NOAA FISHERIES
Protected Resources Division

This Biological Opinion is no longer active.

Please visit http://www.nero.noaa.gov/Protected/section7/BiOps/ to find active Biological Opinions.

ENDANGERED SPECIES ACT SECTION 7 CONSULTATION
BIOLOGICAL OPINION

Action Agency: National Marine Fisheries Service, Northeast Region

Activity: Endangered Species Act Section 7 Consultation on the Atlantic Bluefish Fishery Management Plan [Consultation No. F/NER/2007/09036]

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

Date Issued: OCT 29 2010

Approved by:

TABLE OF CONTENTS

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

NMFS NERO has reinitiated formal intra-service consultation, in accordance with section 7(a)(2) of the ESA and 50 CFR 402.16, given that new information on large whale interactions with and sea turtle bycatch in net gear used or consistent with that used in the Atlantic bluefish fishery (bluefish fishery) reveals that the continued operation of the bluefish fishery may affect listed species in a manner or to an extent not previously considered. This document represents NMFS's biological opinion (Opinion) on the continued operation of the bluefish fishery, within the constraints of the Atlantic Bluefish Fishery Management Plan (Bluefish FMP), and its effects on ESA-listed species under NMFS jurisdiction in accordance with section 7 of the ESA.

Formal intra-service section 7 consultation on the continued operation of the bluefish fishery, within the constraints of the current Bluefish FMP, was reinitiated on December 18, 2007 [Consultation No. F/NER/2007/09036]. This Opinion is based on information developed by NMFS NERO as well as other scientific data and reports cited in the Literature Cited section of this document.

1.0 CONSULTATION HISTORY

The consultation history for the bluefish fishery was reviewed by NMFS in a previous formal consultation initiated on May 10, 1999 and completed on July 2, 1999. The 1999 Opinion issued by NMFS concluded that the continued operation of the bluefish fishery, within the constraints of the Bluefish FMP and proposed Amendment 1 (which was subsequently approved and implemented by NMFS in July 2000), would not jeopardize the continued existence of right, humpback, and fin whales, loggerhead, leatherback, and Kemp's ridley sea turtles, or shortnose sturgeon, and was not likely to adversely modify right whale critical habitat (NMFS 1999a).

However, sea turtles and shortnose sturgeon were expected to experience harassment, injury, or mortality due to interactions with the gear associated with this fishery. Interactions between these species and bluefish fishing gear can include captures or entanglements in net gear (*e.g.*, trawls, gillnets) and, on rarer occasions, hooking (internally or externally) or entanglements in hook and line gear. In accordance with ESA regulations (50 CFR 402.02), such interactions are considered "incidental takes." An Incidental Take Statement (ITS) was provided with the 1999 Opinion along with non-discretionary Reasonable and Prudent Measures (RPMs) to minimize the impacts of incidental take. As described in the ITS, up to 6 loggerhead sea turtles, 6 Kemp's ridley sea turtles, and 1 shortnose sturgeon were anticipated to be captured, entangled, or hooked annually as a result of the continued operation of the bluefish fishery. Of the incidental takes exempted by the ITS, no more than 3 loggerhead takes per year were anticipated to be lethal, although up to 6 Kemp's ridley takes could have been lethal (NMFS 1999a). At the time of the 1999 Opinion, no takes of ESA-listed whales were expected to occur in the bluefish fishery.

In addition to the 1999 formal consultation, informal Section 7 consultations were conducted and completed for the development of the original Bluefish FMP in 1989 and for the proposed rule for withdrawal of Secretarial approval of the FMP in 1996. These informal consultations concluded that the bluefish fishery either had no effect on or might affect, but was not likely to adversely affect, ESA-listed species under NMFS jurisdiction or designated critical habitat.

Cause for Reinitiating

As provided in 50 CFR 402.16, reinitiation of formal consultation is required and shall be requested by the Federal agency or by the Service, where discretionary Federal involvement or control over the action has been retained or is authorized by law and if: (1) the amount or extent of incidental take specified in the ITS is exceeded; (2) new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (3) the action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in the Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action.

In 1999, a right whale mortality in U.S. Atlantic waters was attributed to entanglement in gillnet gear. However, NMFS was unable to determine the origin of the gillnet gear (*i.e.*, the fishery in which the gear was being fished). In addition, other entanglements of ESA-listed large whale species in gillnet gear were observed after completion of the original Opinion on the Bluefish FMP in 1999. There was insufficient information to determine whether any of the entanglements, including the entanglement that caused the death of a right whale in 1999, were the result of effort in the bluefish fishery. Nevertheless, NMFS concluded that the entanglements provide information that reveals effects of the action (the continued operation of the bluefish fishery) that may affect ESA-listed large whales in a manner or to an extent not previously considered.

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 1999a). 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 1999a).

In 2006, the NMFS Northeast Fisheries Science Center (NEFSC) released reference document 06-19 (Murray 2006) that reported on the annual estimated bycatch 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 trips primarily landing bluefish during the period of 2000-2004 was estimated to be 3. NMFS also received an estimate of loggerhead sea turtle bycatch in sink gillnet gear used in the bluefish fishery from the NEFSC in November 2009 (Murray 2009a). In that report, the average annual bycatch of loggerhead sea turtles in sink gillnet gear used in the bluefish fishery, based on Vessel Trip Report (VTR) data and individual species landed, was estimated to be 48 for the period of 2002-2006 with a 95% confidence interval (CI) for the five-year annual average of 23-79 (Murray 2009a). In addition, NMFS has received new information on the effects of the fishery on other ESA-listed sea turtles. The captures of two leatherback sea turtles and one unidentified sea turtle were reported in drift gillnet gear used in the bluefish fishery in 2003 and 2004, records that were verified by NMFS in 2007. Taken together, the loggerhead bycatch estimates for trawls and gillnets reported in Murray (2008, 2009a) and the verified captures of 3 sea turtles in gillnet gear landing bluefish represent new information on the effects of the bluefish fishery on ESA-listed sea turtles.

Based upon the information above, and in accordance with the regulations at 50 CFR 402.16, formal consultation was reinitiated to reconsider the effects of the bluefish fishery on ESA-listed whales (including right, humpback, fin, and also sei whales) and ESA-listed sea turtles (including loggerhead, leatherback, Kemp's ridley, and also green sea turtles). In addition, NMFS will use this Opinion to reconsider the effects of the fishery on shortnose sturgeon.

2.0 DESCRIPTION OF THE PROPOSED ACTION

The proposed action is the continued operation of the bluefish fishery, which is managed by NMFS within the constraints of the current Bluefish FMP. The Bluefish FMP covers fishing by Federal bluefish permit holders in both state and Federal waters, out to the limits of the U.S. Exclusive Economic Zone (EEZ), and also authorizes both the commercial and recreational components of the fishery in Federal waters. 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 often landed concurrently. As a result, for the purposes of this Opinion, fishing effort under the Bluefish FMP will include actions that result in landings of bluefish by federally permitted vessels operating within the action area described below. In order to identify and analyze fishery impacts on ESA-listed species, ideally, documented takes 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% bluefish, 35% haddock (a species managed under the Multispecies FMP), and 25% monkfish would be allocated proportionately to the Bluefish FMP (40%), Multispecies FMP (35%), and Monkfish

FMP (25%). Therefore, the overall estimated bycatch for each FMP is the sum of the proportionally allocated bycatch estimates.

However, at the moment, data on the incidental take of ESA-listed 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, 2009a). The bycatch estimate for trawl fisheries attributes loggerhead takes to the most abundant (by weight) fish species landed per trip (which is used as a proxy for its associated FMP). Alternatively, the gillnet bycatch estimate for loggerhead sea turtles 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 are attributed to the FMP that covers the species which makes up the majority (by weight) of the catch for the trip during which those sea turtles were caught. It should be noted that 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 sea 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 ESA-listed large whales, we can only rarely attribute incidental takes to a specific fishery. Therefore, we attribute takes by gear type and assume that any one of the FMPs that authorizes the use of that gear may be responsible for that take.

In regards to the recreational component of this and other fisheries, 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. Therefore, NMFS is unable to estimate an amount or extent of take occurring in the recreational component of the bluefish fishery at this time and will instead focus the majority of the effects analysis on the commercial component of the fishery.

2.1 Description of the Current Fishery for Bluefish

The current management measures for the bluefish fishery, the history of the fishery, and the general distribution and habitat preferences of bluefish are described in *Status of Fishery Resources off the Northeastern US – Bluefish* (Shepherd 2006), the revised *41st Northeast Regional Stock Assessment Workshop (41st SAW) Assessment Report* (NEFSC 2006a), *2010 Bluefish Specifications, Environmental Assessment, Regulatory Impact Review, and Initial Regulatory Flexibility Analysis* (MAFMC 2009), and *Amendment 1 to the Bluefish Fishery Management Plan* (MAFMC and ASMFC 1998). Additional information on the distribution and habitat characteristics of bluefish can be found in the Essential Fish Habitat source documents for the species (Fahay *et al.* 1999; Shepherd and Packer 2006). A summary of the current fishery and its management history based on these sources is provided below.

Bluefish are a migratory species found in temperate and semi-tropical continental shelf waters of the Northwest Atlantic Ocean from Nova Scotia to Florida (NEFSC 2006a; Shepherd and Packer 2006). Bluefish are described as warm water migrants and usually do not occur in Mid-Atlantic Bight[1] waters at temperatures below 14°-16°C (Shepherd and Packer 2006). They generally move north in spring-summer to centers of abundance in the New York Bight[2] and southern New England, and move south in fall-winter to waters in the South Atlantic Bight[3] as far as southeastern Florida (Shepherd and Packer 2006). However, not all bluefish move to the South Atlantic Bight in the winter. Larger fish may overwinter off North Carolina, where they are often caught in a winter fishery (Shepherd *et al.* 2006).

The fishing year for the bluefish fishery is defined for management purposes as January 1 through December 31 (50 CFR 648.160). Although the management unit for the Bluefish FMP is broadly defined as U.S. waters in the Northwest Atlantic from Maine through Key West, Florida, the fishery does not operate at all times and in all areas of the management unit. In U.S. Atlantic waters, peaks in landings are evident by both season and location. These peaks may be influenced by management measures, market conditions, weather, spawning, and coastal migrations, among other factors.

The bluefish fishery is managed by NMFS under a joint FMP collaboratively developed by the Mid-Atlantic Fishery Management Council (MAFMC) and the Atlantic States Marine Fisheries Commission (ASMFC) and implemented in 1990. The management measures presently include an overall annual landings quota in which 17% of the quota is allocated to the commercial fishery and 83% is allocated to the recreational fishery. Up to 3% of the quota may be set aside for research purposes. The total commercial quota is divided into state specific quotas, and there may be a transfer of a portion of the recreational quota to the commercial sector if predicted recreational landings are below the annual allocation (NEFSC 2006a; Shepherd 2006). This most recently occurred in 2009, when the commercial fishery was allocated a larger percentage of the total quota (just over 33%) due to lower projected landings from the recreational sector.

As indicated above, the bluefish fishery is primarily a recreational fishery. The recreational bluefish fishery accounted for approximately 72% of total bluefish landings from 2004-2008 (MAFMC 2009). Rod and reel, handline, pot, trap, and spear gear are used in the recreational fishery, with rod and reel being the predominant gear type used. Recreational fishers are limited by Federal regulations to possessing up to 15 bluefish per person (50 CFR 648.164). Much of the recreational fishery occurs in state waters. Both recreational and commercial fishermen must comply with state regulations when fishing in state waters. These include, for example, more restrictive state restrictions on bluefish possession limits and recreational size limits.

[1] The Mid-Atlantic Bight is defined as the coastal ocean area between Cape Hatteras, NC and Long Island, NY.

[2] The New York Bight is defined as the coastal ocean area along the south shore of Long Island and the east shore of NJ.

[3] The South Atlantic Bight is defined as the coastal ocean area between West Palm Beach, FL and Cape Hatteras, NC.

Effort in terms of landings and state quota allocations for the commercial sector of the fishery reflect the predominance of bluefish within portions of the management unit. Nearly all of the commercial fishery bluefish landings are in waters from Massachusetts through North Carolina as well as Florida (MAFMC 2009). Relative to total landings value, bluefish are most important in New York and North Carolina, contributing the largest percentage of ex-vessel value of all commercial landings in those states (MAFMC 2009). Allocations of the bluefish quota are not equally divided amongst the states. North Carolina receives the greatest percentage of the quota (approximately 32%) while Georgia, South Carolina, New Hampshire, and Maine (in that order) receive the least with less than 1% each of the bluefish quota (NMFS 2010a). Florida receives approximately 10% of the annual quota but has not come close to fully harvesting its quota share in recent years (NMFS 2010a). Gillnets and bottom otter trawl gear account for the vast majority of bluefish landed in the commercial fishery. In 2008, these two gear types accounted for 97.1% of the total commercial directed catch and 79.6% of the total commercial trips targeting bluefish (MAFMC 2009). Other gear types currently authorized for use in the commercial bluefish fishery are longline, handline, bandit, rod and reel, pot, trap, seine, and dredge gear (50 CFR 600.725(v)), although as evidenced above are rarely used. As these gear types are seldom used in the commercial fishery, their effects on ESA-listed species are believed to be discountable.

2.1.1 Summary of the Fishery

Bluefish are one of the most sought after species in recreational fisheries along the U.S. Atlantic coast. In 2008, recreational anglers along the Atlantic coast landed over 8,600 metric tons (mt) of bluefish, second only to striped bass (12,300 mt landed). Recreational catch of bluefish averaged 15,680 mt per year from 1981-2008, but only 8,237 mt per year from 2004-2008 (MAFMC 2009). Landings from the commercial bluefish fishery have been consistently lower than the recreational fishery. Regional variations in commercial fishing activity are linked to the seasonal migration of bluefish. Bluefish are most abundant along the North and Mid-Atlantic coasts of the U.S. from late spring to early fall, when the majority of commercial fishing activity for bluefish in these areas occurs. In the late fall and winter, bluefish move southward and landings peak along the South Atlantic coast. Annually, the majority of commercial landings are taken in the Mid- and South Atlantic regions where approximately 87% of the coastwide total landings have occurred since 1950 (NEFSC 2006a). In 2008, the commercial bluefish fishery in U.S. Atlantic waters landed 2,711 mt, down from a peak of 7,463 mt in 1981 (MAFMC 2009). In recent years (1999-2008), commercial landings have remained more or less stable while recreational landings have increased. In all, baseline conditions (*i.e.*, the distribution and intensity of bottom otter trawling and gillnetting) in the commercial bluefish fishery have not changed significantly since 2001 (MAFMC 2009).

NMFS manages the bluefish fishery under Amendment 1 to the Bluefish FMP. The FMP defines the management unit as bluefish occurring in U.S. waters of the Northwest Atlantic Ocean and is considered a single stock of fish. The FMP allows a state-by-state commercial quota system and recreational harvest limit to reduce fishing mortality. The ASMFC and MAFMC adjust both quotas annually by the specification setting process (NEFSC 2006a).

The ASMFC and MAFMC jointly developed the Bluefish FMP and adopted the plan in 1989. The Secretary of Commerce approved the Bluefish FMP in March 1990. The ASMFC and MAFMC approved Amendment 1 to the Bluefish FMP in October 1998 and NMFS published the final rule to implement the Amendment 1 measures in July 2000. Amendment 1 implemented an annual coastwide quota to control bluefish landings. The ASMFC and MAFMC adjust the quota and harvest limit annually using the specification setting process detailed in Amendment 1.

2.2 Action Area

The action area for an Opinion is defined as all of the areas directly or indirectly affected by the Federal action, and not merely the immediate area involved in the action. For the purposes of this Opinion, the action area encompasses the area in which the bluefish fishery operates, broadly defined as all U.S. EEZ waters from Maine through Key West, Florida and the adjoining state waters that are affected through the regulation of activities of Federal bluefish permit holders fishing in those waters. The direct and indirect effects of the bluefish fishery on ESA-listed species in the action area have been summarized as impacts resulting from: (1) entanglement, capture, or hooking of these species in bluefish fishing gear, (2) the operation of vessels utilized in the fishery, and (3) changes to these species' habitats and prey as a result of bottom trawl and gillnet gear utilized in the fishery.

3.0 STATUS OF LISTED SPECIES AND CRITICAL HABITAT

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

Common name	Scientific name	ESA Status
North Atlantic right whale	*Eubalaena glacialis*	Endangered
Humpback whale	*Megaptera novaeangliae*	Endangered
Fin whale	*Balaenoptera physalus*	Endangered
Sei whale	*Balaenoptera borealis*	Endangered
Loggerhead sea turtle	*Caretta caretta*	Threatened
Leatherback sea turtle	*Dermochelys coriacea*	Endangered
Kemp's ridley sea turtle	*Lepidochelys kempii*	Endangered
Green sea turtle	*Chelonia mydas*	Endangered[4]

NMFS has determined that the action being considered in the Opinion is no longer likely to adversely affect the shortnose sturgeon (*Acipenser brevirostrum*), nor is it likely to adversely affect the Gulf of Maine distinct population segment (DPS) of Atlantic salmon (*Salmo salar*), the smalltooth sawfish (*Pristis pectinata*) DPS, hawksbill sea turtles (*Eretmochelys imbricata*), blue whales (*Balaenoptera musculus*), and sperm whales (*Physeter macrocephalus*), all of which are

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

listed as endangered under the ESA. NMFS has also determined that the action being considered in the Opinion is not likely to adversely affect Johnson's seagrass (*Halophila johnsonii*), elkhorn coral (*Acropora palmata*), or staghorn coral (*Acropora cervicornis*), each of which are listed as threatened. The following discussions are NMFS's rationale for these determinations.

Shortnose sturgeon are benthic fish that occur in large coastal rivers of eastern North America. They range from as far south as the St. Johns River, Florida (possibly extirpated from this system) to as far north as the Saint John River in New Brunswick, Canada. The species is anadromous in the southern portion of its range (*i.e.*, south of Chesapeake Bay), while some northern populations are amphidromous (NMFS 1998a). Given the range of the species (remaining mostly in the river systems, with some coastal migrations between rivers), and the proposed action occurring in more offshore ocean areas, shortnose sturgeon are not expected to be present in areas where the commercial and recreational sectors of the bluefish fishery primarily operate. While it is possible for the range of the species and the bluefish fishery to overlap in certain estuarine areas along the U.S. Atlantic coast, the probability of an entanglement or capture occurring is discountable. Shortnose sturgeon most often occur in estuarine areas during the winter months, the time of year when fishing effort in the bluefish fishery is at a minimum. In addition, adverse effects are not expected since interactions with shortnose sturgeon have never been documented from the bluefish fishery. Therefore, although the 1999 Opinion anticipated the annual incidental take of one shortnose sturgeon in the bluefish fishery annually, there is no information available indicating that this level of incidental take is currently occurring or is likely to occur.

The naturally spawned and conservation hatchery populations of anadromous Atlantic salmon whose freshwater range occurs in the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River, including those that were already listed in November 2000, are listed as endangered under the ESA (NMFS 2009a, 2009b). These populations include those in the Dennys, East Machias, Machias, Pleasant, Narraguagus, Ducktrap, Sheepscot, Penobscot, Androscoggin, and Kennebec Rivers as well as Cove Brook. Juvenile salmon in New England rivers typically migrate to sea in May after a two- to three-year period of development in freshwater streams, and remain at sea for two winters before returning to their U.S. natal rivers to spawn (Reddin 2006). The preferred habitat of post-smolt salmon in the open ocean is principally the upper 10 meters (m) of the water column, although there is evidence of forays into deeper water for shorter periods. In contrast, adult Atlantic salmon demonstrate a wider depth profile (ICES 2005). Results from a 2001-2003 post-smolt trawl survey in the nearshore waters of the Gulf of Maine indicate that Atlantic salmon post-smolts are prevalent in the upper water column throughout this area in mid to late May (Lacroix and Knox 2005). Therefore, fishing close to the bottom with relatively large mesh gear, as is most often practiced in the commercial bluefish fishery, reduces the potential for catching Atlantic salmon as either post-smolts or adults.

In its report on salmon bycatch, the Working Group for North Atlantic Salmon (WGNAS) concluded that bycatch of Atlantic salmon in Northeast Atlantic commercial fisheries was not an obvious concern. The 2006 WGNAS report also discussed potential salmon bycatch implications from these fisheries and indicated there was insufficient information to quantify bycatch, although based on information reviewed so far, there was no evidence of major bycatch of salmon in Northeast fisheries (ICES 2006). 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 given that operation of the commercial and recreational components of the bluefish fishery is not prevalent in or near rivers where concentrations of Atlantic salmon are likely to be found. Also, the commercial bluefish quota allocated to Maine is less than 1% and the recreational bluefish catch in Maine is also negligible compared to other U.S. Atlantic states further south. Thus, neither this species nor its designated critical habitat will be considered further in this Opinion.

Smalltooth sawfish generally inhabit shallow coastal waters very close to shore in muddy and sandy bottoms, and are often found in sheltered bays, on shallow banks, and in estuaries or river mouths. Based on the 2000 status review, the 2003 listing rule, and the 2009 recovery plan, the smalltooth sawfish DPS has a very limited range off the extreme southwestern portion of Florida, from Charlotte Harbor to the Dry Tortugas and Florida Bay (NMFS 2000, 2003a, 2009c). Since the bluefish fishery in the Atlantic likely does not extend west or inshore of Key West, Florida, the likelihood of the fishery overlapping with the smalltooth sawfish DPS is discountable. In the unlikely event that the fishery and the DPS did overlap, the use of fishing gear known to be most detrimental to smalltooth sawfish (*e.g.*, gillnets and trawls) would be minimal in those areas. Florida has banned most types of gillnetting in state waters and smalltooth sawfish almost always occur at depths that are likely too shallow for bottom trawling. As a result of these factors, the likelihood of an interaction occurring between bluefish fishing gear and a smalltooth sawfish within the range of the DPS is insignificant. Designated critical habitat for the smalltooth sawfish DPS, which includes the Charlotte Harbor Estuary Unit and the Ten Thousand Islands/Everglades Unit (NMFS 2009d), also occurs only in Florida waters west and inshore of Key West. Since the bluefish fishery does not extend into these areas, the likelihood of the fishery impacting the species' designated critical habitat is also discountable.

The hawksbill sea turtle is uncommon in the waters of the continental U.S. Hawksbills prefer tropical, coral reef habitats, such as those found in the Caribbean and Central America. The waters surrounding Mona and Monito Islands (Puerto Rico) are designated as critical habitat for the species, and Buck Island (St. Croix, U.S. Virgin Islands) also contains especially important foraging and nesting habitat for hawksbills. Within the continental U.S., nesting is restricted to the southeast coast of Florida and the Florida Keys, but nesting in these areas is rare (0-4 nests per year). Hawksbills have been recorded from all U.S. states adjacent to the Gulf of Mexico and along the east coast of the U.S. as far north as Massachusetts, although sightings north of southern Florida are highly infrequent. Aside from Florida, Texas is the only other U.S. state where hawksbills are sighted with any regularity. Due to the species' tropical distribution, the rarity of nesting adjacent to the action area, and the fact that bluefish fishing effort is centered in the Mid-Atlantic (commercial and recreational landings in Florida in 2008 represented only 3% of all landings), it is highly unlikely that the fishery will adversely affect hawksbill sea turtles.

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 bluefish fishery operates. Blue whales feed on euphausiids (krill) (Sears 2002) which are too small to be captured in bluefish fishing gear. Given that the species is unlikely to occur in areas where the bluefish fishery operates, and given that the operation of the bluefish 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 bluefish 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 bluefish fishery operates in continental shelf waters. The average depth of sperm whale sightings observed during the CeTAP surveys was 1,792 m (CeTAP 1982). Female sperm whales and young males almost always inhabit waters deeper than 1,000 m 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 bluefish fishery operates. Given that sperm whales are unlikely to occur in areas (based on water depth) where the bluefish fishery operates, and given that the operation of the bluefish 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 bluefish 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 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 bluefish fishery will not affect the availability of copepods for foraging right whales because copepods are very small organisms that will pass through bluefish fishing gear rather than being captured in it. Nearshore waters off Georgia and northeastern Florida were designated as critical habitat for right whales due to their importance as winter calving and nursery grounds for the species. The environmental features that have been correlated with the distribution of right whales in these waters include preferred water depths, water temperature, and the distribution of right whale cow/calf pairs and the distance from shore to the 40 m isobath (Kraus *et al.* 1993; Kenney 2002). Currently there is no evidence that the bluefish fishery and its associated gear types are likely to impact water depth, water temperature, or distance from shore. Since the action being considered in this Opinion is not likely to affect the physical and biological features that characterize both the feeding and calving habitat for right whales, 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 bluefish 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 bluefish fishery will not affect the availability of copepods for foraging sei whales because copepods are very small organisms that will pass through bluefish 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 bluefish 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 bluefish fishery. Humpback whales and fin whales feed on krill as well as small schooling fish (*e.g.*, sand lance, herring, mackerel) (Aguilar 2002; Clapham 2002). Although small schooling fish species may be caught in net or hook and line gear targeting bluefish, the numbers of individuals caught are likely insignificant as recent assessments of the bluefish fishery have indicated that commercial bycatch is minimal (NEFSC 2006a). In addition, bluefish trawl and gillnet gear routinely operate on or very near the bottom. Fish species caught in these gears 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 bluefish fishery will not affect the availability of prey for foraging humpback or fin whales. Although the bluefish fishery operates in waters off the southeast U.S. where right whale calving and nursing is known to occur (Kenney 2002), the continued operation of the fishery will not affect the oceanographic conditions that are conducive for these behaviors. Calving and nursing grounds for the other species of large whales discussed in this Opinion are located outside the geographic range of the bluefish fishery, and thus will not be affected by the fishery.

Johnson's seagrass prefers to grow in coastal lagoons in the intertidal zone, and is found in coarse sand and muddy substrates and in areas of turbid waters and high tidal currents. It has a very limited distribution and is the least abundant seagrass within its range. This seagrass has only been found growing in inshore lagoons along approximately 200 kilometers (km) of coastline in southeastern Florida between Sebastian Inlet and north Biscayne Bay (NMFS 2002a). Since the primary location and habitats for Johnson's seagrass in the southeastern U.S. do not overlap with offshore areas in which the bluefish fishery primarily operates, NMFS has determined that the bluefish fishery is not likely to adversely affect this species or its designated critical habitat.

Acroporid (*i.e.*, elkhorn and staghorn) corals require relatively clear, well circulated water. Typical water temperatures in which these species occur range from 21°-29°C, but these species are capable of withstanding temperatures above the seasonal maxima for short periods of time. The environmental conditions of most of the U.S. Atlantic EEZ are not suitable for acroporid corals. The northern extent of acroporid coral occurrence off the U.S. east coast is Palm Beach County, Florida. Elkhorn corals commonly grow in turbulent shallow water on the seaward face of reefs in waters ranging from 1-5 m in depth, but have been found to 30 m. Staghorn corals commonly grow in more protected, deeper waters ranging from 5-20 m in depth and have been found in rare instances to 60 m. Elkhorn and staghorn corals have a very limited distribution in waters where the bluefish fishery operates. The Florida Keys National Marine Sanctuary

(FKNMS) and nearshore waters along the southeast coast of Florida north to Palm Beach are the only areas in the U.S. Atlantic EEZ with suitable depth and water quality to support these corals.

Potential effects on *Acropora* corals associated with fishing activities include abrasion and breakage resulting from: (1) vessel groundings, (2) anchoring, (3) damaging fishing practices, and (4) fishing/marine debris. Damaging fishing practices involve gear being dragged along or moved across, directly landing on, or becoming wrapped around coral reef habitat. The density of *Acropora spp.* and fishing gear are primary factors determining whether potential adverse impacts occur. Of the fishing gears utilized in the bluefish fishery, bottom trawls and gillnets have the potential to snag or become wrapped around coral heads. However, bottom trawling is primarily conducted in sandy and muddy bottom habitats where these corals would not occur and gillnets are usually fished so as to not come into contact with corals to avoid damage to the gear.

Regulations are in place in the areas where *Acropora spp.* are most likely to occur to protect them from the potential routes of the effects described above. FKNMS Regulations at 15 CFR 922.163 establish specific prohibitions against injuring corals (including *Acropora* species), anchoring on corals, and grounding vessels on corals. Additionally, this section prohibits the discharge of fishing/marine debris into the waters of the FKNMS. Regulations at 15 CFR 922.164 provide additional protection for corals (including *Acropora* species) occurring within specific management areas of the FKNMS by prohibiting the use of vessel-towed or anchored bottom fishing gears or nets. The low likelihood of *Acropora spp.* occurring where fishing is likely to occur, in combination with the measures in place to protect *Acropora spp.* where they may occur from bluefish fishing practices, make any adverse effects on these species from the proposed action extremely unlikely to occur. Based on this information, effects of the bluefish fishery on ESA-listed *Acropora* corals and their designated critical habitats are discountable.

3.1 Status of Large Whales

All of the large whale 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 whales 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 blue whales, 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 1970s (Perry *et al.* 1999). Today, the greatest known threats to ESA-listed large whales are ship strikes and fishing gear interactions, although the number of each species affected by these activities does vary.

Information on the range-wide status of these large whale species is included to provide information on the status of each species overall. 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 status reviews (NMFS 2006a), recovery plans (NMFS 1991, 2005a, 2006b), Marine Mammal Stock Assessment Reports (SARs) (Waring *et al.* 2009), and other publications (Clapham *et al.* 1999; Perry *et al.* 1999; Best *et al.* 2001).

3.1.1 North Atlantic right whale

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 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 distinct species: the 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 spp.*) 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 migrates 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 2006a). 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). The following information will focus on the western population of North Atlantic right whales, which occurs in the action area.

Habitat and Distribution

The western North Atlantic population of right whales generally occurs 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; Baumgartner and Mate 2005; NMFS 2005a; Waring *et al.* 2009). Right whales are most abundant in Cape Cod Bay between February and April (Watkins and Schevill 1982; Schevill *et al.* 1986; Hamilton and Mayo 1990) and in the Great South Channel in May and June (Kenney *et al.* 1986, 1995; Payne *et al.* 1990; 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 *et al.* 1986). Patrician *et al.* (2009) analyzed photographs of a right whale calf sighted in the Great South Channel in June of 2007 and determined the calf appeared too young to have been born in the known southern calving area. 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 patterns 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 waters 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, 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 of only two sightings this century of a right whale in Norwegian waters, and the first since 1926. Together, these long-range matches indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not presently well described. Similarly, records from the Gulf of Mexico (Moore and Clark 1963; Schmidly *et al.* 1972) represent either geographic anomalies or a more extensive historic range beyond the sole known calving and wintering ground in the waters of the southeastern U.S. 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 (photo-ID) techniques and an assumption of mortality for those whales not seen in seven years, a total of 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 right 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 right whales alive during this period, but with significant variation due to apparent losses exceeding gains during 1998-1999. Mean growth rate for the period of 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 per 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 (1997/1998, 1998/1999, and 1999/2000) 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 preliminary calf count for the 2008/2009 season indicates a new record calving season of 39 calves (Barb Zoodsma, pers. comm.). However, the western North Atlantic stock has also continued to experience losses of calves, juveniles, and adults. As of August 1, 2008, there were 528 individually identified right whales in the photo-ID catalog of which 25 were known to be dead, 135 were presumed to be dead (as they had not been sighted in the past six years), and 368 were presumed to be alive (Hamilton *et al.* 2008).

As is the case with other mammalian species, there is an interest in monitoring the number of females in the 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 6 calves each, and 4 cows had at least seven calves. Two of these cows were at an age which indicated a reproductive life span of at least 31 years (Kraus *et al.* 2007). As described above, the 2000/2001 - 2006/2007 calving seasons had relatively high calf production and included additional first time mothers (*e.g.*, eight new mothers in 2000/2001). These potential "gains" have been offset, however, by 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 and 2007, at least nine were adult females, three of which were carrying near-term fetuses and four 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 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 2006c). 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 2006c).

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-ID 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, the NMFS NEFSC hosted a workshop to review right whale population models to examine: (1) potential bias in the models and (2) changes in the population trend based on new information collected in the late 1990s (Clapham *et al.* 2002). Three different models were used to explore right whale survivability and to address potential sources of bias. Although biases were identified that could negatively affect the results, all three modeling techniques resulted in the same conclusion: survival has continued to decline and seems to 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 October 10, 2008, 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. Recently, the 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 1,000 projections (with the status quo simulation) ended with a smaller total population size than they started and zero projections resulted in extinctions (Pace *in review*). 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 (*e.g.*, 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 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; Leandro *et al.* 2010).

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 2000), 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 on 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 live in an ocean environment, examining right whale carcasses is often very difficult. Some carcasses are discovered floating at sea and cannot be retrieved. Others are in such an advanced stage of decomposition when discovered that a complete examination is not possible. Wave action and post-mortem predation by sharks can also damage carcasses and preclude a thorough examination of all body parts. It should also be noted that mortality and serious injury 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 from 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 an entanglement or vessel collision does not cause direct mortality, it may weaken or otherwise affect an individual 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.* 1999) 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 individual, catalogued right whales were reviewed and 625 separate entanglement interactions were documented between 1980 and 2004. Approximately 358 out of 493 animals (72.6% of the population) were entangled at least once; 185 animals bore scars from a single entanglement, however one animal showed scars from six different entanglement events. The number of male and female right whales bearing entanglement scars was nearly equivalent (142 out of 202 females, 71.8%; 182 out of 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.* (1999) estimated that 6.4% of the North Atlantic right whale population exhibits 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 reported mortalities and one serious injury resulted from right whale ship strikes (Glass *et al.* 2009).

Summary of Status for North Atlantic Right Whales
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 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. Of the recent mortalities, including those in the first half of 2005, six were adult females, three of which were carrying near-term fetuses and four of which were just starting to bear calves (Glass *et al.* 2009).

Over the five-year period of 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 due to entanglements and nine due to ship strikes (Glass *et al.* 2009). This represents an absolute minimum number of 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 suggest 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 calving seasons. Based on the information currently available, for the purposes of this Opinion, NMFS believes that the minimum estimate for the western North Atlantic right whale population is 345 individuals and that the population 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 2010 SAR also assigned a potential biological removal (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 and ship strike records for the period 2004-2008 averaged 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 are consistent with the 2009 SAR and provide additional indications of an increasing population size of and positive growth rate for North Atlantic right whales.

3.1.2 Humpback whale

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.* 2007). Within the Pacific Ocean, NMFS recognizes three management units within the U.S. EEZ for the purposes of managing this species under the MMPA. They are: the eastern North Pacific stock (feeding areas off the U.S. 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 1980s and early 1990s with a best estimate of 8% growth per year (Carretta *et al.* 2009). The minimum population estimate 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, the 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 and resides 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 (Minton *et al.* 2008).

The Southern Hemisphere population of humpback whales is known to feed mainly in the Antarctic, although some have been observed feeding in the Benguela Current ecosystem on the migration route west of South Africa (Reilly *et al.* 2008a). The IWC Scientific Committee recognizes seven major breeding stocks, some of which are tentatively further subdivided into substocks. The seven major breeding stocks, with their respective breeding ground estimates in parentheses, 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 subpopulation's wintering grounds, and the possibility exists that the entire population does not migrate to the wintering grounds (Reilly *et al.* 2008a). Like other large whales, Southern Hemisphere humpback whales were heavily exploited through commercial whaling. Although they were given protection by the IWC in 1963, Soviet whaling data made available in the 1990s 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 (IWC 1995; Zemsky *et al.* 1995; Perry *et al.* 1999).

North Atlantic Ocean (Gulf of Maine). 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 that humpback whales may also feed on euphausiids (krill) as well as capelin (Stevick *et al.* 2006; Waring *et al.* 2009).

In winter, humpback 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 1991).

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 from 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 m 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 2003-2007, 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). This includes incidental fishery interactions, which averaged 2.8 per year (U.S. waters: 2.4, Canadian waters: 0.4). Between 2003 and 2007, humpback whales were involved in 76 confirmed entanglement events and 11 confirmed ship strike events (Glass *et al.* 2009). Over that 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 and 2002 of the caudal peduncles and flukes 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 the Gulf of Maine stock of humpback whales between 2000 and 2002 suggest that 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% of the individuals exhibited raw injuries, presumably a result from agonistic interactions. However, current evidence suggests that breeding ground interactions alone cannot explain the higher frequency of healed scar patterns among male humpbacks from the Gulf of Maine stock (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) provided 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). Since that mass mortality event, there have been three additional known cases of a mass mortality involving large whale species along the U.S. east coast: 2003, 2005, and 2006. In the most recent 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 U.S. 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 Status for Humpback Whales
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 U.S. 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%-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 biased 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 U.S. 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 the following 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 humpbacks in the Southern Hemisphere, and the northern Indian Ocean stock. The 2009 SAR indicates that there are insufficient data to reliably determine current population trends for humpback whales in the North Atlantic overall (Waring *et al.* 2009). However, given the best available information, including that from the Stevick *et al.* (2003) study referenced above, for the purposes of this biological opinion NMFS believes that the humpback whale population is increasing.

Compared to the 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 2004-2008. Incidental fishery entanglement and ship strike records for the period 2004-2008 averaged 3.0 (U.S. waters: 2.8) and 1.6 (U.S. waters: 1.6), respectively, per year. Consistent with the 2009

SAR, the draft 2010 SAR concludes that current population trends for humpback whales in the North Atlantic overall cannot be determined, but that recent data indicate that the Gulf of Maine humpback whale stock is likely steadily increasing in size.

3.1.3 Fin whale

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 2006a). 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.

North Pacific Ocean and Southern Hemisphere. Within U.S. 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 U.S. 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% from 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 trends 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).

The 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 abundance estimates for southern hemisphere fin whales. Since these fin whales do not occur in U.S. waters, there is no recovery plan or stock assessment report for these animals.

North Atlantic Ocean. NMFS has designated one population of fin whales in U.S. waters of the North Atlantic (Waring *et al.* 2009). This population 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). Photoidentification 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 IWC has proposed stock boundaries for North Atlantic fin whales. Fin whales off the eastern U.S., 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 these 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 appears to be from the Great South Channel, along the 50 m 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 the birth of a single calf after an 11-12 month gestation period (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 (*e.g.*, herring, capelin, and sand lance) as well as squid and planktonic crustaceans (Wynne and Schwartz 1999).

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 (CPUE) 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 continental shelf waters of the Northeastern U.S. The Draft 2009 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 western North Atlantic fin whales is 1,678 (Waring *et al.* 2009). However, there are insufficient data at this time to determine population trends for the fin whale (Waring *et al.* 2009).

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 in Greenland (Caulfield 1993; Gambell 1993). However, Iceland reported a catch of 136 whales in the 1988/1989 and 1989/1990 seasons (Perry *et al.* 1999), and 7 in 2006/2007. 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.

Summary of Status for Fin Whales
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, which is believed to be an underestimate. The minimum population estimate for the western North Atlantic fin whale 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 2010 SAR also 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 and ship strike records for the period 2004-2008 averaged 1.2 (U.S. waters: 1.0) and 2.0 (U.S. waters: 1.4), respectively, per year.

3.1.4 Sei whale

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's 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 the U.S. Pacific EEZ (California, Oregon, and Washington waters out to 300 nautical miles) is 46 (CV = 0.61) sei whales (Barlow and Forney 2007; Forney 2007; Carretta *et al.* 2008). No fishery-related serious injuries or mortalities 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 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 large 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 recovery plan or stock assessment report for them.

North Atlantic. Sei whales occur in deep water throughout their range, typically over the continental slope or in basins situated between banks. 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 SARs, and based on a proposed IWC stock definition, NMFS recognizes 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 from June 12 to August 4, 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 2003-2007 show one record with substantial evidence of fishery interactions causing serious injury in April 2006 (Glass *et al.* 2009). Between 2003 and 2007, three ship strike mortalities have been confirmed. The first ship strike was in February 2003; an 11 m long 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 also recorded off Deer Island, Massachusetts (Waring *et al.* 2009). NMFS also has two other human caused sei whale mortalities on record. One 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 m long 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, but is considered a very conservative estimate of abundance for this stock in the North Atlantic (Waring *et al.* 2009). There are insufficient data to determine trends of the Nova Scotian sei whale population. One sei whale serious injury from a fishery interaction and three mortalities from ship strikes have been recorded in U.S. waters from 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 in the 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 (208) 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 and ship strike records for the period 2004-2008 averaged 0.6 and 0.4, respectively, per year.

3.2 Status of Sea Turtles

Sea turtles continue to be affected by many factors occurring on the nesting beaches and in the marine environment. Poaching, habitat modification and destruction, and nest 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).[5] 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 DPSs. 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 (Hirth 1997; Turtle Expert Working Group [TEWG] 1998, 2000, 2007, 2009; NMFS and USFWS 1995; 2007a, 2007b, 2007c, 2007d; Conant *et al.* 2009; NMFS SEFSC 2009), 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 DPSs. 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.

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

[5] 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 authorization of NMFS-managed fisheries under FMPs, 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.

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 both the western and 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, which 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 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: 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, fishery interactions, oil and gas exploration, coastal development, transportation, and marine pollution (*e.g.*, oil spils). For instance, on April 20, 2010, the Deepwater Horizon oil spill occurred in the Gulf of Mexico off the coast of Louisiana. As loggerhead sea turtles are known to migrate through, nest along, and forage in the coastal waters of the Gulf of Mexico, the oil spill is likely to affect the loggerhead population. However, because all the information on sea turtle stranding, deaths, and recoveries has not yet been documented, the effects of the oil spill on the loggerhead population cannot be determined at this time.

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 2002b; 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 2002b). 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 2002b). On February 21, 2003, NMFS issued a final rule in the *Federal Register* to require the use of the larger opening TEDs (68 FR 8456). The rule also provided 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. The 2002 Opinion take estimates are based in part on fishery effort levels. In recent years, low shrimp prices, rising fuel costs, competition with imported products, and the impacts of recent hurricanes in the Gulf of Mexico have all impacted the shrimp fleets; in some cases reducing fishing effort by as much as 50% for offshore waters of the Gulf of Mexico (GMFMC 2007). As a result, loggerhead interactions and mortalities in the Gulf of Mexico have been substantially less than projected in the 2002 Opinion. Currently, the estimated annual number of interactions between loggerheads and shrimp trawls in the Gulf of Mexico shrimp fishery is 23,336, with 647 (2.8%) of those interactions resulting in mortality (Memo from Dr. B. Ponwith, Southeast Fisheries Science Center [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 (50%) 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 trawls 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 group (fish or invertebrate) 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 annual bycatch of loggerheads in U.S. Mid-Atlantic gillnet gear was estimated to average 350 turtles (95% CI over the 12-year period: 234 to 504). Bycatch rates were correlated with latitude, sea surface temperature, and mesh size. The highest predicted bycatch rates occurred in warm waters of the southern Mid-Atlantic in large-mesh gillnets (Murray 2009b).

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. This includes 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 has 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 recently 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 (75 FR 12597, March 16, 2010). NMFS and FWS are accepting 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). However, 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). However, leatherbacks appear to be approaching extinction in Malaysia (Spotila *et al.* 2000). For example, the nesting group on Terengganu, which was once one of the most significant nesting sites in the western Pacific, declined from an estimated 3,103 females in 1968 to 2 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, leatherbacks 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 fully one half, 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 20 females by 2019-2020. An analysis of Costa Rican nesting beaches indicates a decline in nesting during 15 years of monitoring (1988-1989 to 2003-2004) 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, Colombia, Ecuador, and Peru; purse seine fisheries for tuna in the eastern tropical Pacific Ocean; and the California/Oregon drift gillnet fishery. 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). Spotila *et al.* (2000) indicated that leatherback sea turtles have been virtually extinct in Sri Lanka since 1994 and disappeared from India before 1930.

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 (Peter 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* spp.) 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 3.2.1, 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 trend for all of the seven populations or groups of populations with the exception of the Western Caribbean and West Africa. However, caution is also warranted even for those that were identified as stable or increasing. In St. Croix, for example, researchers have noted a declining presence of neophytes (first-time nesters) since 2002 (Garner 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 (Sea Turtle Stranding and Salvage Network [STSSN] database). Leatherback sea turtles tagged in Puerto Rico, Trinidad, and the Virgin Islands have also been subsequently found on U.S. beaches of southern, Mid-Atlantic, and northern states (STSSN database). Animals from the South Atlantic nesting assemblages 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 caught by the U.S. Atlantic tuna and swordfish longline fisheries between 1992-1999, of which 88 were released dead (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 of the leatherbacks were dead upon release and one 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).

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 leatherback sea turtles were reported entangled in a crab pot buoy line inside Hatteras Inlet (NMFS SEFSC 2001). A third leatherback was reported entangled in a crab pot buoy line 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 of five leatherback strandings from 1982 to 1997 were due to entanglement (Boulon 2000), leatherbacks have been observed with their flippers wrapped in the line of West Indian fish traps (Rafe 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 2002b). Leatherbacks are likely to encounter shrimp trawls working in the coastal waters off the U.S. Atlantic coast (from Cape Canaveral, Florida through North Carolina) as they make their annual spring migration north. For many years, TEDs that were required for use in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries were less effective for leatherbacks as compared to the smaller, hard-shelled turtle species, because the TED openings were too small to allow leatherbacks to escape. To address this problem, NMFS issued a final rule on February 21, 2003 to amend the TED regulations (68 FR 8456). 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 3.2.1 above for further information on the shrimp trawl fishery). Currently, an estimated 520 leatherbacks are believed to interact with shrimp trawls in the Gulf of Mexico shrimp fishery annually, with 15 (2.9%) of those interactions resulting in mortality (Memo from Dr. B. Ponwith, SEFSC to Dr. R. Crabtree, SERO, PRD, December 2008).

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 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 in NMFS SEFSC 2001).

Fishing gear interactions are problematic for leatherbacks throughout their range. Entanglements are common in Canadian waters where Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland/Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line, and crab pot line. Leatherbacks are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo *et al.* 1994; Graff 1995). Gillnets are one of the suspected causes for the decline in the leatherback sea turtle population in French Guiana (Chevalier *et al.* 1999), and gillnets targeting green and hawksbill sea turtles in the waters of coastal Nicaragua also incidentally catch leatherback sea turtles (Lagueux 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alio-M. 2000). An estimated 1,000 mature female leatherback sea turtles are caught annually in fishing nets off of Trinidad and Tobago with mortality estimated to be between 50%-95% (Eckert and Lien 1999). However, many of the sea turtles do not die as a result of drowning, but rather because the fishermen butcher the turtles in order to get 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 at 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 (*e.g.*, oil spills) and habitat destruction account for an unknown level of other mortality. For instance, on April 20, 2010 the Deepwater Horizon oil spill occurred in the Gulf of Mexico off the coast of Louisiana. As leatherback sea turtles are known to migrate through and along the coastal waters of the Gulf of Mexico, the oil spill is likely to affect the leatherback population. However, because all the information on sea turtle stranding, deaths, and recoveries has not yet been documented, the effects of the oil spill on the leatherback population cannot be determined at this time. 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 for the leatherback (NMFS and USFWS 2007b).

3.2.3 Kemp's ridley sea turtle

The Kemp's ridley is one of the least abundant of the world's sea turtle species. In contrast to loggerhead, leatherback, and green sea turtles, which are found in multiple oceans of the world, Kemp's ridleys typically occur only in the Gulf of Mexico and the 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% C.I. 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 sea turtles recovered by the STSSN suggests that benthic immature developmental areas occur in many areas along the U.S. coast and that these areas may change given resource quality and quantity (TEWG 2000). 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* sp., *Ovalipes* sp., *Libinia* sp., and *Cancer* sp. 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 there by juveniles of the same size from North Carolina sounds and smaller juveniles from New York and New England to form one of the densest concentrations of Kemp's ridleys outside of the Gulf of Mexico (Epperly *et al.* 1995a, 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 nearshore 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 3.2.1 above, there is a 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 2002b; Lewison *et al.* 2003). The 2002 Opinion on shrimp trawling in the southeastern U.S. concluded that 155,503 Kemp's ridley sea turtles would be taken annually in the fishery with 4,208 of the takes resulting in mortality (NMFS 2002b). Recently, NMFS has discovered that those numbers are likely overestimates. Currently, the annual number of interactions between Kemp's ridleys and shrimp trawls in the Gulf of Mexico shrimp fishery is estimated to be 98,184, with 2,716 (2.8%) of those interactions resulting in mortality (Memo from Dr. B. Ponwith, SEFSC to Dr. R. Crabtree, SERO, PRD, December 2008).

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 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). The five Kemp's ridley carcasses that were found are likely to have been only a minimum count of the number of Kemp's ridleys that were killed or seriously injured as a result of the fishery interaction, since it is unlikely that all of the carcasses washed ashore.

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 (*e.g.*, oil spills), and habitat destruction account for an unknown level of other mortality. For instance, on April 20, 2010 the Deepwater Horizon oil spill occurred in the Gulf of Mexico off the coast of Louisiana. As Kemp's ridley sea turtles are known to migrate through, nest along, and forage in the coastal waters of the Gulf of Mexico, the oil spill is likely to affect the Kemp's ridley population. However, because all the information on sea turtle stranding, deaths, and recoveries has not yet been documented, the effects of the oil spill on the Kemp's ridley population cannot be determined at this time.

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 turtle

Green sea turtles are distributed circumglobally, and can be found in the Pacific, Indian, and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 1991; 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. As noted previously, 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 fibropapillomatosis, which is a viral disease that causes tumors in affected turtles (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.* 2003). Based on a review of the 32 Index Sites used to monitor green sea turtle nesting worldwide, Seminoff (2004) concluded that declines in green sea turtle nesting were evident for many of the Indian Ocean Index Sites. While several of these had not demonstrated further declines in the more recent past, only the Comoros Island Index Site in the Western Indian Ocean showed evidence of increased nesting (Seminoff 2004).

Mediterranean Sea. There are four nesting concentrations of green sea turtles in the Mediterranean from which data are available, and they are located 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. Although this population is depleted from historic levels (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; Yaniv 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 pastures 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 considered 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 Archipelago, Guinea-Bissau (NMFS and USFWS 2007d). Nesting at all of these sites was considered to be stable or increasing with the exception of Bioko Island, which may be declining, and the Bijagos Archipelago, which may be stable; however, the lack of sufficient data precluded a meaningful trend assessment for either site (NMFS and USFWS 2007d).

Seminoff (2004) likewise reviewed green sea turtle nesting data for eight sites in the western, eastern, and central Atlantic, including all of the above 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).

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). Green sea turtle nesting has also been documented to occur in North Carolina at Bald Head Island (just east of the mouth of the Cape Fear River), Onslow Island, and Cape Hatteras National Seashore (Peterson *et al.* 1985; Hawkes *et al.* 2005).

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. Currently, an estimated 11,311 green sea turtles are believed to interact with shrimp trawls in the Gulf of Mexico shrimp fishery annually, with 319 (2.8%) of those interactions resulting in mortality (Memo from Dr. B. Ponwith, SEFSC to Dr. R. Crabtree, SERO, PRD, December 2008). Other activities like dredging, pollution (*e.g.*, oil spills), and habitat destruction account for an unknown level of other mortality. For instance, on April 20, 2010 the Deepwater Horizon oil spill occurred in the Gulf of Mexico off the coast of Louisiana. As green sea turtles are known to migrate through, nest along, and forage in the coastal waters of the Gulf of Mexico, the oil spill is likely to affect the green sea turtle population. However, because all the information on sea turtle stranding, deaths, and recoveries has not yet been documented, the effects of the oil spill on the green sea turtle population cannot be determined at this time. 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[6] distributed globally revealed a 48%-67% decline in the number of mature females nesting annually over the last three generations[7] (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). Despite the apparent global increase in numbers, the positive overall trend should be viewed cautiously because trend data were available for just over half of all sites examined. 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

[6] 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.

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

for green sea turtles, caution is urged regarding the status for any of the nesting groups since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d).

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

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 ESA-listed large whales and 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 whale or sea turtle's ability to function, and (3) recovery activities associated with reducing impacts to whales and sea turtles.

Many of the fisheries and other activities causing death or injury to large whales and 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.*, take reduction teams/plans for large whales, 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 sea turtles.

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

4.1 Fishery Operations

4.1.1 Federal fisheries

Commercial and recreational fisheries in the action area employ gear that is known to harass, injure, and/or kill large whales and sea turtles. Several federally regulated fisheries that use gillnet, longline, trawl, dredge, and pot/trap gear have been documented as unintentionally capturing, entangling, hooking, entraining, or colliding with these species. In some cases, the animals are harmed, injured, or killed as a result of the interaction. Available information suggests that these species can be captured, entangled, hooked, or entrained in these gear types when the operation of the gear overlaps with the distribution of the species.

ESA-listed whales and sea turtles are also known to be killed and injured as a result of being struck by fishing vessels on the water. However, for the following reasons, the operation of commercial fishing vessels used in the aforementioned fisheries will have discountable effects on these species. First, commercial fishing vessels operate at relatively slow speeds, particularly when towing or hauling gear. Thus, large whales and sea turtles in the path of a commercial fishing vessel would likely be able to move out of the vessel's path before being struck. Second, commercial fishing effort for all of the Federal fisheries within the action area is constrained in some way, either through a limited access permit system or by fishing quotas, thus limiting the amount of time that vessels are on the water. The less the time that vessels are on the water, the less opportunity for vessel collisions with these species. Finally, ESA-listed large whales and sea turtles do not occur strictly at or within close proximity to the water surface (Morreale 1999; Baird *et al.* 2000), meaning that they spend part of their time at depths out of range of a collision with fishing vessels. For these reasons, the impacts of commercial fishing vessels on ESA-listed large whales and sea turtles are negligible. There might be a greater potential for interactions between recreational fishing vessels and large whales and/or sea turtles, but there is currently no mechanism for estimating the likelihood of such interactions within the action area.

The types of gear used in the Federal fisheries described below are also expected to have an insignificant effect on large whale and sea 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 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, 1997; USFWS and NMFS 1992). 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 (Lutcavage and Musick 1985; Keinath *et al.* 1987; Dodd 1988; Burke *et al.* 1993, 1994; Morreale and Standora 2005; Seney and Musick 2005, 2007). 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 (Lutcavage and Musick 1985; Keinath *et al.* 1987; 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.

Several of the fisheries described below use bottom otter trawl gear. The Northeast Region Essential Fish Habitat Steering Committee (NREFHSC), a panel of experts in the fields of benthic ecology, fishery ecology, geology, fishing gear technology, and fisheries gear operations, has previously concluded that the effects of even light weight otter trawl gear would include: (1) the scraping or plowing of the doors on the bottom, sometimes creating furrows along their path, (2) sediment suspension resulting from the turbulence caused by the doors and the ground gear on the bottom, (3) the removal or damage to benthic or demersal species, and (4) the removal or damage to structure forming biota. The panel also concluded that the greatest impacts from otter trawls occur in high and low energy gravel habitats and in hard clay outcroppings, and that sand habitats were the least likely to be impacted (NREFHSC 2002). The action area in which these Federal fisheries occur along the U.S. Atlantic coast includes very few habitats that are purely gravel or hard clay (Amato 1994). The foraging distributions 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. Like leatherbacks, factors affecting food availability for large whales are likely to be oceanographic conditions rather than bottom habitat (IWC 1992; Perry *et al.* 1999; Baumgartner *et al.* 2003; Pace and Merrick 2008). Fixed gear (*e.g.*, pots, traps, and sink gillnets) is expected to have less of an effect on bottom habitat than mobile gear. For these reasons and the lack of any evidence that fishing practices affect bottom habitats in degrees that harm or harass ESA-listed species, NMFS finds that while continued bluefish fishing efforts may potentially alter benthic habitats, these alterations will be insignificant to ESA-listed species.

In the Northeast Region (Maine through Virginia), ESA section 7 consultations have been conducted on the American lobster, Atlantic herring, Atlantic mackerel/squid/butterfish, Atlantic sea scallop, monkfish, northeast multispecies, red crab, skate, spiny dogfish, summer flounder/scup/black sea bass, and tilefish fisheries. An ITS has been issued for the incidental take of sea turtles in each of these fisheries. The ITS reflects the incidental take of sea turtles and other ESA-listed species anticipated from the date of the ITS and forward in time. Several of these fisheries have also been determined by NMFS to use gear that may injure or kill right, humpback, fin, or sei whales (Glass *et al.* 2009; Waring *et al.* 2009). In the Southeast Region (North Carolina through Texas), ESA section 7 consultations have been conducted on the coastal migratory pelagics, swordfish/tuna/shark/billfish, snapper/grouper, dolphin/wahoo, southern flounder gillnet, and the Southeast shrimp trawl fisheries. An ITS has been issued for the incidental take of sea turtles in each of these fisheries as well.

The only fishery that has been determined by NMFS to reduce the reproduction, numbers, or distribution of ESA-listed sea turtles, and thereby 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.

A summary of each fishery that has been subject to section 7 consultation is provided below, but more detailed information can be found in the respective biological opinions. The information describes times and areas where the fishery presently operates in order to qualitatively assess the likelihood of overlap between operation of the fishery and distribution of ESA-listed large whales and sea turtles.

As described in Section 1.0, ESA section 7 consultation has also been previously conducted on the Atlantic bluefish fishery - a fishery with a history in U.S. Atlantic waters that dates back to at least the 1950s and possibly earlier (MAFMC and ASMFC 1998; NEFSC 2006a). Therefore, the environmental baseline for this action also includes the effects of the past operation of the Atlantic bluefish fishery.

The *American lobster trap fishery*, which is managed in Federal waters by NMFS under the Atlantic Coastal Fisheries Cooperative Management Act (ACFCMA), has been identified as a source of gear 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 2002c). Loggerheads and leatherbacks 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 loggerheads and leatherbacks in Mid-Atlantic and New England waters and the operation of the lobster fishery, these species, most notably leatherbacks, are expected to overlap with the fishing of lobster pot/trap gear during the months of May through October in waters off of Maine through New Jersey.

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; Glass *et al.* 2009; Waring *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).

American lobsters occur within U.S. waters from Maine to Virginia. They are most abundant from Maine to New Jersey with abundance declining from north to south (ASMFC 1997). 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. In 2006, Maine and Massachusetts produced 90% of the total U.S. landings of American lobster, with Maine accounting for 79% of these landings (NMFS 2007a). Lobster landings in the other New England states as well as New York and New Jersey account for most of the remainder of U.S. American lobster landings. However, declines in lobster abundance and landings have occurred from Rhode Island through New Jersey in recent years. The Mid-Atlantic states from Delaware through North Carolina have been granted *de minimus* status under the Atlantic States Marine Fisheries Commission's (ASMFC) 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% from current levels. Such measures are of benefit to large whales and sea turtles by reducing the amount of gear (specifically buoy lines) in waters where these species occur.

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

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. However, due to new information on the effects of the fishery on sea turtles and right whales, including modifications to the ALWTRP and replacement of both the Seasonal Area Management (SAM) and Dynamic Area Management (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; NEFSC 2006a). Additional information on management of the fishery is provided in Section 2.1.

The bluefish fishery is known to operate in areas where and at times when large whales occur; thus, interactions between whales and the 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 gillnet gear has never been traced back to the bluefish fishery specifically, oftentimes the gear responsible cannot be identified. Gillnet gear used in the bluefish fishery is required to follow regulations set by the ALWTRP.

This fishery is also known to interact with sea turtles, given the time and locations where the fishery occurs. No captures of loggerheads have been reported in bottom otter trawl gear for trips that were targeting bluefish (where >50% of the catch was bluefish) (NMFS 1999a). However, loggerhead captures have been observed in bottom otter trawl gear where bluefish was caught but constituted less than 50% of the catch (NMFS 1999a). 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 VTR data 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 for trips primarily landing bluefish during the period of 2000-2004 was estimated to be 3 loggerheads per year (Murray 2008). The 1999 Opinion on the fishery anticipated the annual incidental take of 6 loggerheads. 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 leatherback sea turtles and one unidentified hard-shelled sea turtle have been reported in gillnet gear used in the bluefish fishery. NMFS also received an estimate of loggerhead sea turtle bycatch in gillnet gear used in the bluefish fishery from the NEFSC in November 2009 (Murray 2009a). In that report, 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 for the 5-year annual average of 23-79 (Murray 2009a). Due to new information on both large whale and sea turtle takes, formal section 7 consultation on the continued operation of the bluefish fishery under the Bluefish FMP was reinitiated on December 18, 2007.

There is no information to suggest that large whale and 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 large whale or sea turtle interactions with bluefish trawl or gillnet gear were insufficient prior to increased observer coverage in recent years. Therefore, it is likely that the effect of the bluefish fishery on ESA-listed species, while only quantified and recognized within the last few years, has been present for decades.

Section 7 consultation was originally completed on the *Atlantic herring fishery* on September 17, 1999 (NMFS 1999b). This fishery is managed under the Northeast Atlantic Herring FMP, which was implemented on December 11, 2000. At the time, NMFS concluded that operation of the Federal herring fishery under the Atlantic Herring FMP was likely to adversely affect green, Kemp's ridley, leatherback, and loggerhead sea turtles, but not likely to jeopardize their continued existence. An ITS for sea turtles was provided with the biological opinion, based on the observed capture of sea turtles in other fisheries using comparable gear. It exempted the annual incidental take of 6 loggerheads, 1 leatherback, 1 Kemp's ridley, and 1 green sea turtle.

Consultation on the Atlantic herring fishery was reinitiated on March 23, 2005 due to new information on the effects of the fishery on the Gulf of Maine DPS of Atlantic salmon and sea turtles. That consultation was recently completed in 2010 and determined that the herring fishery is not likely to adversely affect any ESA-listed species, including sea turtles. Based on analysis of VTR data, Murray (2008) estimated zero sea turtle takes in trawl gear by the Atlantic herring fishery. In addition, over the 5 year period from 2004-2008, higher than normal observer coverage occurred in the herring fishery, without any observed takes of sea turtles.

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 area of this consultation where sea turtles also occur. While squid landings occur year round, the majority of *Loligo* squid landings occur in the fall through winter months and 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 sea turtles in Mid-Atlantic waters. Gillnets account for only 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.

The most recent biological opinion on these fisheries was completed on April 28, 1999. The Opinion concluded that the continued operation of the fisheries under the FMP was likely to adversely affect sea turtles, but not jeopardize their continued existence (NMFS 1999c). Trawl gear is the primary fishing gear for these fisheries, but several other types of gear may also be used, including gillnet, hook and line, pot/trap, dredge, pound net, and bandit gear. Entanglements or entrapments of sea turtles have been recorded in one or more of these gear types. An ITS was provided with the Opinion exempting the annual incidental take of up to 6 loggerhead (no more than 3 lethal), 1 leatherback, 2 Kemp's ridley, and 2 green sea turtles.

In 2008, the NEFSC, using VTR data from 2000-2004, estimated the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear targeting Atlantic mackerel, squid, and butterfish to be 62 loggerhead sea turtles per year for that 5-year period (Murray 2008). Based on this new information on the estimated incidental take of loggerheads in the mackerel, squid, and butterfish fisheries, section 7 consultation on the continued operation of the fishery under the Squid/Mackerel/Butterfish FMP was reinitiated on March 6, 2008. That consultation is ongoing.

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. Vessels that did not qualify for a limited access permit can obtain an open access, general category scallop permit (type 1A or 1B). An increase in active general category permits and the increase in landings by general category permitted vessels prompted the initiation of Amendment 11 to the Scallop FMP. In particular, it was noted that in these last several years there has been an increasing percentage of general category landings by vessels with homeports in the Mid-Atlantic region, and shifts in fishing effort by general category vessels to Mid-Atlantic fishing grounds (NEFMC 2007). Amendment 11 is expected to contribute to the management objectives of the fishery by reducing or constraining effort in the general category sector.

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. Single sea turtle captures in scallop dredge gear were reported in both 1997 and 1999 as well. At the time, each of these events was thought to be an anomaly that only happened on extremely rare occasions. However, 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 at more than a very low level (as was thought due to the single observed takes in 1996, 1997, and 1999), there is no information to suggest that turtle interactions with scallop fishing gear are a new event or are occurring at a greater rate than what has likely occurred in the past. To the contrary, the methods used to detect any sea turtle interactions with scallop fishing gear (dredge or trawl gear) were insufficient prior to increased observer coverage in 2001. In addition, there have been no known changes to the seasonal distribution of loggerhead sea turtles in the Mid-Atlantic north of Cape Hatteras

(CeTAP 1982; Lutcavage and Musick 1985; Keinath *et al.* 1987; Shoop and Kenney 1992; Burke *et al.* 1993, 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 9 or so years, has been present for decades.

Formal section 7 consultation on the continued operation 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 the 2008 Opinion, NMFS determined that the continued operation of the fishery under 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).

The federal *monkfish fishery* occurs in all waters under federal jurisdiction from Maine to the North Carolina/South Carolina border and is jointly managed by the NEFMC and MAFMC under the Monkfish FMP (NEFSC 2005). 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 m with concentrations between 70 and 100 m and at 190 m. The directed monkfish fishery uses several gear types that may incidentally take ESA-listed species, including gillnet and trawl gear.

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 3 loggerhead and 1 non-loggerhead sea turtles in monkfish gillnet gear and one sea turtle (either a loggerhead, leatherback, Kemp's ridley, or green) in monkfish trawl gear. Although the estimated incidental take of sea turtles in monkfish gillnet gear is relatively low, there is concern that much higher levels of interaction could occur. 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 sea 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 sea 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.

Use of gillnet gear in the fishery is also affected by measures implemented under the ALWTRP. In the 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 for the monkfish fishery in the 2001 Opinion, and reissued in the 2003 Opinion, implemented the SAM and DAM programs into 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.

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 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 for trips primarily landing monkfish during the period of 2000-2004 was estimated to be 2 loggerheads per year (Murray 2008). NMFS also received an estimate of loggerhead sea turtle bycatch in gillnet gear used in the monkfish fishery from the NEFSC in November 2009 (Murray 2009a). In that report, 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 for the 5-year annual average of 68-171 (Murray 2009a). This information represents new information on the capture of loggerhead sea turtles in the monkfish fishery. As a result, section 7 consultation has been reinitiated for the monkfish fishery due to the new information received on sea turtle takes in bottom trawl gear as well as changes in management of interactions between endangered large whales and commercial gillnet gear. This consultation is currently ongoing.

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, Kemp's ridley, 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 Amendments 13 and 16 to the Northeast Multispecies FMP. The exact relationship between multispecies fishing effort and the number of large whale and sea turtle 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 large whales or sea turtles to be captured or entangled in multispecies fishing gear.

Gillnet and trap/pot gear in the fishery is also affected by measures implemented under the ALWTRP. In the June 2001 Opinion on the multispecies fishery, 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 into the ALWTRP. Recently, the SAM and DAM programs have been replaced with broad-based gear modifications under the ALWTRP.

In 2008, the NEFSC, using VTR data from 2000-2004, estimated the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear targeting fish belonging to the Northeast multispecies complex to be 43 loggerhead sea turtles per year for that 5-year period (Murray 2008). Additional information on loggerhead sea turtle interactions with gillnet gear is provided in Murray (2009a). In that report, 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 for that 5-year period (Murray 2009a). Given this new information on sea turtle takes as well as 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 Multispecies FMP. That consultation is ongoing.

The *deep-sea red crab fishery* is a pot/trap fishery that occurs in deep waters along the continental slope. The primary fishing zone for red crab, as reported by the fishing industry, is at a depth of 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. Section 7 consultation was completed on the fishery during the proposed implementation of the Red Crab FMP (NMFS 2002d). The Opinion concluded that the action was not likely to result in jeopardy to any ESA-listed species under NMFS jurisdiction. 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 groundlines and/or buoy 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, primarily as bycatch in the fishery for Northeast multispecies. Otter trawls are the primary gear used to land skates in the U.S., with some landings also coming from sink gillnet, longline, and other gear. 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.

This fishery is known to interact with sea turtles, given the time and locations where the fishery occurs. Section 7 consultation on the Skate FMP was completed on July 24, 2003 (NMFS 2003c), and concluded that operation of the skate fishery under the Skate FMP may adversely affect sea turtles as a result of interactions with gillnet and trawl gear. Although there have been no recorded takes of sea turtles in the skate fishery, given that sea turtle interactions with trawl and gillnet gear have been observed in other fisheries, takes in gear used in the skate fishery may be possible where the gear and sea turtle distributions overlap. Subsequently, the NEFSC, using VTR data from 2000-2004, estimated the average annual bycatch (take) of loggerhead sea turtles in bottom otter trawl gear for trips primarily landing skates to be 24 loggerhead sea turtles a year for that 5-year period (Murray 2008). Additional information on loggerhead sea turtle interactions with gillnet gear, including gillnet gear used in the skate fishery, has also been published in Murray (2009a). 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 for the 5-year annual average of 5-15 (Murray 2009a).

ESA-listed large whales have also been known to interact with gillnet gear, thus interactions may occur where the gear and large whale distributions overlap. The 2003 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. Due to new information on both large whale and sea turtle takes, formal section 7 consultation on the continued operation of the skate fishery under the Skate FMP was reinitiated on April 2, 2008. That consultation is ongoing.

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 (NEFSC 2003). The predominance of any one gear type has varied over time (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) (NEFSC 2006b). More recently, data from fish dealer reports for the 2008 fishing year indicate that spiny dogfish landings came mostly from sink gillnets (68.2%), 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 large whales are also known to be seriously injured or killed from interactions 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 large whales 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 number of gear interactions that occur. As a result, the June 14, 2001 Opinion on the fishery concluded that its operation under the Spiny Dogfish FMP may adversely affect but was not likely to jeopardize the continued existence of ESA-listed sea turtles. An 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 bycatch (take) of loggerhead sea turtles in bottom otter trawl gear for trips primarily landing spiny dogfish to be 1 loggerhead sea turtle per year for that 5-year period (Murray 2008). Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the spiny dogfish fishery, has also been published in Murray (2009a). 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 for the 5-year annual average of 0-1 (Murray 2009a). A thorough analysis of sea turtle interactions with trawl and gillnet gear is being included in the ongoing consultation.

The 2001 Opinion also concluded that the continued operation of the spiny dogfish fishery would adversely affect North Atlantic right whales. The Opinion provided an RPA which included components to minimize the overlap of right whales and spiny dogfish gillnet gear (*e.g.*, the SAM and DAM programs 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 replacement of the SAM and DAM programs with broad-based gear modifications under the ALWTRP, which represents new information not previously considered on the effects of the fishery on ESA-listed large 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). In 2001, NMFS prepared an Opinion on the effects of these three fisheries on ESA-listed sea turtles. An ITS was provided for the anticipated capture of sea turtles in trawl and gillnet gear used in these fisheries. It currently exempts the annual incidental take of up to

19 loggerhead or Kemp's ridley sea turtles (up to five lethal takes) 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 (50%) 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. 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 incidental takes causing injury and death in summer flounder, scup, and black sea bass fishing gear.

An estimate of loggerhead sea turtle bycatch in bottom otter trawl gear used in the summer flounder, scup, black sea bass fisheries has been published in a 2008 NEFSC Reference Document (Murray 2008). Using VTR data 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 for trips primarily landing summer flounder, scup, or black sea bass during the period of 2000-2004 was estimated to be 192 loggerhead sea turtles per year (Murray 2008). 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 published in Murray (2009a). 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 for the 5-year annual average of 2-11 (Murray 2009a). This represents new information on the incidental take of loggerhead sea turtles in the summer flounder, scup, and black sea bass fisheries. NMFS has, therefore, reinitiated section 7 consultation on the continued operation of the summer flounder, scup, and black sea bass fisheries. Consultation is on-going.

Gillnets and pots/trap gear are used in the summer flounder, scup, and black sea bass fisheries. These gears represent only a small portion of effort in this fishery and is not expected to adversely affect ESA-listed large whales. All gillnet and pot/trap gear used by the summer flounder, scup, and black sea bass fisheries are subject to complying with the ALWTRP.

A summary of the current *tilefish fishery* is provided in the 48th Northeast Regional Stock Assessment Report (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).

In 2007, NMFS completed a section 7 consultation on the continued operation of the *coastal migratory pelagic* fishery in the Gulf of Mexico and South Atlantic (NMFS 2007b). In the Gulf of Mexico, hook-and-line, gillnet, and cast net gears are used. Gillnets are the primary gear type used by commercial fishermen in the South Atlantic regions as well, while the recreational sector uses hook-and-line gear. The hook-and-line effort is primarily trolling. The biological opinion concluded that green, Kemp's ridley, leatherback, and loggerhead sea turtles may be adversely affected by operation of the fishery. However, the proposed action was not expected to jeopardize the continued existence of any of these species and an ITS was provided. The ITS exempts the incidental take of up to 33 loggerhead, 14 green, 4 Kemp's ridley, and 2 leatherback sea turtles over a three-year period as a result of interactions with gear used in the fishery.

The *Atlantic pelagic fisheries for swordfish, tuna, sharks, and billfish (highly migratory species)* are known to incidentally capture 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 biological 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 operation of the fishery in a manner 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).

A section 7 consultation on the *South Atlantic snapper-grouper fishery* was completed by NMFS in 2006. The fishery uses spear and powerhead, black sea bass pot, and hook-and-line gear. Hook-and-line gear used in the fishery includes commercial bottom longline gear and commercial and recreational vertical line gear (*e.g.*, handline, bandit gear, rod and reel). The consultation found that only hook-and-line gear is likely to adversely affect green, Kemp's ridley, leatherback, and loggerhead sea turtles. The consultation concluded the proposed action was not likely to jeopardize the continued existence of any of these species, and an ITS was provided. The ITS exempts the incidental take of up to 202 loggerhead, 39 green, 25 leatherback, and 19 Kemp's ridley sea turtles over a three-year period as a result of interactions with gear used in the fishery (NMFS 2006d).

An FMP for the South Atlantic *dolphin-wahoo fishery* was approved in December 2003. The stated purpose of the Dolphin and Wahoo FMP is to adopt precautionary management strategies to maintain the current harvest level and historical allocations of dolphin (90% recreational) and ensure no new fisheries develop. NMFS conducted a formal section 7 consultation to consider the effects on sea turtles of authorizing fishing under the FMP (NMFS 2003d). The August 27, 2003 Opinion concluded that the longline component of the fishery may adversely affect but would not jeopardize the continued existence of loggerhead, leatherback, Kemp's ridley, and green sea turtles. An ITS for sea turtles was provided with the Opinion, exempting the incidental take of up to 12 loggerheads, 12 leatherbacks, and any combination of 3 incidental takes for Kemp's ridley, green, and hawksbill sea turtles over a three-year period (NMFS 2003d). Also, pelagic longline vessels can no longer target dolphin-wahoo with smaller hooks because of hook size requirements in the pelagic longline fishery.

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 2002b). 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- and South Atlantic in state waters from Connecticut through Florida, in areas where sea turtles also occur. Incidental takes 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 capture or entangle sea turtles given the gear type, but no interactions have been observed. Similarly, small mesh gillnet fisheries occurring in Virginia state waters are suspected of capturing or entangling sea turtles, but no interactions have been observed. From May to June of 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 sea turtle takes were observed (NMFS 2004c).

In North Carolina, a large-mesh gillnet fishery for summer and southern flounder in the southern portion of Pamlico Sound was found to take sea turtles in gillnet gear. In particular, the North Carolina inshore fall southern flounder gillnet fishery was identified as a source of large numbers of sea turtle mortalities in 1999 and 2000, especially loggerhead sea turtles. In 2000, 2001, and 2002, NMFS issued an ESA section 10 permit to the North Carolina Department of Marine Fisheries (NCDMF) for the take of sea turtles in the Pamlico Sound large-mesh gillnet fishery and provided mitigative measures for the southern flounder fishery. Subsequently, sea turtle mortalities in these fisheries were drastically reduced. The reduction of sea turtle mortalities in these fisheries reduces the negative effects these fisheries have on the environmental baseline. The annual takes for the 2002-2004 fishing seasons as estimated in the 2001 permit were 24 lethal and 164 live takes each of Kemp's ridley, green, and loggerhead sea turtles. NMFS issued another ESA section 10 permit to the NCDMF in 2005 covering incidental takes through 2010. The new permit estimated the take of 41, 168, and 41 for Kemp's ridley, green, and loggerhead sea turtles, respectively. As described in section 4.4.3.1 below, NMFS has also taken regulatory action to address the potential for sea turtle interactions with gillnet gear with ≥7 inch (17.9 cm) stretched mesh fished in Federal waters off of North Carolina and Virginia.

Regulations are in place for nearshore gillnetting off South Carolina, Georgia, and Florida as well. Georgia and South Carolina prohibit gillnets for all but the shad fishery, and Florida banned all but very small nets in state waters. Although many states have imposed strict regulations on gillnetting, the practice still occurs off some states' waters and in Federal waters. The nearshore and inshore gillnet fisheries off North Carolina are of particular concern due to the incidental captures (both lethal and non-lethal) of loggerhead, leatherback, Kemp's ridley, and green sea turtles (Wendy Teas, pers. comm., Joanne Braun-McNeill pers. comm.). In June 2009, 11 sea turtle captures (6 greens, 3 Kemp's ridleys, and 2 loggerheads) occurred over a one-week period in the southern flounder anchored sink gillnet fishery in Core Sound, North Carolina (NEFSC Fisheries Sampling Branch [FSB] database). Illegal gillnet incidental captures have also been reported in South Carolina and Florida (NMFS SEFSC 2001).

An *Atlantic croaker fishery* using trawl and gillnet gear also occurs within the action area and incidental takes of sea turtle have been observed in the fishery. Using VTR data from 2000-2004 and the average annual bycatch of loggerheads as reported in Murray (2006), the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear for trips primarily landing Atlantic croaker was estimated to be 41 loggerhead sea turtles for that 5-year period (Murray 2008). Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the Atlantic croaker fishery, has been published in Murray (2009a). 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 for the 5-year annual average of 3-20 (Murray 2009a). ESA-listed large whales have also been known to interact with gillnet gear, thus interactions may occur where the gear and large whale distributions overlap. A humpback whale mortality was recorded in 2001 as a result of entanglement in sink gillnet gear used in the croaker fishery (Waring *et al.* 2009).

The *weakfish fishery* occurs in both state and Federal waters, but the majority of commercially and recreationally caught weakfish are caught in state waters (ASMFC 2002). The dominant commercial gears include gillnets, pound nets, haul seines, 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 3.2.1, sea turtle bycatch in the weakfish fishery has occurred (Murray 2008, 2009a). The average annual bycatch of loggerhead sea turtles in bottom otter trawl gear for trips primarily landing weakfish during the period of 2000-2004 was estimated to be 4 loggerhead sea turtles per year (Murray 2008). Additional information on sea turtle interactions with gillnet gear, including gillnet gear used in the weakfish fishery, has been published in Murray (2009a). 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 for the 5-year annual average of 0-1 (Murray 2009a). ESA-listed large whales have also been known to interact with gillnet gear; thus, interactions may occur where the gear and large whale 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 2007c). The whelk fishery has been verified as the fishery involved in 13 sea turtle entanglements collectively in Massachusetts, New Jersey, and Virginia from 2002 to 2008. These whelk pot incidental takes have involved 8 leatherbacks, 4 loggerheads, and 1 green sea turtle, and have occurred in the months of May, June, July, August, and October (Northeast Region Sea Turtle Disentanglement Network [STDN] database).

Various *crab fisheries*, such as horseshoe crab and blue crab, also occur in Federal and state waters and may be detrimental to sea turtles as a result of entanglement or entrapment in the pot/trap gear used. The Virginia blue crab fishery has been verified as the fishery involved in three sea turtle entanglements from 2002 to 2008. Two entanglement events involved a leatherback sea turtle and one involved a loggerhead (Northeast Region STDN database).

These crab fisheries may also 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, although it remains a possibility. 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 significantly impacting the forage base for loggerheads in some areas of their range.
An *American lobster trap fishery* also occurs in state waters of New England and the Mid-Atlantic and is managed under the ASMFC's ISFMP. Like the Federal waters component of the fishery, the state waters fishery has also been identified as a source of gear causing injuries to

and mortality of large whales as well as loggerhead and leatherback sea turtles as a result of entanglement in vertical buoy lines of the pot/trap gear. Between 2002 and 2008, the lobster trap fishery in state waters was verified as the fishery involved in at least 27 leatherback entanglements in the Northeast Region. All entanglements involved the vertical line of the gear. These verified/confirmed entanglements occurred in Maine, Massachusetts, and Rhode Island state waters from June through October (Northeast Region STDN database).

The *Virginia pound net fishery* has also been documented as a source of sea turtle interactions. Pound nets with large-mesh leaders set in the Chesapeake Bay have been observed to lethally capture sea turtles as a result of entanglement in the pound net leader. As described in section 4.4.3.3 below, NMFS has taken regulatory action to address sea turtle interactions with the Virginia pound net fishery.

Incidental takes of loggerheads in fish traps have also been reported from several Atlantic coast states (Shoop and Ruckdeschel 1989; Wendy Teas, pers. comm.). Long haul seines and channel nets are also known to incidentally take loggerheads and other sea turtles in sounds and other inshore waters along the U.S. Atlantic coast, although no lethal takes have been reported (NMFS SEFSC 2001).

Observations of state recreational fisheries have shown that loggerhead, leatherback, and green sea turtles are known to bite baited hooks, and loggerheads frequently ingest the hooks. Hooked sea turtles have been reported by the public fishing from boats, piers, beaches, banks, and jetties, and from commercial fishermen fishing for snapper, grouper, and sharks with both single rigs and bottom longlines (NMFS SEFSC 2001). A summary of known impacts of hook-and-line captures on loggerhead sea turtles can be found in the TEWG (1998, 2000, 2009) reports.

4.2 Vessel Activity and Military Operations

Potential sources of adverse effects to large whales and 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 ESA-listed species.

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

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

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

Similarly, operations of vessels by other Federal agencies within the action area (NOAA, EPA, and ACOE) may adversely affect large whales 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. From 2009 on, NOAA research vessels conducting fisheries surveys for the NEFSC are estimated to take no more than nine sea turtles per year (eight alive, one dead). This includes up to seven loggerheads as well as an additional loggerhead, leatherback, Kemp's ridley, or green sea turtle per year during bottom trawl surveys and one loggerhead, leatherback, Kemp's ridley, or green sea turtle per year during scallop dredge surveys (NMFS 2007d).

4.3 Other Activities

4.3.1 Hopper Dredging

The construction and maintenance of federal navigation channels and sand mining ("borrow") areas have also been identified as sources of sea turtle mortality. Hopper dredges move relatively rapidly (compared to sea turtle swimming speeds) and can entrain and kill sea turtles, presumably as the drag arm of the moving dredge overtakes the slower moving sea turtle. Along the Atlantic coast of the southeastern U.S., NMFS estimates that annual observed injury or mortality of sea turtles from hopper dredging may reach 35 loggerhead, 7 green, 7 Kemp's ridley, and 2 hawksbill sea turtles (NMFS 1997b).

Further north, 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 incidental take of eight loggerhead sea turtles and one Kemp's ridley or green sea turtle. Actual dredging did not begin until May 1998, and no sea turtle interactions were observed during the 1998 dredge cycle. In June 2001, the ACOE indicated that the next dredge cycle, which was scheduled to begin in the summer of 2002, would require 1.5 million cy of sand initially, with an anticipated 1.1 million cy every two years thereafter. Although the volume of sand had increased from the previous cycle, NMFS reduced the ITS to five loggerheads and one Kemp's ridley or green sea turtle due to the lack of observed interactions in the previous cycle, along with the levels of anticipated and observed incidental take in hopper dredging projects in nearby locations.

In January 1996, NMFS completed section 7 consultation on the USN'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 incidental take of 10 loggerheads and one Kemp's ridley or green sea turtle during each dredge cycle. However, no interactions were observed during the 1996 cycle. The USN 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 incidental take of four loggerheads and one Kemp's ridley or green sea turtle during each dredge cycle. NMFS concluded that this level of incidental 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 large whales and sea turtles. The effects of fishing vessels, recreational vessels, or other types of commercial vessels on ESA-listed species may involve disturbance or injury/mortality due to collisions or entanglement in anchor lines. It is important to note that minor vessel collisions may not kill an animal directly, but may weaken or otherwise affect it so it is more likely to become vulnerable to effects such as entanglement. Listed species may also be affected by fuel oil spills resulting from vessel accidents. Fuel oil spills could affect animals through the food chain. However, these spills typically involve small amounts of material that are unlikely to adversely affect listed species. Larger oil spills may result from severe accidents (such as the BP Deepwater Horizon oil spill), although these events would be rare. No direct adverse effects on ESA-listed whales and sea turtles resulting from fishing vessel fuel spills have been documented.

NMFS has completed section 7 consultations for the issuance of permits to allow for the construction and operation of three Liquid Natural Gas (LNG) terminals within the action area of this consultation (Broadwater, Neptune, and Northeast Gateway). In a 2008 informal consultation, NMFS concluded that the construction and operation of the Broadwater facility was not likely to adversely affect ESA-listed large whales or sea turtles. NMFS has concluded that the construction and operation of the Neptune and Northeast Gateway facilities is likely to adversely affect, but will not jeopardize the continued existence of ESA-listed right, humpback, and fin whales, and is not likely to adversely affect ESA-listed loggerhead, leatherback, Kemp's ridley, and green sea turtles (NMFS 2007e, 2010b). In regards to ESA-listed sei whales, NMFS has determined that the Neptune LNG facility is likely to adversely affect, but not jeopardize the continued existence of the species while the Northeast Gateway facility is not likely to adversely affect the species (NMFS 2007e, 2010b).

4.3.3 Pollution

Anthropogenic sources of marine pollution, while difficult to attribute to a specific Federal, state, local, or private action, may affect large whales and sea turtles in the action area. Sources of pollutants in the action area include atmospheric loading of pollutants such as PCBs; storm water runoff from coastal towns, cities, and villages; runoff into rivers emptying into bays; groundwater discharges; sewage treatment plant effluents; and oil spills. The pathological effects of oil spills on sea turtles have been documented in several laboratory studies (Vargo *et al.* 1986). Marine debris (*e.g.*, discarded fishing line or lines from boats) can entangle large whales or sea turtles causing serious injury or mortality. Sea 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 species' stomach contents (Magnuson *et al.* 1990).

Nutrient loading from land-based sources, such as coastal communities and agricultural operations, is known to stimulate plankton blooms in closed or semi-closed estuarine systems. The effect to larger embayments is unknown. Contaminants could 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 Global climate change and ocean acidification

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

The effects of global climate change are typically viewed as being detrimental to sea turtles (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 (Nicholls 1998; Church *et al.* 2001; Fish *et al.* 2005; Baker *et al.* 2006; IPCC 2007; 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 (Goldenberg *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 Guyana 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 behaviorally adapt to such changes (Girondot and Fretey 1996; Plaziat and Augustinius 2004; Rivalan *et al.* 2005; Kelle *et al.* 2007; Pike and Stiner 2007; Witt *et al.* 2008).

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. (Rahmstorf 1997, 1999; Stocker and Schmittner 1997; Gagosian 2003; NMFS and USFWS 2007a; 2007b; 2007c; 2007d). 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 *et al.* 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 sea 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 temperatures in the middle third of the incubation period with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25°-35°C (Ackerman 1997). Based on modeling, a 2°C increase in air temperature is expected to result in a sex ratio of over 80% female offspring for loggerhead nesting beaches in the vicinity of Southport, North Carolina. 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 sea 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. (Hamann *et al.* 2007; Hawkes *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).

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.

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.

The impact of climate change on ESA-listed large whales is likely to be related to changes in sea temperatures, potential freshening of sea water due to melting ice and increased rainfall, sea level rise, the loss of polar habitats, and potential shifts in the distribution and abundance of prey species. Of the main factors affecting distribution of large whales, water temperature appears to be the main influence on geographic ranges of these 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 effect 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 effect on sei whales, allowing them to expand their range into higher latitudes (MacLeod 2009).

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

The direct effects of increased CO_2 concentrations, and associated decrease in pH (ocean acidification), on large whales 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 large whale 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 (Tynan and DeMaster 1997; Waluda *et al.* 2001; Learmonth *et al.* 2006). These changes will likely have several indirect effects on large whales, 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 large whales (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 acquisition 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 collects data on dead sea turtles and rescues and rehabilitates live stranded ones. 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 72° 30' 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 35° 46.3' N, north of 35° 00' N, and east of 76° 30' W.

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 sea 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 37° 19' N and west of 76° 13' W, and all waters south of 37° 13' N to the Chesapeake Bay Bridge Tunnel at the mouth of the Chesapeake Bay, and the James and York Rivers downstream of the first bridge in each tributary. 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 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 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 MMPA and has been developed by NMFS. The ALWTRP covers the U.S. Atlantic EEZ from Maine through Florida (26° 46.5' N). The requirements are year-round in the Northeast, and seasonal in the Mid- and South Atlantic.

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

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 exert 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 areas finalized by April 2011;

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

- Development of vertical line and whale distribution co-occurrence overlays. Time frame: by October 2010 for the Northeast and April 2011 and Mid- and South Atlantic;

- 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; and

- 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. Memoranda of Agreement were also issued between NMFS and other Federal agencies to increase the resources available to respond to reports of entangled large whales anywhere along the eastern seaboard of the U.S. 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 yards. 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 yards, 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 U.S., a proposal to the International Maritime Organization (IMO) requesting approval of a mandatory ship reporting system (MSR) in two areas off the U.S. east coast: 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 U.S. 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 U.S. 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). In view of uncertainties these restrictions will have on large whales and the burdens imposed on vessel operators, 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 U.S. Coast Pilots, and mariners have been notified through USCG Notices to Mariners.

Through a joint effort between NOAA and the USCG, the U.S. 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 Stellwagen Bank National Marine Sanctuary (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% from 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 Team (ATGTRT)

In 2006, NMFS convened a take reduction team (TRT) to address the incidental mortality and serious injury of long- and short-finned pilot whales, common dolphins, and white-sided dolphins incidental to bottom and mid-water trawl fisheries operating in both the Northeast and Mid-Atlantic regions. Under section 118 of the MMPA, the ATGTRT is responsible for developing a TRP to reduce mortality and serious injury of pilot whales, common dolphins, and white sided dolphins in the Atlantic trawl gear fishery. The immediate goal of the TRP is to reduce, within six months of implementation, the incidental mortality and serious injury of marine mammals to levels less than the stock's PBR level. The long term goal is to reduce within five years of implementation, the mortality and serious injury of marine mammals to insignificant levels approaching a zero mortality and serious injury rate. In December 2008, the ATGTRT published its take reduction strategy for trawl gear, which can be found at http://www.nero.noaa.gov/prot_res/atgtrp/. This strategy may also benefit ESA-listed whale and sea turtle species in the action area.

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 turtles and large whales are present. However, if closures shift effort to areas with a comparable or higher density of large whales 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 in the action area of future State, tribal, local, or private actions that are reasonably certain to occur. 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 large whales 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, and pollution. While the combination of these activities may affect populations of ESA-listed whales 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. The NRC (1990) 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 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 (Glass *et al.* 2008; Waring *et al.* 2009). 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's STSSN data indicate that vessel interactions are responsible for a large number of sea turtle strandings within the action area each year. In the U.S. Atlantic from 1997-2005, 14.9% of all stranded loggerheads were documented as having sustained some type of propeller or collision injuries (NMFS and USFWS 2007a). The incidence of propeller wounds rose from approximately 10% in the late 1980s to a record high of 20.5% in 2004 (STSSN database). Such collisions are reasonably certain to continue into the future. Collisions with boats can stun, injure, or kill sea turtles, and many live-captured and stranded sea turtles have obvious propeller or collision marks (Dwyer *et al.* 2003). However, it is not always clear whether the collision occurred pre- or post-mortem. As a result, an estimate of the number of sea turtles that will likely be killed by vessels is not possible.

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 4.4.6, 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 or shipping lanes. The program does not require reduced speeds in all areas where right whales may occur. Although these measures are designed to reduce takes of ESA-listed large 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 causing pollution are reasonably certain to continue in the future, as are impacts from them on large whales and sea turtles in the action area. However, the level of impacts cannot be projected. Marine debris (*e.g.*, discarded fishing line or lines from boats) can entangle large whales and sea turtles in the water and potentially drown them. Sea 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, sea turtles are not very easily affected by changes in water quality or increased suspended sediments, but if these alterations make habitat less suitable for them and hinder their capability to forage, eventually they would leave or avoid these less desirable areas (Ruben and Morreale 1999). 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 sea turtles (Vargo *et al.* 1986). There have been a number of documented oil spills in the northeastern U.S.

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 right whales and that inhalation may be an important exposure route (Wise *et al.* 2008). The impacts of biotoxins on large whales 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.* 2010). 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 large whales but may be a concern for other marine organisms, including sea turtles. The potential effects of noise pollution on large whales 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. 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 a large whale's ability to feed and communicate (Richardson *et al.* 1995). Masking is a major concern about shipping, but only a few species of large whales have been observed to demonstrate behavioral changes to low level sounds. NMFS is in the process of developing a comprehensive acoustic policy that will provide guidance on assessing the impacts of anthropogenically produced sound on marine mammals. In the interim, NMFS's current thresholds for determining impacts to marine mammals typically center around root-mean-square (RMS) received levels of 180 dB re 1μPa for potential injury, 160 dB re 1μPa for behavioral disturbance/harassment from an impulsive noise source (*e.g.*, seismic survey), and 120 dB re 1μPa for behavioral disturbance/harassment from a continuous noise source (*e.g.*, dredging). These thresholds are based on a limited number of experimental studies on captive odontocetes and pinnipeds, a limited number of controlled field studies on wild marine mammals, observations of marine mammal behavior in the wild, and inferences from studies of hearing in terrestrial mammals. In addition, marine mammal responses to sound can be highly variable, depending on the individual hearing sensitivity of the animal, the behavioral or motivational state at the time of exposure, past exposure to the noise which may have caused habituation or sensitization, demographic factors, habitat characteristics, environmental factors that affect sound transmission, and non-acoustic characteristics of the sound source, such as whether it is stationary or moving (NRC 2003). 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 effects to sea turtles and large whales 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 large whales, and with global climate change the range of these species may be altered. Ocean acidification may also have an adverse impact on the prey for these species 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 large whales 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.

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 bluefish fishery are analyzed to determine whether the action is likely to jeopardize the continued existence of ESA-listed species in the action area. Past effects of the bluefish fishery are included in this "baseline." To the extent available information allows, this baseline (which does not include the future effects of the bluefish 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 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 large whales and sea turtles 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 a single species classified as "endangered" under the ESA. The 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 and 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, 1997; 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 August 1, 2008, there were 368 individually identified right whales in the photo-identification catalog that were presumed to be alive (Hamilton *et al.* 2008). An additional 135 were presumed to be dead as they had not been sighted in the past six years (Hamilton *et al.* 2008). Examination of the minimum number of right whales alive as calculated from the sightings database indicate 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 October 10, 2008, for the years 1990-2005, 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.* 2009). 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 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 due to entanglements, nine due to ship strikes (Glass *et al.* 2009). Serious injury was documented for an additional three 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 (Angliss and Outlaw

2007; Carretta *et al.* 2007). Recent research efforts via the SPLASH Project estimate the abundance of humpback whales to be just under 20,000 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 1980s and 1990s (Angliss 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 Southern Hemisphere humpbacks were given protection by the IWC in 1963, Soviet whaling data made available in the 1990s revealed that they continued to be hunted through 1980 (IWC 1995; Zemsky *et al.* 1995; Perry *et al.* 1999).

Photographic mark-recapture analyses from the 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). 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).

In light of the above, NMFS considers the trend for humpback whales as a species to be increasing. However, NMFS also recognizes that there are many on-going negative impacts to the species across all areas of its range and to all age classes. Therefore, caution should also be exercised in considering the overall effect to the species given the available information and its classification as an "endangered" species under the ESA.

Fin whales are listed as a single species classified 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 CPUE 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, 2,814, 2,933, and 1,925 in 1995, 1999, 2002, and 2004, respectively (Palka 2006; Waring *et al.* 2009). The 2009 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[8], only 7 were of entanglements (all confirmed), two of which were fatal (Cole *et al.* 2005). Ten ship strikes were reported, five of which were confirmed and proved fatal. Of 61 fin whale events recorded between 2003 and 2007, eight mortalities were associated with vessel interactions, and three 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 a single species classified as "endangered" under the ESA. The IWC only considers one stock of sei whales in the North Pacific (Donovan 1991), but for NMFS's 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

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

(California, Oregon, and Washington waters out to 300 nm) is 46 (C.V. = 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 SARs, 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 (C.V. = 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 reported events for sei whales in the Atlantic from 2003-2007, one was confirmed as a serious injury resulting from entanglement in unidentified gear (Glass *et al.* 2009). The remaining seven 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 a single species classified 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; Bowen and Karl 2007). The Loggerhead 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 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 3.2 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 $1/10^{th}$ 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 peninsular 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 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. The TEWG (2009) report 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 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. 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 a single species classified as "endangered" under the ESA. Leatherbacks are widely distributed throughout the oceans of the world, and are found in waters of the Atlantic, Pacific, and Indian Oceans, the Caribbean Sea, Mediterranean Sea, and the Gulf of Mexico (Ernst and Barbour 1972). Leatherback nesting occurs on beaches of the Atlantic, Pacific, and Indian Oceans as well as in the Caribbean (NMFS and USFWS 2007b).

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

There are some population estimates for leatherback sea turtles although there appears to be considerable uncertainty in the numbers. In 1980, the global population of adult leatherback females was estimated to be approximately 115,000 (Pritchard 1982). By 1995, this global population of adult females was estimated to be 34,500 (Spotila *et al.* 1996). However, the most recent population size estimate for the North Atlantic alone is 34,000-94,000 adult leatherbacks (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 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 trend in the number of nests laid is not a reflection of the overall trend in any leatherback population given that the proportion of adult males to females and the age structure of the population(s) 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 3.2 and 4.0).

Nest count data provides the best available information on the number of adult females nesting each year. As is the case with the other sea turtles species discussed above, nest count data must be interpreted with caution given that these estimates provide a minimum count of the number of nesting Kemp's ridley sea turtles. In addition, the estimates do not account for adult males or juveniles of either sex. Without information on the proportion of adult males to females, and the age structure of the Kemp's ridley population, nest counts cannot be used to estimate the total population size and, similarly, trends in the number of nests laid cannot be used as an indicator of the population trend (whether decreasing, increasing, or stable) (Meylan 1982; Ross 1996; Zurita *et al.* 2003; Hawkes *et al.* 2005; 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 2007c). Nest count data indicate increased nesting and increased numbers of nesting females in the population. NMFS also considers the species' classification of endangered under the ESA, the caveats associated with using nesting data as indicators of population size and population trends, that the estimated number of nesting females in the current population is still far below historical numbers (Stephens and Alvarado-Bremer 2003; NMFS and USFWS 2007c), the many on-going negative impacts to the species, and that the majority of nesting for the species occurs in one area.

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 1991b; 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. Also, the trend in the number of green sea turtle nests laid is not an indication of the overall population trend given that the proportion of adult males to females and the age structure of the population(s) are unknown. Given the late age to maturity for green sea turtles (20 to 50 years) (Balazs 1982; Frazer and Ehrhart 1985; Seminoff 2004), caution is urged regarding the trend for any of the nesting groups since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d).

6.0 EFFECTS OF THE ACTION

Pursuant to Section 7(a)(2) of the ESA (16 U.S.C. 1536), Federal agencies are directed to ensure that 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 ESA-listed species within the action area to determine if the continued operation of the bluefish fishery, within the constraints of the Bluefish FMP, is likely to jeopardize the continued existence of those species. This analysis is done after careful review of the listed species status and the factors that affect the survival and recovery of those species, as described above.

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

As described in Section 3.0, NMFS has determined that ESA-listed right, humpback, fin, and sei whales as well as loggerhead, leatherback, Kemp's ridley, and green sea turtles may be adversely affected by the continued operation of the bluefish fishery as a result of capture and entanglement in gear associated with the fishery. NMFS's assessment of the effects of ESA-listed species interactions with bluefish gear is provided below in order for NMFS to make a determination as to whether the proposed action is likely to jeopardize the continued existence of these species.

6.1 Approach to the Assessment

NMFS generally approaches a jeopardy analysis in three steps. The first step identifies the probable direct and indirect effects of an action on the physical, chemical, and biotic environment of the action area, including the effects on individuals of threatened or endangered species. The second step determines the reasonableness of expecting threatened or endangered species to experience reductions in reproduction, numbers, or distribution in response to these effects. The third step determines if any reductions in a species' reproduction, numbers, or distribution (identified in the second step of the 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 large whales and sea turtles resulting from fishing gear used in the bluefish 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; Glass *et al.* 2009; Waring *et al.* 2009; Northeast Region STDN database), (2) captures of loggerhead sea turtles in bottom otter trawl gear and sink gillnet gear where effort in the bluefish fishery and sea turtle distribution overlap (Murray 2008, 2009a), (3) information on the capture of other sea turtle species in bottom otter trawl and gillnet gear in other fisheries where the bluefish fishery also operates, (4) life history information for large whales and sea turtles, and (5) the effects of fishing gear entanglements on large whales and sea turtles that has been published in a number of documents. These sources include status reviews and biological reports (TEWG 2000, 2007, 2009; NMFS SEFSC 2001; Moore *et al.* 2004; Johnson *et al.* 2005; NMFS and USFWS 2007a, 2007b, 2007c, 2007d; Glass *et al.* 2009; Waring *et al.* 2009), recovery plans (NMFS 1991, 2005; 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

Since 1985, bottom otter trawls and gillnets have been the predominant gear types utilized in the commercial bluefish fishery, while the use of "other" gear types (*e.g.*, haul seines, paired trawls, purse seines, pound nets, troll and hand lines) has consistently remained at low levels or has significantly declined in importance (MAFMC and ASMFC 1998). Recreational fishermen primarily catch bluefish using rod and reel gear, which generally does not pose a significant risk to ESA-listed species except for occasional hooking events or entanglement risks posed by active or discarded lines. Due to the rarity of use of "other" gear types in the commercial fishery and a lack of information on impacts from hook and line gear in the recreational fishery, these gear types will not be analyzed further in this Opinion (aside from a brief discussion in section 6.2.2 on the lack of information on impacts from hook and line gear in U.S. Atlantic waters).

Bottom trawls are typically cone-shaped nets which are towed on the bottom. Large, rectangular doors attached to the two cables keep the net open while deployed. At the bottom of an otter trawl mouth is the footrope or ground rope that can bear many heavy (tens to hundreds of kilograms) steel weights (bobbins) that keep the trawl on the seabed. In addition, bottom trawls may be constructed with large (up to 40 cm diameter) rubber discs or steel bobbins (rockhoppers) that ride over structures such as boulders and coral heads that might otherwise snag the net. Some trawls are constructed with tickler chains that disturb the seabed to flush shrimp or fishes into the water column to be caught by the net. The constricted posterior netting of a trawl is called the codend.

Gillnets are panels of net anchored in some form, with a top rope, referred to as the head rope or floating line, and a bottom rope, referred to as the lead line. As the name implies, floats are attached to floating line while the lead line is weighted to help maintain the vertical profile of the gillnet in the water column. Multiple net panels are typically attached together in series to form a net-string. Buoy lines attached to each end of a net string rise to the surface to mark the location of the gear. Gillnets fish by presenting a wall of netting in which fish are incidentally snagged or entangled. In some areas, fishermen either choose or are required to reduce the vertical profile of their gillnets by using "tie-downs". Tie-downs refer to twine used between the floatline and the lead line as a way to create a pocket or bag of netting to trap fish. Fishermen may use tie-downs in order to better entangle bottom species (monkfish or flounder) in the gillnet or to reduce vertical profile of the net to minimize protected species entanglements.

Data indicate that gillnet gear like that used in the bluefish fishery has the capacity to seriously injure large whales and/or sea turtles. However, it is often difficult to assess gear found on stranded animals or observed on animals at sea and assign it to a specific fishery. Only a fraction of the takes are observed, and the catch rate represented by the majority of takes, which are reported opportunistically, (*i.e.*, not as part of a random sampling program), is unknown. Consequently, documented takes provide an underestimate of interactions and the total level of interactions often cannot be determined through extrapolation, except in the case of the Murray (2009a) bycatch estimate report for loggerheads, which is referred to throughout this Opinion.

As previously described, ESA-listed large whales are not reasonably likely to be captured in bottom otter trawl gear. They may, however, become entangled in lines associated with anchored sink gillnet gear. ESA-listed sea turtles are reasonably likely to be captured or entangled in bottom otter trawl gear as well as sink gillnet gear (either the lines associated with it or the mesh itself) if gear overlaps with the distribution of sea turtles.

Large whale interactions with gillnet gear used in the bluefish fishery can take the form of entanglements of the head, flippers, or fluke. The effects of entanglement can range from no injury to death. Polypropylene (floating) lines between the buoy line and anchor line have been identified as a serious entanglement risk to large whales. Floating line can become entangled in baleen when the animal is moving through the water with the mouth gaped for feeding. Knots in the line further hinder the ability of the line to pass through the baleen. In addition, anchors on the gear offer resistance against which the whale may struggle and result in further entanglement of the fishing gear across the mouth and/or body of the whale.

Sea turtle interactions with trawl and gillnet gear used in the fishery can take the form of entanglements of the head, limbs, or carapace or captures of the entire animal. Captures of sea turtles in gillnets are an extremely severe type of interaction as they most often result in serious injury and death. Gillnets are so effective at catching sea turtles they were commonly used in the historical sea turtle fishery. Drowning may occur immediately as a result of forced submergence or, at a later time, if trailing gear becomes lodged between rocks and ledges below the surface. Although drowning due to forced submergence is the most obvious threat to sea turtles, constriction of a sea turtle's neck and flippers can amputate limbs also leading to death by infection or to impaired foraging or swimming ability. Animals that do escape often retain pieces of gear that can inhibit their foraging or survival. If the animal is cut loose with line attached, the flipper may eventually become occluded, infected, and necrotic.

Documented cases have indicated that entangled large whales and sea turtles may travel for extended periods of time and over long distances before freeing themselves, being disentangled, or dying as a result of the entanglement (Angliss and Demaster 1998). Entanglements may lead to exhaustion and starvation due to increased drag (Wallace 1985). A sustained stress response, such as repeated or prolonged entanglement in gear makes these species less able to fight infection or disease, and may make them more prone to boat/ship strikes and predation (Lutcavage *et al.* 1997). Younger animals are particularly at risk if the entangling gear is tightly wrapped since the gear will become more constricting as they grow.

For trawls specifically, the use of TEDs and the length of the tows can influence the level of harm to sea turtles. For both gear types it is often difficult for an observer to tell if an animal released alive has been injured to the point of influencing its future survival potential. Consequently, since most data on interactions with gear cannot be refined to further detail with respect to the level of effect as described above, the term "take" is sometimes used in this discussion to refer to these different types of potential interactions with bluefish gear.

6.1.2 Description of Incidental Take of Large Whales

ESA-listed large whales are known to become entangled in gear used by the bluefish fishery. The ALWTRT has the most complete and up to date large whale entanglement data set with contribution from the ALWDN. The ALWDN receives reports from a variety of sources, some of which include recreational boaters, commercial fishermen, USCG, NMFS aerial surveys, and research vessels. The MMHSRP also contributes to the collection of fishery interaction data. The Stranding Network evaluates stranded cetaceans and determines if commercial fishing activity was involved. NMFS has collectively analyzed both data sets and a summary is presented below.

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 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 these ESA-listed large whales and the North Atlantic bottom trawl fishery. Their great size and mobility presumably allows them to avoid interactions with the relatively slow moving trawl gear. There have been six (6) documented humpback whale interactions with hook and line gear, none of which were documented as serious injuries or mortalities. Interactions with hook and line gear and right, fin, and sei whales have not been observed.

Between January 1999 and 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 able to be determined (NMFS NERO 2010). Three (3) of the 12 sink gillnet interactions, including the one (1) right whale interaction, resulted in

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., Canadian, 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. Atlantic fisheries. Confirmed serious injury/mortality (SI/M) events, as reported in the U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessment Reports, 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)

serious injuries or mortalities. Within the same time period, an additional 14 (1 right whale and 13 humpback 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 and fishery for observed entanglement events are often not traceable by researchers, 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).

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 estimate 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 this approach does provide a methodology for estimating the full amount of entanglements, including those that result in serious injury or mortality, 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 Description of Incidental Take of Sea Turtles

Sea turtles incidentally taken in fishing gear must be reported to NMFS on VTRs that are required for the bluefish 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 bluefish fishing gear that follows will focus on trawl and gillnet gear. Past observed takes of sea turtles in trawl and gillnet gear were reviewed in the 1999 Opinion for the bluefish fishery. Updated information is provided herein. It is difficult to ascertain the gear types responsible for incidental takes of sea turtles when only portions of the gear or injuries resulting from the takes are observed. Additionally important to note is 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 off 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 Cape Cod. The spatial distribution of sea 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 sea turtles were documented as incidentally caught in Mid-Atlantic sink gillnet gear (Murray 2009a). Sea turtle takes in gillnet gear were not documented in the Northeast (east of Cape Cod and in the Gulf of Maine) during the 1995-2006 time period despite substantial observer coverage of gillnet hauls occurring in the Northeast. Sizes of the observed loggerheads caught in gillnet gear, for which measurements could be taken, ranged between 52 and 101 cm CCL with a mean of 65.3 cm CCL (n=12 turtles). Ten of the 12 (83%) loggerheads measured were under 72 cm CCL, a size considered to be within the juvenile life stage (NMFS and USFWS 2008; Murray 2009b). Approximately 40% of the loggerheads taken in gillnet gear were determined to be dead (Murray 2009a). Documented trawl and gillnet gear takes of loggerheads after the time periods analyzed in the Murray (2008, 2009a) reports are presented in Table 2.

Table 2. Documented incidental takes of loggerhead sea turtles (excluding moderately and severely decomposed sea turtles) in bottom otter trawl gear (fish, scallops, and twin) from 2005-2009 and gillnet gear from 2007-2009 along with the most landed commercial species (by weight) 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 scallops	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 and gillnet gear published in Murray (2008, 2009a) represent the best available information for and analysis of loggerhead bycatch in Mid-Atlantic commercial fisheries. These estimates are described further in Section 6.2.2. Such estimates are not available for leatherback, Kemp's ridley, and green sea turtles. Therefore, fisheries observer data for these species represent the best available information.

Two captures of leatherback sea turtles and one capture of an unidentified sea turtle have been reported in gillnet gear used to land bluefish between 2003 and 2004. The first leatherback was captured on August 28, 2003 by a drift sink gillnet, the second leatherback was captured on July 15, 2004 by a drift floating gillnet, and the unidentified sea turtle was captured on June 28, 2004 by a drift floating gillnet (NEFSC FSB database). The NEFOP has documented the most landed commercial species (by weight) per trip when an incidental take occurs (among many other variables), and that information has been used to look at the relative frequency that individual commercial fish species are associated with the incidental take of leatherback as well as Kemp's ridley and green sea turtles (Table 3).

Table 3. Documented incidental takes of leatherback, Kemp's ridley, green, and unidentified sea turtles (excluding moderately and severely decomposed sea turtles) in bottom otter trawl (fish, scallops, and twin) and gillnet gear from 2000-2009 along with the most landed commercial species (by weight) 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)	Longfin squid	Summer flounder	Atlantic croaker	Silver hake	Pollock	Atlantic sea scallops	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
Kemp's	0	1	1	0	0	0	1	0	5[9]	1	1	0	0	0	0
Green	1	0	0	0	0	0	1	2	12[9]	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 bluefish, using this number as the estimated take would be an underestimate in two ways. First, sea turtle takes could have occurred on trips where bluefish were 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,

[9] Five (5) Kemp's ridley and twelve (12) green sea turtles were observed incidentally taken in 2009 by a state fishery targeting southern flounder with sink gillnet gear in Pamlico Sound. Pamlico Sound is located at the southernmost point of the action area. All of these takes were documented by the NEFOP, and thus will be considered in this Opinion.

Kemp's ridley, and green sea turtles in fishing gear authorized under the Bluefish 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 sea turtles) in bottom otter trawl (BOT: fish and scallops) and gillnet gear from 2000-2009. Gillnet gear includes anchored sink gillnets and drift sink gillnets[10]. Source: NEFSC FSB database.

Species	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).

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. Sea turtles have also been observed to dive to the bottom and hunker down when alarmed by loud noise or gear (Memo to the File, L. Lankshear, December 4, 2007), which could place them in the path of bottom gear such as a 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, 1994; Morreale and Standora 2005; Seney and Musick 2005, 2007). 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 bluefish fishery operates, the sea turtles would be at risk.

[10] See footnote 9.

Tagging studies have shown that leatherbacks, which occur seasonally in western North Atlantic continental shelf waters where the bluefish 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), they are likely to spend more time in the water column than on the bottom. Given that leatherbacks forage primarily within the water column rather than on the bottom, interactions between 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 bottom trawl gear used (Table 4) or consistent with that used in the bluefish 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 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 where the netting of the gear is found. However, pelagic leatherbacks are also prone to becoming entangled in the buoy lines or surface systems of sink gillnets. Sea turtles are unlikely to be able to break off entangling fishing gear. Sea turtles are vulnerable to drowning from forced submergence, although some sea turtles have been recovered alive in sink gillnet gear.

6.1.4 Factors Affecting Large Whale Captures in Bluefish Fishing Gear

As stated in Section 6.1.2, there have been no documented interactions between ESA-listed large whales and the North Atlantic bottom trawl fishery. Their great size and mobility presumably allow them to avoid interactions with 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, large whales may become 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 where the bluefish fishery operates. As described in detail in section 3.1, right, humpback, fin, and sei 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 peaking 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).

Since the highest abundances of 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 gillnet gear for the bluefish fishery. Humpback and fin whales are utilizing the mid-Atlantic waters during October-March with seemingly increasing frequency and low numbers of whales may reside 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 bluefish 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 bluefish fishery operates, 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 Captures in Bluefish Fishing Gear

As described in section 3.2, the occurrence of loggerhead, leatherback, Kemp's ridley, and green sea turtles in New England, Mid-Atlantic, and south Atlantic waters is primarily temperature dependent (Thompson 1984; Keinath *et al.* 1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998, 2005; Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; James *et al.* 2005a). In general, sea turtles move up the U.S. Atlantic coast from southern wintering areas as water temperatures warm in the spring (Keinath *et al.* 1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998, 2005; Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; James *et al.* 2005a). The trend is reversed in the fall as water temperatures cool. By December, sea turtles have passed Cape Hatteras, returning to more southern waters for the winter (Keinath *et al.* 1987; Shoop and Kenney 1992; Musick and Limpus 1997; Morreale and Standora 1998, 2005; Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; James *et al.* 2005a). Recreational anglers have reported sightings of sea turtles in waters defined as inshore waters (bays, inlets, rivers, or sounds; Braun-McNeill and Epperly 2004) as far north as New York as early as March-April, but in relatively low numbers (Braun-McNeill and Epperly 2004). Greater numbers of loggerheads, Kemp's ridleys, and greens are found in inshore, nearshore, and offshore waters of North Carolina and Virginia from May through November and in inshore, nearshore, and offshore waters of New York from June through October (Keinath *et al.* 1987; Morreale and Standora 1993; Braun-McNeill and Epperly 2004). The hard-shelled sea turtles (loggerheads, Kemp's ridleys, and greens) appear to be temperature limited to water no further north than Cape Cod. Leatherback sea turtles have a similar seasonal distribution but have a more extensive range in the Gulf of Maine compared to the hard-shelled species (Shoop and Kenney 1992; Mitchell *et al.* 2003; STSSN database).

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

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

NMFS has also considered other factors that might affect the likelihood that ESA-listed sea turtles will be incidentally taken in bluefish 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 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 2002b). 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 skate gear may increase the risk of interactions between bluefish gear and ESA-listed sea turtles that would be attracted to these areas for feeding. However, at present there is no information to clearly indicate any of these as influencing ESA-listed sea turtle takes in bluefish fishing gear.

Given the seasonal distribution of sea turtles and the times and areas when the bluefish 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 New England. Based on the best, currently available information, sea turtle interactions with bluefish gear are likely at times when and in areas where sea turtle distribution overlaps with operation of the fishery.

6.2 Anticipated Effects of the Proposed Action

NMFS has identified that the proposed action is likely to adversely affect the four species of large whales and four species of sea turtles whenever they come into physical contact with bluefish fishing gear (gillnets and trawls). Such interactions have occurred in other fisheries utilizing the same types of gear and have resulted in injuries and even death to these species. Other effects to large whales and sea turtles as a result of the proposed action, including the effects of vessel strikes and impacts on the availability of prey, are expected to be insignificant or discountable, while the effects of recreational hook and line gear are unable to be assessed at this time due to a lack of information.

In this section of the Opinion, NMFS will determine, given the currently available information, the anticipated number of large whales and sea turtles that will be adversely affected by the continued operation of the bluefish fishery, defining such effects by species.

6.2.1 Anticipated take of large whales in bluefish gear

No method has yet been identified for predicting the level of overall or species-specific large whale bycatch in the bluefish fishery or any particular gear-type component of the fishery. Some large whale mortalities are likely never observed, thus the actual annual number of documented mortalities is likely a subset of the actual number of entanglement related serious injuries and mortalities that occur. Additionally, assignment of a specific fishery to an observed entanglement is rarely possible because 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 Opinion differs in an important way from the reporting in the Marine Mammal SARs. In this Opinion specifically, gear analysis results were used to categorize entanglement events to U.S., Canadian, or undefined origin. In contrast, the SARs 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 provide a confirmed country of origin for the involved gear. The location where an entangled whale is first sighted may be quite a distance from the original location of entanglement.

The objective of the SARs is to report on the status of marine mammal populations. The objective of this Opinion is to assess potential impacts to ESA-listed species due to the proposed action, which in this case is the continued operation of the bluefish fishery under the constraints of the Bluefish 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 attributable to U.S. fisheries or cannot 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)].

Right, humpback, fin, and sei whales occur in the areas where the bluefish fishery takes place. Nevertheless, none of these species 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, (2) the fisheries will not affect the availability of prey for these species, and (3) little (in regards to right whales) to no (in regards to the other three species) bluefish fishing effort occurs in low latitude waters where the vast majority of known calving and nursing occurs for these large whale species. Although large whale entanglements in hook and line gear have been documented, these are also rare events relative to gillnet entanglements and are thus believed to be discountable in terms of their effects.

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

From 1999-2008, gillnet gear was recorded in 26 entanglement events with ESA-listed large whales: two (2) involving right whales and 24 involving humpback whales (NMFS NERO 2010). That equates to a mean annual rate of 0.2 right whale entanglements and 2.4 humpback whale entanglements in gillnet gear used in all U.S. Atlantic fisheries per year. Six (6) of the 26 gillnet interactions resulted in serious injuries or mortalities (NMFS NERO 2010). In regards to sink gillnet gear specifically, there were 12 documented entanglements from gear of U.S. or undocumented origin from 1999-2008. This equates to a mean annual rate of 1.2 large whale entanglements per year in sink gillnet gear for the 1999-2008 time period (0.1 per year for right whales and 1.1 per year for humpback whales). Of those occasions, entanglements determined to be SI/M events resulted in a mean annual rate of 0.3 (0.1 per year for right whales and 0.2 per year for humpback whales). Considering the recent and 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 bluefish gear

Commercial trawls and gillnets. As described earlier in this Opinion, the Murray (2008, 2009a) reports analyze fishery observer data and VTR data from fishermen in order to estimate the takes of loggerhead sea turtles in bottom otter trawl 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 also divide the takes by FMP or fish species landed. These documents represent the most accurate predictor for loggerhead sea turtle takes in the bluefish fishery and other Mid-Atlantic fisheries that use these gear types.

It is important to note that while both reports divide the takes by individual species/species group landed (a proxy for FMP), the two reports use different methodologies. The trawl estimate (Murray 2008) assigned trips (and associated takes) to a single FMP/species group based on the most significant species landed (by weight) for that trip. The gillnet estimate (Murray 2009a) assigned trips (and associated takes) to multiple FMPs/individual species landed based on the distribution of landings for that trip. For example, trips in a certain time and area using 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 single FMP/species group. So if a trip landed 60% summer flounder, 20% spiny dogfish, and 20% weakfish, the trip and its associated takes (calculated using the take rate), were assigned to the summer flounder/scup/black sea bass group and that group 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 species landed – it just provides an average annual take level over the 2000-2004 time period. The gillnet estimate does provide a 95% CI around the annual point estimate for each species landed. Due to this difference, the takes assumed and analyzed for this Opinion are the point estimates for trawl gear and the upper end of the 95% CI for gillnet gear. This difference is also carried through into the Incidental Take Statement, and influences how the takes in the fishery will be monitored.

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

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 trips primarily landing bluefish during 2000-2004 as 3 loggerheads (Murray 2008). This estimate of loggerhead sea turtle takes in bottom otter trawl gear provides 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 3 loggerheads per year averaged over a 5-year period as the best available information for the anticipated take of loggerhead sea turtles in the bottom otter trawl component of the bluefish fishery in future years. Since this take estimate was calculated as a 5-year average from 2000-2004, future takes would be expected to average 3 per year over a 5-year period.

It should be noted that the Murray (2008) trawl bycatch estimate for loggerheads taken in the bluefish fishery could be an underestimate as bluefish taken as bycatch in other fisheries (*e.g.*, summer flounder) may have been attributed to those fisheries since the approach used in the trawl bycatch analysis grouped the catch according to dominant species. So, while it may be a slight underestimate of the loggerhead bycatch here for the Bluefish FMP, it would be offset by a slight overestimate of the loggerhead bycatch estimate for the Summer Flounder, Scup, and Black Sea Bass FMP.

From 2002-2006, the average annual bycatch estimate of loggerheads in Mid-Atlantic sink gillnet gear was 288 turtles (Murray 2009a). For the bluefish fishery, it is estimated that on average 48 loggerhead takes occur per year (with a 95% CI for the 5-year annual average of 23-79). With this CI, it would be expected that anywhere from 23-79 loggerheads could be taken annually over a 5-year period 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. For the purposes of this Opinion, NMFS is assuming that an average of up to 79 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 bluefish fishery.

There are no total bycatch estimates for the incidental take of leatherback, Kemp's ridley, or green sea turtles in either bottom otter trawl or gillnet fishing gear. As stated earlier in Section 6.1.3, NEFOP observers have documented incidental takes of three leatherback, two Kemp's ridley, one green, and five unidentified sea turtles in Mid-Atlantic bottom otter trawl gear and of three leatherback, eight Kemp's ridley, fifteen green, and nine unidentified sea turtles in Mid-Atlantic sink gillnet gear from January 2000 through December 2009 (NEFSC FSB database).

The very low number of observed leatherback incidental takes in bottom otter trawl and sink gillnet gear used in multiple trawl fisheries in the action area suggests that takes of leatherback sea turtles within the action area are rare events. However, given the generally low percentage of trips with observer coverage in the bluefish fishery as well as other trawl and gillnet fisheries in the action area, it is likely that some interactions with leatherback sea turtles, in addition to the two that were reported in 2003 and 2004, 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 bluefish gear, leatherback sea turtles are likely to be taken in either trawl or gillnet gear.

As summarized in Table 4, the annual average number of documented leatherback takes in bottom otter trawl and gillnet gear in the action area has been 0.3 and 0.3, respectively. Since the take of a partial sea turtle is not possible, NMFS anticipates the potential annual take of one leatherback sea turtle in bottom otter trawl and one in gillnet gear (either the netting or vertical lines) used in the skate fishery. Leatherbacks are the sea turtle species most susceptible to vertical line entanglements, and from 2002-2008 there were 41 leatherback interactions with vertical lines where the fishery could not be determined. Thus, it is reasonable to assume that an interaction in bluefish gillnet gear may be attributable to either a vertical line entanglement or entanglement in the mesh used to catch the target fish species. Additionally, because of the average annual take of 0.5 and 0.9 unidentified sea turtles in trawl and gillnet gear, respectively, another 2 sea turtles (which may be leatherbacks, Kemp's ridleys, or greens) are forecasted to be taken in the bluefish fishery annually. Thus, the continued operation of the bluefish fishery is anticipated to result in the annual non-lethal or lethal take of up to four leatherback sea turtles.

Likewise, the low number of observed Kemp's ridley and green sea turtle incidental takes in bottom otter trawl and sink gillnet gear used in multiple trawl fisheries in the action area suggests that takes of these sea turtles species within the action area are rare events. However, given the generally low percentage of trips with observer coverage in the bluefish fishery as well as other trawl and gillnet fisheries in the action area, it is likely that some interactions with Kemp's ridley and 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 these species with operation of bluefish gear, Kemp's ridley and green sea turtles are likely to be taken in trawl or gillnet gear.

As summarized in Table 4, the average annual number of documented Kemp's ridley takes in bottom otter trawl and gillnet gear in the action area has been 0.2 and 0.8, respectively. Since the take of a partial sea turtle is not possible, NMFS anticipates the potential annual take of one Kemp's ridley sea turtle in bottom otter trawl and one in gillnet gear used in the bluefish fishery. Additionally, because of the average annual take of 0.5 and 0.9 unidentified sea turtles in trawl and gillnet gear, respectively, another 2 sea turtles (which may be leatherbacks, Kemp's ridleys, or greens) are forecasted to be taken in the bluefish fishery annually. Thus, the continued operation of the bluefish fishery is anticipated to result in the annual non-lethal or lethal take of up to four Kemp's ridley sea turtles.

As also summarized in Table 4, the average annual number of documented green sea turtle takes in bottom otter trawl and gillnet gear in the action area has been 0.1 and 1.5, respectively. Since the take of a partial sea turtle is not possible, NMFS anticipates the potential annual take of one green sea turtle in bottom otter trawl and two in gillnet gear used in the bluefish fishery. Additionally, because of the average annual take of 0.5 and 0.9 unidentified sea turtles in trawl and gillnet gear, respectively, another 2 sea turtles (which may be leatherbacks, Kemp's ridleys, or greens) are forecasted to be taken in the bluefish fishery annually. Thus, the continued operation of the bluefish fishery is anticipated to result in the annual non-lethal or lethal take of up to five green sea turtles.

Other commercial gear types. As noted earlier, longlines, handlines, bandit gear, rod and reel, pots, traps, seines, and dredges are also used in the commercial bluefish fishery. Smaller marine mammals are known to be entangled or captured in haul seines, but interactions between haul seines and sea turtles have not been documented. Since the commercial fishery is small to begin with and each of these gear types represents a minute proportion of the total effort, it is unlikely that entanglement or capture in these types of gear would occur for any ESA-listed sea turtles.

Recreational fishery. Since the recreational fishery receives 80% of the annual bluefish quota and charter/recreational boats are commonly found throughout the action area, a significant amount of hook and line fishing occurs for bluefish. However, recent data from the MRFSS indicate that only a small percentage of recreational fishing activity for bluefish (an average of 9.9% from 2005-2009, in terms of landings) occurs in Federal waters where NMFS directly regulates the fishery (NMFS unpublished data). In state waters, the federal FMP sets the overall quota, but management of the recreational fishery is administered at the state level.

All four species of sea turtles discussed in this Opinion are known to ingest baited hooks or have their appendages snagged by hooks, both of which have been recorded in the STSSN database. Loggerhead and Kemp's ridley are the species caught most often, and frequently ingest the hooks. Hooked seas turtles have been reported by the public fishing from boats, piers, beaches, banks, and jetties (TEWG 2000). Most sea turtle captures on rod and reel, as reported to the strandings network, have occurred during pier fishing. Fishing piers are suspected to attract sea turtles that learn to forage there for discarded bait and fish carcasses. The amount of persistent debris, including monofilament line, fishing tackle, and other man-made items, has also been found to increase around piers, posing an additional threat to sea turtles in the area.

While there is at least some research on the effects of commercial longline fisheries on the capture of sea turtles, little data exist on the capture of sea turtles as a part of recreational hook and line fisheries. Deceased sea turtles found stranded with hooks in their digestive tract have been reported, though it is assumed that most sea turtles hooked by recreational fishermen are released alive. Some will break free on their own and escape with embedded/ingested hooks and/or trailing line. Others may be cut free by fishermen and intentionally released. These sea turtles will escape with embedded or swallowed hooks, or trailing varying amounts of monofilament fishing line which may cause post-release injury or death. The ingested hook and/or the trailing, monofilament fishing line may ultimately be swallowed and ingested by the animal, potentially leading to constriction and strangulation of the sea turtle's internal digestive organs; or the line may become entangled around the animal's limbs (leading to limb amputations) or around seafloor obstructions, preventing the animal from surfacing (leading to drowning). Thus, some of these hooking/entanglement interactions may eventually prove lethal.

However, the probability of hooking or entanglements in recreational hook and line gear is difficult to ascertain and very little data are available for the U.S. Atlantic to analyze impacts from this type of interaction on individual animals. In addition, it is often impossible to tell if the entangling gear is recreational or commercial. Based on this lack of information on the frequency, nature, or impact of interactions between recreational fishermen and sea turtles, NMFS is unable to determine the amount or extent of effects from recreational hook and line gear on sea turtles in the action area at this time. Nevertheless, it should be noted that the overall anticipated level of take of sea turtles in the bluefish fishery will be an underestimate since analyzable data are only currently available for the commercial component of the fishery.

6.2.2.1 Age classes of sea turtles anticipated to interact with the bluefish fishery

Loggerhead sea turtles. The five life stages recognized for loggerhead sea turtles are: (1) Year One: terrestrial to oceanic, size ≤15 cm straight carapace length (SCL); (2) Juvenile (Stage 1): exclusively oceanic, size range of 15-63 cm SCL; (3) Juvenile (Stage 2): oceanic or neritic, size range of 41-82 cm SCL; (4) Juvenile (Stage 3): oceanic or neritic, size range 63-100 cm SCL; and (5) Adult: neritic or oceanic, size ≥82 cm SCL (TEWG 2009). There are insufficient data available on loggerhead sea turtles incidentally taken in bluefish fishing gear to determine estimated sizes of future takes. However, based on observer measurements and the known distribution of loggerhead sea turtles captured in other Atlantic trawl and gillnet fisheries, NMFS expects that both juvenile and adult loggerheads may be captured in bluefish gear as a result of the continued operation of the fishery because both life stages are present within the action area.

Leatherback sea turtles. NMFS believes that leatherback sea turtles may be incidentally taken in bluefish 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 bluefish fishery operates (NMFS and USFWS 1992; NMFS SEFSC 2001). Tracking of tagged leatherbacks also demonstrate the movement of sexually mature leatherbacks over U.S. continental shelf waters (James *et al.* 2005a, 2005b). Therefore, either immature or sexually mature leatherback sea turtles could be taken in bluefish 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 only immature Kemp's ridley sea turtles are likely to be incidentally taken in bluefish 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). Thus, NMFS expects that only benthic immature green sea turtles are likely to be incidentally taken in bluefish fishing gear as a result of the continued operation of the fishery.

6.2.2.2 Estimated mortality of sea turtles captured in bluefish fishing gear

Capture and/or entanglement of sea turtles in bluefish trawl and gillnet gear likely results in a higher level of sea turtle mortality than is evident based on the number of sea turtles returned to the water alive. Injuries suffered by sea turtles captured in bluefish fishing gear fall into two main categories: (1) submergence injuries characterized by an absence or obvious reduction in breathing and consciousness with no other apparent injury, and (2) contact injuries characterized by entanglement of flippers and/or other body parts in the gear. The following information is provided as an assessment of the extent of these types of injuries likely to occur in the future for sea turtles affected by the continued operation of the bluefish 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 sea turtles rising from 0% for the first 50 minutes of capture to 70% after 90 minutes of capture (Henwood and Stuntz 1987). However, metabolic changes that can impair a sea turtle's ability to function can occur within minutes of a forced submergence. While most voluntary dives appear to be aerobic, showing little if any increases in blood lactate and only minor changes in acid-base status, the story is quite different in forcibly submerged sea turtles, where oxygen stores are rapidly consumed, anaerobic glycolysis is activated, and acid-base balance is disturbed, sometimes to lethal levels (Lutcavage and Lutz 1997). Forced submergence of Kemp's ridley sea turtles in shrimp trawls resulted in an acid-base imbalance after just a few minutes (times that were within the normal dive times for the species) (Stabenau *et al.* 1991). Conversely, recovery times for acid-base levels to return to normal may be prolonged. Henwood and Stuntz (1987) found that it took as long as 20 hours for the acid-base levels of loggerhead sea turtles to return to normal after capture in shrimp trawls for less than 30 minutes. This effect is expected to be worse for sea turtles that are recaptured before metabolic levels have returned to normal.

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

Tows by bottom otter trawl vessels are usually around one to two hours in duration, which should help to reduce the risk of death from forced submergence for sea turtles caught in bluefish trawl gear, but does not eliminate the risk. Assuming that the mortality rate for sea turtles from forced submergence in bluefish trawl gear is comparable to that measured for the shrimp fishery by Epperly *et al.* (2002) and Sasso and Epperly (2006), sea turtles may die as a result of capture and forced submergence in trawl gear used in the bluefish fishery.

As described above, NMFS anticipates the annual incidental capture of up to 82 loggerhead, four leatherback, four Kemp's ridley, and five green sea turtles in bluefish 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 sea 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 sea turtles that survive interactions with bottom otter trawl gear. Therefore, an estimated 43% of the estimated 3 loggerhead sea turtles taken per year (averaged over 5 years) in bluefish fishery trawl gear will be injuries or mortalities, which translates to an anticipated lethal take of 1.3 loggerhead sea turtles annually (averaged over a 5-year period). Since a part of a sea turtle cannot be taken, this number is rounded up to an average of 2 lethal loggerhead sea turtle takes in bluefish trawl gear per year averaged over a 5-year period. However, it should be noted that this is likely an over-estimate of the actual mortality rate as some injuries may be minor and not result in mortality.

As also discussed above, NMFS anticipates the annual take of up to 79 loggerhead sea turtles in gillnet gear. Murray (2009b) stated that about 40% of the observed loggerheads in gillnet fisheries from 1995 through 2006 were dead. Therefore, NMFS anticipates that 32 loggerhead takes in the gillnet component of the bluefish fishery could result in a lethal take.

For leatherback, Kemp's ridley, and green sea turtle takes 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.

6.2.2.3 Qualitative impacts of the bluefish fishery on sea turtles by region

New England region. In contrast to the Mid- and South Atlantic portions of the fishery, most commercial landings of bluefish in New England are from effort in the EEZ while the recreational fishery only occurs in New England waters from July through October (peaking in the 2-month period from August to September). The bluefish fishery is expected to adversely affect ESA-listed sea turtles in this region, although expected interactions in New England waters would be much lower than in the Mid-Atlantic because (1) overall fishing effort is much lower and (2) the abundance of sea turtles in this region throughout the year is lower as well.

Mid-Atlantic region. The bluefish fishery would be expected to have the most impact on sea turtles in this region. Comparing the Mid-Atlantic bluefish fishery with the Atlantic croaker, summer flounder, and monkfish fisheries, those which are the most similar to the bluefish fishery and for which NMFS has sufficient information, data in the NEFSC sea sampling database from 2003 to 2008 showed multiple captures or entanglements of sea turtles in gear targeting these species or which contained these species in the catch. During the times of year when sea turtles are present in areas being fished, bluefish fishing gear is just as capable of capturing sea turtles. Bycatch patterns logically follow the seasonal nature of both the fisheries and the sea turtles, with interactions being more prevalent off North Carolina in the winter and off New York and New Jersey during the summer. The time of year that would be expected to result in the greatest number of sea turtles interactions in the Mid-Atlantic would be from spring through fall.

South Atlantic region. The bluefish fishery would be expected to have the least impact on sea turtles in this region of the fishery. This is because effort for bluefish is comparatively low overall in the southeast U.S. compared to the other two regions, sea turtles are protected from trawlers in this fishery by TEDs in North Carolina (based upon their meeting the summer flounder definition), and gillnets are banned in most South Carolina, Georgia, and Florida state waters.

6.3 Summary of anticipated incidental take of whales and sea turtles in the bluefish fishery

The primary gear types used in the bluefish fishery are bottom trawls and sink gillnets. Large whale entanglements in trawls are believed to be extremely rare events relative to gillnet entanglements, especially since gillnets are the most frequently used gear type in the fishery.

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 whale 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 (NMFS NERO 2010; Table 1). All of the entanglement records for fin and sei whales over this 10-year period were with undetermined gear types. These entanglement rates for fin and sei whales are both below the PBR for each stock as referenced in the 2009 SAR. However, the total U.S. fishery-related mortality and serious injury for these stocks derived from the available records is not less than 10% of the calculated PBR, and therefore cannot be considered insignificant and approaching the Zero Mortality Rate Goal (ZMRG).

As shown above in Table 1, the annual SI/M for right and humpback whales from entanglements 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 (NMFS NERO 2010). The 2009 Marine Mammal SAR provides an annual mean rate of SI/M fishery gear entanglements to be 0.6 and 2.4, respectively for right and humpback whales in U.S. waters between 2003 and 2007 (Waring *et al.* 2009). Sink gillnet gear, a gear used in the bluefish fishery, has been documented in entanglements with right and humpback whales. Sink gillnet gear was verified to be involved with entanglements of one 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.

Based on the discussions above in this Opinion, including analysis of observer data and comparison to similar fisheries, the bluefish fishery is likely to have its greatest effect on sea turtles in the Mid-Atlantic area from spring through fall. These incidental takes are likely to occur during the use of both bottom otter trawl and gillnet gear.

Based on the best available information regarding sea turtle takes in gear utilized in the bluefish fishery, NMFS anticipates the annual incidental take of up to 3 loggerheads in trawl gear and 79 loggerheads in gillnet gear over a 5-year average as a result of the continued operation of the bluefish fishery. For leatherback and Kemp's ridley sea turtles, NMFS anticipates the annual incidental take of up to four of each species in either trawl or gillnet gear. For green sea turtles, NMFS anticipates up to five individuals to be taken annually in either trawl or gillnet gear.

Loggerhead and leatherback sea turtles taken in bluefish fishing gear are expected to include benthic immature and sexually mature individuals, while Kemp's ridley and green sea turtles taken in bluefish fishing gear are expected to include only benthic immature individuals. Loggerheads originating from nesting beaches of each of the five recognized nesting groups in the western North Atlantic may be taken in gear used in the bluefish fishery.

Murray (2008, 2009b) reported that 43% and 40% of observed loggerhead sea turtle takes in trawl and gillnet gear, respectively, resulted in mortality. For the analysis in this Opinion, it is assumed that 34 of the loggerhead sea turtles taken in the bluefish fishery will die or otherwise be seriously injured as a result of the capture. For leatherback, Kemp's ridley, and green sea turtles, it is assumed that each possible incidental take will lead to serious injury or mortality.

7.0 INTEGRATION AND SYNTHESIS OF EFFECTS

The *Status of Listed Species*, *Environmental Baseline*, and *Cumulative Effects* sections of this Opinion discuss the natural and human-related phenomena that caused ESA-listed species to become threatened or endangered and may continue to place those 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 continued operation of the bluefish fishery within the constraints of the current Bluefish FMP (described in Section 6.2) in the context of information presented in the status of the species, environmental baseline, and cumulative effects sections to determine: (a) if those effects due to the fishery would be expected to reduce the reproduction, numbers, or distribution of large whales and sea turtles, and (b) if any reduction in the reproduction, numbers, or distribution of large whales and sea turtles causes an appreciable reduction in the likelihood of these species surviving and recovering in the wild.

7.1 Integration and Synthesis of Effects on Large Whales and Sea Turtles

This Opinion has identified in Section 6 (*Effects of the Action*) that the proposed action, the continued operation of the bluefish fishery within the constraints of the current Bluefish FMP, may adversely affect right, humpback, fin, and sei whales as a result of entanglement in sink gillnet fished in the bluefish fishery. No other direct or indirect effects to ESA-listed large whales are expected as a result of the activity. This Opinion has also identified that the proposed action may adversely 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 bluefish fishery. No other direct or indirect effects to ESA-listed sea turtles are expected or, in the case of recreational hook and line fishing, are able to be quantified as a result of this activity. The following discussions in Sections 7.1.1 through 7.1.7 below provide NMFS's 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.

7.1.1 North Atlantic right whale

As described in the *Status of Listed 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 per year (U.S. waters, 0.6; Canadian waters, 0.2) (Waring *et al.* 2009). Most frequently an entanglement report does not contain the necessary information to assign the entanglements to a particular fishery. From 1999-2008, gillnet gear of U.S. or undocumented origin was recorded in two entanglement events with right whales, both of which led to SI/M (Table 1). Of those two events, sink gillnet gear was verified to be involved in one right whale entanglement.

For the purposes of this assessment, we are assuming that one 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 bluefish fishery, our assessment for this Opinion assumes that the serious injury or morality could and would occur as a result of the bluefish fishery.

As described in the *Status of Listed Species* section of this Opinion, the latest stock assessment report indicates that the population of North Atlantic right whales has grown at a rate of 1.8%

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 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 right whale (random life stage and sex) per year and one 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 animal (random life stage and sex), one 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 1.3% for status quo conditions to 2.1% for reductions in mortality equivalent to three animals per year.
- Status quo projections suggest a very low likelihood of extinction. No extinctions or quasi-extinctions were observed in the 1,000 projections.
- Only 2 of 1,000 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 less mortality per year, that probability went up to 14.7%, and with one less adult female mortality per year, the probability improved to 24.6%.

Effects on Survival and Recovery

In the NMFS/FWS 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 very low likelihood of the North Atlantic right whale going extinct or reaching a quasi-extinction level. None of the model projections actually predicted extinction or quasi-extinction. 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 (Franklin 1980; Shaffer 1981), 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 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 (0%) 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.

This model assumes 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 the North Atlantic Right Whale 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 1990s (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 criterion 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 criterion will be met. The population appears to be on the correct trajectory to meet the downlisting criterion 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 as identified in the species' recovery plan. It does not appear that the bluefish 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 Listed 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 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 will result in an accelerated rate of recovery.

Based on the analysis described above, the serious injury or mortality of one 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 in the North Atlantic. From 1999-2008, a total of 24 humpback whale entanglements were recorded in gillnet gear, four of which led to serious injury or mortality. This equates to an annual average of 0.4 SI/M events for humpbacks in gillnet gear, with 0.2 SI/M events per year attributable specifically to sink gillnet gear. During that same time period, the average number of documented SI/M events for humpbacks in all fishing gear of U.S. or unidentified origin was 3.4 per year (NMFS NERO 2010). Another accounting of SI/M 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 5-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 large 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.

The 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 SARs. 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 levels of take in U.S. fisheries are appreciably reducing the survival and recovery of the Gulf of Maine stock of humpback whales. If so, then NMFS 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 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 1992-2000, 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 SAR 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 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. A subsequent project, More of North Atlantic Humpbacks (MoNAH), was conducted from 2003-2005 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, 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 humpback 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 SAR 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 (Stevick *et al.* 2003; Waring *et al.* 2009). Additionally, the draft 2010 SAR reports that the 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 SAR does conclude that human impacts (vessel collisions and entanglements) may be slowing the recovery of humpback whale populations. The question for this Opinion is whether impacts associated with fishing authorized under the Bluefish FMP are likely to result in an appreciable reduction in the recovery of humpback whales. The goal of the 1991 recovery plan for humpback whales is to assist humpback whale populations to grow and to reoccupy areas where they were historically found (NMFS 1991). The long-term numerical goal of the recovery 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 recovery plan used the 1986 population estimate for the Gulf of Maine feeding aggregation of humpback whales, which was 240 whales (95% CI = 147 to 333) (NMFS 1991). The most recent best estimate of abundance for Gulf of Maine humpback whales is 847 animals (C.V. = 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 (Waring *et al.* 2009). 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 the large-scale MoNAH assessment, 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 1990s (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 ASMFC and in evaluation by NMFS (NMFS 2010c). While some fishing effort may increase in the future as fisheries stocks respond to management measures to rebuild them, as will occur in the skate fishery in FY 2010-2011, 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 serious injuries and mortalities due to humpback whale entanglement. The most recent data indicates 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 Bluefish 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

Entanglements leading to serious injury and mortality of fin and sei whales have been documented to occur at a level below the 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 its optimum sustainable population. Additionally, broad based gear modifications of the ALWTRP have been implemented and preliminary data in the draft 2010 SAR show a greater and a stable population size for fin and sei whales, respectively. While takes of fin and sei whales continue to be possible through the continued operation of the fishery under the Bluefish FMP, the levels of take are 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

Based on information from Murray (2008, 2009a), NMFS anticipates the taking of up to 82 loggerhead sea turtles annually in gear as a result of the continued operation of the bluefish fishery. Loggerhead sea turtles "taken" in bluefish fishing gear (which for the purposes of this Opinion include trawl and gillnet gear only) are those that are captured or entangled in the gear. Murray (2008) stated that up to an annual average of 3 loggerhead sea turtles are expected to be taken over a 5-year period in bluefish trawl gear based on VTR data from 2000-2004. In addition, an average of up to 79 loggerhead sea turtles are expected to be taken annually over a 5-year period in bluefish gillnet gear, based on upper end of the 95% CI for the bycatch estimate in Murray (2009a). Forty-three percent (1.2, which is rounded up to 2) of the annual takes in trawl gear and 40% (32) of the annual takes in gillnet gear are expected to lead to mortality or serious injury. As a result of being captured in the fishing gear, up to 34 of the 82 loggerhead sea turtles captured annually are expected to die or sustain serious injuries leading 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 final revised recovery plan for loggerhead sea turtles in the Northwest Atlantic includes several objective and measurable recovery criteria which, when met, would result in a determination that the species be removed from the List of Endangered and Threatened Wildlife (NMFS and USFWS 2008). Recovery criteria can be viewed as targets, or values, by which progress toward achievement of recovery objectives can be measured. Recovery criteria may include such things as population numbers and sizes, management or elimination of threats by specific mechanisms, and specific habitat conditions. As a result, there is a need to frame recovery criteria in terms of both population parameters (Demographic Recovery Criteria) and the five listing factors (Listing Factor Recovery Criteria). The nesting beach Demographic Recovery Criteria are specific to recovery units. The remaining criteria cannot be delineated by recovery unit because individuals in the recovery units mix in the marine environment; therefore, these criteria are applicable to all recovery units. Recovery criteria must be met for all recovery units (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 up to 34 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 34 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 the species, that show limited evidence of interbreeding. The 2008 recovery plan compiled the most recent information on the mean number of loggerhead nests and the approximated counts of nesting females per year for four of the five identified recovery units (*i.e.*, nesting groups). They are: (1) for the NRU, a mean of 5,215 nests per year with approximately 1,272 females nesting per year; (2) for the PFRU, a mean of 64,513 nests per year with approximately 15,735 females nesting per year; (3) for the DTRU, a mean of 246 nests per year with approximately 60 females nesting per year; and (4) for the NGMRU, a mean of 906 nests per year with approximately 221 females nesting per year. For the GCRU, the only estimate available for the number of loggerhead nests per year is from Quintana Roo, Yucatán, Mexico, where a range of 903-2,331 nests per year was estimated from 1987-2001 (NMFS and USFWS 2007a). There are no annual nest estimates available for the Yucatán since 2001 or for any other regions in the GCRU, nor are there any estimates of the number of nesting females per year for any nesting assemblage in this recovery unit. However, the 2008 recovery plan indicates that the Yucatán nesting aggregation has at least 1,000 nesting females annually. It should be noted here, and it is explained further below, that the above numbers only include nesting females (*i.e.*, do not include non-nesting adult females, adult males, or juvenile males or females in the population).

It is likely that loggerheads taken in the bluefish fishery originate from several of the recovery units. Limited information is available on the genetic makeup of loggerheads 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 loggerheads 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 loggerheads 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% of the juveniles and sub-adults utilizing the foraging habitat originated from the south Florida nesting population, 12% from the northern subpopulation, 6% from the Yucatán subpopulation, and 2% from other rookeries (including the Florida Panhandle, Dry Tortugas, Brazil, Greece, and Turkey). The previously defined loggerhead subpopulations do not share the exact delineations of the recovery units identified in the 2008 recovery plan. However, the PFRU encompasses the south Florida subpopulation, 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 Yucatán 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 NGMRU likely to occur in the action area, it is extremely unlikely that any of the up to 34 loggerheads that are likely to be seriously injured or killed due to bluefish fishing operations are likely to have originated from either of these recovery units. The majority, at least 80% of the loggerheads anticipated to be seriously injured or killed, are likely to have originated from the PFRU, with the remainder from the NRU and GCRU. As such, 28 of the loggerheads are expected to be from the PFRU, 4 from the NRU, and 2 from the GCRU. The best available information indicates that the proportion of the takes from each recovery unit are 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 will be disproportionately impacted by the take in the bluefish fishery. Therefore, our discussion of the impacts of the bluefish fishery will focus on the overall western North Atlantic population of loggerhead sea turtles, which comprises these recovery units.

In determining whether the continued operation of the bluefish fishery would reduce appreciably the likelihood of survival and recovery of loggerhead sea turtles, NMFS has considered the 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 2008b), 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 bluefish 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 bluefish fishery bycatch of loggerhead sea turtles is estimated to be up to 82 individuals, resulting in up to 34 mortalities, which includes both male and female individuals, as well as juveniles and adults. Mortalities for each gear component of the fishery are calculated independently. A mortality rate of 43% is expected due to trawl gear (3 total turtles, mortality of 2 turtles), while a 40% mortality rate is expected from gillnet gear (total of 79 turtles, 32 mortalities), for a final total of 34 loggerhead sea turtle mortalities. The adult female equivalent of the 34 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 34, and thus less than 20% of the 102 adult female equivalent mortalities estimated for the Atlantic sea scallop fishery.

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 (data from takes in the scallop fishery supports this assumption), 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 bluefish fishery would result in the removal of 0.09% of the adult female loggerhead population in the Western Atlantic (17 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 operation of the bluefish fishery under the Bluefish FMP will cause an appreciable reduction in the

Northwest Atlantic or worldwide population. Therefore, the loss of up to 34 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 bluefish 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 bluefish fishery is estimated to result in the mortality of less than 20% of adult female equivalents compared to the scallop fishery.

The above information also supports the conclusion that continued operation of the bluefish 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 operation of the bluefish 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 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 bluefish 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 or able to be quantified as a result of the proposed action. The continued operation of the bluefish 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 bluefish fishery within the constraints of the current Bluefish FMP will cause no appreciable reduction in the ability to achieve any of the Listing Factor Recovery Criteria.

7.1.5 Leatherback sea turtle

There have been two recently documented captures of leatherback sea turtles in gillnet gear where the primary species landed was bluefish (NEFSC FSB database). Takes of leatherback sea turtles in the bluefish fishery are reasonably likely to occur given: (1) that the distribution of leatherbacks overlaps with operation of bluefish gear, and (2) two leatherback sea turtles were observed captured in bottom otter trawl gear used in the *Loligo* squid fishery and the summer flounder fishery operating in Mid-Atlantic waters where the bluefish fishery also operates. From 2000-2009 there were three (3) confirmed takes of leatherback sea turtles in bottom otter trawl gear in the Mid-Atlantic and three (3) additional confirmed takes in mid-Atlantic sink gillnet gear (NEFSC FSB database). Based on observer data, the take of leatherback sea turtles in any gear (fixed or mobile) operating within the action area, including bluefish gear, would be a rare event. However, given the low percentage of trips with observer coverage in the bluefish 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 bluefish trawl gear could result in death due to forced submergence, given that there are no regulatory controls on tow-times in the bluefish fishery and some trawl tows that have been observed to capture loggerhead 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 leatherback sea turtles.

The lethal removal of up to 4 leatherback sea turtles annually, whether male or female or immature or mature, would be expected to reduce the number of Atlantic leatherback sea turtles as compared to the number of leatherback sea turtles in the Atlantic that would have been present in the absence of the proposed action assuming all other variables remained the same. The loss of 2 female leatherback sea turtles, annually, would be expected to reduce the reproduction of Atlantic leatherback sea turtles as compared to the reproductive output of leatherback sea turtles in the Atlantic in the absence of the proposed action. The lethal removal of 4 leatherback sea turtles annually from the Atlantic as a result of the continued operation of the bluefish fishery will not appreciably reduce the likelihood of survival for the species for the following reasons. Unlike leatherbacks in the Pacific, the nesting trend (in terms of number of nests laid) for leatherbacks in the Atlantic is stable or increasing for nearly all Atlantic leatherback nesting sites. The TEWG (2007) report identified seven leatherback populations or groups of populations in the Atlantic: Florida, North Caribbean, Western Caribbean, Southern Caribbean, West Africa, South Africa, and Brazil. The Leatherback TEWG concluded that there was an increasing or stable trend in nesting for all of these with the exception of the Western Caribbean and West Africa. For example, the Florida Statewide Nesting Beach Survey Program has documented an increase in leatherback nesting numbers in that state from 98 in 1988 to between 800 and 900 nests in the early 2000s (NMFS and USFWS 2007b). In 2001, the number of nests for Suriname and French Guiana, the largest known nesting areas for leatherbacks worldwide, was 60,000 (Hilterman and Goverse 2004).

This is one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). A stable trend in nesting suggests that leatherbacks are able to maintain current levels of nesting as well as current numbers of adult females despite on-going activities as described in the *Environmental Baseline*, *Cumulative Effects*, and the *Status of Listed 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 Listed 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 up to 4 leatherback sea turtles annually in the Atlantic as a result of the continued operation of the bluefish 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 bluefish fishery has no effects on leatherback sea turtles that occur outside of the Atlantic. Therefore, since the continued operation of the bluefish 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; (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 bluefish fishery is expected to kill up to 4 leatherback sea turtles annually. No other effects to leatherbacks are expected as a result of the proposed action. The continued operation of the bluefish 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 bluefish fishery within the constraints of the FMP will have no effect on recovery criteria #2 and #3.

The lethal take of up to 4 leatherback sea turtles, 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 lethal take if the leatherbacks are female. These conclusions are relevant to recovery criteria #1 of the 1992 recovery plan for leatherbacks in the Atlantic. As described in the 5-year status review, the number of nests counted in Puerto Rico increased from 9 in 1978 to a minimum of 469-882 nests recorded each year from 2000-2005. Based on the nesting numbers, the annual female population growth rate was positive for the 28-year time period from 1978-2005. In 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 4 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 bluefish fishery within the constraints of the current Bluefish FMP will not appreciably reduce the likelihood of recovery for leatherback sea turtles in the Atlantic. Since the bluefish 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

There have been no documented incidental takes of Kemp's ridley sea turtles in gear targeting bluefish. However, the distribution of Kemp's ridleys overlaps seasonally with the bluefish fishery, and Kemp's ridley sea turtles are taken in gear types (*e.g.*, gillnets, trawls) which are routinely used in the bluefish fishery. From 2000-2009 there were two confirmed takes of Kemp's ridley sea turtles in bottom otter trawl gear in the Mid-Atlantic (NEFSC FSB database). Additionally, from 2000-2009 there were eight confirmed takes of Kemp's ridley sea turtles in Mid-Atlantic sink gillnet gear (Murray 2009a; NEFSC FSB database). However, five of these gillnet takes occurred in Pamlico Sound and are discounted in this Opinion.

Based on recent observer data, takes of Kemp's ridley sea turtles in fixed or mobile gear operating within the action area, including bluefish gear, is infrequent yet still likely. However, given the low percentage of trips with observer coverage in the bluefish 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 lethal or non-lethal take of up to 4 Kemp's ridleys as a result of the continued operation of the bluefish fishery.

Based on results from the U.S. south Atlantic and Gulf of Mexico shrimp trawl fisheries (Epperly *et al.* 2002; Sasso and Epperly 2006), any capture of a Kemp's ridley sea turtle in bluefish trawl gear could result in death due to forced submergence, given that there are no regulatory controls on tow-times in the bluefish trawl fishery and some trawl tows that have been observed to take loggerhead sea turtles have exceeded one hour in duration (NEFSC FSB database). It is assumed that there is an equal chance of lethally taking a male or female Kemp's ridley sea turtle since available information suggests that both sexes occur in the action area. Kemp's ridley sea turtles taken as a result of bluefish gear in the Mid-Atlantic and New England regions are expected to be immature individuals.

The lethal removal of up to 4 Kemp's ridley sea turtles annually, whether males or females, 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 4 female Kemp's ridley sea turtles annually would be expected to reduce the future 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 4 Kemp's ridley sea turtles annually as a result of the continued operation of the bluefish fishery under the Bluefish FMP will not appreciably reduce the likelihood of survival for the species. 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 Listed 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 Listed 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 4 Kemp's ridley sea turtles annually as a result of the continued operation of the bluefish 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 bluefish fishery within the constraints of the Bluefish 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 bluefish fishery is expected to kill up to 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 up to 4 Kemp's ridleys annually is not expected to modify, curtail, or destroy their range. The bluefish fishery does not utilize Kemp's ridleys for recreational, scientific, or commercial purposes, or affect the adequacy of existing regulatory mechanisms to protect them. Therefore, the continued operation of the bluefish fishery within the constraints of the current Bluefish FMP will have no effect on ESA listing criteria #1 through #4.

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

7.1.7 Green sea turtle

There have been no documented takes of green sea turtles in gear targeting bluefish. There has been one observed take of a green sea turtle in bottom otter trawl gear in the Northeast and Mid-Atlantic regions from 2000-2009. Also during that period, fifteen green sea turtles were observed to be taken 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 bluefish gear, and green sea turtles have been captured in gear (*e.g.*, gillnets, trawls) which is utilized by the bluefish fishery. Based on observer data, the incidental take of green sea turtles in any gear (fixed or mobile) operating within the action area, including bluefish gear, would be a rare event. However, given the low level of observer coverage in the bluefish 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 information in section 6.2.2, NMFS anticipates that up to 5 lethal or non-lethal green sea turtle takes in bluefish gear will occur annually as a result of the continued operation of the bluefish fishery. It is assumed that there is an equal chance of lethally taking male and female green sea turtles since available information suggests that both sexes occur in the action area. Green sea turtles incidentally taken in bluefish trawl or gillnet gear are only expected to be immature individuals.

The lethal removal of up to 5 green sea turtles annually from the Atlantic, whether males or females, 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 5 female green sea turtles annually would be expected to reduce the future 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 5 green sea turtles annually from the Atlantic as a result of the continued operation of the bluefish 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 Listed 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 Listed 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 5 green sea turtles annually in the Atlantic as a result of the continued operation of the bluefish fishery under the Bluefish 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 bluefish fishery has no effects on green sea turtles that occur outside of the Atlantic. Therefore, since the continued operation of the bluefish 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, completed in 2007, reviewed the recovery criteria provided with the 1991 recovery plan for green sea turtles in the Atlantic, and the progress made in meeting each objective (NMFS and USFWS 2007d). These are that the U.S. population of green 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; and (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 bluefish fishery is expected to kill up to 5 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 bluefish 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 bluefish fishery within the constraints of the current Bluefish FMP will have no effect on recovery criteria #2 and #4.

The lethal taking of up to 5 green sea turtles annually in the bluefish 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 future green sea turtle reproduction in the Atlantic as a result of the lethal takes if the green sea turtles are female. 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 5 green sea turtles, which may be immature males or females, 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 sea 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). 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, New York, Morreale *et al.* (2005) documented the capture of more than twice as many green sea turtles in 2003 and 2004 with less pound net gear fished, compared to the number of green sea turtles captured in pound net gear in the area during the 1990s. Yet other studies have found no difference in the abundance (decreasing or increasing) of green sea turtles on foraging grounds in the Atlantic (Bjorndal *et al.* 2005; Epperly *et al.* 2007). The annual loss of up to 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 bluefish fishery will not appreciably reduce the likelihood of recovery for green sea turtles in the Atlantic. Since the bluefish fishery has no effects on green sea turtles that occur outside of the Atlantic, the continued operation of the bluefish fishery within the constraints of the current Bluefish FMP will not appreciably reduce the likelihood of recovery for the species.

8.0 CONCLUSION

After reviewing the current status of the species, the environmental baseline and cumulative effects in the action area, and the effects of the continued operation of the bluefish fishery within the constraints of the current Bluefish FMP, it is NMFS's biological opinion that the proposed action may adversely affect, but is not likely to jeopardize the continued existence of right, humpback, fin, and sei whales or loggerhead, leatherback, Kemp's ridley, and green sea turtles.

Proposed Rule to List Loggerhead Sea Turtles

As explained in *Status of Listed Species* section of this Opinion, on March 16, 2010, NMFS published a proposed rule to list two DPSs of loggerhead sea turtles as threatened and seven DPSs 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 34 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 34 loggerhead sea turtles each year as a result of the continued operation of the bluefish 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 0.09% 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 0.09% 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. A 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, a conference is not required for this proposed action. Additionally, as the 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 ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, unless a special exemption has been granted. Take is defined as "to harass, harm, pursue, hunt, shoot, capture, or collect, or to attempt to engage in any such conduct." Incidental take is defined as take that is incidental to, and not the purpose of, the execution of an otherwise lawful activity. Under the terms of sections 7(b)(4) and 7(o)(2), taking that is incidental to and not intended as part of the action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement (ITS).

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 of section 7(o)(2).

Anticipated Amount or Extent of Incidental Take

Based on the Murray (2008, 2009a) reports, incidental capture data from observer reports for the bluefish and other similar fisheries, and the distribution and abundance of sea turtles in the action area, NMFS anticipates that the continued operation of the commercial component of the bluefish fishery may result in the incidental take of sea turtles as follows[11]:

- for loggerhead sea turtles, NMFS anticipates (a) the annual take of up to 3 individuals over a 5-year average in trawl gear, of which up to 2 per year may be lethal and (b) the annual take of up to 79 individuals over a 5-year average in gillnet gear, of which up to 32 per year may be lethal;

- for leatherback sea turtles, NMFS anticipates the annual lethal or non-lethal take of up to 4 individuals in trawl and gillnet gear combined;

- for Kemp's ridley sea turtles, NMFS anticipates the annual lethal or non-lethal take of up to 4 individuals in trawl and gillnet gear combined;

- for green sea turtles, NMFS anticipates the annual lethal or non-lethal take of up to 5 individuals in trawl and gillnet gear combined.

The anticipated level of incidental take of sea turtles for the recreational component of the bluefish fishery cannot be estimated at this time. In addition, 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.

---

[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.

Anticipated Impact of Incidental Take

NMFS has concluded that the continued operation of the bluefish fishery within the constraints of the current Bluefish FMP 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 bluefish 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. Some of these measures were included as RPMs with the July 2, 1999 Opinion. They are repeated here because they still meet the criteria for a RPM and reflect work in progress to minimize the taking of sea turtles in bluefish fishing gear.

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 bluefish fishery:

1. NMFS must seek to ensure that any sea turtles incidentally taken in bluefish 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 bluefish 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 bluefish 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 bluefish 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 distribute information to Federal bluefish 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 Northeast

Region STDN Disentanglement Guidelines to those permit holders that use gillnet gear. 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 dredge, trawl, and gillnet fisheries to document and estimate incidental bycatch of loggerhead sea turtles. Monthly summaries and an annual report of sea turtle takes in New England and Mid-Atlantic fisheries, including trips where bluefish 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 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 Northeast 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 develop a specialized survey for estimating recreational sea turtle takes. The survey must be developed by January 2011 and implemented no later than January 2012.

8. 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.

9. To comply with RPM #3 above, NMFS will continue to investigate modifications of trawl and gillnet gear and its 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.

10. To comply with RPM #4 above, NMFS must continue to review all data available on the observed/documented take of sea turtles in Mid-Atlantic dredge, 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 bluefish 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 bluefish fishery. Observer coverage has been used as the principal means to estimate sea turtle bycatch in the bluefish fishery and to monitor incidental take levels. NMFS will continue to use observer coverage to monitor sea turtle bycatch in commercial gillnet and trawl gear that is authorized by the Bluefish 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 should 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 bluefish fishery using appropriate statistical methods. A new bycatch estimate for loggerhead sea turtles caught in bottom trawl gear is scheduled to be completed in 2010. A revised estimate for gillnet gear is planned to be completed within 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.

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 conservation:

1. NMFS should continue to collect and analyze biological samples from sea turtles incidentally taken in fishing gear targeting bluefish to determine the nesting origin of sea turtles taken in the bluefish fishery in order to better assess the effects of the fishery on nesting groups and recovery units 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 bluefish 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 the seasonal distribution and abundance of ESA-listed species 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 ESA-listed species 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 should 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.

11.0 REINITIATING CONSULTATION

This concludes formal consultation on the continued operation of the bluefish fishery within the constraints of the current Bluefish FMP. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In the event that the amount or extent of incidental take is exceeded, NMFS NERO must immediately request reinitiation of formal consultation.

Literature Cited

Ackerman, R.A. 1997. The nest environment and embryonic development of sea turtles. Pages 83-106 in P.L. Lutz and J.A. Musick, eds. The Biology of Sea Turtles. New York: CRC Press.

Agler, B.A., R.L. Schooley, S.E. Frohock, S.K. Katona, and I.E. Seipt. 1993. Reproduction of photographically identified fin whales, *Balaenoptera physalus*, from the Gulf of Maine. Journal of Mammalogy 74:577-587.

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

Aguilar, A., and C. Lockyer. 1987. Growth, physical maturity and mortality of fin whales (*Balaenoptera physalus*) inhabiting the temperate waters of the Northeast Atlantic. Canadian Journal of Zoology 65:253-264.

Allen, B.M., and Angliss R.P. 2010. Alaska marine mammal stock assessments, 2009. NOAA Technical Memorandum NMFS-AFSC-206:1-276.

Amato, R.V. 1994. Sand and gravel maps of the Atlantic Continental Shelf with explanatory text. U.S. Minerals Management Service OCS Monograph MMS 93-0037, scale 1:1,000,000, 35 pp.

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

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

Angliss, R.P., and D.P. DeMaster. 1998. Differentiating serious and non-serious injury of marine mammals taken incidental to commercial fishing operations: Report of the serious injury workshop 1-2 April 1997, Silver Spring, Maryland. NOAA Technical Memorandum NMFS-OPR-13. 48 pp.

Angliss, R.P., and R.B. Outlaw. 2007. Alaska marine mammal stock assessments, 2006. NOAA Technical Memorandum NMFS-AFSC-168:1-244.

Angliss, R.P., D.P. DeMaster, and A.L. Lopez. 2001. Alaska marine mammal stock assessments, 2001. NOAA Technical Memorandum NMFS-AFSC-124:1-206.

ASMFC (Atlantic States Marine Fisheries Commission). 1997. Amendment 3 to the Interstate Fishery Management Plan for American Lobster. Fishery Management Report No. 29. Washington, D.C.: Atlantic States Marine Fisheries Commission.

ASMFC (Atlantic States Marine Fisheries Commission). 2002. Amendment 4 to the Interstate Fishery Management Plan for weakfish. Fishery Management Report No. 39. Washington, D.C.: Atlantic States Marine Fisheries Commission.

ASMFC (Atlantic States Marine Fisheries Commission). 2004. Horseshoe crab 2004 stock assessment. Washington, D.C.: Atlantic States Marine Fisheries Commission.

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

Baird, R.W., A.D. Ligon, and S.K. Hooker. 2000. Sub-surface and night-time behavior of humpback whales off Maui, Hawaii: A preliminary report. Report prepared under Contract # 40ABNC050729 from the Hawaiian Islands Humpback Whale National Marine Sanctuary, Kihei, Hawaii, to the Hawaii Wildlife Fund, Paia, Hawaii. 18 pp.

Baker, J.D., C.L. Littnan, and D.W. Johnston. 2006. Potential effects of sea level rise on the terrestrial habitats of endangered and endemic megafauna in the Northwestern Hawaiian Islands. Endangered Species Research 2:21-30.

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

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

Baldwin, R., G.R. Hughes, and R.T. Prince. 2003. Loggerhead turtles in the Indian Ocean. Pages 218-232 in A.B. Bolten and B.E. Witherington, eds. Loggerhead Sea Turtles. Washington, D.C.: Smithsonian Books.

Barco, S., W.A. McLellan, J. Allen, R. Asmutis, R. Mallon-Day, E. Meagher, D.A. Pabst, J. Robbins, R. Seton, R.M. Swingle, M.T. Weinrich, and P. Clapham. 2002. Population identity of humpback whales in the waters of the U.S. mid-Atlantic states. Journal of Cetacean Research and Management 4:135-141.

Barlow, J., and P.J. Clapham. 1997. A new birth-interval approach to estimating demographic parameters of humpback whales. Ecology 78:535-546.

Barlow, J., and K.A. Forney. 2007. Abundance and population density of cetaceans in the California Current ecosystem. Fishery Bulletin 105:509-526.

Bass, A.L., S.P. Epperly, and J. Braun-McNeill. 2004 Multiyear analysis of stock composition of a loggerhead turtle (*Caretta caretta*) foraging habitat using maximum likelihood and Bayesian methods. Conservation Genetics 5:783-796.

Baumgartner, M.F., and B.R. Mate. 2005. Summer and fall habitat of North Atlantic right whales (*Eubalaena glacialis*) inferred from satellite telemetry. Canadian Journal of Fisheries and Aquatic Sciences 62:527-543.

Baumgartner, M.F., T.V.N. Cole, R.G. Campbell, G.J. Teegarden, and E.G. Durbin. 2003. Associations between North Atlantic right whales and their prey, *Calanus finmarchicus,* over diel and tidal time scales. Marine Ecology Progress Series 264:155-166.

Bérubé, M., A. Aguilar, D. Dendanto, F. Larsen, G. Notarbatolo di Sciara, R. Sears, J. Sigurjonsson, J. Urban-R, and P. Palsboll. 1998. Population genetic structure of North Atlantic, Mediterranean Sea and Sea of Cortez fin whales: analysis of mitochondrial and nuclear loci. Molecular Ecology 7:585-599.

Best, P.B., J.L. Bannister, R.L. Brownell Jr., and G.P. Donovan (eds.). 2001. Right whales: worldwide status. Journal of Cetacean Research and Management, Special Issue 2. 309 pp.

Bjork, M., F. Short, E. Mcleod, and S. Beer. 2008. Managing seagrasses for resilience to climate change. Gland, Switzerland: IUCN. 56 pp.

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

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

Bjorndal, K.A., A.B. Bolten, and M.Y. Chaloupka. 2005. Evaluating trends in abundance of immature Green Turtles, *Chelonia mydas*, in the greater Caribbean. Ecological Applications 15:304-314.

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

Bolten, A.B. 2003. Variation in sea turtle life history patterns: neritic vs. oceanic developmental stages. Pages 243-257 in P.L. Lutz, J.A. Musick, and J. Wyneken, eds. The Biology of Sea Turtles, Vol. 2. Boca Raton, Florida: CRC Press.

Bolten, A.B., J.A. Wetherall, G.H. Balazs, and S.G. Pooley, compilers. 1996. Status of marine turtles in the Pacific Ocean relevant to incidental take in the Hawaii-based pelagic longline fishery. NOAA Technical Memorandum NMFS-SWFSC-230.

Boulon, R.H., Jr. 2000. Trends in sea turtle strandings, U.S. Virgin Islands: 1982 to 1997. Pages 261-262 in F.A. Abreu-Grobois, R. Briseño-Dueñas, R. Márquez-Millán, and L. Sarti-Martínez, compilers. Proceedings of the Eighteenth International Sea Turtle Symposium. NOAA Technical Memorandum NMFS-SEFSC-436.

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

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

Bowen, B.W., A.L. Bass, S.-M. Chow, M. Bostrom, K.A. Bjorndal, A.B. Bolten, T. Okuyama, B.M. Bolker., S. Epperly, E. Lacasella, D. Shaver, M. Dodd, S.R. Hopkins-Murphy, J.A. Musick, M. Swingle, K. Rankin-Baransky, W. Teas, W.N. Witzell, and P.H. Dutton. 2004. Natal homing in juvenile loggerhead turtles (*Caretta caretta*). Molecular Ecology 13:3797-3808.

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

Bowman, R., E. Lyman, D. Mattila, C. Mayo, and M. Brown. 2003. Habitat Management Lessons From a Satellite-Tracked Right Whale. Presentation to the ARGOS Animal Tracking Symposium. March 24-26, 2003. Annapolis, Maryland.

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

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

Brown, M.W., and M.K. Marx. 2000. Surveillance, Monitoring and Management of North Atlantic Right Whales, *Eubalaena glacialis*, in Cape Cod Bay, Massachusetts: January to Mid-May, 2000. Final report. Contract No. SCFWE3000-8365027. 52 pp.

Brown, M.W., O.C. Nichols, M.K. Marx, and J.N. Ciano. 2002. Surveillance of North Atlantic right whales in Cape Cod Bay and adjacent waters - 2002. Final report to the Division of Marine Fisheries, Commonwealth of Massachusetts. 29 pp.

Brown, S.G. 1986. Twentieth-century records of right whales (*Eubalaena glacialis*) in the northeast Atlantic Ocean. Pages 121-128 in R.L. Brownell Jr., P.B. Best, and J.H. Prescott, eds. Right whales: Past and Present Status. IWC Special Issue No. 10.

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

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

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

Calambokidis, J., E.A. Falcone, T.J. Quinn., A.M. Burdin, P.J. Clapham, J.K.B. Ford, C.M. Cabriele, R. LeDuc, D. Matilla, L. Rojas-Bracho, J.M. Straley, B.L. Taylor, J. Urban, D. Weller, B.H. Witteveen, M.Yamaguchi, A. Bendlin, D. Camacho, K. Flynn, A. Havron, J. Huggins, and N. Maloney. 2008. SPLASH: Structure of Populations, Levels of Abundance and Status of Humpback Whales in the North Pacific. Final Report for Contract AB133F-03-RP-00078. 57 pp.

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

Carretta, J.V., K.A. Forney, M.M. Muto, J. Barlow, J. Baker, B. Hanson, and M. Lowry. 2007. U.S. Pacific marine mammal stock assessments: 2006. NOAA Technical Memorandum NMFS-SWFSC-398. 316 pp.

Carretta, J.V., K.A. Forney, M.S. Lowry, J. Barlow, J. Baker, D. Johnston, B. Hanson, M.M. Muto, D. Lynch, and L. Carswell. 2008. U.S. Pacific marine mammal stock assessments: 2008. NOAA Technical Memorandum NMFS-SWFSC-434. 340 pp.

Carretta, J.V., K.A. Forney, M.S. Lowry, J. Barlow, J. Baker, D. Johnston, B. Hanson, R.L. Brownell Jr., J. Robbins, D.K. Matilla, K. Ralls, M.M. Muto, D. Lynch, and L. Carswell. 2009. U.S. Pacific marine mammal stock assessments: 2009. NOAA Technical Memorandum NMFS-SWFSC-453. 341 pp.

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

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

Caswell, H., M. Fujiwara, and S. Brault. 1999. Declining survival probability threatens the North Atlantic right whale. Proceedings of the National Academy of Sciences 96:3308-3313.

Caulfield, R.A. 1993. Aboriginal subsistence whaling in Greenland: the case of Qeqertarsuaq municipality in West Greenland. Arctic 46:144-155.

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

Chaloupka, M., and C. Limpus. 2001. Trends in the abundance of sea turtles resident in southern Great Barrier Reef waters. Biological Conservation 102:235-249.

Chan, E.H., and H.C. Liew. 1996. Decline of the leatherback population in Terengganu, Malaysia, 1956-1995. Chelonian Conservation and Biology 2(2):192-203.

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

Church, J., J.M. Gregory, P. Huybrechts, M. Kuhn, K. Lambeck, M.T. Nhuan, D. Qin, and P.L. Woodworth. 2001. Changes in sea level. Pages 639-694 in J.T. Houghton, Y. Ding, D.J. Griggs, M. Noguer, P.J. Vander Linden, X. Dai, K. Maskell, and C.A. Johnson CA, eds. Climate change 2001: the scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge: Cambridge University Press.

Clapham, P.J. 1992. Age at attainment of sexual maturity in humpback whales, *Megaptera novaengliae*. Canadian Journal of Zoology 70:1470-1472.

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

Clapham, P.J., and C.A. Mayo. 1990. Reproduction of humpback whales (*Megaptera novaengliae*) observed in the Gulf of Maine. Reports of the International Whaling Commission, Special Issue 12:171-175.

Clapham, P.J., S.B. Young, and R.L. Brownell, Jr. 1999. Baleen whales: conservation issues and the status of the most endangered populations. Mammal Review 29(1):35-60.

Clapham, P., S. Brault, H. Caswell, M. Fujiwara, S. Kraus, R. Pace and P. Wade. 2002. Report of the Working Group on Survival Estimation for North Atlantic Right Whales.

Clapham, P., J. Barlow, M. Bessinger, T. Cole, D. Mattila, R. Pace, D. Palka, J. Robbins, and R. Seton. 2003. Abundance and demographic parameters of humpback whales from the Gulf of Maine, and stock definition relative to the Scotian Shelf. Journal of Cetacean Research and Management 5:13-22.

Clark, C.W. 1995. Application of U.S. Navy underwater hydrophone arrays for scientific research on whales. Reports of the International Whaling Commission 45:210-212.

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

Cole, T.V.N., D.L. Hartley, and R.L. Merrick. 2005. Mortality and serious injury determinations for large whale stocks along the eastern seaboard of the United States, 1999-2003. NEFSC Reference Document 05-08. 18 pp.

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

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

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

Donovan, G.P. 1991. A review of IWC stock boundaries. Reports of the International Whaling Commission, Special Issue 13:39-63.

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

Durbin, E., G. Teegarden, R. Campbell, A. Cembella, M.F. Baumgartner, and B.R. Mate. 2002. North Atlantic right whales, *Eubalaena glacialis*, exposed to Paralytic Shellfish Poisoning (PSP) toxins via a zooplankton vector, *Calanus finmarchicus*. Harmful Algae 1:243-251.

Dutton, P.H., B.W. Bowen, D.W. Owens, A. Barragan, and S.K. Davis. 1999. Global phylogeography of the leatherback turtle (*Dermochelys coriacea*). Journal of Zoology 248:397-409.

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

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

Dwyer, K.L., C.E. Ryder, and R. Prescott. 2003. Anthropogenic mortality of leatherback sea turtles in Massachusetts waters. Page 260 in J.A. Seminoff, compiler. Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503.

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

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

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

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

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

Encyclopedia Britannica. 2010. Neritic Zone. Accessed 12 January 2010. http://www.britannica.com/eb/article-9055318.

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

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

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

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

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

Epperly, S., L. Avens, L. Garrison, T. Henwood, W. Hoggard, J. Mitchell, J. Nance, J. Poffenberger, C. Sasso, E. Scott-Denton, and C. Yeung. 2002. Analysis of sea turtle bycatch in the commercial shrimp fisheries of southeast U.S. waters and the Gulf of Mexico. NOAA Technical Memorandum NMFS-SEFSC-490:1-88.

Epperly, S., L. Stokes, and S. Dick. 2004. Careful release protocols for sea turtle release with minimal injury. NOAA Technical Memorandum NMFS-SEFSC-524:1-42.

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

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

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

Fahay, M.P., P.L. Berrien, D.L. Johnson, and W.W. Morse. 1999. Essential Fish Habitat Source Document: Bluefish, *Pomatomus saltatrix*, Life History and Habitat Characteristics. NOAA Technical Memorandum NMFS-NE-144.

Feely, R.A., C.L. Sabine, J.M. Hernandez-Ayon, D. Ianson, and B. Hales. 2008. Evidence for upwelling of corrosive "acidified" water onto the continental shelf. Science 320:1490-1492.

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

Fish, M.R., I.M. Cote, J.A. Gill, A.P. Jones, S. Renshoff, and A.R.Watkinson. 2005. Predicting the impact of sea-level rise on Caribbean sea turtle nesting habitat. Conservation Biology 19:482-491.

Flinn, R.D., A.W. Trites, E.J. Gregr, and R.I. Perry. 2002. Diets of fin, sei, and sperm whales in British Columbia: an analysis of commercial whaling records, 1963-1967. Marine Mammal Science 18(3):663-679.

Forney, K.A. 2007. Preliminary estimates of cetacean abundance along the U.S. west coast and within four National Marine Sanctuaries during 2005. NOAA Technical Memorandum NMFS-SWFSC-406:1-27.

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

Franklin, I.R. 1980. Evolutionary change in small populations. Pages 135-150 in M.E. Soule and B.A. Wilcox, eds. Conservation Biology, an Evolutionary-Ecological Perspective. Sunderland, Massachusetts: Sinauer.

Frasier, T.R., B.A. McLeod, R.M. Gillett, M.W. Brown, and B.N. White. 2007. Right Whales Past and Present as Revealed by Their Genes. Pages 200-231 in S.D. Kraus and R.M. Rolland, eds. The Urban Whale. Cambridge, Massachusetts and London, England: Harvard University Press.

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

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

Fujiwara, M., and H. Caswell. 2001. Demography of the endangered North Atlantic right whale. Nature 414:537-541.

Gagosian, R.B. 2003. Abrupt climate change: should we be worried? Prepared for a panel on abrupt climate change at the World Economic Forum, Davos, Switzerland, January 27, 2003. 9pp.

Gambell, R. 1993. International management of whales and whaling: an historical review of the regulation of commercial and aboriginal subsistence whaling. Arctic 46:97-107.

Garner, J.A., and S.A. Garner. 2007. Tagging and nesting research on leatherback sea turtles (*Dermochelys coriacea*) on Sandy Point, St. Croix, U.S. Virgin Islands, 2007. Annual Report of the Virgin Islands Department of Planning and Natural Resources, Division of Fish and Wildlife. 48 pp.

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

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

Geraci, J.R., D.M. Anderson, R.J. Timperi, D.J. St. Aubin, G.A. Early, J.H.Prescott, and C.A. Mayo. 1989. Humpback whales (*Megaptera novaeangliae*) fatally poisoned by dinoflagellate toxin. Canadian Journal of Fisheries and Aquatic Sciences 46(11):1895-1898.

Girondot, M., and J. Fretey. 1996. Leatherback turtles, *Dermochelys coriacea*, nesting in French Guiana 1978-1995. Chelonian Conservation and Biology 2:204-208.

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

Glass, A.H., T.V.N. Cole, M. Garron, R.L. Merrick, and R.M. Pace III. 2008. Mortality and serious injury determinations for baleen whale stocks along the United States eastern seaboard and adjacent Canadian Maritimes, 2002-2006. NEFSC Reference Document 08-04; 18 pp.

Glass, A.H., T.V.N. Cole, and M. Garron. 2009. Mortality and serious injury determinations for baleen whale stocks along the United States eastern seaboard and adjacent Canadian Maritimes, 2003-2007 (2nd Edition). NEFSC Reference Document 09-04; 19 pp.

Glen, F., A.C. Broderick, B.J. Godley, and G.C. Hays. 2003. Incubation environment affects phenotype of naturally incubated green turtle hatchlings. Journal of the Marine Biological Association of the United Kingdom 83(5):1183-1186.

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

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

Goldenberg, S.B., C.W. Landsea, A.M. Mestas-Nunez, and W.M. Gray. 2001. The recent increase in Atlantic hurricane activity: causes and implications. Science 293:474-479.

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

Greene, C.H., and A.J. Pershing. 2004. Climate and the conservation biology of North Atlantic right whales: the right whale at the wrong time? Frontiers in Ecology and the Environment 2(1):29-34.

Greene, C.H., A.J. Pershing, R.D. Kenney, and J.W. Jossi. 2003. Impact of climate variability on the recovery of endangered North Atlantic right whales. Oceanography 16: 96-101.

Hain, J.H.W., M.A. Hyman, R.D. Kenney, and H.E. Winn. 1985. The role of cetaceans in the shelf-edge region of the northeastern United States. Marine Fisheries Review 47(1):13-17.

Hain, J.H.W., M.J. Ratnaswamy, R.D. Kenney, and H.E. Winn. 1992. The fin whale, *Balaenoptera physalus*, in waters of the northeastern United States continental shelf. Reports of the International Whaling Commission 42:653-669.

Hamann, M., C.J. Limpus, and M.A. Read. 2007. Vulnerability of marine reptiles in the Great Barrier Reef to climate change. Pages 465-496 in J.E. Johnson and P.A. Marshall, eds. Climate change and the Great Barrier Reef: a vulnerability assessment. Hobart: Great Barrier Reef Marine Park Authority and Australia Greenhouse Office.

Hamilton, P.K., and C.A. Mayo. 1990. Population characteristics of right whales (*Eubalaena glacialis*) observed in Cape Cod and Massachusetts Bays, 1978-1986. Reports of the International Whaling Commission, Special Issue 12:203-208.

Hamilton, P.K., M.K. Marx, and S.D. Kraus. 1999. Scarification analysis of North Atlantic right whales (*Eubalaena glacialis*) as a method of assessing human impacts. Final report to the NMFS Northeast Fisheries Science Center. Contract No. 4EANF-6-0004. 39 pp.

Hamilton, P.K., A.R. Knowlton, and S.D. Kraus. 2008. Maintenance of the North Atlanic right whale catalog: 1 January-31 December 2007. Final report to the NMFS Northeast Fisheries Science Center. Contract No. EA133F-05-CN-1231. Edgerton Research Laboratory, New England Aquarium. 27 pp.

Hatase, H., M. Kinoshita, T. Bando, N. Kamezaki, K. Sato, Y. Matsuzawa, K. Goto, K. Omuta, Y. Nakashima, H. Takeshita, and W. Sakamoto. 2002. Population structure of loggerhead turtles, *Caretta caretta*, nesting in Japan: Bottlenecks on the Pacific population. Marine Biology 141:299-305.

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

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

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

Hawkes, L.A., A.C. Broderick, M.H. Godfrey, and B.J. Godley. 2009. Climate change and marine turtles. Endangered Species Research 7:137-159.

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

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

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

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

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

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

ICES (International Council for the Exploration of the Sea). 2005. Study Group on the Bycatch of Salmon in Pelagic Trawl Fisheries (SGBYSAL). ICES CM 2005 (ACFM:13), 41 pp.

ICES (International Council for the Exploration of the Sea). 2006. Report of the Working Group on North Atlantic Salmon (WGNAS). ICES CM 2006 (ACFM:23), 254 pp.

IPCC (Intergovernmental Panel on Climate Change). 2007. Summary for Policymakers. Cambridge: Cambridge University Press.

IWC (International Whaling Commission). 1979. Report of the sub-committee on protected species, Annex G, Appendix I. Report of the International Whaling Commission 29:84-86.

IWC (International Whaling Commission). 1986. Right whales: past and present status. Reports of the International Whaling Commission, Special Issue 10:1-33.

IWC (International Whaling Commission). 1992. Report of the comprehensive assessment special meeting on North Atlantic fin whales. Reports of the International Whaling Commission 42:595-644.

IWC (International Whaling Commission). 1995. Report of the Scientific Committee, Annex E. Reports of the International Whaling Commission 45:121-138.

IWC (International Whaling Commission). 1997. Report of the IWC workshop on climate change and cetaceans. Reports of the International Whaling Commission 47:293-313.

IWC (International Whaling Commission). 2001. Report of the workshop on the comprehensive assessment of right whales: A worldwide comparison. Reports of the International Whaling Commission, Special Issue 2:1-60.

IWC (International Whaling Commission). 2002. Report of the subcommittee on the Comprehensive Assessment of North Atlantic humpback whales. Journal of Cetacean Research and Management 4:230-260.

IWC (International Whaling Commission). 2003. Report of the subcommittee on the Comprehensive Assessment of North Atlantic humpback whales. Journal of Cetacean Research and Management 5:293-323.

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

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

Jefferson, T.A., M.A. Webber, and R.L. Pitman. 2008. Marine mammals of the world. Amsterdam: Elsevier. 573 pp.

Johnson, A, G. Salvador, J. Kenney, J. Robbins, S. Kraus, S. Landry, and P. Clapham. 2005. Fishing gear involved in entanglements of right and humpback whales. Marine Mammal Science 21(4):635-645.

Johnson, J.H., and A.A. Wolman. 1984. The humpback whale, *Megaptera novaengliae*. Marine Fisheries Review 46(4):30-37.

Jones, A.R., W. Gladstone, and N.J. Hacking. 2007. Australian sandy beach ecosystems and climate change: ecology and management. Australian Zoology 34:190-202.

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

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

Kelle, L., N. Gratiot, I. Nolibos, J. Therese, R. Wongsopawiro, and B. DeThoisy. 2007. Monitoring of nesting leatherback turtles (*Dermochelys coriacea*): contribution of remote-sensing for real time assessment of beach coverage in French Guiana. Chelonian Conservation and Biology 6:142-149.

Kenney, R.D. 2000. What's wrong with North Atlantic right whales? Whalewatcher 32:1-8.

Kenney, R.D. 2001. Anomalous 1992 spring and summer right whale (*Eubalaena glacialis*) distribution in the Gulf of Maine. Journal of Cetacean Research and Management, Special Issue 2:209-223.

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

Kenney, R.D., M.A.M. Hyman, R.E. Owen, G.P. Scott, and H.E. Winn. 1986. Estimation of prey densities required by Western North Atlantic right whales. Marine Mammal Science 2(1):1-13.

Kenney, R.D., H.E. Winn, and M.C. Macaulay. 1995. Cetaceans in the Great South Channel, 1979-1989: right whale (*Eubalaena glacialis*). Continental Shelf Research 15:385-414.

Khan, C., T.V.N. Cole, P. Duley, A. Glass, M. Niemeyer, and C. Christman. 2009. North Atlantic Right Whale Sighting Survey (NARWSS) and Right Whale Sighting Advisory System (RWSAS) 2008 Results Summary. NEFSC Reference Document 09-05; 7 pp.

Khan, C., T. Cole, P. Duley, A. Glass, and J. Gatzke. 2010. North Atlantic Right Whale Sighting Survey (NARWSS) and Right Whale Sighting Advisory System (RWSAS) 2009 Results Summary. NEFSC Reference Document 10-07; 7 pp.

Knowlton, A.R., J. Sigurjonsson, J.N. Ciano, and S.D. Kraus. 1992. Long-distance movements of North Atlantic right whales (*Eubalaena glacialis*). Marine Mammal Science 8(4):397-405.

Knowlton, A.R., S.D. Kraus, and R.D. Kenney. 1994. Reproduction in North Atlantic right whales, *Eubalaena glacialis*. Canadian Journal of Zoology 72:1297-1305.

Knowlton, A.R., M.K. Marx, H.M. Pettis, P.K. Hamilton and S.D. Kraus. 2005. Analysis of scarring on North Atlantic right whales (*Eubalaena glacialis*): monitoring rates of entanglement interaction 1980-2002. Report to the National Marine Fisheries Service. Contract No. 43EANF030107.

Knowlton, A.R., L.A. Cooper, P.K. Hamilton, M.K. Marx, H.M. Pettis, and S.D. Kraus. 2008. Analysis of scarring on North Atlantic right whales (*Eubalaena glacialis*): Monitoring rate of entanglement interaction—1980-2004. Final report to the NMFS Northeast Fisheries Science Center. Contract No. EA133F-03-SE-0323. 25 pp.

Kraus, S.D., J.H. Prescott, and A.R. Knowlton. 1986. Wintering right whales (*Eubalaena glacialis*) along the southeastern coast of the United States, 1984-1986. Boston: New England Aquarium. 15 pp.

Kraus, S.D., M.J. Crone, and A.R. Knowlton. 1988. The North Atlantic right whale. Pages 684-698 in W.J. Chandler, ed. Audbon wildlife report 1988/1989. San Diego: Academic Press.

Kraus, S.D., R.D. Kenney, A.R. Knowlton, and J.N. Ciano. 1993. Endangered Right Whales of the Southwestern North Atlantic. Report to the Minerals Management Service. Contract No. 14-35-0001-30486. 69 pp.

Kraus, S.D., P.K. Hamilton, R.D. Kenney, A.R. Knowlton, and C.K. Slay. 2001. Reproductive parameters of the North Atlantic right whale. Journal of Cetacean Research and Management 2: 231-236.

Kraus, S.D., M.W. Brown, H. Caswell, C.W. Clark, M. Fujiwara, P.K. Hamilton, R.D. Kenney, A.R. Knowlton, S. Landry, C.A. Mayo, W.A. McLellan, M.J. Moore, D.P. Nowacek, D.A. Pabst, A.J. Read, and R.M. Rolland. 2005. North Atlantic right whales in crisis. Science 309:561-562.

Kraus, S.D., R.M. Pace III, and T.R. Frasier. 2007. High investment low return: the strange case of reproduction in *Eubalaena glacialis*. Pages 172-199 in S.D. Kraus and R.M. Rolland, eds. The Urban Whale. Cambridge, Massachusetts and London, England: Harvard University Press.

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

Lacroix, G.L., and D. Knox. 2005. Distribution of Atlantic salmon (*Salmo salar*) postsmolts of different origins in the Bay of Fundy and Gulf of Maine and evaluation of factors affecting migration, growth, and survival. Canadian Journal of Fisheries and Aquatic Sciences 62:1363-1376.

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

Laist, D.W., A.R. Knowlton, J.G. Mead, A.S. Collet, and M. Podesta. 2001. Collisions between ships and whales. Marine Mammal Science 17(1):35-75.

Lalli, C.M., and T.R. Parsons. 1997. Biological oceanography: An introduction. 2nd Edition. Oxford: Butterworth-Heinemann Publications.

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

Leandro, L.F.; R.M. Rolland; P.B. Roth; N. Undholm; Z. Wang, and G.J. Doucette. 2010. Exposure of the North Atlantic right whale *Eubalaena glacialis* to the marine algal biotoxin, domoic acid. Marine Ecology Progress Series 398:287-303.

Learmonth, J.A., C.D. MacLeod, M.B. Santos, G.J. Pierce, H.Q.P. Crick, and R.A. Robinson. 2006. Potential effects of climate change on marine mammals. Oceanography and Marine Biology: An Annual Review 44:431-464.

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

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

Limpus, C.J., and D.J. Limpus. 2000. Mangroves in the diet of *Chelonia mydas* in Queensland, Australia. Marine Turtle Newsletter 89:13-15.

Limpus, C.J., and D.J. Limpus. 2003. Loggerhead turtles in the equatorial Pacific and southern Pacific Ocean: a species in decline. Pages 93-113 in A.B. Bolten and B.E. Witherington, eds. Loggerhead Sea Turtles. Washington, D.C.: Smithsonian Institution.

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

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

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

MacLeod, C.D. 2009. Global climate change, range changes and potential implications for the conservation of marine cetaceans: a review and synthesis. Endangered Species Research 7:125-136.

MAFMC (Mid-Atlantic Fishery Management Council). 2007a. 2008 Atlantic mackerel, squid, and butterfish specifications, including an Environmental Assessment, Regulatory Impact Review, and Initial Flexibility Analysis. Mid-Atlantic Fishery Management Council. November 2007.

MAFMC (Mid-Atlantic Fishery Management Council). 2007b. 2008 Summer flounder, scup, and black sea bass specifications including an Environmental Assessment, Regulatory Impact Review, Initial Regulatory Flexibility Analysis and Essential Fish Habitat Assessment. Mid-Atlantic Fishery Management Council. November 2007.

MAFMC (Mid-Atlantic Fishery Management Council). 2009. 2010 Bluefish specifications, Environmental Assessment, Regulatory Impact Review, and Initial Regulatory Flexibility Analysis. Mid-Atlantic Fishery Management Council. December 2009.

MAFMC (Mid-Atlantic Fishery Management Council). 2010. Spiny Dogfish Specifications, Environmental Assessment, Regulatory Impact Review, and Initial Regulatory Flexibility Analysis. Mid-Atlantic Fishery Management Council. March 2010.

MAFMC (Mid-Atlantic Fishery Management Council) and ASMFC (Atlantic States Marine Fisheries Commission). 1998. Amendment 1 to the Bluefish Fishery Management Plan (includes Environmental Impact Statement and Regulatory Impact Review). Mid-Atlantic Fishery Management Council and Atlantic States Marine Fisheries Commission. October 1998.

Magnuson, J.J., K.A. Bjorndal, W.D. DuPaul, G.L. Graham, D.W. Owens, P.C.H. Pritchard, J.I. Richardson, G.E. Saul, and C.W. West. 1990. Decline of the sea turtles: causes and prevention. National Academy Press, Washington, D.C. 274 pp.

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

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

Malik, S., M.W. Brown, S.D. Kraus, and B.N. White. 2000. Analysis of mitochondrial DNA diversity within and between North and South Atlantic right whales. Marine Mammal Science 16:545-558.

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

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

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

Marcovaldi, M.A., and M. Chaloupka. 2007. Conservation status of the loggerhead sea turtle in Brazil: an encouraging outlook. Endangered Species Research 3:133-143.

Margaritoulis, D., R. Argano, I. Baran, F. Bentivegna, M.N. Bradai, J.A. Camiñas, P. Casale, G. De Metrio, A. Demetropoulos, G. Gerosa, B.J. Godley, D.A. Haddoud, J. Houghton, L. Laurent, and B. Lazar. 2003. Loggerhead turtles in the Mediterranean Sea: present knowledge and conservation perspectives. Pages 175-198 in A.B. Bolten and B.E. Witherington, eds. Loggerhead Sea Turtles. Washington, D.C.: Smithsonian Institution.

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

Martin, R.E. 1996. Storm impacts on loggerhead turtle reproductive success. Marine Turtle Newsletter 73:10-12.

Mate, B.M., S.L. Nieukirk, R. Mescar, and T. Martin. 1992. Application of remote sensing methods for tracking large cetaceans: North Atlantic right whales (*Eubalaena glacialis*). Final Report to the Minerals Management Service, Contract No. 14-12-0001-30411. 167 pp.

Mate, B.M., S.L. Nieukirk, and S.D. Kraus. 1997. Satellite monitored movements of the North Atlantic right whale. Journal of Wildlife Management 61:1393-1405.

Mazaris, A.D., G. Mastinos, and J.D. Pantis. 2009. Evaluating the impacts of coastal squeeze on sea turtle nesting. Ocean and Coastal Management 52:139-145.

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

Merrick, R., and H. Haas. 2008. Analysis of Atlantic sea scallop (*Placopecten magellanicus*) fishery impacts on the North Atlantic population of loggerhead sea turtles (*Caretta caretta*). NOAA Technical Memorandum NMFS-NE-207:1-22.

Meylan, A. 1982. Estimation of population size in sea turtles. Pages 135-138 in K.A. Bjorndal, ed. Biology and Conservation of Sea Turtles. Washington, D.C.: Smithsonian Institution Press.

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

Meylan, A., B.E. Witherington, B. Brost, R. Rivero, and P.S. Kubilis. 2006. Sea turtle nesting in Florida, USA: Assessments of abundance and trends for regionally significant populations of *Caretta*, *Chelonia*, and *Dermochelys*. Pages 306-307 in M. Frick, A. Panagopoulou, A. Rees, and K. Williams, compilers. Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation, Book of Abstracts.

Minton, G., T. Collins, C. Pomilla, K.P. Findlay, H. Rosenbaum, R. Baldwin, and R.L. Brownell Jr. 2008. *Megaptera novaeangliae* (Arabian Sea subpopulation). In 2010 IUCN Red List of Threatened Species, Version 2010.2. Accessed 23 August 2010. http://www.iucnredlist.org/apps/redlist/details/132835/0.

Mitchell, E., V.M. Kozicki, and R.R. Reeves. 1986. Sightings of right whales, *Eubalaena glacialis*, on the Scotian Shelf, 1966-1972. Reports of the International Whaling Commission, Special Issue 10:83-107.

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

Mizroch, S.A., and A.E. York. 1984. Have pregnancy rates of Southern Hemisphere fin whales, *Balaenoptera physalus*, increased? Reports of the International Whaling Commission, Special Issue 6:401-410.

Moore, J.C., and E. Clark. 1963. Discovery of right whales in the Gulf of Mexico. Science 141:269.

Moore, M.J., A.R., Knowlton, S.D. Kraus, W.A. McLellan, and R.K. Bonde. 2004. Morphometry, gross morphology and available histopathology in North Atlantic right whale (*Eubalaena glacialis*) mortalities (1970-2002). Journal of Cetacean Research and Management 6(3):199-214.

Moore, M.J., W.A. McLellan, P. Daous, R.K. Bonde, and A.R. Knowlton. 2007. Right whale mortality: a message from the dead to the living. Pages 358-379 in S.D. Kraus and R.M. Rolland, eds. The Urban Whale. Cambridge, Massachusetts and London, England: Harvard University Press.

Morreale, S.J. 1999. Oceanic migrations of sea turtles. Ph.D. diss., Cornell University.

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

NEFMC (New England Fishery Management Council). 1982. Fishery Management Plan, Final Environmental Impact Statement, Regulatory Impact Review for Atlantic sea scallops (*Placopecten magellanicus*). Prepared by the New England Fishery Management Council in consultation with Mid-Atlantic Fishery Management Council and South Atlantic Fishery Management Council. January 1982.

NEFMC (New England Fishery Management Council). 2003. Final Amendment 10 to the Atlantic Sea Scallop Fishery Management Plan with a Supplemental Environmental Impact Statement, Regulatory Impact Review, and Regulatory Flexibility Analysis. New England Fishery Management Council. November 2003.

NEFMC (New England Fishery Management Council). 2007. Final Amendment 11 to the Atlantic Sea Scallop Fishery Management Plan (FMP) including a Final Supplemental Environmental Impact Statement (FSEIS) and Initial Regulatory Flexibility Analysis (IRFA). Prepared by the New England Fishery Management Council in consultation with the National Marine Fisheries Service and the Mid-Atlantic Fishery Management Council. September 2007.

NEFMC (New England Fishery Management Council). 2009. Amendment 16 to the Northeast Multispecies Fishery Management Plan including a Final Supplemental Environmental Impact Statement (FSEIS) and Initial Regulatory Flexibility Analysis (IRFA). New England Fishery Management Council. October 2009.

NEFSC (Northeast Fisheries Science Center). 2003. Spiny dogfish (*Squalus acanthias*). Pages 133-283 in 37th Northeast Regional Stock Assessment Workshop (37th SAW): Stock Assessment Review Committee (SARC) consensus summary of assessments. NEFSC Reference Document 03-16. 597 pp.

NEFSC (Northeast Fisheries Science Center). 2005. Goosefish (Monkfish) Assessment Report. Pages 22-146 in 40th Northeast Regional Stock Assessment Workshop (40th SAW). NEFSC Reference Document 05-04. 146 pp.

NEFSC (Northeast Fisheries Science Center). 2006a. Bluefish Assessment Report. Pages 53-137 in 41st Northeast Regional Stock Assessment Workshop (41st SAW). NEFSC Reference Document 05-14. 237 pp.

NEFSC (Northeast Fisheries Science Center). 2006b. Assessment of Spiny Dogfish. Pages 14-163 in 43rd Northeast Regional Stock Assessment Workshop (43rd SAW). NEFSC Reference Document 06-25. 400 pp.

NEFSC (Northeast Fisheries Science Center). 2007. Assessment of Atlantic Sea Scallops. Pages 139-370 in 45th Northeast Regional Stock Assessment Workshop (45th SAW). NEFSC Reference Document 07-16. 380 pp.

NEFSC (Northeast Fisheries Science Center). 2009. Golden Tilefish Assessment Summary for 2009. Pages 12-25 in 48th Northeast Regional Stock Assessment Workshop (48th SAW). NEFSC Reference Document 09-10. 50 pp.

Nicholls, R.J. 1998. Coastal vulnerability assessment for sea level rise: evaluation and selection of methodologies for implementation. Technical Report R098002, Caribbean Planning for Adaption to Global Climate Change (CPACC) Project. Available at: www.cpacc.org.

NMFS (National Marine Fisheries Service). 1991. Final recovery plan for the humpback whale (*Megaptera novaeangliae*). Prepared by the Humpback Whale Recovery Team for the National Marine Fisheries Service, Silver Spring, Maryland. 105 pp.

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

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

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

NMFS (National Marine Fisheries Service). 1997b. Endangered Species Act Section 7 consultation on the continued hopper dredging of channels and borrow areas in the southeastern United States. Biological Opinion. September 25, 1997.

NMFS (National Marine Fisheries Service). 1998a. Final recovery plan for the shortnose sturgeon *(Acipenser brevirostrum)*. Prepared by the Shortnose Sturgeon Recovery Team for the National Marine Fisheries Service, Silver Spring, Maryland. October 1998.

NMFS (National Marine Fisheries Service). 1998b. Second reinitiation of Endangered Species Act Section 7 consultation on United States Coast Guard vessel and aircraft activities along the Atlantic coast. Biological Opinion. June 8, 1998.

NMFS (National Marine Fisheries Service). 1999a. Endangered Species Act Section 7 Consultation on the Fishery Management Plan for the Atlantic Bluefish Fishery and Amendment 1 to the Fishery Management Plan. Biological Opinion. July 2, 1999.

NMFS (National Marine Fisheries Service). 1999b. Endangered Species Act Section 7 Consultation on Federal Atlantic Herring Fishery Management Plan. Biological Opinion. September 17, 1999.

NMFS (National Marine Fisheries Service). 1999c. Endangered Species Act Section 7 Consultation regarding the Fishery Management Plan of Atlantic Mackerel, Squid, and Atlantic Butterfish Fishery and Amendment 8 to the Fishery Management Plan. Biological Opinion. April 28, 1999.

NMFS (National Marine Fisheries Service). 2000. Status review of smalltooth sawfish (*Pristis pectinata*). Silver Spring, Maryland: National Marine Fisheries Service.

NMFS (National Marine Fisheries Service). 2001a. Endangered Species Act Section 7 Consultation on authorization of fisheries under the Northeast Multispecies Fishery Management Plan. Biological Opinion. June 14, 2001.

NMFS (National Marine Fisheries Service). 2001b. Endangered Species Act Section 7 Consultation on authorization of fisheries under the Spiny Dogfish Fishery Management Plan. Biological Opinion. June 14, 2001.

NMFS (National Marine Fisheries Service). 2001c. Endangered Species Act Section 7 Consultation on authorization of fisheries under the Summer Flounder, Scup, and Black Sea Bass Fishery Management Plan. Biological Opinion. December 16, 2001.

NMFS (National Marine Fisheries Service). 2001d. Endangered Species Act Section 7 Consultation on NMFS' approval of the Tilefish Fishery Management Plan. Biological Opinion. March 13, 2001.

NMFS (National Marine Fisheries Service). 2002a. Final recovery plan for Johnson's seagrass *(Halophila johnsonii* Eiseman). Prepared by the Johnson's Seagrass Recovery Team for the National Marine Fisheries Service, Silver Spring, Maryland. September 2002.

NMFS (National Marine Fisheries Service). 2002b. Endangered Species Act Section 7 Consultation on Shrimp Trawling in the Southeastern United States, under the Sea Turtle Conservation Regulations and as Managed by the Fishery Management Plans for Shrimp in the South Atlantic and Gulf of Mexico. Biological Opinion. December 2, 2002.

NMFS (National Marine Fisheries Service). 2002c. Reinitiation of Consultation on Federal Lobster Management in the Exclusive Economic Zone for Implementation of Historical Participation. Biological Opinion. October 31, 2002.

NMFS (National Marine Fisheries Service). 2002d. Endangered Species Act Section 7 Consultation on the Implementation of the Deep-Sea Red Crab, *Chaceon quinquadens*, Fishery Management Plan. Biological Opinion. February 6, 2002.

NMFS (National Marine Fisheries Service). 2003a. Endangered and Threatened Species; Final Endangered Status for a Distinct Population Segment of Smalltooth Sawfish (*Pristis pectinata*) in the United States. Federal Register Final Rule, 68 FR 15674. April 1, 2003.

NMFS (National Marine Fisheries Service). 2003b. Endangered Species Act Section 7 consultation on authorization of fisheries under Monkfish Fishery Management Plan. Biological Opinion. April 14, 2003.

NMFS (National Marine Fisheries Service). 2003c. Endangered Species Act Section 7 consultation on authorization of fisheries under Skate Fishery Management Plan. Biological Opinion. July 24, 2003.

NMFS (National Marine Fisheries Service). 2003d. Endangered Species Act Section 7 consultation on the Fishery Management Plan for Dolphin and Wahoo fishery of the Atlantic Ocean. Biological Opinion. August 27, 2003.

NMFS (National Marine Fisheries Service). 2004a. Endangered Species Act Section 7 Reinitiated Consultation on the Continued Authorization of the Atlantic Pelagic Longline Fishery under the Fishery Management Plan for Atlantic Tunas, Swordfish, and Sharks (HMS FMP). Biological Opinion. June 1, 2004.

NMFS (National Marine Fisheries Service). 2004b. Endangered Species Act Section 7 Consultation on the Proposed Regulatory Amendments to the Fisheries Management Plan for the Pelagic Fisheries of the Western Pacific. Biological Opinion. February 23, 2004.

NMFS (National Marine Fisheries Service). 2004c. Final Environmental Assessment and Regulatory Impact Review Regulatory Flexibility Act Analysis on Sea Turtle Conservation Measures for the Pound Net Fishery in Virginia Waters of the Chesapeake Bay. Prepared by the National Marine Fisheries Service, Northeast Region. April 2004.

NMFS (National Marine Fisheries Service). 2005a. Recovery Plan for the North Atlantic Right Whale (*Eubalaena glacialis*). National Marine Fisheries Service: Silver Spring, Maryland.

NMFS (National Marine Fisheries Service). 2005b. Endangered Species Act Section 7 Consultation on the Continued Authorization of the Hawaii-based Pelagic, Deep-Set, Tuna Longline Fishery based on the Fishery Management Plan for Pelagic Fisheries of the Western Pacific Region. Biological Opinion. October 4, 2005.

NMFS (National Marine Fisheries Service). 2006a. Review of the Status of the Right Whales in the North Atlantic and North Pacific Oceans. Prepared by the National Marine Fisheries Service. December 2006.

NMFS (National Marine Fisheries Service). 2006b. Draft recovery plan for the fin whale (*Balaenoptera physalus*). National Marine Fisheries Service: Silver Spring, Maryland.

NMFS (National Marine Fisheries Service). 2006c. Draft Environmental Impact Statement (DEIS) to Implement the Operational Measures of the North Atlantic Right Whale Ship Strike Reduction Strategy. National Marine Fisheries Service. July 2006.

NMFS (National Marine Fisheries Service). 2006d. Endangered Species Act Section 7 consultation on the Continued Authorization of Snapper-Grouper Fishing in the U.S. South Atlantic Exclusive Economic Zone (EEZ) as Managed under the Snapper-Grouper Fishery Management Plan (SGFMP) of the South Atlantic Region, including Amendment 13C to the SGFMP. Biological Opinion. June 7, 2006.

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

NMFS (National Marine Fisheries Service). 2007b. Endangered Species Act Section 7 consultation on the Continued Authorization of Fishing under the Fishery Management Plan (FMP) for Coastal Migratory Pelagic Resources in Atlantic and Gulf of Mexico. Biological Opinion. August 13, 2007.

NMFS (National Marine Fisheries Service). 2007c. Final Environmental Impact Statement for amending the Atlantic Large Whale Take Reduction Plan: broad-based gear modifications. Volume I of II.

NMFS (National Marine Fisheries Service). 2007d. Endangered Species Act Section 7 consultation on the Northeast Fisheries Science Center Research Vessel Activities. Biological Opinion. August 20, 2007.

NMFS (National Marine Fisheries Service). 2007e. Reinitiation of Endangered Species Act Section 7 consultation on the issuance of license to Northeast Gateway Energy Bridge, LLC by MARAD to construct, own, and operate an LNG deepwater port. Biological Opinion. November 30, 2007.

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

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

NMFS (National Marine Fisheries Service). 2008c. Endangered Species Act Section 7 Consultation on the U.S. Navy Atlantic Fleet's conduct of active sonar training along the Atlantic Coast of the United States and in the Gulf of Mexico from January 2009 to January 2014. Biological Opinion. January 16, 2008.

NMFS (National Marine Fisheries Service). 2009a. Endangered and Threatened Species; Determination of Endangered Status for the Gulf of Maine Distinct Population Segment of Atlantic Salmon. Federal Register Final Rule, 74 FR 29344. June 19, 2009.

NMFS (National Marine Fisheries Service). 2009b. Endangered and Threatened Species; Designation of Critical Habitat for Atlantic Salmon (*Salmo salar*) Gulf of Maine Distinct Population Segment. Federal Register Final Rule, 74 FR 29300. June 19, 2009.

NMFS (National Marine Fisheries Service). 2009c. Recovery Plan for Smalltooth Sawfish (*Pristis pectinata*). Prepared by the Smalltooth Sawfish Recovery Team for the National Marine Fisheries Service, Silver Spring, Maryland.

NMFS (National Marine Fisheries Service). 2009d. Endangered and Threatened Species; Critical Habitat for the Endangered Distinct Population Segment of Smalltooth Sawfish. Federal Register Final Rule, 74 FR 45353. September 2, 2009.

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

NMFS (National Marine Fisheries Service). 2009f. Endangered Species Act Section 7 Consultation on the U.S. Navy Atlantic Fleet's conduct of active sonar training along the Atlantic Coast of the United States and in the Gulf of Mexico from January 2009 to January 2010. Biological Opinion. January 21, 2009.

NMFS (National Marine Fisheries Service). 2010a. Fisheries of the Northeastern United States; Atlantic Bluefish Fishery; 2010 Atlantic Bluefish Specifications. Federal Register Final rule, 75 FR 27221. May 14, 2010.

NMFS (National Marine Fisheries Service). 2010b. Endangered Species Act Section 7 consultation on the Issuance of license to Neptune LNG by MARAD to own and operate an LNG deepwater port. Biological Opinion. July 12, 2010.

NMFS (National Marine Fisheries Service). 2010c. Draft Environmental Impact Statement for Proposed Effort Control Measures for the American Lobster Fishery. March 2010.

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 1991. Recovery plan for U.S. population of Atlantic green turtle *Chelonia mydas*. Washington, D.C.: National Marine Fisheries Service. 58 pp.

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

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

NMFS (National Marine Fisheries Service) and USFWS (U.S. Fish and Wildlife Service). 1998a. Recovery Plan for U.S. Pacific Populations of the Loggerhead Turtle (*Caretta caretta*). Silver Spring, Maryland: National Marine Fisheries Service. 59 pp.

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

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

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

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

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

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

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

NMFS NERO (Northeast Regional Office). 2010. Preliminary summary of western North Atlantic large whale entanglement with Serious Injury and Mortality determinations. Version 08/19/2010.

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

NMFS SEFSC (Southeast Fisheries Science Center). 2008. Careful release protocols for sea turtle release with minimal injury. NOAA Technical Memorandum NMFS-SEFSC-580. 130 pp.

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

NOAA (National Oceanic and Atmospheric Administration). 2008. High numbers of right whales seen in Gulf of Maine: NOAA researchers identify wintering ground and potential breeding ground. NOAA press release; December 31, 2008.

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

NRC (National Research Council). 2003. Ocean noise and marine mammals. Washington, D.C.: National Academy Press. 204 pp.

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

Pace, R.M., III. *In review*. Scaling the influence of anthropogenic mortality reduction on recovery prospects of North Atlantic right whales. Submitted to Biological Conservation.

Pace, R.M., III, and R.L. Merrick. 2008. Northwest Atlantic Ocean habitats important to the conservation of North Atlantic right whales (*Eubalaena glacialis*). NEFSC Reference Document 08-07. 24 pp.

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

Palka, D.L. 2006. Summer abundance estimates of cetaceans in US North Atlantic Navy Operating Areas. NEFSC Reference Document 06-03; 41 pp.

Patrician, M.R., I.S. Biedron, H. Carter Esch, F.W. Wenzel, L.A. Cooper, P.K. Hamilton, A.H. Glass, and M.F. Baumgartner. 2009. Evidence of a North Atlantic right whale calf (*Eubalaena glacialis*) born in northeastern U.S. waters. Marine Mammal Science 25(2):462-477.

Payne, P.M., J.R. Nicholas, L. O'Brien, and K.D. Powers. 1986. The distribution of the humpback whale, *Megaptera novaeangliae*, on Georges Bank and in the Gulf of Maine in relation to densities of the sand eel, *Ammodytes americanus*. Fishery Bulletin 84(2):271-277.

Payne, P.M., D.N. Wiley, S.B. Young, S. Pittman, P.J. Clapham, and J.W Jossi. 1990. Recent fluctuations in the abundance of baleen whales in the southern Gulf of Maine in relation to changes in selected prey. Fishery Bulletin 88(4):687-696.

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

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

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

Peterson, C., G. Monahan, and F. Schwartz. 1985. Tagged green turtle returns and nests again in North Carolina. Marine Turtle Newsletter 35:5-6.

Pike, D.A., and J.C. Stiner. 2007. Sea turtle species vary in their susceptibility to tropical cyclones. Oecologia 153: 471-478.

Pike, D.A., R.L. Antworth, and J.C. Stiner. 2006. Earlier nesting contributes to shorter nesting seasons for the loggerhead sea turtle, *Caretta caretta*. Journal of Herpetology 40(1):91-94.

Plaziat, J.C., and P.G.E.F. Augustinius. 2004. Evolution of progradation/erosion along the French Guiana mangrove coast: a comparison of mapped shorelines since the 18th century with Holocene data. Marine Geology 208:127-143.

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

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

Prusty, G., S. Dash, and M.P. Singh. 2007. Spatio-temporal analysis of multi-date IRS imageries for turtle habitat dynamics characterisation at Gahirmatha coast, India. International Journal of Remote Sensing 28: 871-883.

Rahmstorf, S. 1997. Risk of sea-change in the Atlantic. Nature 388:825-826.

Rahmstorf, S. 1999. Shifting seas in the greenhouse? Nature 399:523-524.

Read, A.J. 1994. Interactions between cetaceans and gillnet and trap fisheries in the northwest Atlantic. Reports of the International Whaling Commission, Special Issue 15:133-147.

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

Reddin, D.G. 2006. Perspectives on the marine ecology of Atlantic salmon (*Salmo salar*) in the Northwest Atlantic. Canadian Science Advisory Secretariat. Research Document 2006/018.

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

Reilly, S.B., J.L Bannister, P.B. Best, M. Brown, R.L. Brownell Jr., D.S. Butterworth, P.J. Clapham, J. Cooke, G.P. Donovan, J. Urbán, and A.N. Zerbini. 2008a. *Megaptera novaeangliae*. In 2010 IUCN Red List of Threatened Species, Version 2010.2. Accessed 23 August 2010. http://www.iucnredlist.org/apps/redlist/details/13006/0.

Reilly, S.B., J.L Bannister, P.B. Best, M. Brown, R.L. Brownell Jr., D.S. Butterworth, P.J. Clapham, J. Cooke, G.P. Donovan, J. Urbán, and A.N. Zerbini. 2008b. *Eubalaena glacialis*. In 2010 IUCN Red List of Threatened Species, Version 2010.2. Accessed 23 August 2010. http://www.iucnredlist.org/apps/redlist/details/41712/0.

Richardson, W.J., C.I. Malme, C.R. Green, and D.H. Thomson. 1995. Marine Mammals and noise: Volume 1. San Diego: Academic Press.

Rivalan, P., P.H. Dutton, E. Baudry, S.E. Roden, and M. Girondot. 2005. Demographic scenario inferred from genetic data in leatherback turtles nesting in French Guiana and Suriname. Biological Conservation 1:1-9.

Robbins, J. 2007. Structure and dynamics of the Gulf of Maine humpback whale population. PhD thesis, University of St. Andrews.

Robbins, J., and D. Mattila. 2004. Estimating humpback whale (*Megaptera novaeangliae*) entanglement rates on the basis of scar evidence. Report to the Northeast Fisheries Science Center, National Marine Fisheries Service. Order no. 43EANF030121. 21 pp.

Robbins, J., S. Landry, and D.K. Mattilla. 2009. Estimating entanglement related mortality from scar-based studies. Scientific Committee Meeting of the International Whaling Commission. SC/61/BC3.

Rolland, R.M, K.E. Hunt, G.J. Doucette, L.G. Rickard, and S. K. Wasser. 2007. The Inner Whale: Hormones, Biotoxins, and Parasites. Pages 232-272 in S.D. Kraus and R.M. Rolland, eds. The Urban Whale. Harvard University Press, Cambridge, Massachusetts, London, England. vii-xv + 543 pp.

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

Ross, J.P. 2005. Hurricane effects on nesting *Caretta caretta.* Marine Turtle Newsletter 108:13-14.

Ruben, H.J., and S.J. Morreale. 1999. Draft Biological Assessment for sea turtles in the New York and New Jersey Harbor Complex. Prepared for the National Marine Fisheries Service.

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

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

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

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

Schaeff, C.M., S.D. Kraus, M.W. Brown, J.S. Perkins, R. Payne, and B.N. White. 1997. Comparison of genetic variability of North and South Atlantic right whales (*Eubalaena*), using DNA fingerprinting. Canadian Journal of Zoology 75:1073-1080.

Schevill, W.E., W.A. Watkins, and K.E. Moore. 1986. Status of *Eubalaena glacialis* off Cape Cod. Report of the International Whaling Commission, Special Issue 10:79-82.

Schick, R.S., P.N. Halpin, A.J. Read, C.K. Slay, S.D. Kraus, B.R. Mate, M.F. Baumgartner, J.J. Roberts, B.D. Best, C.P. Good, S.R. Loarie, and J.S. Clark. 2009. Striking the right balance in right whale conservation. Canadian Journal of Fisheries and Aquatic Sciences 66:1399-1403.

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

Schmidly, D.J., C.O. Martin, and G.F. Collins. 1972. First occurrence of a black right whale (*Balaena glacialis*) along the Texas coast. The Southwestern Naturalist 17:214-215.

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

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

Seipt, I., P.J. Clapham, C.A. Mayo, and M.P. Hawvermale. 1990. Population characteristics of individually identified fin whales, *Balaenoptera physalus*, in Massachusetts Bay. Fishery Bulletin 88:271-278.

Sella, I. 1982. Sea turtles in the Eastern Mediterranean and Northern Red Sea. Pages 417-423 in K.A. Bjorndal, ed. Biology and Conservation of Sea Turtles. Washington, D.C.: Smithsonian Institution Press.

Seminoff, J.A. 2004. *Chelonia mydas*. In 2007 IUCN Red List of Threatened Species. Accessed 31 July 2009. http://www.iucnredlist.org/search/details.php/4615/summ.

Seney, E.E., and J.A. Musick. 2005. Diet analysis of Kemp's ridley sea turtles (*Lepidochelys kempii*) in Virginia. Chelonian Conservation and Biology 4(4):864-871.

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

Shaffer, M.L. 1981. Minimum population sizes for species conservation. BioScience 31(2):131-134.

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

Shepherd, G. 2006. Status of Fisheries Resources off Northeastern US - Bluefish. Accessed 10 September 2010. http://www.nefsc.noaa.gov/sos/spsyn/op/bluefish/.

Shepherd, G.R., and D.B. Packer. 2006. Essential Fish Habitat Source Document: Bluefish, *Pomatomus saltatrix*, Life History and Habitat Characteristics (2nd Edition). NOAA Technical Memorandum NMFS-NE-198.

Shepherd, G.R., J. Moser, D. Deuel, and P. Carlsen. 2006. The migration pattern of bluefish (*Pomatomus saltatrix*) along the Atlantic coast determined from tag recoveries. Fishery Bulletin 104(4):559-570.

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

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

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

Short, F.T., and H.A. Neckles. 1999. The effects of global climate change on seagrasses. Aquatic Botany 63:169-196.

Smith, T.D., J. Allen, P.J. Clapham, P.S. Hammond, S. Katona, F. Larsen, J. Lien, D. Mattila, P.J. Palsboll, J. Sigurjonsson, P.T. Stevick, and N. Oien. 1999. An ocean-basin-wide mark-recapture study of the North Atlantic humpback whale (*Megaptera novaeangliae*). Marine Mammal Science 15(1):1-32.

Snover, M. 2005. Population trends and viability analyses for Pacific marine turtles. PIFSC Internal Report IR-05-08:1-33.

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

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

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

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

Stephens, S.H., and J. Alvarado-Bremer. 2003. Preliminary information on the effective population size of the Kemp's ridley (*Lepidochelys kempii*) sea turtle. Page 250 in J.A. Seminoff, compiler. Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503.

Stevick P.T., J. Allen, P.J. Clapham, N. Friday, S.K. Katona, F. Larsen, J. Lien, D.K. Matilla, P.J. Palsboll, J. Sigurjonsson, T.D. Smith, N. Oien, and P.S. Hammond. 2003. North Atlantic humpback whale abundance and rate of increase four decades after protection from whaling. Marine Ecology Progress Series 258:263-273.

Stevick, P.T., J. Allen, P.J. Clapham, S.K. Katona, F. Larsen, J. Lien, D.K. Mattila, P.J. Palsboll, R. Sears, J. Sigurjonsson, T.D. Smith, G. Vikingsson, N. Oien, and P.S. Hammond. 2006. Population spatial structuring on the feeding grounds in North Atlantic humpback whales (*Megaptera novaeangliae*). Journal of Zoology 270:244-255.

Stocker, T.F., and A. Schmittner. 1997. Influence of CO_2 emission rates on the stability of the thermohaline circulation. Nature 388:862-865.

Stone, G.S., L. Flores-Gonzalez, and S. Cotton. 1990. Whale migration record. Nature 346:705.

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

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

Swingle, W.M., S.G. Barco, T.D. Pitchford, W.A. McLellan, and D.A. Pabst. 1993. Appearance of juvenile humpback whales feeding in the nearshore waters of Virginia. Marine Mammal Science 9:309-315.

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

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

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

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

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

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

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

Townsend, D.W., A.C. Thomas, L.M. Mayer, M. Thomas and J. Quinlan. 2006. Oceanography of the Northwest Atlantic Continental Shelf. Pages 119-168 in A.R. Robinson and K.H. Brink, eds. The Sea, Volume 14. Cambridge: Harvard University Press.

Tynan, C.T., and D.P. DeMaster. 1997. Observations and predictions of Arctic climatic change: potential effects on marine mammals. Arctic 50:308-322.

USFWS (U.S. Fish and Wildlife Service) and NMFS (National Marine Fisheries Service). 1992. Recovery plan for the Kemp's ridley sea turtle (*Lepidochelys kempii*). St. Petersburg, Florida: National Marine Fisheries Service. 40 pp.

Van Houton, K.S., and O.L. Bass. 2007. Stormy oceans are associated with declines in sea turtle hatching. Current Biology 17:R590.

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

Wallace, N. 1985. Debris entanglement in the marine environment. Pages 259-277 in R.S. Shomura and H.O. Yoshida, eds. Proceedings of the workshop on the fate and impact of marine debris, 27-29 November, 1984, Honolulu, Hawaii. NOAA Technical Memorandum NMFS-SWFC-54.

Waluda, C.M., P.G. Rodhouse, G.P. Podestá, P.N. Trathan, and G.J. Pierce. 2001. Surface oceanography of the inferred hatching grounds of *Illex argentinus* (Cephalopoda: Ommastrephidae) and influences on recruitment variability. Marine Biology 139:671-679.

Waring, G.T., J.M. Quintal, and C.P. Fairfield (eds.). 2002. U.S. Atlantic and Gulf of Mexico marine mammal stock assessments - 2002. NOAA Technical Memorandum NMFS-NE-169:1-318.

Waring, G.T., E. Josephson, K. Maze-Foley, and P.E. Rosel (eds.). 2009. U.S. Atlantic and Gulf of Mexico marine mammal stock assessments - 2009. NOAA Technical Memorandum NMFS-NE-213:1-540.

Waring, G.T., E. Josephson, K. Maze-Foley, and P.E. Rosel (eds.). 2010. Draft U.S. Atlantic and Gulf of Mexico marine mammal stock assessments - 2010. Available from: http://www.nmfs.noaa.gov/pr/sars/draft.htm.

Watkins, W.A., and W.E. Schevill. 1982. Observations of right whales (*Eubalaena glacialis*) in Cape Cod waters. Fishery Bulletin 80(4):875-880.

Watkins, W.A., K.E. Moore, J. Sigurjonsson, D. Wartzok, and G. Notarbartolo di Sciara. 1984. Fin whale (*Balaenoptera physalus*) tracked by radio in the Irminger Sea. Rit Fiskideildar 8(1):1-14.

Webster, P.J., G.J. Holland, J.A. Curry, and H.R. Chang. 2005. Changes in tropical cyclone number, duration, and intensity in a warming environment. Science 309:1844-1846.

Weisbrod, A.V., D. Shea, M.J. Moore, and J.J. Stegeman. 2000. Organochlorine exposure and bioaccumulation in the endangered Northwest Atlantic right whale (*Eubalaena glacialis*) population. Environmental Toxicology and Chemistry 19(3):654-666.

Weishampel, J.F., D.A. Bagley, and L.M. Ehrhart. 2004. Earlier nesting by loggerhead sea turtles following sea surface warming. Global Change Biology 10:1424-1427.

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

Wilcox, J.R., G. Bouska, J. Gorham. B. Peery, and M. Bresette. 1998. Knee deep in green turtles: Recent trends in capture rates at the St. Lucie nuclear power plant. Pages 147-148 in R. Byles and Y. Fernandez (compilers). Proceedings of the Sixteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-412.

Wiley, D.N., R.A. Asmutis, T.D. Pitchford, and D.P. Gannon. 1995. Stranding and mortality of humpback whales, *Megaptera novaeangliae*, in the mid-Atlantic and southeast United States, 1985-1992. Fishery Bulletin 93(1):196-205.

Winn, H.E., C.A. Price, and P.W. Sorensen. 1986. The distributional biology of the right whale (*Eubalaena glacialis*) in the western North Atlantic. Reports of the International Whaling Commission (Special issue) 10:129-138.

Wise, J.P., S.S. Wise, S. Kraus, R. Shaffley, M. Grau, T.L. Chen, C. Perkins, W.D. Thompson, T. Zhang, Y. Zhang, T. Romano and T. O'Hara. 2008. Hexavalent chromium is cytotoxic and genotoxic to the North Atlantic right whale (*Eubalaena glacialis*) lung and testes fibroblasts. Mutation Research - Genetic Toxicology and Environmental Mutagenesis. 650(1):30-38.

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

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

Witt, M.J., A.C. Broderick, M. Coyne, A. Formia, S. Ngouessono, R.J. Parnell, G.-P. Sounguet, and B.J. Godley. 2008. Satellite tracking highlights difficulties in the design of effective protected areas for critically endangered leatherback turtles *Dermochelys coriacea* during the inter-nesting period. Oryx 42:296-300.

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

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

Zemsky, V., A.A. Berzin, Y.A. Mikhaliev, and D.D. Tormosov. 1995. Soviet Antarctic pelagic whaling after WWII: review of actual catch data. Report of the Sub-committee on Southern Hemisphere baleen whales. Reports of the International Whaling Commission 45:131-135.

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

Zurita, J.C., R. Herrera, A. Arenas, M.E. Torres, C. Calderon, L. Gomez, J.C. Alvarado, and R. Villavicencio. 2003. Nesting loggerhead and green sea turtles in Quintana Roo, Mexico. Pages 125-127 in J.A. Seminoff, compiler. Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503.