brown *et al.* 2002) and offshore waters of the southeastern U.S. (Waring *et al.* 2009).

Generally, Atlantic humpback whales calve and mate in the West Indies after foraging in the northwestern Atlantic during the summer months. Sightings of humpbacks in the New England area are most frequent from mid-March through November, but small numbers of individuals may remain in the area between Cape Cod and Jeffrey's bank year-round (CeTAP 1982). The Mid-Atlantic may also be an important feeding ground for juvenile humpbacks. Since 1989, observations of juvenile humpbacks in the Mid-Atlantic have been peeking in January through March (Swingle *et al.* 1993).

Fin whales are believed to use the North Atlantic water primarily for feeding and more southern waters for calving. Movement of fin whales from the Labrador/Newfoundland region south into the West Indies during the fall have been reported (Clark 1995). However, neonate strandings along the U.S. Mid-Atlantic coast from October through January indicate a possible offshore calving area (Hain *et al.* 1992).

The sei whale is often found in the deeper waters characteristic of the continental shelf edge region (Hain *et al.* 1985), and NMFS aerial surveys found substantial numbers of sei whales in this region, south of Nantucket, in the spring of 2001. Spring is the period of greatest abundance in New England waters, with sightings concentrated along the eastern margin of Georges Bank and into the Northeast Channel area, and along the southwestern edge of Georges Bank in the area of Hydrographer Canyon (CETAP 1982). NMFS aerial surveys in 1999, 2000 and 2001 found concentrations of sei and right whales along the northern edge of Georges Bank in the spring. In years of greater abundance of copepod prey sources, sei whales are reported in more inshore locations, such as the Great South Channel (in 1987 and 1989) and Stellwagen Bank (in 1986) (Waring *et al.* 2009).

Fishing effort for monkfish has occurred during all seasons with trends in landings changing geographically and across years. Seasonal patterns appear to be related to the amount of fishing effort directed on species of groundfish (NEFMC 1998). Patterns of groundfish effort could change due to the changing regulations on the time that a vessel may fish for multispecies. Landings of monkfish bycatch in Virginia appear to peak in the spring, perhaps associated with the spring, offshore summer flounder fishery (NEFMC 1998).

Since the highest abundances of North Atlantic right, humpback, fin, and sei whale populations occur from March through November in New England waters and peak abundances of sei whales have been identified during the spring season, the presence of these whales overlaps peak fishing periods with otter trawl and gill net gear for the monkfish fishery. Humpback and fin whales are utilizing the Mid-Atlantic waters during October-March with seemingly increasing frequency and may have low numbers of whales residing in New England waters through the winters. Because of substantial interannual and geographic variation in whale occurrences and lack of complete data for seasonal distributions, the potential exists for whale interactions with the monkfish fishery throughout the seasons and extent of the action area. However, given the seasonal distribution of ESA-listed whales and the times and areas when the monkfish fishery operates, North Atlantic right, humpback, fin, and sei whales are most likely to overlap with operation of the fishery from May through November in New England waters and throughout the fall and winter in Mid-Atlantic waters.

6.1.5 Factors Affecting Sea Turtle Takes in Monkfish Fishing Gear

As described in sections 3.2.1 – 3.2.4, 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 becoming much less abundant in areas north of 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).

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 monkfish fishery does not operate in bays, inlets, rivers, or sounds, sea turtle distribution would not be expected to overlap with the distribution of monkfish fishing gear until May in nearshore and offshore waters off of North Carolina and Virginia, and until June in nearshore and offshore waters of New York. Given the seasonal distribution of sea turtles and the times and areas when the monkfish 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.

NMFS has also considered other factors that might affect the likelihood that an ESA-listed species will be captured in monkfish fishing gear. These other factors include the behavior of the animals in the presence of fishing gear, as well as the effect of certain oceanographic features and fishery practices on population distributions and abundances. For example, video footage recorded by 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 2002a). Intensity of biological activity in the Gulf of Maine has been associated with oceanographic fronts, including nutrient fluxes and biological productivity. Particular oceanographic features and processes that influence biological activity are vertical mixing by tides; the seasonal cycle of heating and cooling that leads to winter convection and vertical stratification in summer; pressure gradients from density contrasts set up by deep water inflows and lower salinity waters; and influxes of the cold, but fresher waters associated with Scotian Shelf Water (Townsend *et al.* 2006). Such oceanographic features occurring in the same area as the operation of monkfish gear may increase the risk of interactions between monkfish gear and ESA-listed species that would be attracted to these areas for feeding. However, at present there is no information to clearly indicate any of these as influencing ESA-listed species takes in monkfish fishing gear.

Loggerhead bycatch rates in Mid-Atlantic sink gillnet gear are correlated with the mesh size, water temperature, and area fished (Murray 2009b). Murray (2009b) found that gillnet mesh size explained the largest amount of variation in loggerhead bycatch rates among the variables examined, and that estimated bycatch rates of loggerheads in sink gillnet gear were significantly higher in large mesh sizes. Gear targeting monkfish typically use 10" mesh, about the largest size mesh used across all the gillnet fisheries. Based on the best currently available information, cetacean and sea turtle interactions with monkfish gear are likely at times when and in areas where cetacean and sea turtle distribution overlaps with operation of the fishery.

6.2 Anticipated Effects of the Proposed Action

Another significant event that has taken place over the last decade is the reduction in fishing capacity and effort in U.S. Atlantic fisheries. For example, effort in the Northeast multispecies fisheries as a result of Amendment 16 is expected to be reduced by nearly 75% when compared to fishing effort and capacity in the early 1990's (NEFMC 2009). While some fishing effort may increase in the future as fisheries stocks respond to management measures to rebuild them, there are measures in place that will prevent overcapacity from redeveloping (i.e., nearly all U.S. Atlantic commercial fisheries are closed/limited access). Furthermore, as fish stocks increase, a more likely outcome will be increased catches/landings with constant or even reduced fishing effort.

However, since takes of ESA-listed species have been documented as a result of the monkfish fishery, NMFS has determined that the continuation of the monkfish fishery under the authority of the Monkfish FMP is likely to adversely affect ESA-listed cetaceans and sea turtles when the animals come into physical contact with monkfish fishing gear. Such contact can result in injuries, including very severe injuries causing death. No other direct effects to cetaceans or sea turtles are expected as a result of the proposed action. No indirect effects to cetaceans or sea turtles are expected as a result of the proposed action. In this section of the Opinion, NMFS will determine, given the currently available information, the anticipated number of cetaceans and sea turtles that will be affected by the continued operation of the monkfish fishery defining such effects by species.

6.2.1 Anticipated take of cetaceans in monkfish gear

No method has yet been identified for predicting the level of overall or species-specific cetacean bycatch in the monkfish fishery or any particular gear-type component of the fishery. Many whale mortalities may never be observed, thus the actual annual number of documented entanglement related mortalities may be a subset of the actual number of entanglement related mortalities that occur. Additionally, assignment of a specific fishery to an observed entanglement is rarely possible because even in those rare cases where gear is retrieved, identification remains problematic because the same gear (e.g. lines and webbing) is used in multiple fisheries.

It should be noted that the analysis of entanglement events used in this Biological Opinion differs in an important way from the reporting in the NOAA Stock Assessment Reports for Marine Mammals. Specifically, gear analyses results were the criteria used to categorize entanglement events to U.S., Canadian, or undefined origin in this Opinion; in contrast, the NOAA Stock Assessment Reports for Marine Mammals initially use the location the animal was first sighted to categorize the events to "U.S. waters" or "Canadian waters," then re-assign any events when/if gear analyses provides a confirmed country of origin for the involved gear. The location an entangled whale is first sighted may be quite a distance from the original location of entanglement.

The objective of NOAA Stock Assessment Reports for Marine Mammals is to report status of marine mammal populations. The objective of this Biological Opinion is to assess potential impacts to ESA-listed species due to the proposed action, which in this case is the continuation of the monkfish fishery under the authorization of the Monkfish FMP. Thus, for the purposes of this Opinion NMFS has chosen to exclude entanglement events that have been attributed to gear used in Canadian fisheries, and in turn, NMFS has made the decision to focus on entanglement events that are of undetermined origin or confirmed U.S. origin since these events are directly attributed to U.S. fisheries or can't be ruled out as effects of U.S. fisheries. This conservative approach is meant to comply with direction from the U.S. Congress to provide the "benefit of the doubt" to threatened and endangered species [House of Representatives Conference Report No.697, 96th Congress, Second Session, 12 (1979)].

6.2.1.1 Otter trawls

Right, humpback, fin, and sei whales occur in the area where monkfish fisheries take place. Nevertheless, none of these are expected to be affected by the use of bottom otter trawl gear given that: (l) these large cetaceans have the speed and maneuverability to get out of the way of oncoming mobile gear, including trawl gear, (2) the fisheries will not affect the availability of prey for these species, and (3) the monkfish fishery does not occur in low latitude waters where calving and nursing occurs for these large cetacean species.

6.2.1.2 Sink gillnets

The monkfish fishery accounts for a small amount of overall sink gillnet effort in the New England and Mid-Atlantic regions. Although most entanglement reports do not contain enough information to assign the takes to particular fisheries, the gillnet gear is in some instances known to be similar to gear used in the monkfish fishery.

From 1999-2008 gillnet gear of U.S. or undocumented origin was recorded in 26 entanglement events with right and humpback whales (NMFS NERO 2010). Of those 26 events, sink gillnet gear was verified to be involved with entanglements of one (1) right whale and humpback whales. Three (3) of the 12 sink gillnet interactions resulted in serious injuries or mortalities (NMFS NERO 2010). The mean annual rate for whales entangled in sink gillnet gear of U.S. or undetermined origin has been 1.2 for the 1999-2008 time period. Of those occasions, entanglements determined to be serious injury/mortality events resulted in a mean annual rate of 0.3. Considering the continued efforts of the ALWTRP to reduce gillnet impacts on large whales, NMFS anticipates the rates of annual mean entanglements will not increase from those listed in Table 1.

6.2.2 Anticipated take of sea turtles in monkfish gear

Commercial fishery. As described earlier in this Opinion, several reports by Murray (2008, 2009a) have been published that analyze fishery observer data and VTR data from fishermen in order to estimate the takes of loggerhead sea turtles in bottom otter trawl gear and gillnet gear in the Mid-Atlantic. These reports estimate the average number of loggerhead sea turtles taken in each gear type (bottom trawl and sink gillnet, respectively) across all fisheries (i.e., FMPs), and they each also divide the takes by FMP or fish species landed. These documents represent the most accurate predictor for sea turtle takes in the monkfish fishery and other Northeast fisheries that use these gear types.

It is important to note that while both reports divide the takes by individual species landed, the two reports use different methodologies. The trawl estimate (Murray 2008) assigned associated takes to individual species landed based on the most significant species landed (by weight) for that trip. The gillnet estimate (Murray 2009a) assigned and associated takes based on the distribution of landings for that trip. For example, trips in a certain time and area using gillnets or trawls were estimated to have a certain take rate of loggerhead sea turtles (based on the observed takes). In the trawl estimate, each trip in that time/area was assigned to a individual species landed. So if a trip landed 60 percent summer flounder, 20 percent spiny dogfish, and 20 percent weakfish, the trip and its associated takes (calculated using the take rate), were assigned to summer flounder and summer flounder only. In the gillnet estimate, the trip and its associated takes (calculated using the take rate), were assigned to summer flounder, spiny dogfish, and weakfish, in a 60:20:20 ratio. The latter method, used in the gillnet estimate, is meant to reflect the multispecies nature of many of the fisheries in the Mid-Atlantic region.

Another difference between the two estimates is that the trawl estimate does not provide a confidence interval around the point estimate for each target species – it just provides an average annual take level over the 2000-2004 time period. The gillnet estimate does provide a 95 percent confidence interval around the annual point estimate for each target species. Due to this difference, the takes assumed and analyzed for this Opinion are the point estimates for trawl gear, and are the upper end of the 95 percent confidence interval for gillnet gear. This difference is also carried through into the Incidental Take Statement for loggerheads, and influences how the takes in the fishery will be monitored.

The NEFSC is in the process of conducting an updated estimate of loggerhead sea turtle takes in Mid-Atlantic trawl gear, using more recent observer and fisheries data. It is anticipated that the FMP/target species breakdown will be conducted in a manner similar to that done for the gillnet paper (Murray 2009a). The updated trawl estimate is expected to be published in late 2010.

6.2.2.1 Otter trawls

Based on data collected by observers for the reported sea turtle captures in bottom otter trawl gear, the NEFSC estimated the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear for the monkfish fishery from 2000-2004 as two (2) loggerheads (Murray 2008).

This estimate of loggerhead sea turtle takes in the bottom otter trawl gear provide the best available information for determining the anticipated take of loggerhead sea turtles in that component of the fishery. For the purposes of this Opinion, NMFS is using the estimate of two (2) loggerheads per year as the best available information for the anticipated take of loggerhead sea turtles in the bottom otter trawl component of the monkfish fishery in future years.

There are no science-based estimates for the capture of leatherback, Kemp's ridley or green sea turtles in bottom otter trawl fishing gear. As stated earlier in Section 6.1.3, NEFOP observers have documented interactions with three (3) leatherbacks, two (2) Kemp's ridleys, one (1) green, and five (5) unidentified sea turtles in Mid-Atlantic bottom otter trawl gear from January 2000 through December 2009 (NEFSC FSB database).

The very low number of observed leatherback captures in bottom otter trawl gear used in multiple trawl fisheries in the action area suggests that capture of leatherback sea turtles within the action area would be a rare event. However, given the generally low percentage of trips with observer coverage in the monkfish fishery as well as other mobile gear (trawl) fisheries in the action area, it is possible that some interactions with leatherback sea turtles have occurred but were not observed or reported. Given effort in the fishery as a whole, and the seasonal overlap in distribution of this species with operation of monkfish trawl gear, leatherback sea turtles are likely to be captured in monkfish trawl gear.

As summarized in Table 4 in Section 6.1.3, the number of documented leatherback incidental captures in bottom otter trawl gear has been 0.3 annually. Since the take of a partial turtle is not possible, NMFS anticipates the potential annual take of one (1) leatherback sea turtle with bottom otter trawl gear. Additionally, because of the average annual take of 0.5 unidentified turtles in bottom otter trawl gear, another sea turtle (either a leatherback, Kemp's ridley, or green) is forecasted to be taken in the monkfish fishery annually. Thus, the continued operation of the bottom otter trawl gear component of the monkfish fishery is anticipated to result in the annual non-lethal or lethal take of up to two (2) leatherback sea turtles.

The very low number of observed Kemp's ridley captures in bottom otter trawl gear used in multiple trawl fisheries in the action area suggests that capture of Kemp's ridley sea turtles within the action area would be a rare event. However, given the generally low percentage of trips with observer coverage in the monkfish fishery as well as other mobile gear (trawl) fisheries in the action area, it is possible that some interactions with Kemp's ridley sea turtles have occurred but were not observed or reported. Given effort in the fishery as a whole, and the seasonal overlap in distribution of this species with operation of monkfish trawl gear, Kemp's ridleys are likely to be captured in monkfish trawl gear.

As summarized in Table 4 in Section 6.1.3, the number of documented Kemp's ridley incidental captures in bottom otter trawl gear has been 0.2 annually. Since the take of a partial turtle is not possible, NMFS anticipates the potential annual take of one (1) Kemp's ridley sea turtle with bottom otter trawl gear. Additionally, because of the average annual take of 0.5 unidentified turtles in bottom otter trawl gear, another sea turtle (either a leatherback, Kemp's ridley, or green) is forecasted to be taken in the monkfish fishery annually. Thus, the continued operation of the bottom otter trawl gear component of the monkfish fishery is anticipated to result in the annual non-lethal or lethal take of up to two Kemp's ridley sea turtles.

The very low number of observed green sea turtle captures in bottom otter trawl gear used in multiple trawl fisheries in the action area suggests that capture of green sea turtles within the action area would be a rare event. However, given the generally low percentage of trips with observer coverage in the monkfish fishery as well as other mobile gear (trawl) fisheries in the action area, it is possible that some interactions with green sea turtles have occurred but were not observed or reported. Given effort in the fishery as a whole, and the seasonal overlap in distribution of this species with operation of monkfish trawl gear, green sea turtles are likely to be captured in monkfish trawl gear.

As summarized in Table 4 in Section 6.1.3, the number of documented green sea turtles incidental captures in bottom otter trawl gear has been 0.1 annually. Since the take of a partial turtle is not possible, NMFS anticipates the potential annual take of one (1) green sea turtle with bottom otter trawl gear. Additionally, because of the average annual take of 0.5 unidentified turtles in bottom otter trawl gear, another sea turtle (either a leatherback, Kemp's ridley, or green) is forecasted to be taken in the monkfish fishery annually. Thus, the continued operation of the bottom otter trawl gear component of the monkfish fishery is anticipated to result in the annual non-lethal or lethal take of up to two (2) green sea turtles.

6.2.2.2 Sink gillnets

From 2002-2006, the average annual bycatch estimate of loggerheads in Mid-Atlantic sink gillnet gear was 288 turtles (Murray 2009a). For the monkfish fishery, it is estimated that a mean of 118 loggerhead takes occur per year (with a 95% CI for the 5-year annual average of 68-171). With this CI, it would be expected that anywhere from 68-171 loggerheads could be taken on an annual basis and that would be within the range of estimated takes based on past records. This estimate of loggerhead sea turtle takes in the Mid-Atlantic sink gillnet gear provides the best available information for determining the anticipated take of loggerhead sea turtles in that component of the fishery since no predictive variable or set of variables has been found. For the purposes of this Opinion, NMFS is assuming that an average of up to 171 loggerheads per year (the upper end of the 95% CI) over a 5-year period is the best available information for the anticipated take of loggerhead sea turtles in the gillnet component of the monkfish fishery.

Trips landing monkfish had the largest amount of estimated loggerhead bycatch primarily because trips landing monkfish had (a) high predicted bycatch rates (monkfish are mainly caught with ~30 cm mesh gear), and (b) large landings volumes (Murray 2009b). In interpreting these estimates it should be considered that many of the monkfish landed are bycatch of other directed gillnet efforts and were not the major component of fish taken during fishing trips that captured loggerhead sea turtles. This is different than estimates of loggerhead sea turtle takes in Mid-Atlantic trawl gear by Murray (2006, 2008) in that the trawl gear estimates assigned turtle takes to fisheries based on the dominant fish landed per weight. In contrast, the Murray (2009b) assessment for gillnet gear assigns loggerhead takes to fisheries based on percentages of fish landed in each haul that captured a loggerhead turtle. Thus, when multiple species were landed, the estimated bycatch per trip was prorated across all of the species landed based on the proportion (by weight) of the species landings in the trip. For instance, if a vessel landed 600 pounds of skate, 300 pounds of monkfish, and 100 pounds of bluefish, the estimated number of loggerheads for that trip was apportioned among these three species, with monkfish receiving 30% of the total estimated loggerhead bycatch.

There are no total bycatch estimates for the capture of leatherback, Kemp's ridley or green sea turtles in sink gillnet fishing gear. As listed in Table 4, NEFOP observers have documented interactions with three (3) leatherback, eight (8) Kemp's ridley, fifteen (15) green, and nine (9) unidentified sea turtles in Mid-Atlantic sink gillnet gear from 2000 through 2009 (NEFSC FSB database).

The low number of observed leatherback captures in sink gillnet gear used in multiple gillnet fisheries in the action area suggests that capture of leatherback sea turtles within the action area would be a rare event. However, given the generally low percentage of trips with observer coverage in the monkfish fishery as well as other gillnet fisheries in the action area, it is possible that additional interactions with leatherback sea turtles have occurred but were not observed, not reported, or the turtles were not identified as to species. Given effort in the fishery as a whole, and the seasonal overlap in distribution of this species with operation of monkfish gillnet gear, leatherback sea turtles may be captured in monkfish gillnet gear.

As summarized in Table 4 in Section 6.1.3, the number of documented leatherback incidental captures in gillnet gear has resulted in an average of 0.8 per year. Since the take of a partial turtle is not possible, NMFS anticipates the potential annual take of one (1) leatherback sea turtle with sink gillnet gear. Additionally, because of the average annual take of 0.9 unidentified turtles in gillnet gear, another sea turtle (either a leatherback, Kemp's ridley, or green) is forecasted to be taken in the monkfish fishery annually. Thus, the continued operation of the gillnet gear component of the monkfish fishery is anticipated to result in the annual non-lethal or lethal take of up to two (2) leatherback sea turtles.

Likewise, the low number of observed Kemp's ridley captures in sink gillnet gear used in multiple trawl fisheries in the action area suggests that capture of Kemp's ridley sea turtles within the action area would be a rare event. However, given the generally low percentage of trips with observer coverage in the monkfish fishery as well as other gillnet gear fisheries in the action area, it is possible that additional interactions with Kemp's ridley sea turtles have occurred but were not observed, not reported, or the turtles were not identified as to species. Given effort in the fishery as a whole, and the seasonal overlap in distribution of this species with operation of monkfish gillnet gear, Kemp's ridleys may be captured in monkfish gillnet gear.

As summarized in Table 4 in Section 6.1.3, the number of documented Kemp's ridley incidental captures in gillnet gear has resulted in an average of 0.3 per year. Since the take of a partial turtle is not possible, NMFS anticipates the potential annual take of one (1) Kemp's ridley sea turtle with sink gillnet gear. Additionally, because of the average annual take of 0.9 unidentified turtles in gillnet gear, another sea turtle (either a leatherback, Kemp's ridley, or green) is forecasted to be taken in the monkfish fishery annually. Thus, the continued operation of the gillnet gear component of the monkfish fishery is anticipated to result in the annual non-lethal or lethal take of up to two (2) Kemp's ridley sea turtles.

Similarly, the low number of observed green sea turtle captures in sink gillnet gear used in multiple trawl fisheries in the action area suggests that capture of green sea turtles within the action area would be a rare event. However, given the generally low percentage of trips with observer coverage in the monkfish fishery, as well as other gillnet gear fisheries in the action area, it is possible that additional interactions with green sea turtles have occurred but were not observed, not reported, or the turtles were not identified as to species. Given effort in the fishery as a whole, and the seasonal overlap in distribution of this species with operation of monkfish gillnet gear, greens may be captured in monkfish gillnet gear.

As summarized in Table 4 in Section 6.1.3, the number of documented green sea turtles incidentally captured in gillnet gear has resulted in an average of 1.5 per year. Since the take of a partial turtle is not possible, NMFS anticipates the potential take of two (2) green sea turtle with sink gillnet gear. Additionally, because of the average annual take of 0.9 unidentified turtles in gillnet gear, another sea turtle (either a leatherback, Kemp's ridley, or green) is forecasted to be taken in the monkfish fishery annually. Thus, the continued operation of the gillnet gear component of the monkfish fishery is anticipated to result in the annual non-lethal or lethal take of up to three (3) green sea turtles.

6.2.3 Age classes of sea turtles anticipated to interact with the monkfish fishery

Loggerhead sea turtles. The size range in which oceanic juveniles and neritic juveniles overlap has been described as 46-64 cm curved carapace length (CCL) (NMFS and UFWS 2008). The two (2) loggerhead sea turtles taken in 1996 with small mesh sink gillnet gear targeting groundfish had curved carapace lengths (CCL) of 28 cm and 84 cm (NEFSC, FSB Observer Database). Although the two (2) turtles captured seem to fit the oceanic juvenile and neritic juvenile size categories, respectively, NMFS expects that both juvenile and mature loggerhead sea turtles may be captured in monkfish fishing gear because both life stages are present within the action area of the monkfish fishery (NMFS and USFWS 2008).

Leatherback sea turtles. NMFS believes that leatherback sea turtles could be captured in monkfish fishing gear given the presence of leatherback sea turtles in areas where the fishery occurs. Stranding and sighting records suggest that both adult and immature leatherback sea turtles occur within the action area where the monkfish fishery operates (NMFS and USFWS 1992; NMFS SEFSC 2001). Therefore, either immature or sexually mature leatherback sea turtles could be captured in monkfish fishery gear since both age classes occur in areas where the fishery operates.

Kemp's ridley sea turtles. The post-hatchling stage for Kemp's ridley sea turtles was defined by the TEWG as Kemp's ridleys of 5-20 cm (2-8 inches) SCL while turtles 20-60 cm (8-23 inches) SCL were considered to be benthic immature (TEWG 2000). The latter stage is described as sea turtles that have recruited to coastal benthic habitat. Mid-Atlantic and coastal New England waters (as far north as approximately Cape Cod) are known to be developmental foraging habitat for immature Kemp's ridley sea turtles, while adults have been documented from waters and nesting beaches along the South Atlantic coast of the U.S. (Musick and Limpus 1997; TEWG 2000; Morreale and Standora 2005). Given the life history of the species, NMFS expects that both immature and sexually mature Kemp's ridley sea turtles could be captured in monkfish fishery gear as a result of the continued operation of the fishery.

Green sea turtles. Hirth (1997) defined a juvenile green sea turtle as a post-hatchling up to 40 cm (16 inches) 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 inches) SCL (Hirth 1997). 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). NMFS expects that benthic immature and/or sexually mature green sea turtles could be captured in monkfish fishing gear as a result of the continued operation of the fishery.

6.2.4 Estimated mortality of sea turtles that interact with monkfish fishing gear

The following information is provided as an assessment of the types of injuries likely to occur in the future for sea turtles affected by the continued operation of the monkfish 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%). Intermediate tow times (10-200 minutes in summer and 10-150 minutes in winter) result in a rapid escalation of mortality, and eventually reach a plateau of high mortality, but will not equal 100%, as a 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).

Epperly *et al.* (2002) and Sasso and Epperly (2006) found that, in general, otter trawl 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% as a turtle caught within the last hour of a long tow will likely survive (Epperly *et al.* 2002; Sasso and Epperly 2006). Murray (2009b) found that tow times of bottom otter trawl gear that resulted in sea turtle bycatch ranged from 0.5 to over 5 hours.

As described in section 6.2.2.1, NMFS anticipates the annual capture of up to two (2) loggerhead sea turtles, two (2) leatherbacks, two (2) Kemp's ridleys, and two (2) green sea turtles in monkfish bottom otter trawl gear. Of 66 documented loggerhead interactions with bottom otter trawl gear from 1994-2004, 38 (57%) were alive and uninjured, and 28 (43%) were dead, injured, resuscitated, or of unknown condition (Murray 2008). The percentage of turtles in the injured, resuscitated, and unknown condition categories that actually survive and return to a fully functional individual is unknown.

NMFS is using the 57% number listed above as the estimate of loggerhead turtles that survive the interactions with bottom otter trawl gear. Therefore, an estimated 43% of the estimated two (2) loggerhead sea turtles taken per year in monkfish fishery trawl gear results in an anticipated lethal take of 0.86 loggerhead sea turtles annually. NMFS interprets these calculations to represent that one (1) loggerhead take per year in monkfish trawl gear will result in lethaltakes.

As discussed in Section 6.2.2.2, NMFS anticipates the annual take of up to 171 loggerhead sea turtles in monkfish gillnet gear. Murray (2009a) stated that about 40% of the observed loggerheads in gillnet fisheries from 1995 through 2006 were dead. Therefore, NMFS anticipates that 69 loggerhead takes in the gillnet component of the monkfish fishery could result in lethal takes.

For leatherback, Kemp's ridley,.and green sea turtles taken in trawl and gillnet gear, the low occurrence of these observations does not allow valid determinations on the anticipated levels of lethal takes for these events, therefore, these takes could be lethal or non-lethal.

6.3 Summary of anticipated incidental take of cetaceans and sea turtles in the monkfish fishery

The primary gear types used in the monkfish fishery are bottom trawls and sink gillnets, which account for approximately 95% of effort in the fishery. Although large whale entanglements in trawl gear have been documented, these are rare events relative to gillnet entanglements. Based on results from large whale entanglement analyses, NMFS believes the greatest risk to whales from the monkfish fishery is entanglements in gillnet gear.

As described previously, the six species of ESA-listed whales found in the action area for this consultation are the right, humpback, fin, sei, blue, and sperm whales. Interactions with blue and sperm whales would be unlikely since those species prefer deep waters and are infrequently encountered within the action area. Based on the NMFS large whales entanglement data for the years 1999-2008, the annual mean rates of fin whale and sei whale entanglements have been 0.8 and 0.3, respectively (Table 1). One hundred percent (100%) of the entanglement records for fin whales and sei whales are with undetermined gear types.

As shown above in Table 1, the annual serious injury and mortality for right and humpback whales from entanglement in U.S. Atlantic coast fisheries (i.e., gear that was not confirmed as Canadian) averaged 1.0 and 3.4 animals, respectively for 1999-2008. The 2009 Marine Mammal Stock Assessment Report provides an annual mean rate of serious injury or mortality (SI/M) fishery gear entanglements to be 0.6 and 2.4, respectively for right and humpback whales in U.S. waters between 2003-2007. Sink gillnet gear, a gear used in the monkfish fishery, has been documented in entanglements with right and humpback whales. Sink gillnet gear was verified to be involved with entanglements of one (1) right whale and 11 humpback whales over the 1999-2008 time period. The continued implementation and development of ALWTRP measures provide cause to anticipate the number of right and humpback whale entanglements should decline or, at least, not increase.

As a result of the continued operation of the monkfish fishery, NMFS anticipates the annual trawl gear capture of up to two (2) loggerhead sea turtles, two (2) leatherbacks, two (2) Kemp's ridleys, and two (2) green sea turtles. One (1) of the captured loggerheads is likely to result in a lethal take. Takes of leatherbacks, Kemp's ridley, and green sea turtles could be lethal or non-lethal. For gillnet gear, NMFS anticipates the take of 171 loggerheads, two (2) leatherbacks, two (2) Kemp's ridleys, and three (3) green sea turtles. Sixty nine (69) loggerhead takes are expected to be lethal and all other species takes in gillnet gear could 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 have caused right, humpback, fin, and sei whales as well as loggerhead, leatherback, Kemp's ridley, and green sea turtles to become endangered or threatened and may continue to place the 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 CFR 402.02). The present section of this Opinion applies that definition by examining the effects of the proposed action in the context of information presented in the status of the species, environmental baseline, and cumulative effects sections to determine: (a) If the effects of the proposed action would be expected to reduce the reproduction, numbers, or distribution of the previously listed cetaceans and sea turtles, and (b) if any reduction in the reproduction, numbers, or distribution of the previously listed cetaceans and sea turtles causes an appreciable reduction in the species' likelihood of surviving and recovering in the wild.

7.1 Integration and Synthesis of Effects on Cetaceans and Sea Turtles

This Opinion has identified in Section 6 (*Effects of the Action*) that the proposed action--continued operation of the fishery under the Monkfish FMP may directly affect right, humpback, fin and sei whales as a result of entanglement in sink gillnet gear fished in the monkfish fishery. No other direct or indirect effects to ESA-listed cetaceans are expected as a result of the activity. This Opinion has also identified that the proposed action may directly affect loggerhead, leatherback, Kemp's ridley, and green sea turtles as a result of capture in bottom otter trawl gear and gillnet gear used in the monkfish fishery. No other direct or indirect effects to ESA-listed sea turtles are expected as a result of this activity. The following discussion in Sections 7.1.1 through 7.1.7 below provide NMFS' determinations of whether there is a reasonable expectation that right, humpback, fin, and sei whales as well as 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. It is important to consider that the assessments in Sections 7.1.1 through 7.1.7 are based on historical data and do not fully account for the trend in reduction of effort in the monkfish fishery and other fisheries. Thus, the assessments in these Sections could be considered worst case expectations as the relatively recent and proposed future reductions in commercial fisheries effort should result in decreased opportunities for entanglements and captures of ESA-listed species.

7.1.1 North Atlantic Right Whale

As described in the Status of Species section of this Opinion, for 2003-2007, the average reported mortality and serious injury to right whales due to fishery entanglement was 0.8 whales per year (U.S. waters, 0.6; Canadian waters, 0.2) (Waring *et al.* 2009). In the majority of cases an entanglement report does not contain the necessary information to assign the event to a particular fishery. From 1999-2008, gillnet gear of U.S. or undocumented origin was recorded in six (6) SI/M entanglement events with right and humpback whales (Table 1). Of those six (6) events, sink gillnet gear was verified to be involved with the entanglement of one (1) right whale.

For the purposes of this assessment, we are assuming that one (1) right whale is seriously injured or killed each year as a result of U.S. fisheries. Because the serious injury or mortality could happen from the monkfish fishery, our assessment for this Opinion assumes that the serious injury or morality could and would occur as a result of the monkfish fishery.

As described in the Status of Species section of this Opinion, the latest final stock assessment report indicates that the population of North Atlantic right whales has grown at a rate of 1.8 percent over the 1990-2005 time period (Waring *et al.* 2009). In order to assess the impact of fisheries mortality on the North Atlantic right whale population, NMFS Northeast Fisheries Science Center (NEFSC) developed a population viability analysis (PVA) to examine the influence of anthropogenic mortality reduction on survival and recovery for the species (Pace, in review). The PVA included simulation models that re-sampled from observed calving records and a set of survival rates estimated from re-sightings histories of cataloged individuals collected over a 28 year period, and used these to assess the influence that simple and per capita reductions in anthropogenic mortality might have on population trajectories. Status quo simulations project forward assuming conditions are similar to those experienced from 1997 to 2006 – i.e., without any reductions in mortality from entanglements or ship strikes, continuing the observed population trends experienced over the past 28 year period into the future. Basically, the PVA evaluated how the populations would fare without entanglement mortalities compared to the status quo (i.e., with entanglement mortalities). The PVA evaluated several scenarios, including removing the mortality of one (1) right whale (random life stage and sex) per year and one (1) adult female per year. The PVA also evaluated the removal of right whale mortality on a per capita basis (meaning that as the population went up or down, the mortality reduction would go up or down relative to the population size). The three per capita scenarios evaluated the effect of the removal of the mortality of one (1) animal (random life stage and sex), one (1) adult female, and three animals (random life stage and sex).

The entire PVA is attached as an appendix to this Opinion, but some of the relevant results can be summarized as follows:

- Median overall growth rates for the simulated populations ranged from 1.30% for status quo conditions to 2.10% for reductions in mortality equivalent to three animals per year.
- Status quo projections suggest a zero probability of extinction. No extinctions or quasi-extinctions were observed in the 1000 projections.
- Only 2 of 1000 projections (with status quo simulation over a 100 year period) ended with a smaller total population size than they started with (345) after the 100 year projection, and those were just marginally smaller.
- The status quo showed an 8.6% probability of achieving a 2.0% growth rate over the next 35 years. With one (1) less mortality per year, that probability went up to 14.7%, with one (1) less adult female mortality per year, the probability improved to 24.6%.

Effects on Survival and Recovery

In the NMFS/U.S. Fish and Wildlife Section 7 Handbook, *Survival* is defined as follows:

> For determination of jeopardy/adverse modification: the species' persistence as listed or as a recovery unit, beyond the conditions leading to its endangerment, with sufficient resilience to allow for the potential recovery from endangerment. Said another way, survival is the condition in which a species continues to exist into the future while retaining the potential for recovery. This condition is characterized by a species with a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, which exists in an environment providing all requirements for completion of the species' entire life cycle, including reproduction, sustenance, and shelter.

Recovery is defined as:

Improvement in the status of listed species to the point at which listing is no longer appropriate under the criteria set out in Section 4(a)(1) of the Act.

The modeling done by Pace (in review) indicates that under the status quo there is a zero probability of the North Atlantic right whale going extinct, or reaching a quasi-extinction level. Regarding quasi-extinction for North Atlantic right whales, the criteria for quasi-extinction, i.e., population numbers, structure and trends, have not yet been developed. 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). The population analysis conducted by Pace (in review) used a quasi-extinction level of 50 adult female right whales, for the following reasons: (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), and (2) the International Union for Conservation of Nature (IUCN)(Reilly *et al.* 2008b) considers this to be one of the two threshold numerical values for a "critically endangered" population category (IUCN 2008). IUCN uses 250 mature animals as an alternative threshold value for "critically endangered" populations when there is evidence of a population decline. Given the population increase currently observed for the species (1.8% increase from 1990-2005, or as Pace (in review) found 1.3% based on the parameters and time series in his model), it is reasonable to use 50 as the threshold value for quasi-extinction. As described above, using 50 adult females as the quasi-extinction threshold, Pace (in review) found a zero percent chance of quasi-extinction for North Atlantic right whales over the next 100 years, both including and excluding the serious injuries and mortalities assumed to be occurring due to entanglements in U.S. fishing gear.

These models assume that conditions experienced in the future will be similar to conditions experienced in the past. Over the last 30 years there have been periods of very low calving rates. Recent information indicates that the periods of low calving rates may be associated with periods of lower availability of copepods in suitable densities for feeding. We are limited in our ability to influence and manage copepod density, and if copepod densities were to decrease (perhaps due to climate change, pollution, or other factors), this could negatively affect the ability of the populations to successfully reproduce.

The goal of the 2005 revised Recovery Plan for North Atlantic Right Whale is to recover North Atlantic right whales to a level sufficient to warrant their removal from the List of Endangered and Threatened Wildlife and Plants under the ESA. The intermediate goal is to reclassify the species from endangered to threatened. The revised Recovery Plan states that North Atlantic right whales may be considered for *reclassifying to threatened* when all of the following have been met: 1) The population ecology (range, distribution, age structure, and gender ratios, etc.) and vital rates (age-specific survival, age-specific reproduction, and lifetime reproductive success) of right whales are indicative of an increasing population; 2) The population has increased for a period of 35 years at an average rate of increase equal to or greater than 2% per year; 3) None of the known threats to North Atlantic right whales (summarized in the five listing factors) are known to limit the population's growth rate; and 4) Given current and projected threats and environmental conditions, the right whale population has no more than a 1% chance of quasi-extinction in 100 years.

The revised Recovery Plan for North Atlantic Right Whales states that the most significant need for North Atlantic right whale recovery is to reduce or eliminate deaths and injuries from anthropogenic activities, namely shipping and commercial fishing operations. As described in this Opinion, there are numerous management and regulatory initiatives implemented and underway to meet this need. Several significant management measures have been implemented recently, and their effects would not yet be expected to be seen in the population in terms of an increased population growth rate. Two of the more significant measures designed to reduce the risk from these anthropogenic activities are the implementation of the ALWTRP measures in 2009 (e.g., broad based gear modifications requiring the use of sinking groundlines for gillnet and pot/trap gear) and the Ship Strike Reduction Program, including the 2008 regulations requiring large ships to reduce speeds to ten knots in areas where right whales feed and reproduce, as well as along migratory routes. Any positive impacts on right whales from these measures would not be observed for some time in the population, and were not assumed in the model developed by Pace (in review), nor are they included in the latest stock assessment report (Waring et al 2009). Another, significant event that has taken place over the last decade is the reduction in fishing capacity and effort in U.S. Atlantic fisheries. For example, effort in the Northeast multispecies fisheries as a result of Amendment 16 is expected to be reduced by nearly 75% when compared to fishing effort and capacity in the early 1990's (NEFMC 2009). While some fishing effort may increase in the future as fisheries stocks respond to management measures to rebuild them, there are measures in place that will prevent overcapacity from redeveloping (i.e., nearly all U.S. Atlantic commercial fisheries are closed/limited access). Furthermore, as fish stocks increase, another possible outcome will be increased catches/landings with constant or even reduced fishing effort.

As stated previously, the most recent groundline regulations under the ALWTRP and the ship strike measures have not been in place long enough for there to be an opportunity to detect and evaluate their effect on the population of North Atlantic right whales. Similarly, the projections produced by the PVA conducted by Pace (in review), because it uses conditions experienced during the December 1, 1979- November 30, 2005 time period to project forward, do not reflect the effects of these most recent actions.

The threshold of achieving a 2.0% growth rate over a 35 year period is a downlisting and not a recovery threshold. Downlisting criteria identify conditions which when reached indicate that the population is no longer endangered (at risk of extinction) and is more properly classified as threatened (likely to become endangered). The PVA projects a 1.3% population growth and under all scenarios modeled by Pace (in review), the North Atlantic right whale is not likely (<50% probability) to move from an endangered status to a threatened status. When one looks at the actual observed growth rate in the population (1.8%), however, the rate is closer to and is approaching the 2.0% downlisting threshold. It is important to note that the median growth rates (including under the status quo) in Pace (in review) are based on model simulations, while the population growth rate of 1.8 % in Waring et al (2009) is an observed growth rate in the population. The modeling uses a longer timeframe which incorporates years of poorer calving rates which results in more pessimistic forward projections. Decisions regarding downlisting or delisting would be made on the basis of observed growth rates rather than model projections. As stated previously, the downlisting criteria is a 2% growth rate over 35 years. The observed mean growth rate of 1.8% over a 15 year period (1990 – 2005) indicates that if the status quo continues and this growth rate is maintained, or slightly increased to 2%, the downlisting criteria will be met. The population appears to be on the correct trajectory to meet the downlisting criteria if the status quo can be maintained. Any improvements in the status quo would increase the population growth and increase the rate of recovery or decrease the time period to recovery.

Another important factor to consider is that both the observed and modeled population growth rates for the status quo do not take into account any benefits to the species as a result of the recently implemented regulations to reduce the risk of entanglement from groundlines under the ALWTRP, nor do they consider the benefits from the ship speed regulations. These actions have been implemented, but have not been in place long enough for their full beneficial effect to be realized in the population. It is anticipated that it would take at least five years after implementation to be able to detect any changes in the population as a result of these management measures. The vertical line strategy that is being developed under the ALWTRP, when implemented, would also benefit the population. While the details of the vertical line strategy are still being developed in consultation with the ALWTRT, there is a commitment by NMFS to its implementation within a given time schedule (as described in Section 4.4.4.1).

As described above and as indicated in Pace (in review), North Atlantic right whales have a very low risk (zero model projections) of going extinct or reaching quasi extinction over the next 100 years under status quo conditions, including the serious injuries and mortalities caused by U.S. fishing gear. The actual population is increasing at a rate that is approaching the growth rate targeted for downlisting (if maintained for 35 years) as identified in the species' recovery plan. It does not appear that the multispecies fishery is appreciably reducing the survival and recovery of North Atlantic right whales. The species has persisted and is projected to do so into the future and the projected and observed mean population growth for the past fifteen years provides evidence that the species has sufficient resiliency to allow for recovery from endangerment.

As mentioned in the *Status of the Species*, the Draft 2010 SAR indicates an increase in the North Atlantic right whale population size and growth rate. In addition, it is worth noting that these positive population trends have been calculated and realized without consideration of the beneficial effects of recently implemented regulations designed to reduce the risk of ship strikes and entanglement in fishing gear. Considering the likely beneficial, yet unrealized and yet to be modeled effects of these recent regulations, the population of North Atlantic right whales is likely to grow at a faster rate than that modeled by Pace (in review) and currently observed which would result in an accelerated rate of recovery.

Based on the analysis described above, the serious injury or mortality of one (1) right whale per year, as a result of fisheries entanglement is not likely to reduce appreciably the likelihood of both survival and recovery of the North Atlantic right whale population.

7.1.2 Humpback Whale

As established in the above discussions in this Opinion, the use of gillnet gear for the proposed activity is expected to adversely affect humpback whales as a result of entanglement in the gear. Entanglements of humpback whales in gillnet gear have been documented. An annual average of 0.4 SI/M events of humpbacks in gillnet gear has been documented from 1999-2008 (NMFS NERO 2010). During that same time period, the complete documented SI/M events for humpbacks in all fishing gear were 3.4 annually (NMFS NERO 2010). Another accounting of serious injury/mortality events for humpback whales from 2003-2007 indicates the annual rate of documented occurrences with all commercial fishing gear types in U.S. waters has been 2.4 (Waring *et al.* 2009). This annual rate as calculated over a five year period has remained relatively stable with the estimate in the 2008 assessment being 2.6 (covering 2002-2006) and the estimate in the 2007 assessment being 2.4 (covering 2001-2005). Levels of interactions with whales prior to 2006 were calculated through a different method, as described in Waring *et al.* (2009), and therefore are not directly comparable to post-2006 estimates.

Potential biological removal (PBR) for the Gulf of Maine humpback whale stock is 1.1 whales (Waring *et al.* 2009) which has been consistent in the 2007 through 2009 stock assessment reports. As indicated above, while the annual average rate of documented serious injury/mortality events for humpback whales attributable to gillnet gear is less than PBR (0.4 < 1.1), the overall annual rate of documented serious injury/mortality events with all commercial fishing gear types in the U.S. for humpback whales is 2.4, which exceeds the PBR value of 1.1. The term "potential biological removal level" means the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimum sustainable population. It is important to note that optimum sustainable population is a population level that is significantly higher than survival and recovery. The 2009 SAR indicates that the level of serious injuries or mortalities of Gulf of Maine humpback whales attributable to U.S. commercial fisheries is higher than the level necessary to allow for growth to the optimum sustainable population level. The next question is whether the level of take in U.S. fisheries is appreciably reducing the survival and recovery of the Gulf of Maine stock of humpback whales. If so, then we would have to determine if that appreciable reduction in survival and recovery for the Gulf of Maine stock resulted in an appreciable reduction in survival and recovery for humpback whales, which as previously noted are listed as a single global species.

According to the latest final stock assessment report, the best abundance estimate for Gulf of Maine humpback whales was 847 animals and the minimum population estimate is 549 animals. The Gulf of Maine feeding population is estimated to be increasing at a rate of 6.5% for the period 1979-1991 (Barlow and Clapham, 1997). However, using data from 1992 through 2000, the population showed a lower growth rate of 0-4% (Clapham *et al.*, 2003). A more precise estimate was not possible with available data; the lower estimate assumed a calf survival rate of 0.51 and the higher estimate was based on a calf survival rate of 0.875. The authors hypothesized that the apparent decline in growth rate during this later period could have resulted from a shift in humpback whale distribution to areas less sampled, a reduction in adult female survival, increased interbirth intervals or high mortality of first-year whales (such as off the Mid-Atlantic coast (Barco *et al.*, 2002; Clapham *et al.*, 2003). They considered reduced calf survival to be the most likely explanation and noted an apparent improvement after 1996. A subsequent study confirmed both low average reproductive rates and calf survival during much of that period (Robbins, 2007). The average estimated calf survival rate for the period 2000-2005 (0.664, 95% CI: 0.517-0.784) fell between the values assumed by Clapham *et al.* (2003), and did not include neonatal mortality prior to arrival on the feeding ground (Robbins, 2007). Regardless of the cause of lower calf survival between 1992 and 1995, Clapham *et al.* (2003) conclude that calf survival appears to have returned to near-previous levels beginning in 1996 and that it is likely that population growth is now comparable to that observed between 1979 and 1991 (6.5%). Given all of the available data, the 2009 stock assessment concludes that the Gulf of Maine humpback whale stock is steadily increasing in size. The current levels of fishery impacts to the Gulf of Maine humpback whale stock do not, therefore, appear to be causing an appreciable reduction in survival. Despite these impacts the stock is steadily increasing.

In 1992, a large scale international research collaboration called the Year of the North Atlantic Humpback Whale (YONAH) was initiated to study North Atlantic humpback whales on their principal West Indies breeding grounds and high-latitude feeding grounds (Smith *et al.*, 1999). Sampling included two years of photographic identification and biopsy sampling for genetic analysis. Results from YONAH helped to clarify the population structure of North Atlantic humpback whales by providing detailed information on exchange between breeding and feeding areas. The YONAH project also produced the first basin-wide population estimate (see 'Abundance' section). A subsequent project, MONAH (More North Atlantic Humpbacks) was conducted from 2003 to 2005 to provide a second estimate of abundance from which growth rates can be calculated. Results from this project are expected within the next year.

In 2001 and 2002, the IWC Scientific Committee conducted a Comprehensive Assessment of North Atlantic humpback whales (IWC 2002; IWC 2003). The Committee reviewed existing knowledge of this population and, through examination of whaling records and recent sighting, photographic identification and genetic information, attempted to assess the current status of humpack whales in the North Atlantic (relative to estimated pre-exploitation levels).

Based on these collaborative research efforts, a status review is currently being conducted to evaluate stock structure, distribution and abundance and threats to humpback whales globally. The results of this status review will be used to determine if any listing status change is warranted for humpback whales.

In the interim, the 2009 stock assessment also concludes that the North Atlantic population of humpback whales overall had an estimated average population increase of 3.1% over the time period 1979-1993 (Waring *et al.* 2009; Stevich *et al.* 2003). Additionally, the draft 2010 SAR reports that humpback whale population is steadily increasing (Waring *et al.* 2010). Given that U.S. commercial fishery takes are not currently threatening the survival of the Gulf of Maine stock of humpback whales, it is logical to conclude that they are not threatening the survival of the overall stock of North Atlantic humpback whales, particularly in light of the increasing population trend.

The stock assessment does conclude that human impacts (vessel collisions and entanglements) may be slowing recovery of humpback whale populations. The question for this Opinion is whether impacts associated with fishing authorized under the Monkfish FMP are likely to result in an appreciable reduction in recovery of humpback whales. The goal of the 1991 Recovery Plan for the Humpback Whale (Plan) is to assist humpback whale populations to grow and to reoccupy areas where they were historically found. The long-term numerical goal of the Plan is to increase humpback whale populations to at least 60% of the number of existing before commercial exploitation or of current environmental carrying capacity. With those levels undetermined, an intermediate goal was specified as a "doubling of extant populations within the next 20 years."

The 1991 Plan used the 1986 population estimate for the Gulf of Maine feeding aggregation of humpback whales which was 240 (95% CI = 147 to 333) (NMFS 1991b). The most recent best estimate of abundance for Gulf of Maine humpback whales is 847 animals (CV =0.55). The current minimum population estimate is 549 animals (Waring *et al.* 2009). Based on these numbers, it does appear that the Gulf of Maine stock of humpback whales has more than doubled in the past 20 years.

The Recovery Plan for Humpback Whales set out four major objectives to proceed on a path toward recovery; one of the four objectives specifically addresses fishery interactions by identifying the need to, "identify and reduce human-related mortality, injury, and disturbance," to humpback whales. As described in this Opinion, there are numerous management and regulatory initiatives implemented and underway to meet this need. Several significant management measures have been implemented recently, and their effects would not yet be expected to be seen in the population in terms of an increased population growth rate. Two of the more significant measures designed to reduce the risk from these anthropogenic activities are the implementation of the ALWTRP measures in 2009 (e.g., broad based gear modifications requiring the use of sinking groundlines for gillnet and pot/trap gear) and the Ship Strike Reduction Program, including the 2008 regulations requiring large ships to reduce speeds to ten knots in areas where right whales feed and reproduce, as well as along migratory routes. Any positive impacts on humpback whales from these measures would not be observed for some time in the population nor are they included in the latest stock assessment report.

The vertical line strategy developed under the ALWTRP, when implemented, will also benefit the population. While the details of the vertical line strategy are still being developed in consultation with the ALWTRT, there is a commitment to its implementation within a given time schedule.

As part of a large-scale assessment called More of North Atlantic Humpbacks (MoNAH) project, extensive sampling was conducted on humpbacks in the Gulf of Maine/Scotian Shelf region and the primary wintering ground on Silver Bank during 2004-2005. These data are being analyzed along with additional data from the U.S. Mid-Atlantic to estimate abundance and refine knowledge of population structure. This work is intended to update the YONAH population estimate and is being used in an ongoing status review under the ESA.

Another, significant event that has taken place over the last decade is the reduction in fishing capacity and effort in U.S. Atlantic fisheries. For example, effort in the Northeast multispecies fisheries as a result of Amendment 16 is expected to be reduced by nearly 75% when compared to fishing effort and capacity in the early 1990's (NEFMC 2009). Fishing effort in the American lobster fishery is expected to be reduced as a result of lobster trap effort control and trap transferability measures approved by the Atlantic States Marine Fisheries Commission and in evaluation by the NMFS (NMFS 2010). While some fishing effort may increase in the future as fisheries stocks respond to management measures to rebuild them, there are measures in place that will prevent overcapacity from redeveloping (i.e., nearly all U.S. Atlantic commercial fisheries are closed/limited access). Furthermore, as fish stocks increase, another possible outcome will be increased catches/landings with constant or even reduced fishing effort.

Specific downlisting criteria for humpback whales have not been developed. However, the estimated increases in the Gulf of Maine stock and the North Atlantic populations of humpback whales indicate that these populations are recovering despite continued interactions with commercial fisheries inside the U.S. EEZ. Additionally, there are indications of increasing abundance for the eastern and central North Pacific stocks (Waring *et al.* 2009).

The rate of humpback entanglements in fishing gear continues to be of concern to resource managers. The relatively new broad based gear modifications of the ALWTRP are expected to reduce the risk of SI/M due to humpback whale entanglement. The most recent data strongly suggests the humpback whale population is steadily increasing despite the anthropogenic and cumulative effects previously discussed in this Opinion. While takes of humpback whales continue to be possible under the continued authorization of the Monkfish FMP, the level of take is not expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of this species.

7.1.3 Fin and Sei Whales

Serious injury and mortality entanglements of fin and sei whales have been documented but occur at a level below PBR for both species (Waring *et al.* 2009). This indicates that the level of serious injuries or mortalities of fin and sei whales attributable to U.S. commercial fisheries still allows these stocks to maintain population levels and growth rates needed to reach or maintain their optimum sustainable population. Additionally, broad based gear modifications of the ALWTRP have been implemented, and preliminary data in the Draft 2010 SAR shows a greater and a stable population size for fin and sei whales respectively. While takes of fin and sei whales continue to be possible under the continued authorization of the Monkfish FMP, the level of take is not expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of these species.

7.1.4 Loggerhead Sea Turtle

As described above, the use of bottom otter trawl gear and gillnet gear for the proposed activity is expected to adversely affect loggerhead sea turtles as a result of being incidental taken in the gear. This Opinion has identified in Section 6.2.2 that the proposed activity, continued operation of the fishery under the Monkfish FMP, will directly affect loggerhead sea turtles by capturing up to two (2) loggerhead sea turtles in bottom otter trawl gear and another 171 loggerheads in gillnet gear. As a result of being captured in the fishing gear, up to 70 of the 173 loggerhead sea turtles captured annually are expected to die or sustain serious injuries to death or failure to reproduce. The towing of trawl gear on benthic habitat, and the temporary removal of loggerhead prey from the environment (which may be returned to the water alive or dead) as a result of the fishing activities will have an insignificant effect on loggerhead sea turtles, as discussed in Section 4.1.1. No other direct or indirect effects to loggerhead sea turtles are expected as a result of the proposed action.

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

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

The lethal take of 70 loggerhead sea turtles from the Atlantic every year will reduce the number of loggerhead sea turtles as compared to the number that would have been present in the absence of the proposed action (assuming all other variables remained the same). Assuming some or all of those 70 lethal takes are females, the loss of female loggerhead sea turtles as a result of the proposed action is expected to reduce the reproduction of loggerheads in the Atlantic compared to the reproductive output of Atlantic loggerheads in the absence of the proposed action. These losses are relevant to the Demographic Recovery Criteria for nests and nesting females. Nesting data demonstrate recent declines in the number of nests laid for most of the Northwest Atlantic recovery units. 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 (letter to J. Lecky, NMFS Office of Protected Resources, from N. Thompson, NMFS Northeast Fisheries Science Center, December 4, 2007; NMFS and USFWS 2008).

As previously stated, loggerheads exist as five subpopulations in the western Atlantic, recognized as recovery units in the 2008 recovery plan for this species, that show limited evidence of interbreeding. The 2008 recovery plan compiled the most recent information on mean number of loggerhead nests and the approximated counts of nesting females per year for four of the five identified recovery units (i.e., nesting groups). They are: (1) for the NRU, a mean of 5,215 loggerhead nests per year with approximately 1,272 females nesting per year; (2) for the PFRU, a mean of 64,513 nests per year with approximately 15,735 females nesting per year; (3) for the DTRU, a mean of 246 nests per year with approximately 60 females nesting per year; and (4) for the NGMRU, a mean of 906 nests per year with approximately 221 females nesting per year. For the GCRU, the only estimate available for the number of loggerhead nests per year is from Quintana Roo, Yucatan, Mexico, where a range of 903-2,331 nests per year was estimated from 1987-2001 (NMFS and USFWS 2007a). There are no annual nest estimates available for the Yucatan since 2001 or for any other regions in the GCRU, nor are there any estimates of the number of nesting females per year for any nesting assemblage in this recovery unit; however, the 2008 recovery plan indicates that the Yucatan nesting aggregation has at least 1,000 nesting females annually. It should be noted here, and it is explained further below, that the above numbers include nesting females (i.e., do not include non-nesting adult females, adult males, or juvenile males or females in the population).

It is likely that the sea turtles taken in the monkfish fishery originate from several of the recovery units. Limited information is available on the genetic makeup of sea turtles in the Mid-Atlantic.

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

Based on the genetic analysis presented in Bass et al. (2004) and the small number of loggerheads from the DTRU or the NGMRU likely to occur in the action area it is extremely unlikely that any of the up to 70 loggerheads that are likely to be seriously injured or killed due to monkfish fishing operations are likely to have originated from either of these recovery units. The majority, at least 80% of the loggerheads seriously injured or killed, are likely to have originated from the PFRU, with the remainder from the NRU and GCRU. As such, approximately 56 of the sea turtles are expected to be from the PFRU, 9 from the NRU and 5 from the GCRU. The best available information indicates the likelihood of the take being from a particular recovery unit is consistent with the relative sizes of the nesting colonies/recovery units, and we conclude, based on the available evidence, that none of the recovery units are disproportionately impacted by the take in the fisheries for monkfish. Therefore, our discussion of the impacts of the monkfish fishery will focus on the overall western North Atlantic population of loggerhead sea turtles, which comprises these recovery units.

In determining whether the continued authorization of the monkfish fishery would reduce appreciably the likelihood of survival and recovery of loggerhead sea turtles, NMFS has considered the population viability analysis (PVA) for loggerhead sea turtles based on the impacts of the Atlantic sea scallop fishery (Merrick and Haas 2008). The PVA is similar to 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 2005b; Snover 2005). The PVA used to assess the effect of the continued authorization of the Atlantic sea scallop fishery and the Hawaii deep-set pelagic longline fishery on ESA-listed turtles in the Pacific assessed the female portion of the populations, only. 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 the Biological Opinion for the Atlantic Sea Scallop FMP (NMFS 2009c), 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;
- used 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,
- used 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 were not available to develop a PVA model for each of the nesting groups. Second, it was unclear how PVA outputs at a nesting group level could have been reconciled to assess the effects of a 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 were 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, 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 would 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 interannual 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 an action, 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 differences between the projected extinction risk with and without the takes from the Monkfish 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, any real effects of the fishery on quasi-extinction of adult female loggerhead sea turtles in the Atlantic are more likely to be discovered 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 Atlantic sea scallop fishery PVA did not address loggerheads that nest in Greece, Turkey and Brazil since the PVA was performed 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. In short, the PVA established a baseline using the rate of change of the adult female population (which implicitly included the mortalities from the scallop 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 re-running the PVA. The results of these two analyses were then compared. Values for inputs were used throughout such that the PVA would have been 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 Atlantic sea scallop fishery, resulting in mortalities of loggerhead sea turtles, would not have an appreciable effect on the number of adult female loggerhead sea turtles in the western Atlantic over a future 100 years. While a statistically significant difference was detected in the number of iterations out of 1,000 with quasi-extinction probabilities at 100 years greater than 5%, the differences smoothed out over the 1,000 iterations and, taken together, the probability of quasi-extinction at 100 years 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, the PVA results indicated loggerhead sea turtles in the western Atlantic would not go extinct within the future 100 years regardless of the continued authorization of the scallop fishery.

The PVA demonstrated that the continued authorization of the Atlantic sea 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. The annual sea scallop fishery bycatch mortality of adult female loggerheads was estimated to be 102 turtles (Merrick and Haas 2008). The annual monkfish fishery bycatch of loggerhead sea turtles is estimated to be up to 173 individuals, resulting in 70 mortalities, which includes both male and female individuals, as well as juveniles and adults. The adult female equivalent of the 70 total mortalities has not been calculated, but assuming that approximately half of the takes are females, and that some portion of the takes are juveniles, the number of adult female equivalent mortalities is less than half of 70, and thus less than 35% of the 102 adult female equivalent mortalities estimated for the Atlantic sea scallop fishery.

Stranded loggerhead turtles have not been attributed to any operations of the multispecies fishery. However, stranded turtles are rarely attributed to any particular fishery due to lack of information. Therefore, NMFS anticipates the continued authorization of the monkfish fishery will have an insignificant effect on the trends in strandings relative to in-water abundance as listed in Demographic Criteria #3 of the 2008 recovery plan.

It is unclear whether nesting beach trends, in-water abundance trends, or some combination of both, best represents the actual status of loggerhead sea turtle populations in the Atlantic. Regardless, we believe the proposed action will not have a measurable negative effect on either of these trends. Estimates of the total loggerhead population in the Atlantic are not currently available. However, the 1998 TEWG report estimated the total loggerhead population of benthic individuals in U.S. waters – a subset of the whole Western Atlantic population – at over 200,000. Also, a recent loggerhead assessment prepared by NMFS states that the loggerhead adult female population in the western North Atlantic ranges from 20,000 to 40,000 or more, with a 95% CI of 18,333-68,192 individuals (NMFS SEFSC 2009). Although there is much uncertainty in these population estimates, they provide some context for evaluating the size of the likely population of loggerheads in the Atlantic. Assuming that half the loggerheads taken in the fishery are females, and assuming that all the takes are of adults to assume a worst case scenario as far as reproductive value to the population, the loggerhead mortality as a result of the monkfish fishery would result in the removal of 0.1 percent of the adult female loggerhead population in the Western Atlantic (35 out of 18,333, using the low end of the 95% CI from NMFS SEFSC 2009).

In general, while the loss of a small number of individuals from a subpopulation or species may have an appreciable reduction on the numbers, reproduction and distribution of the species, this is likely to occur only when there are very few individuals in a population, the individuals occur in a very limited geographic range or the species has extremely low levels of genetic diversity. This situation is not likely in the case of loggerhead sea turtles because: the species is widely geographically distributed, it is not known to have low levels of genetic diversity, and there are several thousand individuals in the population and subpopulations.

Scaled against the likely size of the population and the magnitude of the trends noted above, NMFS does not believe the level of lethal take projected annually from the continued authorization of the Monkfish FMP (70 individuals) will have an appreciable reduction in the Northwest Atlantic or worldwide population. Therefore, the loss of up to 70 individuals per year is unlikely to cause an appreciable reduction in the species' likelihood of survival and recovery.

This conclusion is supported by comparing the impacts of the Monkfish fishery to that of the scallop fishery. The PVA done for the scallop fishery, as described above, demonstrated that the continued authorization of that fishery would not appreciably reduce the number of adult females in the western Atlantic. The operation of the Monkfish FMP is estimated to result in the mortality of less than 35 percent of adult female equivalents compared to the scallop fishery.

The above information also supports the conclusion that continued authorization of the monkfish fishery will have an insignificant effect on the number of nests and number of nesting females as listed in Demographic Criteria #1 of the 2008 recovery plan for loggerhead sea turtles in the

Atlantic, as referenced earlier in this section. Likewise, this information supports the conclusion that the continued authorization of the monkfish fishery will have an insignificant effect on the trends in abundance on foraging grounds as listed in Demographic Criteria #2 of the 2008 recovery plan.

The Listing Factor Recovery Criteria contained in the recovery plan include programs and strategies that should be implemented to respond to the following five listing factors that have caused loggerheads to be listed as a threatened species under the ESA: (1) 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) inadequacy of existing regulatory mechanisms, and (5) other natural or manmade factors affecting its continued existence. These programs involve both terrestrial and marine components (NMFS and USFWS 2008).

As described above and elsewhere in this Opinion, the continued operation of the monkfish fishery is expected to harass, injure, or kill loggerhead sea turtles as a result of physical contact between the sea turtles and the fishing gear. No other effects to loggerhead sea turtles are expected as a result of the proposed action. The continued operation of the fishery will not affect the protection of nests, nesting beaches, and the marine environment nor will it compromise the ability of researchers to conduct scientific studies or management officials to enact peer-review strategies or legislative policy. Therefore, the continued operation of the monkfish fishery within the constraints of the current Monkfish FMP will have no appreciable reduction in the ability to achieve the Listing Factor Recovery Criteria.

7.1.5 Leatherback Sea Turtle

In the time period 2000-2009 there were three (3) confirmed interactions with leatherback sea turtles and bottom otter trawl gear in the Mid-Atlantic (NEFSC FSB database). Additionally, from 2000-2009 there have been three (3) confirmed interactions of leatherback sea turtles with Mid-Atlantic sink gillnet gear, (NEFSC FSB database). Turtle captures in gillnet gear were not observed in the Northeast region (east of Cape Cod and in the Gulf of Maine) during the 1995-2006 time period. In the Murray (2008) report the Mid-Atlantic region is described as the region from the shoreline below 41°30'N66°W to ~35°N/75°30'W.

Takes of leatherback sea turtles in the monkfish fishery are reasonably likely to occur given: (1) that the distribution of leatherbacks overlaps with operation of monkfish fishery, and (2) takes of leatherback sea turtles in bottom otter trawl gear and gillnet gear have been observed during commercial fishing trips operating in Mid-Atlantic waters. Based on observer data, the capture of leatherback sea turtles in any gear (fixed or mobile) operating within the action area, including monkfish gear, would be a rare event. However, given the low percentage of trips with observer coverage in the monkfish fishery as well as other fisheries in the action area, it is likely that some interactions have occurred but were not observed or reported.

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 monkfish trawl gear could result in death due to forced submergence, given that there are no regulations that control tow-times in the monkfish fishery and some trawl tows that have been observed to capture sea turtles have exceeded one hour in duration (NEFSC FSB database). As described in Section 6.2.2, NMFS anticipates the annual non-lethal or lethal take of up to four (4) leatherback sea turtles.

Nest counts in many areas of the Atlantic Ocean show increasing trends, including for beaches in Suriname and French Guiana which support the majority of leatherback nesting (NMFS and USFWS 2007b) 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*, *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 the U.S. South Atlantic and Gulf of Mexico shrimp fisheries (measures implemented in 2002). 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 four leatherback sea turtles annually in the Atlantic as a result of the continued operation of the monkfish 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 monkfish fishery has no effects on leatherback sea turtles that occur outside of the Atlantic. Therefore, since the continued operation of the monkfish 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. Virgin Islands), and along the East coast of Florida; (2) nesting habitat encompassing at least 75% of nesting activity in Puerto Rico, U.S. Virgin Islands, and Florida is in public ownership; and (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 operation of the monkfish fishery is expected to kill up to four leatherback sea turtles annually. No other effects to leatherbacks are expected as a result of the proposed action. The continued operation of the 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 operation of the monkfish fishery within the constraints of the Monkfish FMP will have no effect on recovery criteria #2 and #3.

The lethal take of up to four (4) 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 leatherbacks are females. 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 nine (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 St. Croix, U.S. Virgin Islands, 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 up to four leatherback sea turtles, together with an increase in nesting, is not expected to affect the positive growth rate in the female population of leatherback sea turtles nesting in Puerto Rico, St. Croix, and Florida. Therefore, the continued operation of the monkfish fishery within the constraints of the current Monkfish FMP will not appreciably reduce the likelihood of recovery for leatherback sea turtles in the Atlantic. Since the monkfish fishery has no effects on leatherback sea turtles that occur outside of the Atlantic, its continued operation will not appreciably reduce the likelihood of recovery for the species.

7.1.6 Kemp's ridley Sea Turtle

In the time period 2000-2009 there were two (2) confirmed interactions with Kemp's ridley sea turtles and bottom otter trawl gear in the Mid-Atlantic (NEFSC FSB database). Additionally, from 2000-2009 there have been eight (8) confirmed interactions of Kemp's ridley sea turtles with sink gillnet gear (NEFSC FSB database). Turtle captures in gillnet gear were not observed in the Northeast region (east of Cape Cod and in the Gulf of Maine) during the 2000-2009 time period despite substantial observer coverage of the gillnet fishery in the Northeast region.

Takes of Kemp's ridley sea turtles in the monkfish fishery are reasonably likely to occur given: (1) that the distribution of Kemp's ridleys overlaps seasonally with operation of the monkfish fishery, and (2) takes of Kemp's ridley sea turtles captured in bottom otter trawl gear and gillnet gear have been observed during commercial fishing trips operating in Mid-Atlantic waters. Based on recent observer data, the capture of Kemp's ridley sea turtles in any gear (fixed or mobile) operating within the action area, including monkfish gear, would be a rare event. However, given the low percentage of trips with observer coverage in the monkfish fishery as well as other fisheries in the action area, it is likely that some interactions that have occurred were not observed or reported. As described in Section 6.2.2, NMFS anticipates the annual non-lethal or lethal take of up to four (4) Kemp's ridley sea turtles.

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 trawl gear could result in death due to forced submergence, given that there are no regulatory controls on tow-times in the Monkfish FMP and some trawl tows that have been observed to take sea turtles have exceeded one hour in duration (NEFSC FSB database). 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.

The lethal removal of up to four (4) 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 four (4) 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. The lethal removal of up to four (4) Kemp's ridley sea turtles annually as a result of the continued operation of the monkfish fishery under the Monkfish FMP 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*, *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 four (4) Kemp's ridley sea turtle annually as a result of the continued operation of the monkfish fishery will not appreciably reduce the likelihood of survival for Kemp's ridley sea turtles given both the increased nesting trend and ongoing measures that reduce the number of Kemp's ridley sea turtles injured and killed (which should result in increases to the numbers of Kemp's ridley sea turtles that would not have occurred in the absence of those regulatory measures).

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, or (5) other natural or manmade factors affecting its continued existence. NMFS is using these factors to assess whether the continued operation of the monkfish fishery within the constraints of the Monkfish FMP 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 set out in section 4(a)(1) of the ESA (50 CFR 402.02). As described in this Opinion, the continued operation of the monkfish fishery is expected to kill up to four (4) Kemp's ridley sea turtles annually. No other effects to Kemp's ridley sea turtles, such as on habitat, or due to disease, predation, and other natural influences on survival, are expected as a result of the proposed action. The loss of four (4) Kemp's ridleys annually is not expected to modify, curtail, or destroy their range. The monkfish fishery does not utilize Kemp's ridleys for recreational, scientific, or commercial purposes. Adequate regulatory mechanisms are in place to protect Kemp's ridley sea turtles. Therefore, the continued operation of the monkfish fishery within the constraints of the current Monkfish FMP will have no effect on ESA listing criteria #1 through #4.

The lethal taking of up to four (4) Kemp's ridley sea turtles annually in the monkfish 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 continued loss of up to four (4) Kemp's ridleys annually is not expected to change the trend in increased nesting. With an increasing trend, the loss of four (4) Kemp's ridleys will not compromise the continued existence of the species, which is the focus of the listing factor #5. Therefore, the continued operation of the monkfish fishery within the constraints of the current Monkfish FMP will not appreciably reduce the likelihood of recovery for the species.

7.1.7 Green Sea Turtle

There has been one (1) observed take of a green sea turtle in bottom otter trawl gear in the Northeast and Mid-Atlantic regions during the period 2000-2009 (NEFSC FSB database). During 2000-2009, three (3) green turtles were observed incidentally caught in Mid-Atlantic sink gillnet gear within the action area (NEFSC FSB database).

The distribution of green sea turtles overlaps seasonally with the use of monkfish gear. Based on observer data, the capture of green sea turtles in any gear (fixed or mobile) operating within the action area, including monkfish gear, would be a rare event. However, given the low percentage of trips with observer coverage in the monkfish fishery as well as other fisheries in the action area, it is likely that some interactions have occurred but were not observed or reported. Based on the average of the number of takes per year in gear capable of catching monkfish for the period 2000-2009, the take of up to five (5) green sea turtles in monkfish gear is anticipated to occur annually as a result of the continued operation of the monkfish fishery within the constraints of the Monkfish FMP. It is assumed that there is an equal chance of lethally taking a male or female green sea turtle since available information suggests that both sexes occur in the action area.

The lethal removal of up to five (5) 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 five (5) 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. The lethal removal of up to five (5) green sea turtles annually from the Atlantic as a result of the continued operation of the monkfish 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 1970s 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*, *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 five (5) green sea turtle annually in the Atlantic as a result of the continued operation of the monkfish fishery under the Monkfish FMP 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 monkfish fishery has no effects on green sea turtles that occur outside of the Atlantic. Therefore, since the continued operation of the monkfish 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). The recovery criteria state that the U.S. population of green sea 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 1991). As described in this Opinion, the continued operation of the monkfish fishery is expected to kill up to five (5) Atlantic green sea turtles annually. No other effects to green sea turtles are expected as a result of the proposed action. The continued operation of the monkfish 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 operation of the monkfish fishery within the constraints of the current Monkfish FMP will have no effect on recovery criteria #2 and #4.

The lethal taking of up to five (5) green sea turtles annually in the monkfish 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 five (5) green sea turtle 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 2,084%, 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.* (2005) documented the capture of more than twice as many green sea turtles in 2003 and 2004 with less pound net gear fished, compared to the number of green sea turtles captured in pound net gear in the area during the 1990s. Yet other studies have found no difference in the abundance (decreasing or increasing) of green sea turtles on foraging grounds in the Atlantic (Bjorndal *et al.* 2005; Epperly *et al.* 2007). The annual loss of five (5) green sea turtles, together with an increase in nesting, is not expected to materially affect the increasing to stable trend in the number of green sea turtles on the foraging grounds in the Atlantic. Therefore, the continued operation of the monkfish fishery will not appreciably reduce the likelihood of recovery for green sea turtles in the Atlantic. Since the monkfish fishery has no effects on green sea turtles that occur outside of the Atlantic, the continued operation of the monkfish fishery within the constraints of the current Monkfish FMP will not appreciably reduce the likelihood of recovery for the species.

8.0 CONCLUSION

After reviewing the current status of right, humpback, fin, and sei whales as well as loggerhead, leatherback, Kemp's ridley, and green sea turtles, the environmental baseline and cumulative effects in the action area, the effects of the continued operation of the Monkfish FMP, in compliance with the requirements of the ALWTRP, it is NMFS' biological opinion that the proposed activity is likely to adversely affect, but not jeopardize the continued existence of these species.

Proposed Rule to List Loggerhead Sea Turtles

As explained in *Status of Affected Species* section of this Opinion, on March 16, 2010, NMFS published a proposed rule to list two distinct population segments of loggerhead sea turtles as threatened and seven distinct population segments of loggerhead sea turtles as endangered. This rule, when finalized, would replace the existing listing for loggerhead sea turtles. Currently, the species is listed as threatened range-wide. Once a species is proposed for listing, the conference provisions of the ESA apply. As stated at 50 CFR 402.10, "Federal agencies are required to confer with NMFS on any action which is likely to jeopardize the continued existence of any proposed species or result in the destruction or adverse modification of proposed critical habitat. The conference is designed to assist the Federal agency and any applicant in identifying and resolving potential conflicts at an early stage in the planning process."

As described in this Opinion, the proposed action is anticipated to result in the death of no more than 49 loggerhead sea turtles on an annual basis. In this Opinion, NMFS concludes that this level of take is not likely to reduce appreciably the likelihood of both the survival and recovery of the species in the wild by reducing the reproduction, numbers, or distribution of that species and that, therefore, the action is not likely to jeopardize the continued existence of loggerhead sea turtles.

As explained in the Opinion, the takes and mortalities caused by the proposed action are all likely to fall within the Northwest Atlantic DPS, one of the seven DPSs proposed to be listed as endangered in the March 16, 2010 proposed rule.

In this Opinion, NMFS determined that the loss of these individuals would not be detectable at the population (Western North Atlantic) level or at the species as whole (i.e., range-wide) and that the death of up to 46 loggerhead sea turtles each year as a result of the continued operation of the monkfish fishery will not appreciably reduce the likelihood of survival (i.e., it will not increase the risk of extinction faced by this species) or recovery for loggerhead sea turtles. As explained in the Opinion, the individuals likely to be killed represent .001 percent of the adult females in the Northwest Atlantic. The proposed Northwest Atlantic DPS is roughly equivalent to the Northwest Atlantic population, as defined in the Recovery Plan. Thus, the individuals likely to be killed represent no more than 0.001% of the adult female loggerhead sea turtles in the proposed Northwest Atlantic DPS. In this Opinion NMFS determines that the loss of these individuals from the population (as defined in the Recovery Plan) was likely to be undetectable; as such, and given that the proposed DPS is roughly equivalent, it is reasonable to expect that the conclusions reached for the Northwest Atlantic population and current range-wide listing would be the same as for the proposed Northwest Atlantic DPS. Conference is only required when an action is likely to jeopardize the continued existence of any proposed species, and, based on the above information, it is unlikely that the effects of the proposed action would result in jeopardy for the proposed Northwest Atlantic DPS. Thus, conference is not required for this proposed action. Additionally, an ITS included with this Opinion contains all terms and conditions and reasonable and prudent measures necessary and appropriate to minimize and monitor take of loggerhead sea turtles, it is unlikely that a conference would identify or resolve additional conflicts or provide additional means to minimize or monitor take of loggerhead sea turtles.

9.0 INCIDENTAL TAKE STATEMENT

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 monkfish fishery, estimates of sea turtle take in gear used in the monkfish fishery, and the distribution and abundance of turtles in the action area, NMFS anticipates that the continued implementation of the Monkfish FMP, may result in the taking of sea turtles as follows[11]:

- for loggerhead sea turtles, NMFS anticipates (a) the annual take of up to 171 individuals over a 5-year average in gillnet gear, of which up to 69 per year may be lethal and (b) the

[11] For sea turtle species other than loggerheads, the estimated observed take is for combined gear type. Effort within the fishery may shift from year to year between gear types and therefore we believe it is most appropriate to have a total estimated observed take number. Because we have a small sample size for these observed takes, we are not including an estimate of how many of these takes are likely to result in serious injury or mortality. Instead, we are assuming that all of them could be mortalities. For loggerhead sea turtles, the incidental take statement includes estimates separately for trawls and for gillnets. This is due to the fact that the take estimates for the gear types are currently calculated differently. The trawl estimate is provided as a point estimate where as the gillnet estimate is a range. Since these are calculated differently, we are not combining them in the incidental take statement, but we would intend to do so in the future if the format of the estimates was similar across gear types.

annual take of up to two (2) individual over a 5-year average in trawl gear, of which up to one (1) per year may be lethal;

- for leatherback sea turtles, NMFS anticipates the annual lethal or non-lethal take of up to four (4) individuals in trawl gear and gillnet gear combined;
- for Kemp's ridley sea turtles, NMFS anticipates the annual lethal or non-lethal take of up to four (4) individual in trawl gear and gillnet gear combined;
- for green sea turtles, NMFS anticipates the annual lethal or non-lethal take of up to five (5) individuals in trawl gear gillnet gear combined;

NMFS is not including an incidental take authorization for right, humpback, fin, and sei whales at this time because the incidental take of ESA-listed whales has not been authorized under section 101(a)(5) of the MMPA. Following the issuance of such authorizations, NMFS may amend this Opinion to include an incidental take allowance for these species, as appropriate.

Anticipated Impact of Incidental Take

NMFS has concluded that the continued operation of the monkfish 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 monkfish 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.

Reasonable and Prudent Measures

NMFS has determined that the following RPMs are necessary or appropriate to minimize impacts of the incidental take of sea turtles in the monkfish fishery:

1. NMFS must seek to ensure that any sea turtles incidentally taken in monkfish fishing gear are handled in such a way as to minimize stress to the animal and increase its survival rate.

2. NMFS must seek to ensure that monitoring and reporting of any sea turtles encountered in monkfish fishing gear: (1) detects any adverse effects such as injury or mortality; (2) assesses the realized level of incidental take in comparison with the anticipated incidental take documented in this Opinion; (3) detects whether the anticipated level of take has occurred or been exceeded; and (4) collects data from individual encounters.

3. NMFS must continue to investigate and implement, within a reasonable time frame following sound research, gear modifications for gear used in the monkfish fishery to reduce incidental takes of sea turtles and/or the severity of the interactions that occur.

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

Terms and Conditions

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

1. To comply with RPM #1 above, NMFS must distribute information to monkfish permit holders specifying handling or resuscitation requirements fishermen must undertake for any sea turtles taken. At a minimum, handling and resuscitation requirements listed in 50 CFR 223.206(d)(1) must be implemented. NMFS must also distribute the NER STDN Disentanglement Guidelines to monkfish permit holders. Use of the sea turtle handling and release protocols described in Epperly *et al.* (2004) and NMFS SEFSC (2008) should also be considered. Implementation of these requirements must occur as soon as operationally feasible and no later than March 31, 2011.

2. To also comply with RPM #1 above, NMFS must develop and implement an outreach program to train commercial fishermen in the use of any sea turtle release equipment and/or sea turtle handling protocols and guidelines implemented. The Northeast Fisheries Observer Program (NEFOP) has acknowledged that they would be willing to help with this initiative. In developing and implementing this outreach program, the HMS pelagic longline educational outreach program should be used as a model. The outreach program must be implemented in conjunction with term and condition # 1.

3. To comply with RPM #2 above, NMFS will continue to ensure that there is adequate observer coverage in Mid-Atlantic trawl, dredge and gillnet fisheries to document and estimate incidental bycatch of loggerhead sea turtles. Monthly summaries and an annual report of observed sea turtle takes in New England and Mid-Atlantic fisheries, including trips where monkfish are landed, should continue to be provided to the NERO Protected Resources Division.

4. To also comply with RPM #2 above, observers must continue to tag and take tissue samples from incidentally captured sea turtles as stipulated under their ESA Section 10 permit. The current NEFOP protocols are to tag any sea turtles caught that are larger than 26 centimeters (cm) in notch-to-tip carapace length and to collect tissue samples for genetic analysis from any sea turtles caught that are larger than 25 cm in notch-to-tip carapace length. The NEFSC shall be the clearinghouse for any genetic samples taken.

5. To also comply with RPM #2 above, NMFS must continue to develop and implement sea turtle serious injury criteria for fisheries in the NE Region in order to better assess and

evaluate injuries sustained by sea turtles in fishing gear, and their potential impact on sea turtle populations.

6. Bycatch estimates need to be combined with quantitative stock assessments to provide improved understanding of how listed species are adversely affected by estimated bycatch levels. Thus, to also comply with RPM #2 above, NMFS must improve its quantitative stock assessment of incidentally caught species. A sufficient quantitative stock assessment includes, but is not limited to, an integrative modeling framework for quantitative stock assessment and the necessary fishery independent data needed to support such assessments. Progress towards this goal must be reported on annually.

7. To also comply with RPM #2 above, NMFS must require that disentanglement responders collect detailed information on the gear involved in entanglements, and submit all information on the gear to NMFS. NMFS must evaluate the gear information regarding entanglements, and produce an annual report on the entanglements that were reported in the previous year.

8. To comply with RPM #3 above, NMFS will continue to investigate modifications of trawl and gillnet fisheries and effects on sea turtles through research and development, as resources allow. Within a reasonable amount of time following completion of an experimental gear trial from or by any source, NMFS will 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.

9. To comply with RPM #4 above, NMFS must continue to review all data available on the observed/documented take of sea turtles in trawl and gillnet fisheries and other suitable information (i.e., data on observed sea turtle interactions for other fisheries, vertical line density information, sea turtle distribution information, or fishery surveys in the area where the monkfish fishery operates) to assess whether there is sufficient information to undertake any additional analysis to attempt to identify correlations with environmental conditions or other drivers of incidental take within some or all of the action area. If such additional analysis is deemed appropriate, within a reasonable amount of time after completing the review, NMFS will take appropriate action to reduce sea turtle interactions and/or their impacts.

Monitoring

NMFS must continue to monitor levels of sea turtle bycatch in the monkfish fishery. Observer coverage has been used as the principal means to estimate sea turtle bycatch in the monkfish fishery and to monitor incidental take levels. NMFS will continue to use observer coverage to monitor sea turtle bycatch in commercial net, trawl, and longline gear that is authorized by the Monkfish FMP. NMFS should also continue to support NEFOP's development of a video monitoring pilot project to evaluate its utility for various fishing gear types including bottom otter trawls and gillnets. If video monitoring proves to be a feasible supplement to observer coverage, the utility of video in identifying sea turtle bycatch events could be investigated. In the future, video could potentially be used to evaluate compliance with VTR requirements for incidentally taken sea turtles.

For the purposes of monitoring this ITS, NMFS will continue to use observer coverage as the primary means of collecting incidental take information. The loggerhead sea turtle take estimates in the Opinion were generated using statistical estimates that are not feasible to conduct on an annual basis. Conducting such statistical estimates are infeasible on an annual basis due to the data needs, length of time to develop, review, and finalize the estimates, and methodology used. As these estimates depend on take rate information over a several year period, re-examination after one year is not likely to produce any noticeable change in the take rate. For these reasons, approximately every 5 years, NMFS will re-estimate takes in the monkfish fishery using appropriate statistical methods. A new bycatch estimate for loggerhead sea turtles caught in trawl gear is scheduled to be completed in 2010. A revised estimate for gillnet gear will be completed within 3-5 years since the publication of Murray (2009a). For species other than loggerheads, NMFS will use all available information (*e.g.*, observed takes, changes in fishing effort, etc.) to determine if the annual incidental take level in this Opinion has been meet or exceeded. NMFS will append each year's determination and the five year review for loggerheads to this Opinion.

10.0 CONSERVATION RECOMMENDATIONS

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

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

2. NMFS should establish a protocol for bringing to shore any sea turtle incidentally taken in monkfish fishing gear that is fresh dead, that dies on the vessel shortly after the gear is retrieved, or dies following attempts at resuscitation in accordance with the regulations. Such protocol should include the steps to be taken to ensure that the carcass can be safely and properly stored on the vessel, properly transferred to appropriate personnel for

examination, as well as identify the purpose for examining the carcass and the samples to be collected. Port samplers and observers should also be trained in the protocols for notification of the appropriate personnel in the event that a vessel comes into port with a sea turtle carcass.

3. 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 ESA-listed species.

4. NMFS should support studies on seasonal ESA-listed species distribution and abundance in the action area, behavioral studies to improve our understanding of ESA-listed species interactions with fishing gear, foraging studies including prey abundance/distribution studies (which may influence distribution), as well as studies and analysis necessary to develop population estimates for sea turtles.

5. NMFS should continue to monitor and evaluate the effectiveness of the ALWTRP, particularly the impacts of the broad based gear requirements implemented in 2008 and 2009, as well as the implementation of the vertical line strategy. As part of the monitoring plan for the ALWTRP, NMFS' goal should be to detect a change in the frequency of entanglements and/or serious injuries and mortalities associated with entanglements. Metrics to consider in detecting this change could include: observed time lapses between detected large whale entanglements, known large whale serious injuries and mortalities due to entanglement, and analysis of whale scarring data.

6. NMFS should continue to undertake and support aerial surveys, passive acoustic monitoring, and the sighting advisory system.

7. NMFS should continue to develop and implement measures to reduce the risk of ship strikes of large whales.

8. NMFS sould continue to undertake and support disentanglement activities, in coordination with the states, other members of the disentanglement and stranding network and with Canada.

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

12.0 REINITIATING CONSULTATION

This concludes formal consultation on the continued operation of the monkfish fishery as it operates under the Monkfish 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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